Regulatory Impact Analysis of the Proposed Revisions to the National Ambient Air Quality Standards for Ground-Level Ozone

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Overview

In setting primary national ambient air quality standards (NAAQS), the EPA's responsibility under the law is to establish standards that protect public health. The Clean Air Act (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 base this decision on health considerations only; economic factors cannot be considered. The prohibition against considering cost in the setting of the primary air quality standards does not mean that costs, benefits or other economic considerations are unimportant. The Agency believes that consideration of costs and benefits is an essential decision-making tool for the efficient implementation of these standards. The impacts of costs, benefits, and efficiency are considered by the States when they make decisions regarding what timelines, strategies, and policies are appropriate for their circumstances.

The EPA is proposing to revise the level of the ozone NAAQS to within a range of 65 ppb to 70 ppb and is soliciting comment on alternative standard levels below 65 ppb, as low as 60 ppb. The EPA is also proposing to revise the level of the secondary standard to within the range of 65 ppb to 70 ppb to provide increased protection against vegetation-related effects on public welfare.[1] The EPA performed an illustrative analysis of the potential costs, human health benefits, and welfare co-benefits of nationally attaining primary alternative ozone standard levels and did not estimate any incremental costs and benefits associated with attaining a revised secondary standard. Per Executive Order 12866 and the guidelines of OMB Circular A-4, this Regulatory Impact Analysis (RIA) presents the analyses of the following alternative standard levels -- 70 ppb, 65 ppb, and 60 ppb. The cost and benefit estimates below are calculated incremental to a 2025 baseline that incorporates air quality improvements achieved through the projected implementation of existing regulations and full attainment of the existing ozone

[1] As an initial matter, the EPA is proposing that ambient ozone concentrations in terms of a three-year average W126 index value within the range from 13 parts per million-hours (ppm-hours) to 17 ppm-hours would provide the requisite protection against known or anticipated adverse effects to the public welfare, which data analyses indicate would provide air quality in terms of three-year average W126 index values of a range at or below 13 ppm-hours to 17 ppm-hours. Data analyses also indicate that actions taken to attain a standard in the range of 65 ppb to 70 ppb would also improve air quality as measured by the W126 metric.

NAAQS (75 ppb). The 2025 baseline reflects, among other existing regulations, the Mercury and Air Toxics Standard, the Clean Air Interstate Rule, the Tier 3 Motor Vehicle Emission and Fuel Standards, and adjustments for the Clean Power Plan, all of which will help many areas move toward attainment of the existing ozone standard (see Chapter 3, Section 3.1.3 for additional information).

In this RIA we present the primary costs and benefits estimates for 2025. We assume that potential nonattainment areas everywhere in the U.S., excluding California, will be designated such that they are required to reach attainment by 2025, and we developed our projected baselines for emissions, air quality, and populations for 2025.

The EPA will likely finalize designations for a revised ozone NAAQS in late 2017. Depending on the precise timing of the effective date of those designations, nonattainment areas classified as Marginal will likely have to attain in either late 2020 or early 2021. Nonattainment areas classified as Moderate will likely have to attain in either late 2023 or early 2024. If a Moderate nonattainment area qualifies for two 1-year extensions, the area may have as late as 2026 to attain. Lastly, Serious nonattainment areas will likely have to attain in late 2026 or early 2027. We selected 2025 as the primary year of analysis because most areas of the U.S. will likely be required to meet a revised ozone standard by 2025 and because it provided a good representation of the remaining air quality concerns that Moderate nonattainment areas would face; states with areas classified as Moderate and higher are required to develop attainment demonstration plans for those nonattainment areas.

In estimating the incremental costs and benefits of potential alternative standards, we recognize that there are several areas that are not required to meet the existing ozone standard by 2025. The Clean Air Act allows areas with more significant air quality problems to take additional time to reach the existing standard. Several areas in California are not required to meet the existing standard by 2025 and may not be required to meet a revised standard until sometime between 2032 and 2037.[2] We were not able to project emissions and air quality

[2] The EPA will likely finalize designations for a revised ozone NAAQS in late 2017. Depending on the precise timing of the effective date of those designations, nonattainment areas classified as Severe 15 will likely have to

beyond 2025 for California, however, we adjusted baseline air quality to reflect mobile source emissions reductions for California that would occur between 2025 and 2030; these emissions reductions were the result of mobile source regulations expected to be fully implemented by 2030. While there is uncertainty about the precise timing of emissions reductions and related costs for California, we assume costs occur through the end of 2037 and beginning of 2038. In addition, we model benefits for California using projected population demographics for 2038.

Because of the different timing for incurring costs and accruing benefits and for ease of discussion throughout the analyses, we refer to the different time periods for potential attainment as 2025 and post-2025 to reflect that (1) we did not project emissions and air quality for any year other than 2025; (2) for California, emissions controls and associated costs are assumed to occur through the end of 2037 and beginning of 2038; and (3) for California benefits are modeled using population demographics in 2038. It is not straightforward to discount the post-2025 results for California to compare with or add to the 2025 results for the rest of the U.S. While we estimate benefits using 2038 information, we do not have good information on precisely when the costs of controls will be incurred. Because of these differences in timing related to California attaining a revised standard, the separate costs and benefits estimates for post-2025 should not be added to the primary estimates for 2025.

ES.1 Overview of Analytical Approach

This RIA consists of multiple analyses including an assessment of the nature and sources of ambient ozone (Chapter 2 – Defining the Air Quality Problem); estimates of current and future emissions of relevant precursors that contribute to the problem; air quality analyses of baseline and alternative control strategies (Chapter 3 – Air Quality Modeling and Analysis); development of illustrative control strategies to attain the primary alternative standard levels (Chapter 4 – Control Strategies and Emissions Reductions); estimates of the incremental benefits of attaining the primary alternative standard levels (Chapter 5 – Human Health Benefits); a qualitative discussion of the welfare co-benefits of attaining the primary alternative standard levels (Chapter 6 – Welfare Co-Benefits of the Primary Standard); estimates of the incremental

attain sometime between late 2032 and early 2033 and nonattainment areas classified as Extreme will likely have to attain by December 31, 2037.

costs of attaining the primary alternative standard levels (Chapter 7 – Engineering Cost Analysis and Economic Impacts); a comparison and discussion of the benefits and costs (Chapter 8 – Comparison of Costs and Benefits); an analysis of the impacts of the relevant statutory and executive orders (Chapter 9 – Statutory and Executive Order Impact Analysis); and a discussion of the theoretical framework used to analyze regulation-induced employment impacts, as well as information on employment related to installation of NOx controls on coal and gas-fired electric generating units, industrial boilers, and cement kilns (Chapter 10 – Qualitative Discussion of Employment Impacts of Air Quality).

Because States are ultimately responsible for implementing strategies to meet revised standards, this RIA provides insights and analysis of a limited number of illustrative control strategies that states might adopt to meet a revised standard. The goal of this RIA is to provide estimates of the costs and benefits of the illustrative attainment strategies to the meet each alternative standard level. The flowchart below (Figure ES-1) outlines the analytical steps taken to illustrate attainment with the potential alternative standard levels, and the following discussion, by primary flowchart section, describes the steps taken.

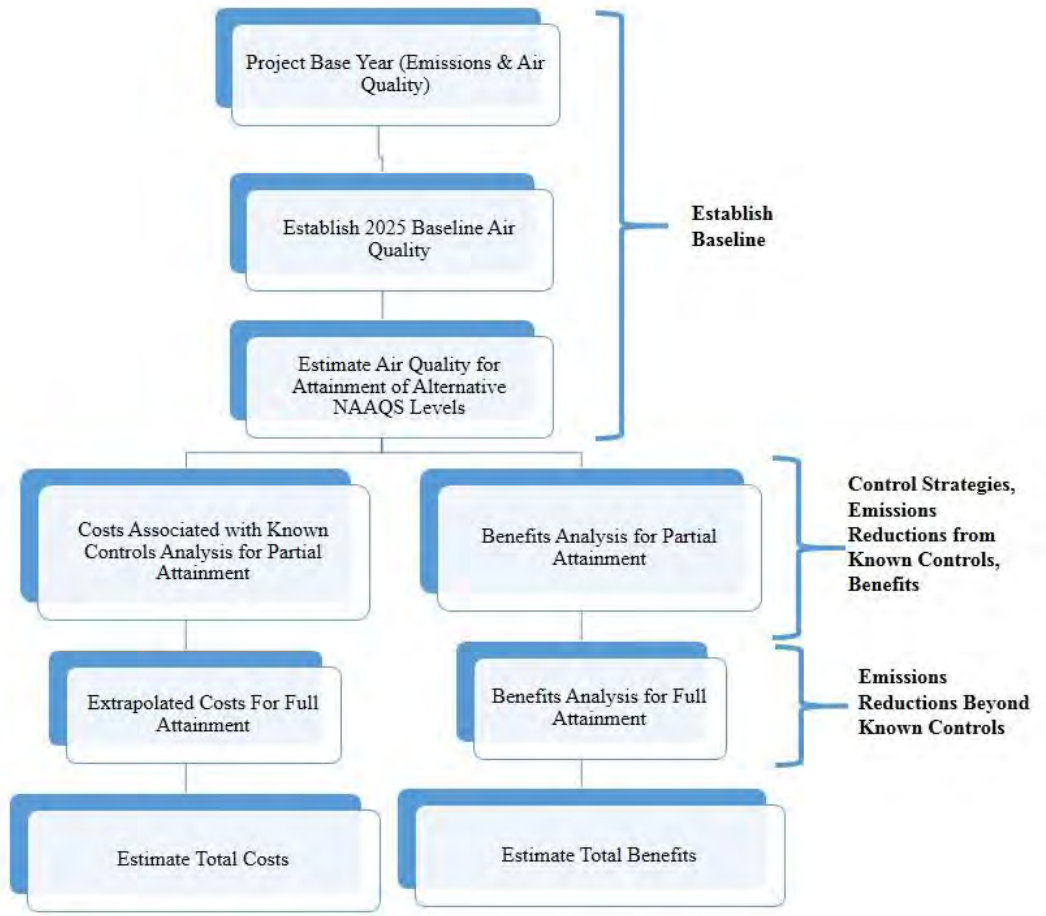

Figure ES-1. Analytical Flowchart for Primary Standards Analyses

ES.1.1 Establishing the Baseline

The future year base case reflects emissions projected from 2011 to 2025 and incorporates current state and federal programs, including the Tier 3 Motor Vehicle Emission and Fuel Standards (U.S. EPA, 2014a) (see Chapter 3, Section 3.1.3 for a discussion of the rules included in the base case). The base case does not include control programs specifically for the purpose of attaining the existing ozone standard (75 ppb). The baseline builds on the future year base case and reflects the additional emissions reductions needed to reach attainment of the current ozone standard (75 ppb), as well as adjustments for the Clean Power Plan (U.S. EPA, 2014b).

We performed a national scale air quality modeling analysis to estimate ozone concentrations for the future base case year of 2025. To accomplish this, we modeled multiple emissions cases for 2025, including the 2025 base case and twelve 2025 emissions sensitivity simulations. The twelve emissions sensitivity simulations were used to develop ozone sensitivity factors (ppb/ton) from the modeled response of ozone to changes in NOx and VOC emissions from various sources and locations. These ozone sensitivity factors were then used to determine the amount of emissions reductions needed to reach the 2025 baseline and evaluate potential alternative standard levels of 70, 65, and 60 ppb incremental to the baseline. We used the estimated emissions reductions needed to reach each of these standard levels to analyze the costs and benefits of alternative standard levels.

ES.1.2 Control Strategies and Emissions Reductions

The EPA analyzed illustrative control strategies that areas across the U.S. might employ to attain alternative revised primary ozone standard levels of 70, 65, and 60 ppb. The EPA analyzed the impact that additional emissions control technologies and measures, across numerous sectors, would have on predicted ambient ozone concentrations incremental to the baseline. These control measures, also referred to as known controls, are based on information available at the time of this analysis and include primarily end-of-pipe control technologies. In addition, to attain some of the alternative primary standard levels analyzed, some areas needed additional emissions reductions beyond the known controls, and we refer to these as unknown controls (see Chapter 7, Section 7.2 for additional information).

Using average ozone response factors, we estimated the portion of the emissions reductions required to meet the baseline, including any additional emissions reductions beyond known controls. We then estimated the emissions reductions incremental to the baseline that were needed to meet the alternative standard levels of 70, 65, and 60 ppb. Costs of controls incremental to (i.e., over and above) the baseline emissions reductions are attributed to the costs of meeting the alternative standard levels. These emissions reductions can come from both specific known controls, as well as unknown controls. The baseline shows that by 2025, while ozone air quality would be significantly better than today under current requirements, depending on the alternative standard level analyzed, several areas in the Eastern, Central, and Western U.S. would need to develop and adopt additional controls to attain alternative standard levels (see Chapter 4, Section 4.3).

ES.1.2.1 Emissions Reductions from Known Controls in 2025

Figure ES-2 shows the counties projected to exceed the alternative standard levels analyzed for 2025 for areas other than California. For the 70 ppb alternative standard level, emissions reductions were required for monitors in the Central and Northeast regions. For the 65 and 60 ppb alternative standard levels, emissions reductions were applied in all regions with projected baseline design values (DVs) above these levels.[3] For the 60 ppb alternative standard level, additional VOC emissions reductions were identified in Chicago because some sites in that area experienced NO_x disbenefits, meaning that the regional NO_x emissions reductions resulted in ozone increases from below 60 ppb to above 60 ppb. Tables ES-1 through ES-3 show the emissions reductions from known controls for the alternative standard levels analyzed.

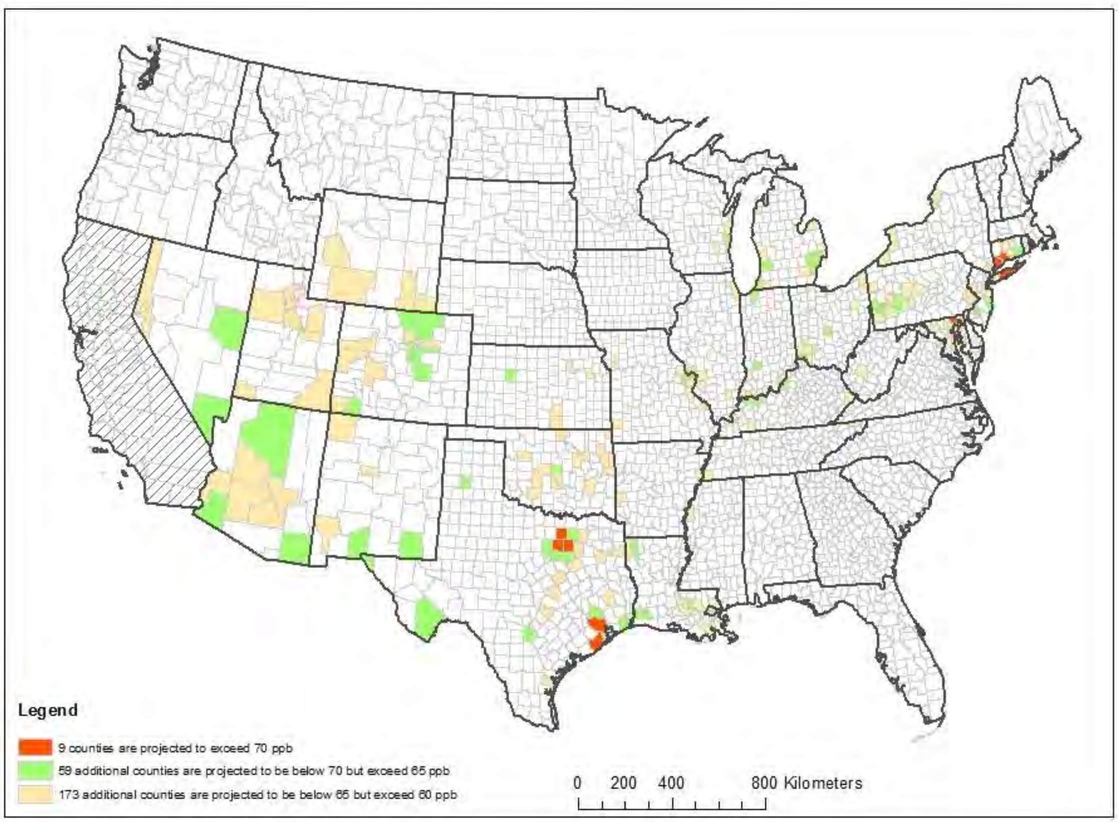

Figure ES-2. Projected Ozone Design Values in the 2025 Baseline Scenario

[3] A design value is a statistic that describes the air quality status of a given area relative to the level of the NAAQS. Design values are typically used to classify nonattainment areas, assess progress toward meeting the NAAQS, and develop control strategies.

Table ES-1. Summary of Emission Reductions by Sector for Known Controls for 70 ppb Proposed Alternative Standard Level for 2025, except California (1,000 tons/year)[a]

Geographic Area	Emissions Sector	NO_x	VOC
East	EGU	25	-
	Non-EGU Point	210	0.98
	Nonpoint	260	54
	Nonroad	5	-
	Total	**490**	**55**
West	EGU	-	-
	Non-EGU Point	-	-
	Nonpoint	-	-
	Nonroad	-	-
	Total	**-**	**-**

[a] Estimates are rounded to two significant figures.

Table ES-2. Summary of Emission Reductions by Sector for Known Controls for 65 ppb Proposed Alternative Standard Level for 2025 - except California (1,000 tons/year)[a]

Geographic Area	Emissions Sector	NO_x	VOC
East	EGU	170	-
	Non-EGU Point	410	3.6
	Nonpoint	420	95
	Nonroad	12	-
	Total	**1,000**	**99**
West	EGU	36	-
	Non-EGU Point	38	0.47
	Nonpoint	37	6.6
	Nonroad	1.3	-
	Total	**110**	**7**

[a] Estimates are rounded to two significant figures.

Table ES-3. Summary of Emission Reductions by Sector for Known Controls for 60 ppb Alternative Standard Level for 2025 - except California (1,000 tons/year)[a]

Geographic Area	Emissions Sector	NO_x	VOC
East	EGU	170	-
	Non-EGU Point	410	4.2
	Nonpoint	420	99
	Nonroad	12	-
	Total	**1,000**	**100**
West	EGU	62	-
	Non-EGU Point	48	0.47
	Nonpoint	39	6.6
	Nonroad	1.3	-
	Total	**150**	**7**

[a] Estimates are rounded to two significant figures.

ES.1.2.2 Emissions Reductions beyond Known Controls in 2025

There were several areas where known controls did not achieve enough emissions reductions to attain the proposed alternative standard levels of 70 and 65 as well as the more stringent alternative standard level of 60 ppb. To complete the analysis, the EPA then estimated the additional emissions reductions beyond known controls needed to reach attainment (i.e., unknown controls). Table ES-4 shows the emissions reductions needed from unknown controls in 2025 for the U.S., except California, for the alternative standard levels analyzed.

Table ES-4. Summary of Emissions Reductions by Alternative Standard for Unknown Controls for 2025 - except California (1,000 tons/year)[a]

	Region	NO_x	VOC
Proposed Alternative Standard Levels			
70 ppb	**East**	**150**	**-**
	West	**-**	**-**
65 ppb	**East**	**750**	**-**
	West	**-**	**-**
Alternative Standard Level			
60 ppb	East	1,900	41
	West	350	-

ES.1.2.3 Emissions Reductions beyond Known Controls for Post-2025

Figure ES-3 shows the counties projected to exceed the alternative standard levels analyzed for the post-2025 analysis for California. For the California post-2025 alternative standard level analyses, all known controls were applied in the baseline, so incremental emissions reductions are from unknown controls. Table ES-5 shows the emissions reductions needed from unknown controls for post-2025 for California for the alternative standard levels analyzed.

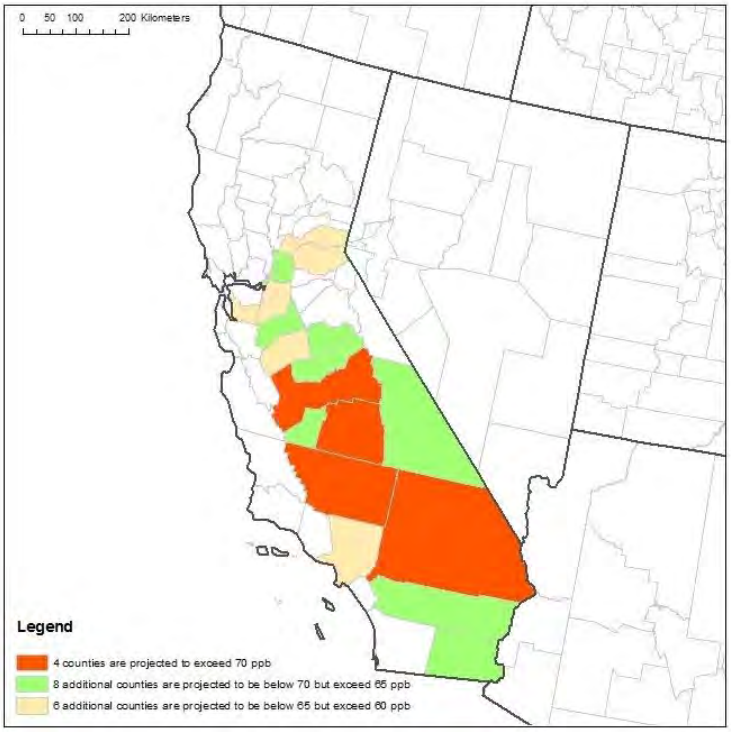

Figure ES-3. Projected Ozone Design Values in the post-2025 Baseline Scenario

Table ES-5. Summary of Emissions Reductions by Alternative Standard Level for Unknown Controls for post-2025 - California (1,000 tons/year)[a]

	Region	NO_x	VOC
Proposed Alternative Standard Levels			
70 ppb	**CA**	**53**	**-**
65 ppb	**CA**	**110**	**-**
Alternative Standard Level			
60 ppb	CA	140	-

[a] Estimates are rounded to two significant figures.

ES.1.3 Human Health Benefits

To estimate benefits, we follow a “damage-function” approach in calculating total benefits of the modeled changes in environmental quality. This approach estimates changes in individual health endpoints (specific effects that can be associated with changes in air quality) and assigns values to those changes assuming independence of the values for those individual endpoints. Total benefits are calculated as the sum of the values for all non-overlapping health 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 values 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, an 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 is limited to those health effects that are directly linked to ambient levels of air pollution and specifically to those linked to ozone and $PM_{2.5}$.

Benefits estimates for ozone were generated using the damage function approach outlined above wherein potential changes in ambient ozone levels (associated with future attainment of alternative standard levels) were explicitly modeled and then translated into reductions in the incidence of specific health endpoints. In generating ozone benefits estimates for the two attainment timeframes considered in the RIA (2025 and post-2025), we used three distinct benefits simulations including one completed for 2025 and two completed for 2038. The way in which these three benefits simulations were used to generate estimates for the two timeframes is detailed in Chapter 5, Section 5.4.3.

In contrast to ozone, we used a benefit-per-ton approach in modeling $PM_{2.5}$ co-benefits. With this approach, we use the results of previous benefits analysis simulations focusing on $PM_{2.5}$ to derive benefits-per-ton estimates for NOx.[4] We then combine these dollar-per-ton estimates with projected reductions in NOx associated with meeting a given alternative standard level to project cobenefits associated with $PM_{2.5}$. We acknowledge increased uncertainty associated with the dollar-per-ton approach for $PM_{2.5}$, relative to explicitly modeling benefits using gridded $PM_{2.5}$ surfaces specific to the baseline and alternative standard levels (see Appendix 5A, Table 5A-1 for additional discussion).

In addition to ozone and $PM_{2.5}$ benefits, implementing emissions controls to reach some of the alternative ozone standard levels would reduce other ambient pollutants. However, because the methods used in this analysis to simulate attainment do not account for changes in ambient

[4] In addition to dollar-per-ton estimates for NOx, we also used incidence-per-ton values (also for NOx) for specific health endpoints to generate incidence reduction estimates associated with the dollar benefits.

concentrations of other pollutants, we were not able to quantify the co-benefits of reduced exposure to these pollutants. In addition, due to data and methodology limitations, we were unable to estimate some anticipated health benefits associated with exposure to ozone and $PM_{2.5}$.

ES.1.4 Welfare Co-Benefits of the Primary Standard

Section 109 of the Clean Air Act defines welfare effects to include any non-health effects, including direct economic damages in the form of lost productivity of crops and trees, indirect damages through alteration of ecosystem functions, indirect economic damages through the loss in value of recreational experiences or the existence value of important resources, and direct damages to property, either through impacts on material structures or by soiling of surfaces (42 U.S.C. 7409). Ozone can affect ecological systems, leading to changes in the ecological community and influencing the diversity, health, and vigor of individual species (U.S. EPA, 2013). Ozone causes discernible injury to a wide array of vegetation (U.S. EPA, 2013). In terms of forest productivity and ecosystem diversity, ozone may be the pollutant with the greatest potential for region-scale forest impacts (U.S. EPA, 2013). Studies have demonstrated repeatedly that ozone concentrations observed in polluted areas can have substantial impacts on plant function (De Steiguer et al,. 1990; Pye, 1988).

In this RIA, we are able to quantify only a small portion of the welfare impacts associated with reductions in ozone concentrations to meet alternative ozone standards. Using a model of commercial agriculture and forest markets, we are able to analyze the effects on consumers and producers of forest and agricultural products of changes in the W126 index resulting from meeting alternative standards within the proposed range of 70 to 65 ppb, as well as a lower standard level of 60 ppb. We also assess the effects of those changes in commercial agricultural and forest yields on carbon sequestration and storage. This analysis provides limited quantitative information on the welfare co-benefits of meeting these alternative standards, focused only on one subset of ecosystem services. Commercial and non-commercial forests provide a number of additional services, including medicinal uses, non-commercial food and fiber production, arts and crafts uses, habitat, recreational uses, and cultural uses for Native American tribes. A more complete discussion of these additional ecosystem services is provided in the final Welfare Risk and Exposure Assessment for Ozone (U.S. EPA, 2014c).

ES.2 Results of Benefit-Cost Analysis

Below in Table ES-6, we present the primary costs and benefits estimates for 2025 for all areas except California. In addition, Tables 5-1 and 5-23 in Chapter 5 provide a breakdown of ozone-only and $PM_{2.5}$-only benefits, as well as total benefits at 3 percent. We anticipate that benefits and costs will likely begin occurring earlier, as states begin implementing control measures to show progress towards attainment. In these tables, ranges within the total benefits rows reflect variability in the studies upon which the estimates associated with premature mortality were derived. $PM_{2.5}$ co-benefits account for approximately two-thirds to three-quarters of the estimated benefits, depending on the standard analyzed and on the choice of ozone and PM mortality functions used. In addition for 2025, Table ES-7 presents the numbers of premature deaths avoided for the alternative standard levels analyzed, as well as the other health effects avoided. Table ES-8 provides information on the costs by geographic region for the U.S., except California in 2025, and Table ES-9 provides a regional breakdown of benefits for 2025.

In the RIA we provide estimates of costs of emissions reductions to attain the proposed standards in three regions -- California, the rest of the western U.S., and the eastern U.S. In addition, we provide estimates of the benefits that accrue to each of these three regions resulting from (i) control strategies applied within the region, (ii) reductions in transport of ozone associated with emissions reductions in other regions, and (iii) the control strategies for which the regional cost estimates are generated. These benefits are not directly comparable to the costs of control strategies in a region because the benefits include benefits not associated with those control strategies.

The net benefits of emissions reductions strategies in a specific region would be the benefits of the emissions reductions occurring both within and outside of the region minus the costs of the emissions reductions. Because the air quality modeling is done the national level, we do not estimate separately the nationwide benefits associated with the emissions reductions occurring in any specific region.[5] As a result, we are only able to provide net benefits estimates at the national level. The difference between the costs for a specific region and the benefits

[5] For California, we provide separate estimates of the costs and nationwide estimates of benefits, so it is appropriate to calculate net benefits. As such, we provide net benefits for the post-2025 California analysis.

accruing to that region is not an estimate of net benefits of the emissions reductions in that region.

Table ES-6. Total Annual Costs and Benefits[a] for U.S., except California in 2025 (billions of 2011$, 7% Discount Rate)[b]

	Proposed Alternative Standard Levels		Alternative Standard Level
	70 ppb	**65 ppb**	60 ppb
Total Costs (7%)	**$3.9**	**$15**	$39
Total Health Benefits (7%)[c]	**$6.4 to $13.0**	**$19 to $38**	$34 to $70
Net Benefits (7%)	**$2.5 to $9.1**	**$4 to $23**	($5) to $31

[a] Benefits are nationwide benefits of attainment everywhere except California.
[b] EPA believes that providing comparisons of social costs and social benefits at 3 and 7 percent is appropriate. Estimating multiple years of costs and benefits is not possible for this RIA due to data and resource limitations. As a result, we provide a snapshot of costs and benefits in 2025, using the best available information to approximate social costs and social benefits recognizing uncertainties and limitations in those estimates.
[c] The benefits range reflects the LOW and UPPER core estimates of short-term ozone and long-term PM mortality.

EPA believes that providing comparisons of social costs and social benefits at 3 and 7 percent is appropriate. Ideally, streams of social costs and social benefits over time would be estimated and the net present values of each would be compared to determine net benefits of the illustrative attainment strategies. The three different uses of discounting in the RIA – (i) construction of annualized engineering costs, (ii) adjusting the value of mortality risk for lags in mortality risk decreases, and (iii) adjusting the cost of illness for non-fatal heart attacks to adjust for lags in follow up costs -- are all appropriate. Our estimates of net benefits are the approximations of the net value (in 2025) of benefits attributable to emissions reductions needed to attain just for the year 2025.

Table ES-7. Summary of Total Number of Annual Ozone and PM-Related Premature Mortalities and Premature Morbidity: 2025 National Benefits [a]

	Proposed Alternative Standard Levels (95th percentile confidence intervals)[b]		Alternative Standard Level (95th percentile confidence intervals)
	70 ppb	**65 ppb**	60 ppb
Short-term exposure-related premature deaths avoided (all ages) (Ozone – 2 studies)	200 to 340 (97 to 300) (180 to 490)	630 to 1,000 (310 to 940) (560 to 1,500)	1,100 to 1,900 (560 to 1,700) (1,000 to 2,800)
Long-term exposure-related premature deaths avoided (age 30+) (PM – 2 studies)	O_3: 680 (230 to 1,100) $PM_{2.5}$: 510 to 1,100[c]	O_3: 2,100 (710 to 3,500) $PM_{2.5}$: 1,400 to 3,300[c]	O_3: 3,900 (1,300 to 6,400) $PM_{2.5}$: 2,600 to 6,000[c]
Other health effects avoided[d]			
Non-fatal heart attacks (age 18-99) (5 studies) [PM]	64 to 600	180 to 1,700	330 to 3,100

	Proposed Alternative Standard Levels (95th percentile confidence intervals)[b]		Alternative Standard Level (95th percentile confidence intervals)
	70 ppb	**65 ppb**	60 ppb
Respiratory hospital admissions (age 0-99)[O3, PM]	510	1,500	2,900
Cardiovascular hospital admissions (age 18-99) [PM]	180	530	950
Asthma emergency department visits (age 0-99) [O3, PM]	1,400	4,300	8,000
Acute bronchitis (age 8-12) [PM]	790	2,300	4,100
Asthma exacerbation (age 6-18) [O3, PM]	320,000	960,000	1,800,000
Lost work days (age 18-65) [PM]	65,000	180,000	340,000
Minor restricted activity days (age 18-65) [O3, PM]	1,300,000	4,000,000	7,300,000
Upper & lower respiratory symptoms (children 7-14) [PM]	24,000	70,000	130,000
School loss days (age 5-17) [O3]	330,000	1,000,000	1,900,000

[a] Nationwide benefits of attainment everywhere except California.
[b] We present a confidence interval in parentheses for each study on short-term or long-term ozone-related mortality.
[c] These estimates were generated using benefit-per-ton estimates and confidence intervals are not available. In general, the 95th percentile confidence interval for the health impact function alone ranges from ± 30 percent for mortality incidence based on Krewski et al. (2009) and ± 46 percent based on Lepeule et al. (2012).
[d] See Table 5-19 in Chapter 5 for detailed information on confidence intervals related to ozone-related morbidity incidence estimates. The $PM_{2.5}$ morbidity incidence estimates were generated using benefit-per-ton estimates and confidence intervals are not available.

Table ES-8. Summary of Total Control Costs (Known and Extrapolated) by Alternative Level for 2025 - U.S., except California (billions of 2011$, 7% Discount Rate)[a]

Alternative Level	Geographic Area	Total Control Costs (Known and Extrapolated)
70 ppb	East	3.9
	West	-
	Total	**$3.9**
65 ppb	East	15
	West	0.40
	Total	**$15**
60 ppb	East	33
	West	5.8
	Total	**$39**

[a] All values are rounded to two significant figures. Extrapolated costs are based on the average-cost methodology.

Table ES-9. Regional Breakdown of Monetized Ozone-Specific Benefits Results for the 2025 Scenario (nationwide benefits of attaining each alternative standard everywhere in the U.S. except California) – Full Attainment [a]

Region	Proposed and Alterative Standards		
	70 ppb	65 ppb	60 ppb
East [b]	99%	96%	92%
California	0%	0%	0%
Rest of West	1%	4%	7%

[a] Because we use benefit-per-ton estimates to calculate the $PM_{2.5}$ co-benefits, a regional breakdown for the co-benefits is not available. Therefore, this table only reflects the ozone benefits.
[b] Includes Texas and those states to the north and east. Several recent rules such as Tier 3 will have substantially reduced ozone concentrations by 2025 in the East, thus few additional controls would be needed to reach 70 ppb.

To understand possible additional costs and benefits of fully attaining in California in a post-2025 timeframe, we provide separate results for California in Table ES-10. In addition, Tables 5-2 and 5-30 in Chapter 5 provide a breakdown of ozone-only and $PM_{2.5}$-only benefits, as well as total benefits at 3 percent. Relative to the primary cost and benefits estimates, the California cost estimates are between 5 and 20 percent and the benefits estimates are between 8 and 15 percent of the national estimates. Because of the differences in the timing of achieving needed emissions reductions, incurring costs, and accruing benefits for California, the separate costs and benefits estimates for post-2025 should not be added to the primary estimates for 2025. For the post-2025 timeframe, Table ES-11 presents the numbers of premature deaths avoided for the alternative standard levels analyzed, as well as the other health effects avoided. Table ES-12 provides information on the costs for California for post-2025, and Table ES-13 provides a regional breakdown of benefits for post-2025.

The EPA presents separate costs and benefits results for California because forcing attainment in an earlier year than would be required under the Clean Air Act would likely lead to an overstatement of costs because California might benefit from some existing federal or state programs that would be implemented between 2025 and the ultimate attainment years; because additional new technologies may become available between 2025 and the attainment years; and because the cost of existing technologies might fall over time. As such, we use the best available data to estimate costs and benefits for California in a post-2025 timeframe, but because of data limitations and additional uncertainty associated with not projecting emissions and air quality beyond 2025, we recognize that the estimates of costs and benefits for California in a post-2025 timeframe are likely to be relatively more uncertain than the national attainment estimates for 2025.

Table ES-10. Total Annual Costs and Benefits[a] of Control Strategies Applied in California, post-2025 (billions of 2011$, 7% Discount Rate)[b]

	Proposed Alternative Standard Levels		Alternative Standard Level
	70 ppb	**65 ppb**	60 ppb
Total Costs (7%)	$0.80	$1.6	$2.2
Total Health Benefits (7%)[c]	$1.1 to $2	$2.2 to $4.1	$3.2 to $5.9
Net Benefits (7%)	$0.3 to $1.2	$0.60 to $2.5	$1 to $3.7

[a] Benefits are nationwide benefits of attainment in California.
[b] EPA believes that providing comparisons of social costs and social benefits at 3 and 7 percent is appropriate. Estimating multiple years of costs and benefits is not possible for this RIA due to data and resource limitations. As a result, we provide a snapshot of costs and benefits in 2025, using the best available information to approximate social costs and social benefits recognizing uncertainties and limitations in those estimates.
[c] The benefits range reflects the LOW and UPPER core estimates of short-term ozone and long-term PM mortality.

Table ES-11. Summary of Total Number of Annual Ozone and PM-Related Premature Mortalities and Premature Morbidity: Post-2025[a]

	Proposed Alternative Standard Levels (95th percentile confidence intervals)[b]		Alternative Standard Level (95th percentile confidence intervals)
	70 ppb	**65 ppb**	60 ppb
Short-term exposure-related premature deaths avoided (all ages) (Ozone – 2 studies)	65 to 110 (31 to 97) (57 to 160)	140 to 230 (68 to 210) (120 to 340)	210 to 350 (100 to 320) (190 to 510)
Long-term exposure-related premature deaths avoided (age 30+) (PM – 2 studies)	O_3: 260 (88 to 430) $PM_{2.5}$: 45 to 100[c]	O_3: 560 (190 to 930) $PM_{2.5}$: 89 to 200[c]	O_3: 840 (290 to 1,400) $PM_{2.5}$: 120 to 280[c]
Other health effects avoided[d]			
Non-fatal heart attacks (age 18-99) (5 studies) [PM]	6 to 54	11 to 110	16 to 140
Respiratory hospital admissions (age 0-99)[O3, PM]	130	290	430
Cardiovascular hospital admissions (age 18-99) [PM]	16	32	45
Asthma emergency department visits (age 0-99) [O3, PM]	340	740	1,100
Acute bronchitis (age 8-12) [PM]	67	130	180
Asthma exacerbation (age 6-18) [O3, PM]	99,000	210,000	320,000
Lost work days (age 18-65) [PM]	5,500	11,000	15,000
Minor restricted activity days (age 18-65) [O3, PM]	320,000	690,000	1,000,000
Upper & lower respiratory symptoms (children 7-14) [PM]	2,100	4,100	5,600
School loss days (age 5-17) [O3]	110,000	230,000	350,000

[a] Nationwide benefits of attainment in California.
[b] We present a confidence interval in parentheses for each study on short-term or long-term ozone-related mortality.

[c] These estimates were generated using benefit-per-ton estimates and confidence intervals are not available. In general, the 95th percentile confidence interval for the health impact function alone ranges from + 30 percent for mortality incidence based on Krewski et al. (2009) and + 46 percent based on Lepeule et al. (2012).
[d] See Table 5-26 in Chapter 5 for detailed information on confidence intervals related to ozone-related morbidity incidence estimates. The PM2.5 morbidity incidence estimates were generated using benefit-per-ton estimates and confidence intervals are not available.

Table ES-12. Summary of Total Control Costs (Known and Extrapolated) by Alternative Level for post-2025 - California (billions of 2011$, 7% Discount Rate)[a]

Alternative Level	Geographic Area	Total Control Costs (Known and Extrapolated)
70 ppb	California	$0.80
65 ppb	California	$1.6
60 ppb	California	$2.2

[a] All values are rounded to two significant figures. Extrapolated costs are based on the average-cost methodology.

Table ES-13. Regional Breakdown of Monetized Ozone-Specific Benefits Results for the post-2025 Scenario (nationwide benefits of attaining each alternative standard just in California) – Full Attainment[a]

Region	Proposed and Alterative Standards		
	70 ppb	**65 ppb**	**60 ppb**
East	0%	0%	0%
California	93%	94%	94%
Rest of West	6%	6%	6%

[a] Because we use benefit-per-ton estimates to calculate the $PM_{2.5}$ co-benefits, a regional breakdown for the co-benefits is not available. Therefore, this table only reflects the ozone benefits.

Despite uncertainties inherent in any complex, quantitative analysis, the overall underlying analytical methods used in this RIA have been peer-reviewed. For a detailed discussion on uncertainty associated with developing illustrative control strategies to attain the alternative standard levels, see Chapter 4, Section 4.4. For a description of the key assumptions and uncertainties related to the modeling of ozone benefits, see Chapter 5, Section 5.7.3, and for an additional qualitative discussion of sources of uncertainty associated with both the modeling of ozone-related benefits and $PM_{2.5}$-related co-benefits, see Appendix 5A. For a discussion of the limitations and uncertainties in the engineering cost analyses, see Chapter 7, Section 7.7. For a discussion about generally framing uncertainty, see Chapter 8, Section 8.3.

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CHAPTER 1: INTRODUCTION AND BACKGROUND

Overview

The EPA Administrator is proposing to revise the level of the ozone National Ambient Air Quality Standards (NAAQS) to within a range of 65 to 70 ppb and is soliciting comment on alternative standard levels below 65 ppb, as low as 60 ppb. This chapter summarizes the purpose and background of this Regulatory Impact Analysis (RIA). In the RIA we estimate the human health and welfare benefits and costs of alternative standards of 65 ppb and 70 ppb, which represent the lower and upper bounds of the range of proposed levels, as well as a more stringent alternative level of 60 ppb. According to the Clean Air Act ("the Act"), the Environmental Protection Agency (EPA) must use health-based criteria in setting the NAAQS and cannot consider estimates of compliance cost. The EPA is producing this RIA both to provide the public a sense of the benefits and costs of meeting a revised ozone NAAQS and to meet the requirements of Executive Orders 12866 and 13563.

1.1 Background

1.1.1 NAAQS

Two sections of the Act govern the establishment and revision of NAAQS. Section 108 (42 U.S.C. 7408) directs the Administrator to identify pollutants that "may reasonably be anticipated to endanger public health or welfare" and to issue air quality criteria for them. These air quality criteria are intended to "accurately reflect the latest scientific knowledge useful in indicating the kind and extent of all identifiable effects on public health or welfare which may be expected from the presence of [a] pollutant in the ambient air." Ozone 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 an ambient air quality standard "the attainment and maintenance of which in the judgment of the Administrator, based on [the] criteria and allowing an adequate margin of safety, [is] 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, is 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.

1.1.2 Ozone NAAQS

The EPA initiated the current ozone NAAQS review in September 2008. Between 2008 and 2014, the EPA prepared draft and final versions of the Integrated Science Assessment, the Health and Welfare Risk and Exposure Assessments, and the Policy Assessment. Multiple drafts of these documents were available for public review and comment, and as required by the Clean Air Act, were peer-reviewed by the Clean Air Scientific Advisory Committee (CASAC), the Administrator's independent advisory committee established by the CAA. The final documents reflect the EPA staff's consideration of the comments and recommendations made by CASAC and the public on draft versions of these documents.

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

1.2.1 Legislative Roles

The EPA Administrator is proposing to revise the level of the ozone National Ambient Air Quality Standards (NAAQS) to within a range of 65 to 70 ppb and is soliciting comment on alternative standard levels below 65 ppb, as low as 60 ppb. The EPA Administrator is also proposing to revise the level of the current secondary standard to within the range of 65 ppb to 70 ppb. As such, the RIA analyzes a range of potential alternative primary standard levels. In setting primary ambient air quality standards, the EPA's responsibility under the law is to establish standards that protect public health, regardless of the costs of implementing those standards. The Act requires the EPA, for each criteria pollutant, to set standards that protect public health with "an adequate margin of safety." As interpreted by the Agency and the courts, the Act requires the EPA to create standards based on health considerations only.

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

1.2.2 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 (U.S. OMB, 2003). In accordance with these guidelines, the RIA analyzes the benefits and costs associated with emissions controls to attain the upper and lower bounds of the proposed 8-hour ozone standard of 65 parts per billion (ppb) to 70 ppb in ambient air, incremental to a baseline of attaining the existing standard (8-hour ozone standard of 75 ppb). OMB Circular A-4 requires analysis of one potential alternative standard level more stringent than the proposed range and one less stringent than the proposed range. In this RIA, we analyze a more stringent alternative standard level of 60 ppb. The existing standard of 75 ppb represents the less stringent alternative standard and the costs and benefits of this standard were presented in the 2008 ozone NAAQS RIA (U.S. EPA, 2008a). The available scientific evidence and quantitative exposure and risk information indicate that reducing ambient ozone concentrations will reduce the occurrence of harmful health effects. As discussed in the Notice, this evidence and information provide strong support for considering alternative standard levels from 65 to 70 ppb, but do not identify a bright line within this range that indicates exactly where to set a standard. Similarly, the available scientific information does not provide a basis for identifying any specific standard level between 70 and 75 ppb for analysis in the RIA.

The control strategies presented in this RIA are illustrative and represent one set of control strategies states might choose to implement in order to meet the final standards. As a result, benefit and cost estimates provided in the RIA are 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.

1.2.3 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.” A market failure or externality occurs when one party’s actions impose uncompensated costs on another party. Environmental problems are a classic case of an externality. Setting and implementing primary and secondary air quality standards is one way the government can address an externality and thereby increase air quality and improve overall public health and welfare.

1.2.4 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.

1.3 Overview and Design of the RIA

The RIA evaluates the costs and benefits of hypothetical national control strategies to attain three alternative ozone standard levels of 60, 65 and 70 ppb.

1.3.1 Existing and Revised Ozone National Ambient Air Quality Standards

The EPA is proposing to retain the indicator, averaging time and form of the existing primary ozone standard and is proposing to revise the level of that standard to within the range of 65 ppb to 70 ppb. The EPA is proposing this revision to increase public health protection, including for "at-risk" populations such as children, older adults, and people with asthma or other lung diseases, against an array of ozone-related adverse health effects. For short-term ozone exposures, these effects include decreased lung function, increased respiratory symptoms and pulmonary inflammation, effects that result in serious indicators of respiratory morbidity, such as emergency department visits and hospital admissions, and all-cause (total non-accidental) mortality. For long-term ozone exposures, these health effects include a variety of respiratory morbidity effects and respiratory mortality. In recognition that levels as low as 60 ppb could potentially be supported, but would place very little weight on the uncertainties in the health effects evidence and exposure/risk information, the EPA is also soliciting comment on alternative standard levels below 65 ppb, as low as 60 ppb. In addition, the EPA is taking comment on the option of retaining the current 8-hour primary ozone standard of 75 ppb.

The EPA is proposing to revise the level of the secondary standard to within the range of 65 ppb to 70 ppb to provide increased protection against vegetation-related effects on public welfare. As an initial matter, the EPA is proposing that ambient ozone concentrations in terms of a three-year average W126 index value within the range from 13 parts per million-hours (ppm-hours) to 17 ppm-hours would provide the requisite protection against known or anticipated adverse effects to the public welfare, which data analyses indicate would provide air quality in terms of three-year average W126 index values of a range at or below 13 ppm-hours to 17 ppm-hours. Data analyses also indicate that actions taken to attain a standard in the range of 65 ppb to 70 ppb would also improve air quality as measured by the W126 metric. The quantitative analysis assesses the welfare benefits of strategies to attain the proposed secondary standard levels of 65 to 70 ppb.

1.3.2 Establishing Attainment with the Current Ozone National Ambient Air Quality Standard

The RIA is intended to evaluate the costs and benefits of reaching attainment with alternative ozone standard levels. To develop and evaluate control strategies for attaining a more stringent primary standard, it is important to first estimate ozone levels in the future after attaining the current NAAQS (75 ppb) and taking into account projections of future air quality reflecting on-the-books Federal regulations, enforcement actions, state regulations, and population and economic growth. This allows us to then estimate the incremental costs and benefits of attaining alternative primary standard levels.

Attaining 75 ppb reflects emissions reductions already achieved as a result of national regulations, emissions reductions expected prior to 2025 from recently promulgated national regulations (i.e., reductions that were not realized before promulgation of the previous standard, but are expected prior to attainment of the existing ozone standard), and reductions from additional controls that the EPA estimates need to be included to attain the existing standard (75 ppb). Emissions reductions achieved as a result of state and local agency regulations and voluntary programs are reflected to the extent that they are represented in emissions inventory information submitted to the EPA by state and local agencies. We took two steps to develop the baseline reflecting attainment of 75 ppb. First, national ozone concentrations were projected based on population and economic growth and the application of emissions controls resulting from national rules promulgated prior to this analysis, as well as state programs and enforcement actions. Second, we apply an illustrative control strategy to estimate emissions reductions for the current standard of 75 ppb, also referred to as the baseline.

Below is a list of some of the national rules reflected in the baseline. For a more complete list, please see the Technical Support Document: Preparation of Emissions Inventories for the Version 6.1, 2011 Emissions Modeling Platform (US EPA, 2014a).

- Carbon Pollution Emission Guidelines for Existing Stationary Sources: Electric Utility Generating Units (U.S. EPA, 2014b)
- Tier 3 Motor Vehicle Emission and Fuel Standards (U.S. EPA, 2014c)
- Mercury and Air Toxics Standards (U.S. EPA, 2011)
- Reciprocating Internal Combustion Engines (RICE) NESHAPs (U.S. EPA, 2010)
- Hospital/Medical/Infectious Waste Incinerators: New Source Performance Standards and Emission Guidelines: Final Rule Amendments (U.S. EPA, 2009)
- C3 Oceangoing Vessels (U.S. EPA, 2010)
- Emissions Standards for Locomotives and Marine Compression-Ignition Engines (U.S. EPA, 2008b)
- Control of Emissions for Nonroad Spark Ignition Engines and Equipment (U.S. EPA, 2008c)
- 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)
- Clean Air Nonroad Diesel Rule (U.S. EPA, 2004)
- Heavy Duty Diesel Rule (U.S. EPA, 2000)
- Light-Duty Vehicle Tier 2 Rule (U.S. EPA, 1999)

The baseline for this analysis does not assume emissions controls that might be implemented to meet the other NAAQS for $PM_{2.5}$, NO_2, or SO_2. We did not conduct this analysis incremental to controls applied as part of previous NAAQS analyses because the data and modeling on which these previous analyses were based are now considered outdated and are not compatible with the current ozone NAAQS analysis.[6] 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.

[6] There were no additional NOx controls applied in the $PM_{2.5}$ NAAQS RIA, and therefore there would be little to no impact on the controls selected as part of this analysis. In addition, the only geographic areas that exceed the alternative ozone standard levels analyzed in this RIA and in the 2012 $PM_{2.5}$ NAAQS RIA are in California. The attainment dates for a new $PM_{2.5}$ NAAQS would likely precede attainment dates for a revised ozone NAAQS. While the 2012 $PM_{2.5}$ NAAQS RIA concluded that controls on directly emitted $PM_{2.5}$ were the most cost-effective on a $/ug basis, states may choose to adopt different control options. These options could include NOx controls. It is difficult to determine the impact on costs and benefits for this RIA because it is highly dependent upon the control measures that would be chosen and the costs of these measures.

1.3.3 Establishing the Baseline for Evaluation of Alternative Standards

The RIA evaluates, to the extent possible, the costs and benefits of attaining the proposed and alternative ozone standards incremental to attaining the existing ozone standard and implementing existing and expected regulations. We assume that potential nonattainment areas everywhere in the U.S., excluding California, will be designated such that they are required to reach attainment by 2025, and we developed our projected baselines for emissions, air quality, and populations for 2025.

The EPA will likely finalize designations for a revised ozone NAAQS in late 2017. Depending on the precise timing of the effective date of those designations, nonattainment areas classified as Marginal will likely have to attain in either late 2020 or early 2021. Nonattainment areas classified as Moderate will likely have to attain in either late 2023 or early 2024. If a Moderate nonattainment area qualifies for two 1-year extensions, the area may have as late as early 2026 to attain. Lastly, Serious nonattainment areas will likely have to attain in late 2026 or early 2027. We selected 2025 as the primary year of analysis because it provided a good representation of the remaining air quality concerns that moderate nonattainment areas would face and because most areas of the U.S. will likely be required to meet a revised ozone standard by 2025. States with areas classified as Moderate and higher are required to develop attainment demonstration plans for those nonattainment areas. In this RIA we present the primary costs and benefits estimates for 2025.

In estimating the incremental costs and benefits of potential alternative standards, we recognize that there are several areas that are not required to meet the existing ozone standard by 2025. The Clean Air Act allows areas with more significant air quality problems to take additional time to reach the existing standard. Several areas in California are not required to meet the existing standard by 2025 and may not be required to meet a revised standard until sometime between 2032 and 2037.[7] We were not able to project emissions and air quality

[7] The EPA will likely finalize designations for a revised ozone NAAQS in late 2017. Depending on the precise timing of the effective date of those designations, nonattainment areas classified as Severe 15 will likely have to attain sometime between late 2032 and early 2033 and nonattainment areas classified as Extreme will likely have to attain sometime between late 2037 and early 2038cember 31,.

beyond 2025 for California, however, we adjusted baseline air quality to reflect mobile source emissions reductions for California that would occur between 2025 and 2030; these emissions reductions were the result of mobile source regulations expected to be fully implemented by 2030. While there is uncertainty about the precise timing of emissions reductions and related costs for California, we assume costs occur through the end of 2037 and beginning of 2038. In addition, we model benefits for California using projected population demographics for 2038.

Because of the different timing for incurring costs and accruing benefits and for ease of discussion throughout the analyses, we refer to the different time periods for potential attainment as 2025 and post-2025 to reflect that (1) we did not project emissions and air quality for any year other than 2025; (2) for California, emissions controls and associated costs are assumed to occur through the end of 2037 and beginning of 2038; and (3) for California benefits are modeled using population demographics in 2038. It is not straightforward to discount the post-2025 results for California to compare with or add to the 2025 results for the rest of the U.S. While we estimate benefits using 2038 information, we do not have good information on precisely when the costs of controls will be incurred. Because of these differences in timing related to California attaining a revised standard, the separate costs and benefits estimates for post-2025 should not be added to the primary estimates for 2025.

1.4 Health and Welfare Benefits Analysis Approach

1.4.1 Health Benefits

The EPA estimated human health (e.g., mortality and morbidity effects) under both partial and full attainment of the three alternative ozone standards. We considered an array of health impacts attributable to changes in ozone and $PM_{2.5}$ exposure and estimated these benefits using the BenMAP tool (US EPA, 2014), which has been used in many recent RIAs (e.g., U.S. EPA, 2006, 2011a, 2011b), and *The Benefits and Costs of the Clean Air Act 1990 to 2020* (U.S. EPA, 2011c). The 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 epidemiology studies evaluating mortality and morbidity associated with ozone and $PM_{2.5}$ exposure, and an expanded uncertainty assessment. Each of these updates is fully described in the health benefits chapter (Chapter 5). In addition, unquantified health benefits are also discussed in Chapter 5.

1.4.2 Welfare Co-Benefits

Even though the primary standards are designed to protect against adverse effects to human health, the emissions reductions would 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. Welfare co-benefits of the primary ozone standard include reduced vegetation effects resulting from ozone exposure, reduced ecological effects from particulate matter deposition and from nitrogen emissions, reduced climate effects, and changes in visibility. Both welfare co-benefits are discussed further in Chapter 6.

1.5 Cost Analysis Approach

The EPA estimated total costs under partial and full attainment of the three alternative ozone standards. These cost estimates reflect only engineering costs, which 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 engineering costs associated with applying end-of-pipe controls, or known controls. Costs for full attainment include estimates for the costs associated with the additional emissions reductions that are needed beyond known controls, referred to as unknown controls. The EPA recognizes that the portion of the cost estimates from unknown controls reflects substantial uncertainty about which sectors and which technologies might become available for cost-effective application in the future.

1.6 Organization of this Regulatory Impact Analysis

This RIA includes the following ten chapters:

- *Chapter 1: Introduction and Background*. This chapter introduces the purpose of the RIA.
- *Chapter 2: Defining the Ozone Air Quality Problem*. This chapter characterizes the nature, scope, and magnitude of the current-year ozone 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 ozone concentrations in 2025 after applying the control strategies.
- *Chapter 5: Human Health Benefits Analysis*. This chapter quantifies the health-related benefits of the ozone-related air quality improvements associated with several alternative standards.
- *Chapter 6*: *Welfare Co-Benefits of the Primary Standard*. This chapter quantifies and monetizes selected other welfare effects, including vegetation effects from ozone exposure, ecological effects from nitrogen and sulfur emissions, changes in visibility, materials damage, ecological effects from PM deposition, ecological effects from mercury deposition, and climate effects.
- *Chapter 7*: *Engineering Cost Analysis*. This chapter summarizes the data sources and methodology used to estimate the engineering costs of partial and full attainment of several alternative standards.
- *Chapter 8: Comparison of Benefits and Costs*. This chapter compares estimates of the total benefits with total costs and summarizes the net benefits of several alternative standards.
- *Chapter 9*: *Statutory and Executive Order Impact Analyses*. This chapter summarizes the Statutory and Executive Order impact analyses.
- *Chapter 10*: *Qualitative Discussion of Employment Impacts of Air Quality.* This chapter provides a discussion of employment impacts of reducing emissions of ozone precursors.

1.7 References

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CHAPTER 2: DEFINING THE OZONE AIR QUALITY PROBLEM

Overview

This section provides overviews of ozone precursor emissions and atmospheric chemistry (section 2.1); ambient ozone concentrations (section 2.2); ambient ozone monitoring in the U.S. (section 2.3); and available evidence and information related to background ozone (section 2.4).

2.1 Emissions and Atmospheric Chemistry

Ozone is formed through photochemical reactions of precursor gases and is not directly emitted from specific sources. In the stratosphere, ozone occurs naturally and provides protection against harmful solar ultraviolet radiation. In the troposphere, near ground level, ozone forms through atmospheric reactions involving two main classes of precursor pollutants: volatile organic compounds (VOCs) and nitrogen oxides (NO_X). Carbon monoxide (CO) and methane (CH_4) are also important for ozone formation over longer time periods (US EPA, 2013, section 3.2.2).

Emissions of ozone precursor compounds can be divided into anthropogenic and natural source categories, with natural sources further divided into biogenic emissions (from vegetation, microbes, and animals) and abiotic emissions (from biomass burning, lightning, and geogenic sources). Anthropogenic sources, including mobile sources and power plants, account for the majority of NO_X and CO emissions. Anthropogenic sources are also important for VOC emissions, though in some locations and at certain times of the year (e.g., southeastern states during summer) the majority of VOC emissions comes from vegetation (US EPA, 2013, section 3.2.1).

Rather than varying directly with emissions of its precursors, ozone changes in a nonlinear fashion with the concentrations of its precursors. NO_X emissions lead to both the formation and destruction of ozone, depending on the local quantities of NO_X, VOC, free radicals, and sunlight. In areas dominated by fresh emissions of NO_X, radicals are removed, which lowers the ozone formation rate. In addition, the scavenging of ozone by reaction with NO is called “titration” and is often found in downtown metropolitan areas, especially near busy streets and roads, as well as in power plant plumes. This short-lived titration results in localized areas in which ozone concentrations are suppressed compared to surrounding areas, but which contain NO_2 that contributes to subsequent ozone formation further downwind. The NOx titration effect is most pronounced in urban core areas that have a high volume of mobile source NOx emissions from vehicles. In areas with relatively low NO_X concentrations, such as those found in remote continental areas and rural and suburban areas downwind of urban centers, ozone production typically responds linearly to NO_X concentrations (e.g., ozone decreases with decreasing NO_X emissions). Consequently, ozone response to reductions in NO_X emissions is complex and may include ozone decreases at some times and locations and increases of ozone at other times and locations. As a general rule, as NOx emissions reductions occur, you can expect lower ozone values to increase while the higher ozone values would be expected to decrease. NOx reductions are expected to result in a compressed ozone distribution, relative to current conditions (EPA, 2014a).

The formation of ozone from precursor emissions is also affected by meteorological parameters such as the intensity of sunlight and atmospheric mixing. Major episodes of high ground-level ozone concentrations in the eastern United States are often associated with slow-moving high pressure systems. High pressure systems during the warmer seasons are associated with the sinking of air, resulting in warm, generally cloudless skies, with light winds. The sinking of air results in the development of stable conditions near the surface that inhibit or reduce the vertical mixing of ozone precursors. The combination of inhibited vertical mixing and light winds minimizes the dispersal of pollutants, allowing their concentrations to build up. In addition, in some parts of the United States (e.g., in Los Angeles), mountain barriers limit mixing and result in a higher frequency and duration of days with elevated ozone concentrations. Photochemical activity involving precursors is enhanced during warmer seasons because of the greater availability of sunlight and higher temperatures (US EPA, 2013, section 3.2).

Elevated wintertime ozone concentrations have recently been measured in mountain valleys in the Western U.S. (Schnell et al, 2009; Rappengluck et al., 2014; Helmig et al., 2014). Hourly ozone concentrations during these winter events have been observed to reach 160 ppb. This phenomenon is believed to result from the combination of several factors: 1) strong wintertime inversions or “cold pools”, which trap air in a shallow layer close to the ground, 2) substantial emissions of NOx and VOC from nearby oil and gas operations, 3) high albedo of deep snow, which leads to enhanced UV intensity and photochemical activity, and 4) possible uncharacterized sources of radicals. These wintertime ozone events have currently only been observed in a limited number of locations in Wyoming, Utah, and Colorado. Events can last for multiple days and can occur several times a year, but do not occur every winter in these locations.

Ozone concentrations in a region are affected both by local formation and by transport of ozone and its precursors from upwind areas. Ozone transport occurs on many spatial scales including local transport between cities, regional transport over large regions of the U.S. and international/long-range transport. In addition, ozone can be transferred into the troposphere from the stratosphere, which is rich in ozone, through stratosphere-troposphere exchange (STE). These intrusions usually occur behind cold fronts, bringing stratospheric air with them and typically affect ozone concentrations in higher elevation areas (e.g. > 1500 m) more than areas at lower elevations (U.S. EPA, 2013, section 3.4.1.1). The role of long-range transport of ozone and other elements of ozone background are discussed in more detail in Section 2.4.

2.2 Spatial and Temporal Variations in Ambient Ozone Concentrations

Because ozone is a secondary pollutant formed in the atmosphere from precursor emissions, concentrations are generally more regionally homogeneous than concentrations of primary pollutants emitted directly from stationary and mobile sources (US EPA, 2013, section 3.6.2.1). However, variation in local emissions characteristics, meteorological conditions, and topography can result in daily and seasonal temporal variability in ambient ozone concentrations, as well as local and national-scale spatial variability.

Temporal variation in ambient ozone concentrations results largely from daily and seasonal patterns in sunlight, precursor emissions, atmospheric stability, wind direction, and temperature (US EPA, 2013, section 3.7.5). On average, ambient ozone concentrations follow well-recognized daily and seasonal patterns, particularly in urban areas. Specifically, daily maximum 1-hour ozone concentrations in urban areas tend to occur in mid-afternoon, with more pronounced peaks in the warm months of the ozone season than in the colder months (US EPA, 2013, Figures 3-54, 3-156 to 3-157). Rural sites also follow this general pattern, though it is less pronounced in colder months (US EPA, 2013, Figure 3-55). With regard to day-to-day variability, median maximum daily 8-hour average (MDA8) ozone concentrations in U.S. cities from 2007 through 2009 were approximately 47 ppb, with typical ranges between 35 to 60 ppb and the highest MDA8 concentrations above 100 ppb in several U.S. cities (as noted further below).

In addition to temporal variability, there is considerable spatial variability in ambient ozone concentrations within cities and across different cities in the United States. With regard to spatial variability within a city, local emissions characteristics, geography, and topography can have important impacts. For example, as noted above, fresh NO emissions from motor vehicles titrate ozone present in the urban background air, resulting in an ozone gradient around roadways with ozone concentrations increasing as distance from the road increases (US EPA, 2013, section 3.6.2.1). Measured ozone concentrations are relatively uniform and well-correlated within some cities (e.g., Atlanta) while they are more variable in others (e.g., Los Angeles) (US EPA, 2013, section 3.6.2.1 and Figures 3-28 to 3-36).

Ozone concentrations also vary considerably across cities. Several cities had very high measured ozone concentrations in 2007 through 2009 when the maximum recorded MDA8 was 137 ppb in Los Angeles, and was near or above 120 ppb in Atlanta, Baltimore, Dallas, New York City, Philadelphia, and St. Louis (US EPA, 2013, Table 3-10). These same cities also had high 98th percentile ozone concentrations, with Los Angeles recording the highest 98th percentile concentration (91 ppb) and many eastern and southern cities reporting 98th percentile concentrations near or above 75 ppb. In contrast, somewhat lower 98th percentile ozone concentrations were recorded in cities in the western United States outside of California (US EPA, 2013, Table 3-10).

Rural sites can be affected by transport of ozone or ozone precursors from upwind urban areas and by local anthropogenic sources such as motor vehicles, power generation, biomass combustion, or oil and gas operations (US EPA, 2013, section 3.6.2.2). In addition, ozone tends to persist longer in rural than in urban areas due to lower rates of chemical scavenging in non-urban environments. At higher elevations, increased ozone concentrations can also result from stratospheric intrusions (US EPA, 2013, sections 3.4, 3.6.2.2). As a result, ozone concentrations measured in some rural sites can be higher than those measured in nearby urban areas (US EPA, 2013, section 3.6.2.2).

2.3 Ozone Monitoring

2.3.1 Ozone Monitoring Network

To monitor compliance with the National Ambient Air Quality Standards (NAAQS), state and local environmental agencies operate ozone monitoring sites at various locations, depending on the population of the area and typical peak ozone concentrations.[8] All of the state and local monitoring stations that report data to the EPA Air Quality System (AQS) use ultraviolet (UV) Federal Equivalent Methods (FEMs). In 2013, there were over 1,300 state, local, and tribal ozone monitors reporting concentrations to EPA. The "State and Local Monitoring Stations" (SLAMS) minimum monitoring requirements to meet the ozone design criteria are specified in 40 CFR Part 58, Appendix D. The requirements are both population and design value based.[9] The minimum number of ozone monitors required in a Metropolitan Statistical Area (MSA) ranges from zero for areas with a population of at least 50,000 and under 350,000 with no recent history of an ozone design value greater than 85 percent of the NAAQS, to four for areas with a population greater than 10 million and an ozone design value greater than 85 percent of the NAAQS. At least one site for each MSA, or Combined Statistical Area (CSA), must be sited to record the maximum concentration for that particular metropolitan area. Since highest ozone concentrations tend to be associated with particular seasons for various locations, EPA requires ozone monitoring during specific ozone monitoring seasons, which vary by state.[10]

Figure 2-1 shows the locations of the U.S. ambient ozone monitoring sites reporting data to EPA at any time during the 2009-2013 period. The gray dots that make up over 80% of the ozone monitoring network are SLAMS monitors, which are operated by state and local governments to meet regulatory requirements and provide air quality information to public health agencies. Thus, the SLAMS monitoring sites are largely focused on urban and suburban areas. The blue dots highlight two important subsets of monitoring sites within the SLAMS network:

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[8] The minimum ozone monitoring network requirements for urban areas are listed in Table D-2 of Appendix D to 40 CFR Part 58.

[9] A design value is a statistic that describes the air quality status of a given area relative to the level of the NAAQS. Design values are typically used to classify nonattainment areas, assess progress towards meeting the NAAQS, and develop control strategies. See http://epa.gov/airtrends/values.html (U, 2010, 677582) for guidance on how these values are defined.

[10] The required ozone monitoring seasons for each state are listed in Table D-3 of Appendix D to 40 CFR Part 58. Revised monitoring seasons are being proposed along with the proposed revision of the ozone NAAQS level.

the "National Core" (NCore) multi-pollutant monitoring network and the "Photochemical Assessment Monitoring Stations" (PAMS) network.

While the existing U.S. ozone monitoring network has a largely urban focus, to address ecosystem impacts of ozone, such as biomass loss and foliar injury, it is equally important to focus on ozone monitoring in rural areas. The green dots in Figure 2-1 represent the Clean Air Status and Trends Network (CASTNET) monitors, which are located in rural areas. There were about 80 CASTNET sites operating in 2013, with sites in the eastern U.S. being operated by EPA and sites in the western U.S. being operated by the National Park Service (NPS).[11] In total, there were about 120 rural ozone monitoring sites operating in the U.S. in 2013.

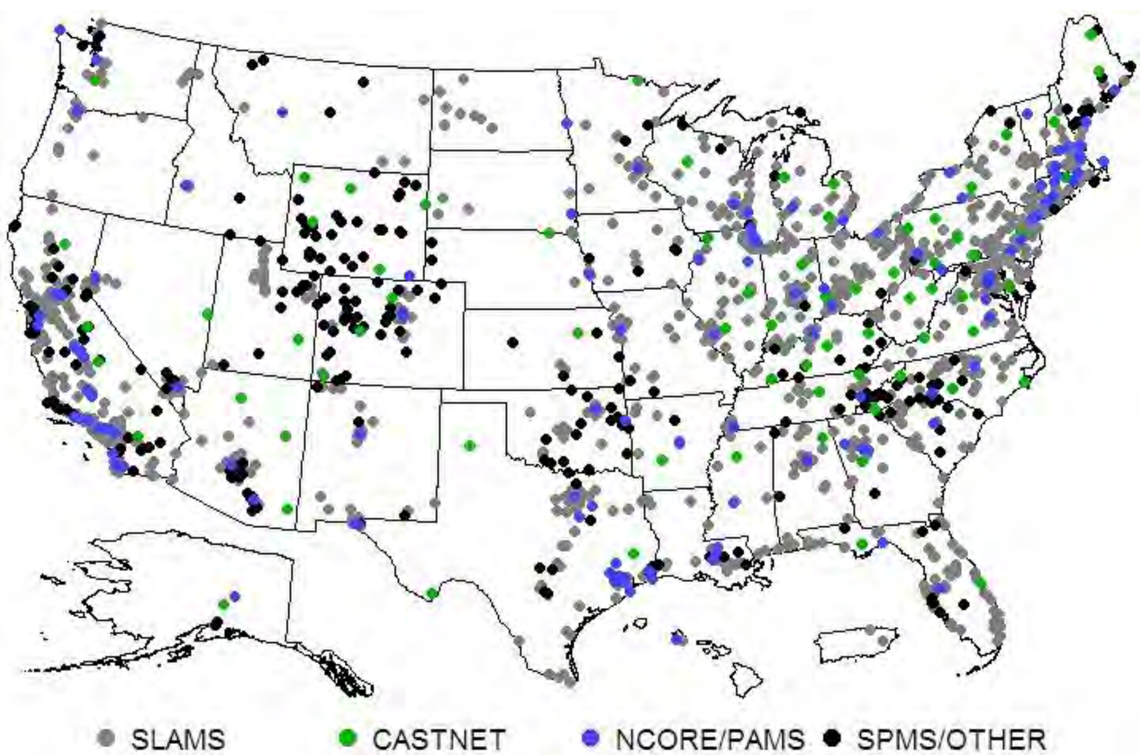

Figure 2-1. Map of U.S. Ambient O_3 Monitoring Sites Reporting Data to EPA During the 2009-2013 Period

[11] Additionally, the black dots represent "Special Purpose Monitoring Stations" (SPMS), which include about 20 rural monitors as part of the "Portable O_3 Monitoring System" (POMS) network operated by the NPS.

2.3.2 *Recent Ozone Monitoring Data and Trends*

To determine whether or not the ozone NAAQS has been met at an ambient monitoring site, a statistic commonly referred to as a "design value" must be calculated based on three consecutive years of data collected from that site. The form of the existing ozone NAAQS design value (DV) statistic is the 3-year average of the annual 4th highest daily maximum 8-hour ozone concentration in parts per billion (ppb), with decimal digits truncated. The existing primary and secondary ozone NAAQS are met at an ambient monitoring site when the DV is less than or equal to 75 ppb.[12] In counties or other geographic areas with multiple monitoring sites, the area-wide DV is defined as the DV at the highest individual monitoring site, and the area is said to have met the NAAQS only if all monitoring sites in the area are meeting the NAAQS.

Figure 2-2 shows the trend in the annual 4th highest daily maximum 8-hour ozone concentrations in ppb based on 910 "trends" sites with complete data records over the 2000 to 2013 period. The center line in this figure represents the median value across the trends sites, while the dashed lines represent the 25th and 75th percentiles, and the bottom and top lines represent the 10th and 90th percentiles. Figure 2-3 shows a map of the ozone DVs (in ppb) averaged across the 2009-2011, 2010-2012, and 2011-2013 periods at all monitoring sites in the contiguous U.S.[13] The trend figure shows that the annual 4th highest daily maximum values decreased for the vast majority of monitoring sites in the U.S. between 2000 and 2013. The decreasing trend is especially sharp from 2002 to 2004, when EPA implemented the "NO_X SIP Call", a program designed to reduce summertime emissions of NO_X in the eastern U.S., but has continued to decrease since then, in part due to ongoing reductions in mobile source NOx emissions. Within the overall downward trend, there are periodic short-term increases. These variations from the overall trend are the result of inter-annual variability in meteorological conditions.

[12] For more details on the data handling procedures used to calculate design values for the existing ozone NAAQS, see 40 CFR Part 50, Appendix P.

[13] All monitoring sites in Alaska, Hawaii, and Puerto Rico had DVs below 60 ppb.

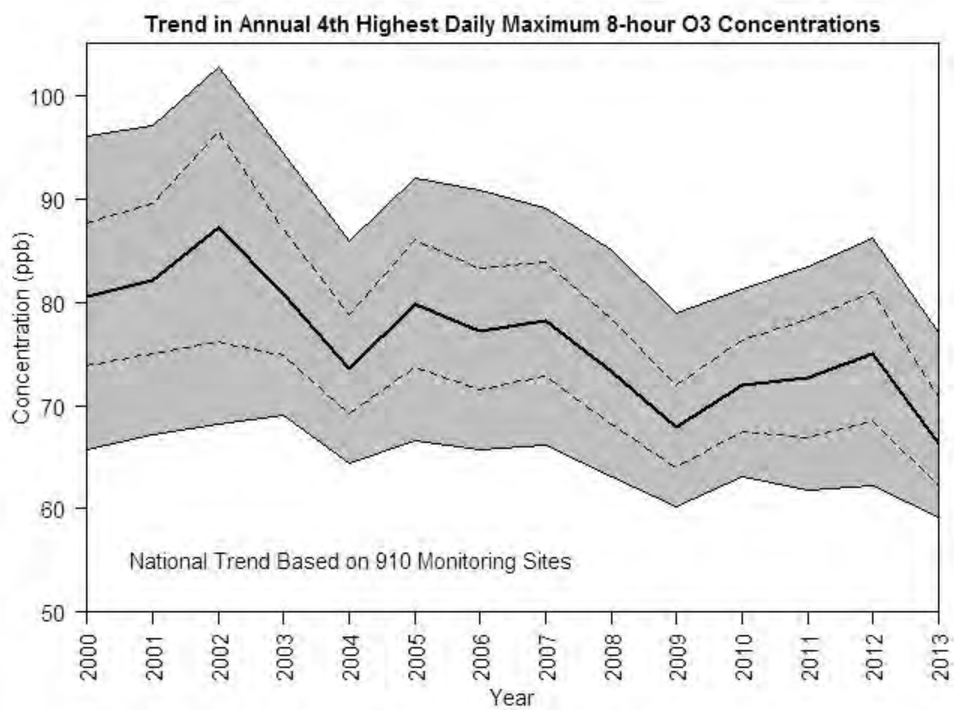

Figure 2-2. Trend in U.S. Annual 4th Highest Daily Maximum 8-hour Ozone Concentrations in ppb, 2000 to 2013. Solid center line represents the median value across monitoring sites, dashed lines represent 25th and 75th percentile values, and top/bottom lines represent 10th and 90th percentile values.

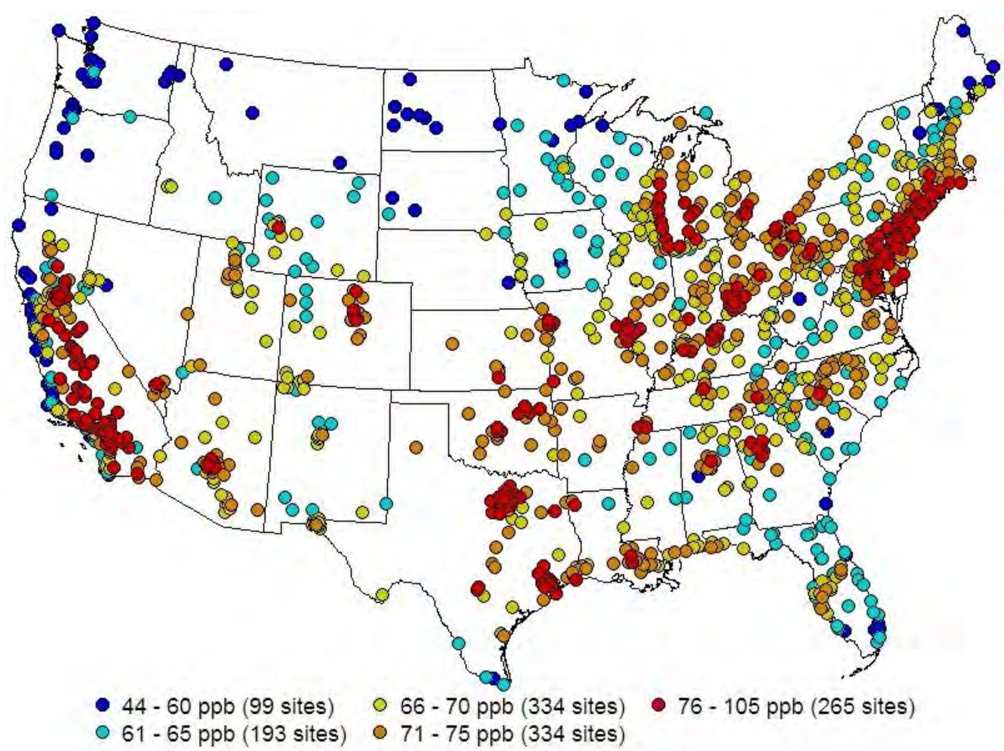

Figure 2-3. Map of 8-hour Ozone Design Values in ppb, Averaged Across the 2009-2011, 2010-2012, and 2011-2013 Periods

In addition to the DV described above, another ozone metric of interest is the W126 index value, which has been found to correlate with ozone-related damage to plants and ecosystems (EPA, 2014c). The W126 metric is a seasonal aggregate of daytime (8:00 AM to 8:00 PM) hourly ozone concentrations designed to measure the cumulative effects of ozone exposure on plant and tree species, with units in parts per million-hours (ppm-hrs). The W126 metric uses a logistic weighting function to place less emphasis on exposure to low hourly ozone concentrations and more emphasis on exposure to high hourly ozone concentrations (Lefohn et al, 1988).

Figure 2-4 shows the trend in annual W126 concentrations in ppm-hrs based on 900 "trends" sites with complete data records over the 2000 to 2013 period. The center line in this figure represents the median value across the trends sites, while the dashed lines represent the 25th and 75th percentiles, and the bottom and top lines represent the 10th and 90th percentiles. Figure 2-5 shows a map of the 3-year average annual W126 concentrations in ppm-hrs averaged across the 2009-2011, 2010-2012, and 2011-2013 periods at all monitoring sites in the contiguous U.S. The general patterns seen in these figures are similar to those seen in the DV metric for the existing standard.

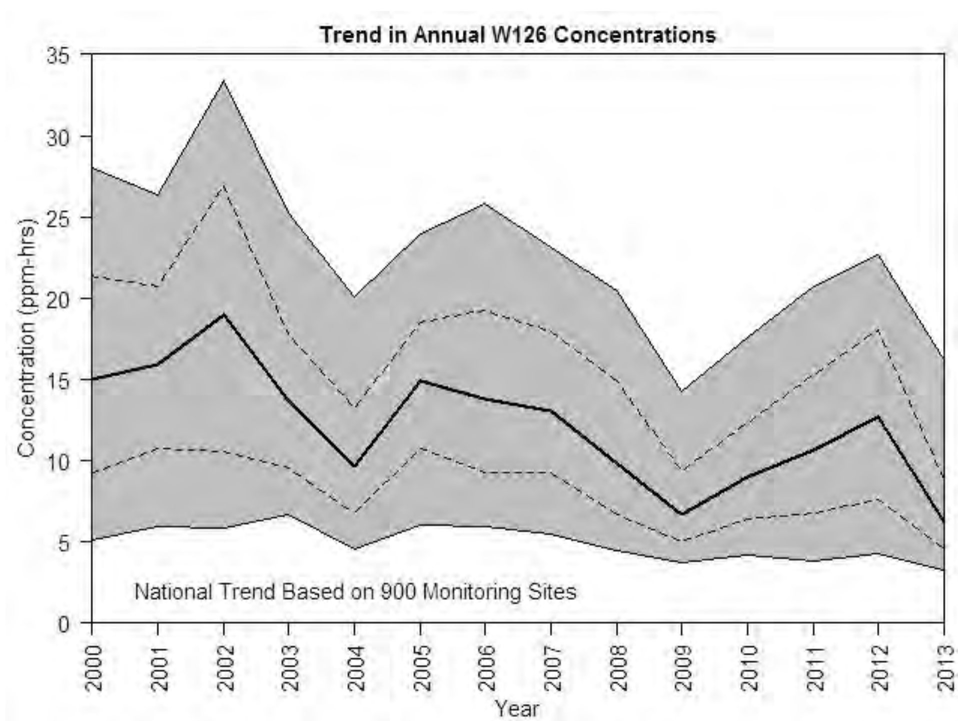

Figure 2-4. Trend in U.S. Annual W126 Concentrations in ppm-hrs, 2000 to 2013. Solid center line represents the median value across monitoring sites, dashed lines represent 25th and 75th percentile values, and top/bottom lines represent 10th and 90th percentile values.

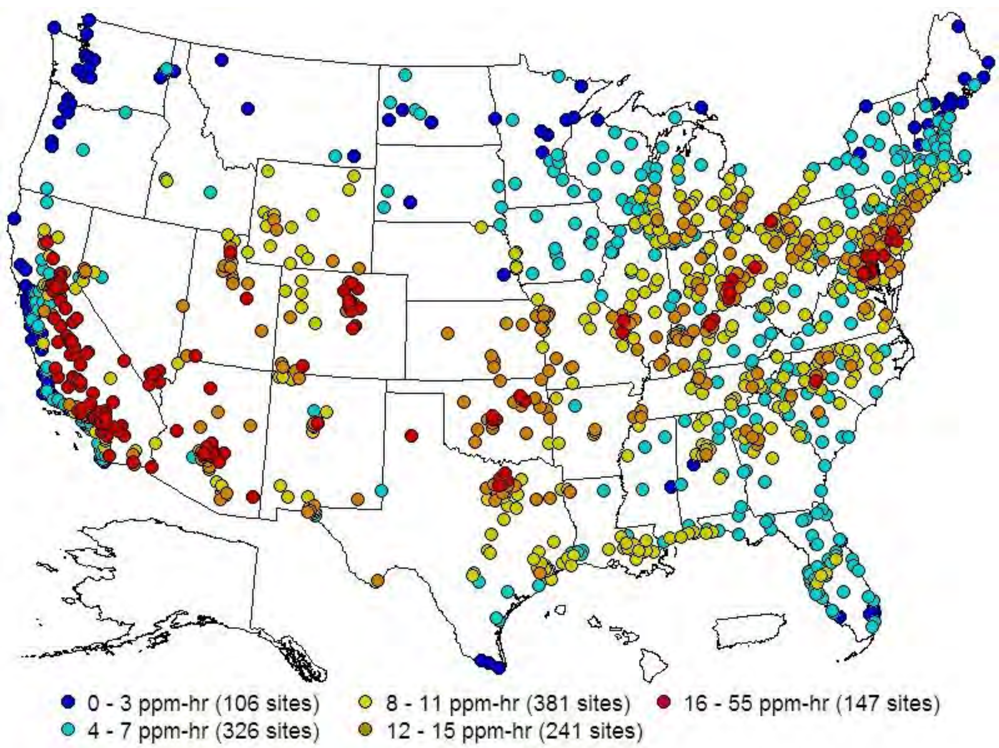

Figure 2-5. Map of 3-year Average W126 Values in ppm-hrs, Averaged Across the 2009-2011, 2010-2012, and 2011-2013 Periods

2.4 Background Ozone

One of the aspects of ozone that is unusual relative to the other pollutants with NAAQS is that, periodically, in some locations, an appreciable fraction of the observed ozone results from sources or processes other than local and domestic regional anthropogenic emissions of ozone precursors (Fiore *et al.*, 2002). Any ozone formed by processes other than the chemical conversion of local or regional ozone precursor emissions is generically referred to as "background" ozone. Background ozone can originate from natural sources of ozone and ozone precursors, as well as from manmade international emissions of ozone precursors. Natural sources of ozone precursor emissions such as wildfires, lightning, and vegetation can lead to ozone formation by chemical reactions with other natural sources. Another important component of background is ozone that is naturally formed in the stratosphere through interactions of ultraviolet light with molecular oxygen. Stratospheric ozone can mix down to the surface at high concentrations in discrete events called intrusions, especially at higher-altitude locations. The manmade portion of the background includes any ozone formed due to anthropogenic sources of ozone precursors emitted far away from the local area (e.g., international emissions). Finally, both biogenic and international anthropogenic emissions of methane, which can be chemically converted to ozone over relatively long time scales, can also contribute to global background ozone levels. Away from the surface, ozone can have an atmospheric lifetime on the order of weeks. As a result, background ozone can be transported long distances in the upper troposphere and, when meteorological conditions are favorable, be available to mix down to the surface and add to the ozone loading from non-background sources.

The definition of background ozone can vary depending upon context, but it generally refers to ozone that is formed by sources or processes that cannot be influenced by actions within the jurisdiction of concern. In the Policy Assessment for the Review of the Ozone National Ambient Air Quality Standards (US EPA, 2014c), EPA identified three specific definitions of background ozone: natural background (NB), North American background (NAB), and United States background (USB). Natural background is the narrowest definition of background, and it is defined as the ozone that would exist in the absence of any manmade ozone precursor emissions. The other two definitions of background are based on a presumption that the U.S. has little influence over anthropogenic emissions outside either our continental or domestic borders. North American background is defined as that ozone that would exist in the absence of any manmade ozone precursor emissions from North America. U.S. background is defined as that ozone that would exist in the absence of any manmade emissions inside the United States.

Modeling studies have estimated what background levels would be in the absence of certain sets of emissions by simply assessing the remaining ozone in a simulation in which certain emissions were removed (Zhang et al. (2011), Emery et al. (2012), US EPA (2014c)). This basic approach is often referred to as “zero-out” modeling or “emissions perturbation” modeling. While the zero-out approach has traditionally been used to estimate natural background, North American background, and U.S. background, the methodology has an acknowledged limitation. It cannot answer the question of how much of the existing observed ozone results from background sources or processes.

A separate modeling technique can be used to estimate the contribution of background ozone and other contributing source terms to total ozone within a model. This approach, referred to as "source apportionment" modeling, has been described and evaluated in the peer-reviewed literature (Dunker et al., 2002; Kemball-Cook et al., 2009). Source apportionment modeling has frequently been used in other regulatory settings to estimate the "contribution" to ozone of certain sets of emissions (EPA 2005, EPA 2011). The source apportionment technique provides a means of estimating the contributions of each user-identified source category to ozone formation in a single model simulation. This is achieved by using multiple tracer species to track the fate of ozone precursor emissions (VOC and NO_X) and the ozone formation resulting from these emissions. The methodology is designed so that all ozone and precursor concentrations are tracked and apportioned to the selected source categories at all times without perturbing the inherent chemistry. The primary limitation of the source apportionment modeling is that its estimations of background ozone are explicitly linked to the emissions scenarios modeled and would change with different emissions scenarios.

2.4.1 Seasonal Mean Background Ozone in the U.S.

The ISA (US EPA 2013, section 3.4) previously established that background ozone concentrations vary spatially and temporally and that simulated mean background concentrations are highest at high-elevation sites within the western U.S. Background levels typically are greatest over the U.S. in the spring and early summer. EPA modeling presented in the Policy Assessment for the Review of the Ozone National Ambient Air Quality Standards (US EPA, 2014c) focused on the months from April to October for 2007 (note that the 2007 modeling is separate from the 2011 modeling described in Chapter 3 and used as the basis of cost and benefit numbers in this RIA). Emissions and model set-up for the 2007 analysis are described in more detail in the Policy Assessment. Briefly, the emissions for 2007 were derived from the 2008 National Emissions Inventory but included 2007 year-specific emissions where available. Wildfire emissions were based on a multi-year climatological average as this analysis was meant to capture seasonal mean background and typical ranges rather than explicitly quantify background ozone on specific days. Figure 2-6 displays the spatial patterns of seasonal mean natural background ozone as estimated by a 2007 zero-out scenario using the Community Multiscale Air Quality (CMAQ) model. Seasonal means are computed over those seven months. This figure shows the average daily maximum 8-hour ozone concentration that would exist in the absence of any anthropogenic ozone precursor emissions at monitor locations. As shown, seasonal mean NB levels range from approximately 15-35 ppb (i.e., +/- 1 standard deviation) with the highest values at higher-elevation sites in the western U.S. The median value over these locations is 24.2 ppb, and more than 50 percent of the locations have natural background levels of 20-25 ppb. The highest modeled estimate of seasonal average, natural background, 8-hr daily maximum ozone is 34.3 ppb at the high-elevation CASTNET site (Gothic) in Gunnison County, CO. Natural background ozone levels are higher at these high-elevation locations primarily because of natural stratospheric ozone impacts and international transport impacts that increase with altitude (where ozone lifetimes are longer).

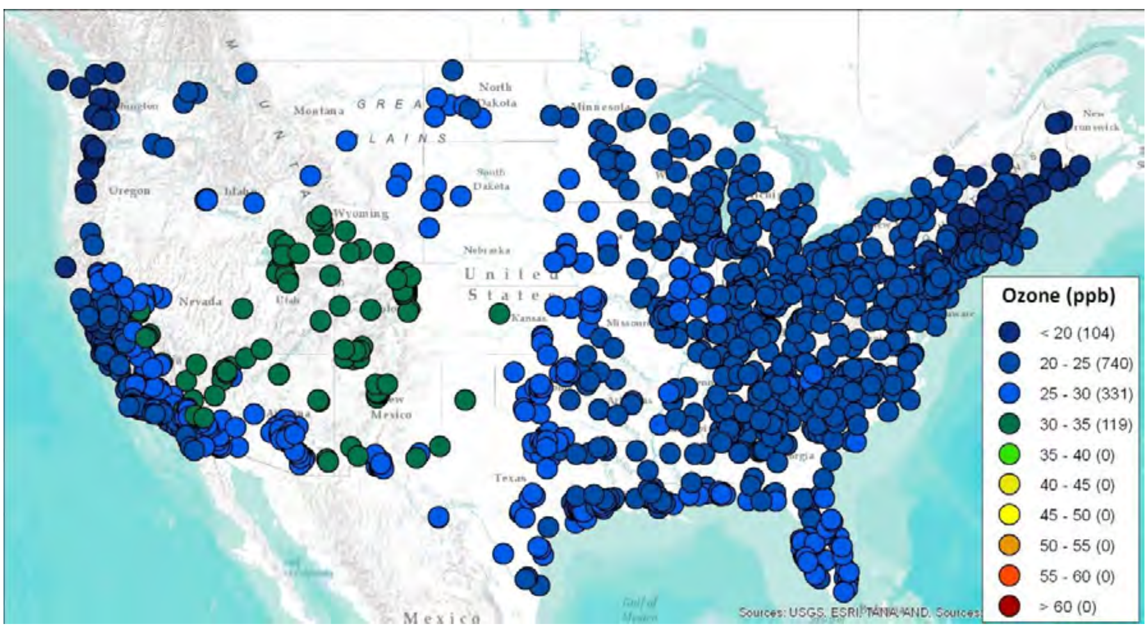

Figure 2-6. Map of 2007 CMAQ-estimated Seasonal Mean of 8-hour Daily Maximum Ozone from Natural Background (ppb) based on Zero-Out Modeling

Figures 2-7 and 2-8 show the same information for the NAB and USB scenarios. In these model runs, all anthropogenic ozone precursor emissions were removed from the U.S., Canada, and Mexico portions of the modeling domain (NAB scenario) and then only from the U.S. (USB scenario). The figures show that there is not a large difference between the NAB and USB scenarios. Seasonal mean NAB and USB ozone levels range from 25-50 ppb, with the most frequent values estimated in the 30-35 ppb range. The median seasonal mean background levels are 31.5 and 32.7 ppb (NAB and USB, respectively). Again, the highest levels of seasonal mean background ozone are predicted over the intermountain western U.S. Locations with NAB and USB concentrations greater than 40 ppb are confined to Colorado, Nevada, Utah, Wyoming, northern Arizona, eastern California, and parts of New Mexico. The 2007 EPA modeling suggests that seasonal mean USB concentrations are on average 1-3 ppb higher than NAB background. These results were similar to those reported by Wang et al. (2009). From a seasonal mean perspective, background ozone levels are below the NAAQS thresholds.

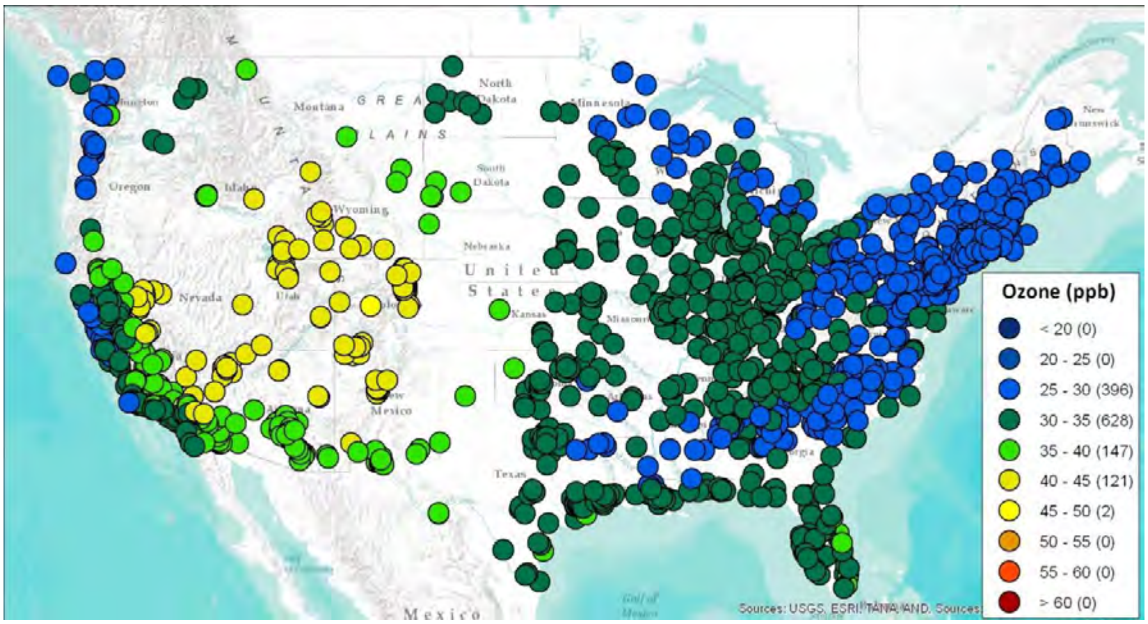

Figure 2-7. Map of 2007 CMAQ-estimated Seasonal Mean of 8-hour Daily Maximum Ozone from North American Background (ppb) based on Zero-out Modeling

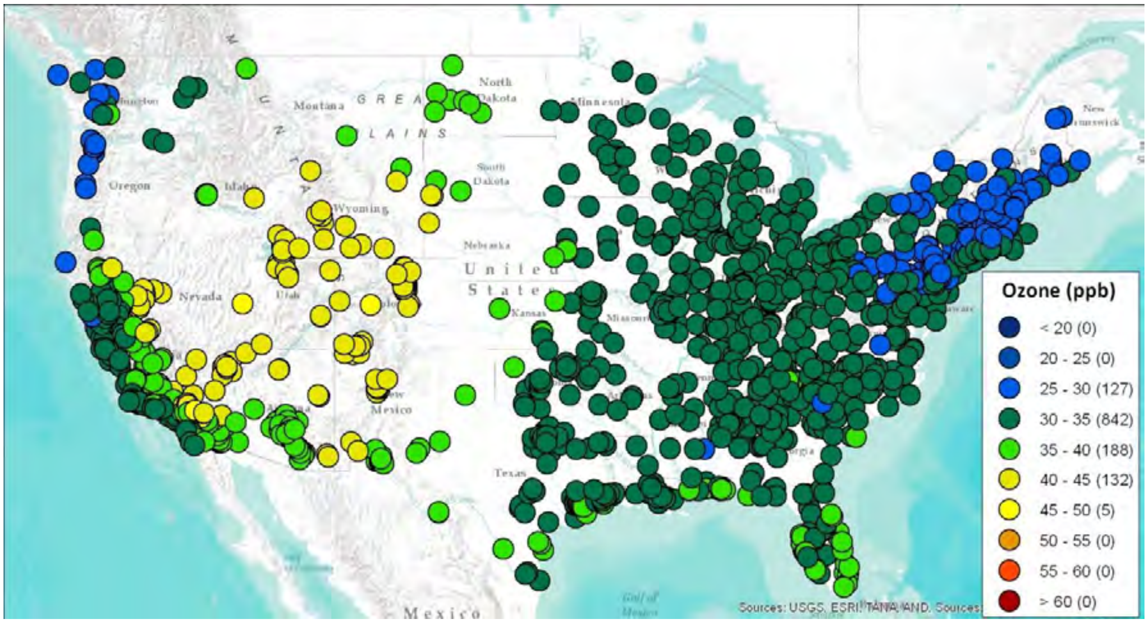

Figure 2-8. Map of 2007 CMAQ-estimated Seasonal Mean of 8-hour Daily Maximum Ozone from United States Background (ppb) based on Zero-Out Modeling

2.4.2 Seasonal Mean Background Ozone in the U.S. as a Proportion of Total Ozone

Another informative way to assess the importance of background ozone as part of seasonal mean ozone levels across the U.S. is to consider the ratios of NB, NAB, and USB to total modeled ozone at each monitoring location. Considering the proportional impact of background ozone allows for an initial assessment of the relative importance of background and non-background sources. Because ozone chemistry is non-linear, one should not assume that individual perturbations (e.g., zero-out runs) are additive in all locations. Figures 2-9 and 2-10 show the ratio of U.S. background to total ozone using the metric of the seasonal mean 8-hr daily maximum ozone concentrations as estimated by both the zero-out and source apportionment modeling methodologies. Recall that the terms NB, NAB, and USB are explicitly linked to the zero-out modeling approach. For comparison, in Figure 2-10 we are extending the definition of USB to also include the source apportionment model estimates of the ozone that are *attributable to sources other than U.S. anthropogenic emissions*. To preserve the original definition of USB, this second term will be hereafter referred to as "apportionment-based USB". As noted earlier, the advantage of the source apportionment modeling is that all of the modeled ozone is attributed to various source terms without perturbing the inherent chemistry. Thus, this approach is not affected by the confounding occurrences of background ozone values exceeding the base ozone values as can happen in the zero-out modeling (i.e., background proportions > 100%).

Consequently, one would expect the fractional background levels to be lower in the source apportionment methodology as a result of removing this artifact.

When averaged over all sites, ozone from sources other than U.S. anthropogenic emissions is estimated to comprise 66 (zero-out) and 59 (source apportionment) percent of the total seasonal ozone mean. The spatial patterns of USB and apportionment-based USB are similar across the two modeling exercises. Background ozone is a relatively larger percentage (e.g., 70-80%) of the total seasonal mean ozone in locations within the intermountain western U.S. and along the U.S. border. In locations where ozone levels are generally higher, like California and the eastern U.S., the seasonal mean background fractions are relatively smaller (e.g., 40-60%). The additional 2007 modeling confirms that background ozone, while generally not approaching levels of the ozone standard, can comprise a considerable fraction of total seasonal mean ozone across the U.S (EPA, 2014c).

2.4.3 Daily Distributions of Background Ozone within the Seasonal Mean

As a first-order understanding, it is valuable to be able to characterize seasonal mean levels of background ozone. However, it is well established that background levels can vary substantially from day-to-day within the seasonal mean. From an implementation perspective, the values of background ozone on possible exceedance days are a more meaningful consideration. The Policy Assessment for the Review of the Ozone National Ambient Air Quality Standards (US EPA, 2014c) concluded that “anthropogenic sources within the U.S. are largely responsible for 4^{th} highest 8-hour daily maximum O_3 concentrations” based on modeling using a 2007 base year and the two distinct modeling methodologies described above. Figure 2-11 and 2-12 show the distribution of daily MDA8 apportionment-based USB levels (absolute magnitudes and relative fractions, respectively) from the CAMx simulation. The 2007 modeling shows that the days with highest ozone levels have similar distributions (i.e., means, inter-quartile ranges) of background ozone levels as days with lower values, down to approximately 40 ppb. As a result, the proportion of total ozone that has background origins is smaller on high ozone days (e.g., days > 60 ppb) than on the more common lower ozone days that tend to drive seasonal means. Figure 2-11 also indicates that there are cases in which the model predicts much larger background proportions, as shown by the upper outliers in the figure. These infrequent episodes usually occur in relation to a specific event, and occur more often in specific geographical locations, such as at high elevations or wildfire prone areas during the local dry season.

It should be noted here that EPA has policies for treatment of air quality monitoring data affected by these types of events. EPA's exceptional events policy allows exclusion of certain air quality monitoring data from regulatory determinations if a State adequately demonstrates that an exceptional event has caused the exceedance or violation of a NAAQS. In addition, Section 179B of the Clean Air Act (CAA) also provides for treatment of air quality data from international transport when an exceedance or violation of a NAAQS would not have occurred but for the emissions emanating from outside of the United States. Finally, CAA section 182(h) authorizes the EPA Administrator to determine that an area designated nonattainment can be treated as a "rural transport area". In accordance with the statute, a nonattainment area may qualify for this distinction if it meets the following criteria: 1) the area does not contain emissions sources that make a significant contribution to monitored ozone concentrations in the area, or in other areas; and 2) the area does not include and is not adjacent to an MSA. More information regarding how background ozone is addressed in Clean Air Act implementation is provided in Section VII.F of the notice of proposed rulemaking.

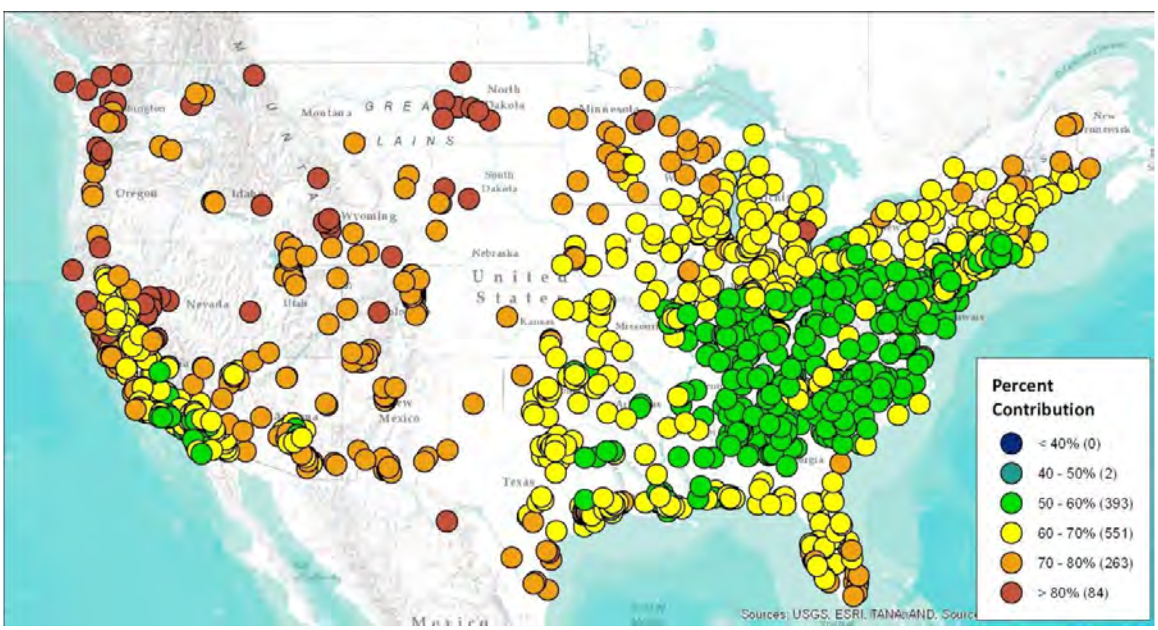

Figure 2-9. Map of Site-Specific Ratios of U.S. Background to Total Seasonal Mean Ozone based on 2007 CMAQ Zero-Out Modeling

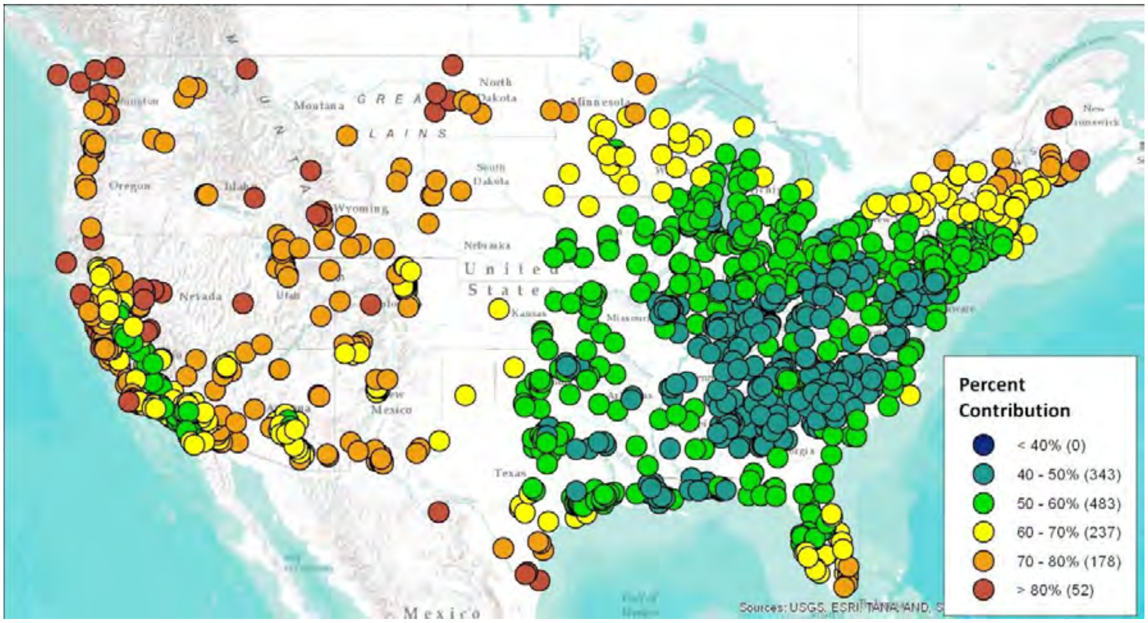

Figure 2-10. Map of Site-Specific Ratios of Apportionment-Based U.S. Background to Seasonal Mean Ozone based on 2007 CAMx Source Apportionment Modeling

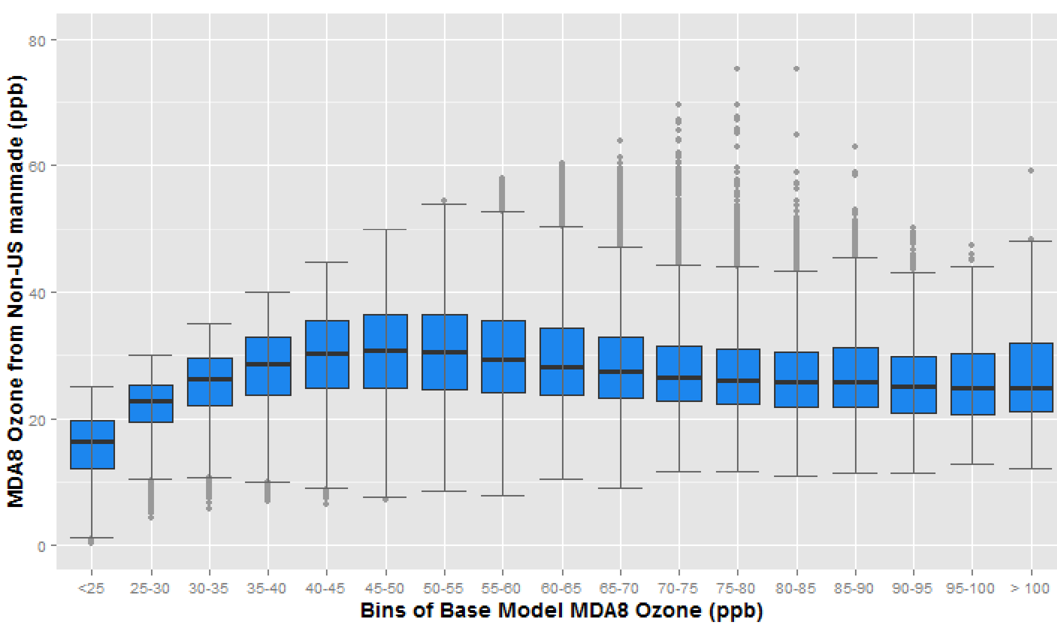

Figure 2-11. Distributions of Absolute Estimates of Apportionment-Based U.S. Background (all site-days), Binned by Modeled MDA8 from the 2007 Source Apportionment Simulation

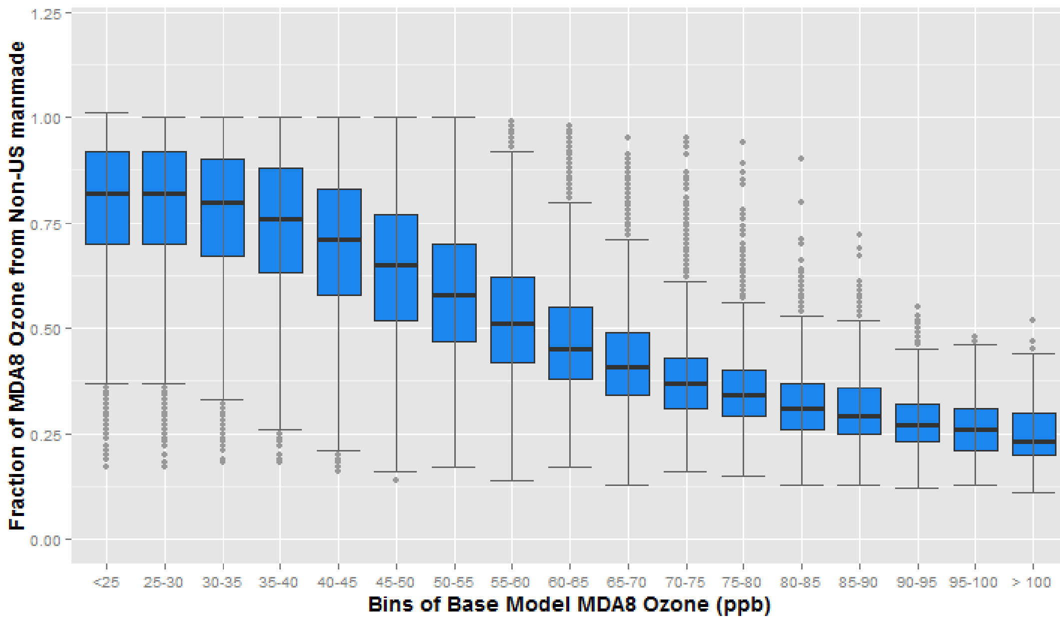

Figure 2-12. Distributions of the Relative Proportion of Apportionment-Based U.S. Background to Total Ozone (all site-days), Binned by Modeled MDA8 from the 2007 Source Apportionment Simulation

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CHAPTER 3: AIR QUALITY MODELING AND ANALYSIS

Overview

This regulatory impacts analysis (RIA) evaluates the costs as well as the health and environmental impacts associated with complying with alterative National Ambient Air Quality Standards (NAAQS) for ozone. For this purpose, we use air quality modeling to project ozone concentrations into the future. This chapter describes the data, tools and methodology used for the analysis, as well as the post-processing techniques used to produce a number of ozone metrics necessary for this analysis.

Throughout this chapter, the base year modeling refers to model simulations conducted for 2011 while the 2025 base case simulation refers to a photochemical model run conducted with emissions projected to the year 2025 assuming all current on-the-books federal regulations will apply[14]. A series of 2025 emissions sensitivity cases are created to determine ozone response to emissions changes incremental to the 2025 base case. Finally, a set of four scenarios are developed based on the 2025 base case and emissions sensitivity cases: the baseline scenario (a scenario which applies additional controls to the 2025 base case that would be required to meet the current standard of 75 ppb), and 3 alternative standard scenarios which represent incremental emissions reductions beyond the baseline to meet potential standard levels of 70, 65, and 60 ppb.

Section 3.1 describes the air quality modeling simulations, section 3.2 describes how current and future ozone design values are calculated, section 3.3 describes the methodology for determining necessary emissions reductions for meeting various alternative NAAQS levels, and section 3.4 describes the creation of spatial surfaces that act as inputs to health and welfare benefits calculations.

[14] The 2012 PM NAAQS is not included in the 2025 base case because the scenarios modeled in PM NAAQS RIA did not reflect any NOx emissions reductions (US EPA, 2012)

3.1 Modeling Ozone Levels in the Future

A national scale air quality modeling analysis was performed to estimate ozone concentrations for the future year of 2025. Ozone sensitivity factors were developed using the modeled response of ozone to changes in NO_x and VOC emissions from various sources and locations. The sensitivity factors were used to calculate ratios of changes in ozone to changes in emissions in order to determine the amount of emissions reductions needed to reach the baseline and evaluate potential alternative standard levels of 70, 65, and 60 ppb incremental to the baseline. The resulting emissions reductions were then used to estimate how health- and welfare-related ozone concentration metrics would change under each scenario. The metrics were used as inputs to the calculation of expected costs and benefits associated with the precursor emissions and ozone concentration changes resulting from just attaining the alternative ozone standards.

As described in section 3.2, air quality modeling was used in a relative sense to project future concentrations of ozone. As part of this approach, ozone predictions from the 2011 base year simulation are coupled with predictions from the 2025 modeling to calculate the relative change (between 2011 and 2025) in concentrations. These relative response factors (RRFs) were applied to the corresponding measured design values[15] (DVs) to predict future DVs. Multiple emissions cases were modeled for 2025 including a 2025 base case and twelve 2025 emissions sensitivity simulations. Details on the 2011-based air quality modeling platform, the 2025 base case and emissions sensitivity simulations, along with the methods and results for attaining these NAAQS levels are provided below.

3.1.1 Selection of Future Analytic Year

The RIA evaluates, to the extent possible, the costs and benefits of attaining the proposed alternative ozone standards, incremental to attaining the existing 75 ppb ozone standard and implementing existing and expected regulations. We selected 2025 as the primary year of analysis because most areas of the U.S. will likely be required to meet a revised ozone standard by 2025. We assumed that potential nonattainment areas everywhere in the U.S., excluding

[15] The design value is the metric that is compared to the standard level to determine whether a monitor is violating the NAAQS. The ozone design value is described in more detail in section 3.2.

California, will be designated such that they are required to attain by 2025, and we developed our projected baselines for emissions, ozone, and populations for 2025.

In estimating the incremental costs and benefits of potential alternative standards, we recognize that there are several areas that are not required to meet the existing ozone standard of 75 ppb set in 2008 by the year 2025. The Clean Air Act allows areas with more significant air quality problems to take additional time to reach the existing standard. Several areas in California are not required to meet the existing standard by 2025 and may not be required to meet a revised standard until sometime between 2032 and 2037.

We projected emissions for and modeled a single future base case year (2025), however for California we adjusted the future baseline ozone concentrations to reflect the effects of mobile source emissions reductions that will occur in California between 2025 and 2030 as described in Section 3.3. While there is uncertainty about the precise timing of emissions reductions and related costs for California, we assume costs occur through the end of 2037 and beginning of 2038. In addition, as described in Chapter 5, we model benefits for California using projected population demographics for 2038.

Because of the different timing for incurring costs and accruing benefits in California and for ease of discussion throughout the analyses, we refer to the different time periods for potential attainment as 2025 and post-2025, to reflect that (1) we did not project emissions and air quality for any year other than 2025; (2) costs in California are assumed to be incurred starting in 2032 and later; and (3) benefits from attainment of alternative standards in California are modeled using population demographics in 2038.

3.1.2 Air Quality Modeling Platform

The 2011-based air quality modeling platform was used to provide emissions, meteorology and other inputs to the 2011 and 2025 air quality model simulations. This platform was chosen because it represents the most recent, complete set of base year emissions information currently available for national-scale modeling.

We use the Comprehensive Air Quality Model with Extensions (CAMx version 6.1) for photochemical model simulations performed for the RIA. CAMx 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.) (Environ, 2014). Consideration of the different processes (e.g., transport and deposition) that affect primary (directly emitted) and secondary (formed by atmospheric processes) pollutants at the regional scale in different locations is fundamental to understanding and assessing the effects of emissions control measures that affect air quality concentrations. Because it accounts for spatial and temporal variations as well as differences in the reactivity of emissions, CAMx is useful for evaluating the impacts of the control strategies on ozone concentrations. CAMx is applied with the carbon-bond 6 revision 2 (CB6r2) gas-phase chemistry mechanism (Ruiz and Yarwood, 2013).

Figure 3-1. Map of the CAMx Modeling Domain Used for Ozone NAAQS RIA

Figure 3-1 shows the geographic extent of the modeling domain that was used for air quality modeling in this analysis. The domain covers the 48 contiguous states along with the southern portions of Canada and the northern portions of Mexico. This modeling domain contains 25 vertical layers with a top at about 17,600 meters, or 50 millibars (mb), and horizontal resolution of 12 km x 12 km. The model simulations produce hourly air quality concentrations for each 12 km grid cell across the modeling domain.

CAMx 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 2011 base year, the 2025 base case, and the 2025 emissions sensitivity simulations. All other inputs (i.e. meteorological fields, initial conditions, and boundary conditions) were specified for the 2011 base year model application and remained unchanged for each future-year modeling simulation. The assumption of constant meteorology and boundary conditions was applied for two reasons: 1) this allows us to isolate the impacts of U.S. emissions changes, and 2) there is considerable uncertainty in the direction and magnitude in any changes in these parameters. EPA recognizes that changes in climate and international emissions may impact these model inputs. Specifically, climate change may lead to temperature increases, higher stagnation frequency, and increased wildfire activity, all of which could lead to higher ozone concentrations. In the western U.S. over the last 15 years, increasing wildfires have already been observed (Dennison et al., 2014). Potential future elevated ozone concentrations could, in turn, necessitate more stringent emissions reductions. However, there are significant uncertainties regarding the precise location and timing of climate change impacts on ambient air quality. Generally, climate projections are most robust for periods at least several decades in the future because the forcing mechanisms that drive near-term natural variability in climate patterns (e.g., El Nino, North American Oscillation) have substantially larger signals over short time spans than the driving forces related to long-term climate change. Boundary conditions, which are impacted by international emissions and may also influence future ozone concentrations, are held constant in this analysis based on a similar rationale regarding the significant uncertainty in estimating future levels.

CAMx 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 and precursors to secondary pollutants. The annual emission inventories, described in Section 3.1.3, were preprocessed into CAMx-ready inputs using the Sparse Matrix Operator Kernel Emissions (SMOKE) modeling system (Houyoux et al., 2000).

Meteorological inputs reflecting 2011 conditions across the contiguous U.S. were derived from Version 3.4 of the Weather Research Forecasting Model (WRF) (Skamarock, 2008). 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 2011 meteorological model simulation and evaluation are provided in a separate technical support document (US EPA, 2014a).

The lateral boundary and initial species concentrations are provided by a three-dimensional global atmospheric chemistry model, GEOS-Chem (Yantosca, 2004) standard version 8-03-02 with 8-02-01 chemistry. The global GEOS-Chem model simulates atmospheric chemical and physical processes driven by assimilated meteorological observations from the NASA's Goddard Earth Observing System (GEOS-5; additional information available at: http://gmao.gsfc.nasa.gov/GEOS/ and http://wiki.seas.harvard.edu/geos-chem/index.php/GEOS-5). This model was run for 2011 with a grid resolution of 2.0 degrees x 2.5 degrees (latitude-longitude). The predictions were used to provide one-way dynamic boundary conditions at one-hour intervals and an initial concentration field for the CAMx simulations. A model evaluation was conducted to validate the appropriateness of this version and model configuration of GEOS-Chem for predicting selected measurements relevant to their use as boundary conditions for CAMx. This evaluation included using satellite retrievals paired with GEOS-Chem grid cell concentrations (Henderson, 2014). More information is available about the GEOS-Chem model and other applications using this tool at: http://www-as.harvard.edu/chemistry/trop/geos.

An operational model performance evaluation for ozone was performed to estimate the ability of the CAMx modeling system to replicate 2011 measured concentrations. This evaluation focused on statistical assessments of model predictions versus observations paired in time and space depending on the sampling period of measured data. Details on the evaluation methodology and the calculation of performance statistics are provided in Appendix 3A. Overall, the model performance statistics for ozone from the CAMx 2011 simulation are within or close to the ranges found in other recent peer-reviewed applications (Simon et al, 2012). These model performance results give us confidence that our application of CAMx using this 2011 modeling platform provides a scientifically credible approach for assessing ozone concentrations for the purposes of the RIA.

3.1.3 Emissions Inventories

The 2011 base year and 2025 base case emissions inventories are described in the Technical Support Document: Preparation of Emissions Inventories for the Version 6.1, 2011 Emissions Modeling Platform (US EPA, 2014b). Section 4 of the technical support document (TSD) summarizes the control and growth assumptions by source type that were used to create the U.S. 2025 base case emissions inventory, and includes a table of such assumptions for each major source sector. Below we summarize the characteristics of the 2025 base case emissions for each major source category.

The 2025 electric generating unit (EGU) projected inventory represents demand growth, fuel resource availability, generating technology cost and performance, and other economic factors affecting power sector behavior. The EGU emissions were developed using the Integrated Planning Model (IPM) version 5.13 (http://www.epa.gov/powersectormodeling/BaseCasev513.html). IPM is a multiregional, dynamic, deterministic linear programming model of the U.S. electric power sector. IPM reflects the expected 2025 emissions accounting for the effects of environmental rules and regulations, consent decrees and settlements, plant closures, units built, control devices installed, and forecast unit construction through the calendar year 2025. In this analysis, the projected EGU emissions include impacts from the Final Mercury and Air Toxics Standard (MATS) announced on December 21, 2011 and the Clean Air Interstate Rule (CAIR) issued March 10, 2005[16].

[16] A sensitivity case described in Section 3.1.4 also included a representation of EPA's proposed carbon pollution guidelines under section 111(d) of the Clean Air Act (CAA)

Projections for most stationary emission sources other than EGUs (i.e., non-EGUs) were developed by using the EPA Control Strategy Tool (CoST) to create future year inventories. CoST is described at http://www.epa.gov/ttnecas1/cost.htm. The 2025 base case non-EGU stationary source emissions inventory includes all enforceable national rules and programs including the Reciprocating Internal Combustion Engines (RICE) and cement manufacturing National Emissions Standards for Hazardous Air Pollutants (NESHAPs) and Boiler Maximum Achievable Control Technology (MACT) reconsideration reductions. Projection factors and percent reductions for non-EGU point sources reflect comments previously received by EPA, along with emissions reductions due to national and local rules, control programs, plant closures, consent decrees and settlements. Projection approaches for corn ethanol and biodiesel plants, refineries and upstream impacts represent the Energy Independence and Security Act (EISA) renewable fuel standards mandate in the Renewable Fuel Standards Program (RFS2). Airport-specific terminal area forecast (TAF) data were used for aircraft to account for projected changes in landing/takeoff activity.

Regional projection factors for point and nonpoint oil and gas emissions were developed by product type using Annual Energy Outlook (AEO) 2013 projections to year 2025 (http://www.eia.gov/forecasts/aeo/). Stationary engine criteria air pollutant (CAP) co-benefit reductions (i.e., from the RICE NESHAP) and New Source Performance Standards (NSPS) VOC controls are reflected for oil and gas sources.

Projection factors for livestock are based on expected changes in animal population from 2005 Department of Agriculture data, updated according to EPA experts in July 2012; fertilizer application NH3 emissions projections include upstream impacts representing EISA. Area fugitive dust projection factors for categories related to livestock estimates are based on expected changes in animal population and upstream impacts from EISA. Residential Wood Combustion (RWC) projection factors reflect assumed growth of wood burning appliances based on sales data, equipment replacement rates and change outs. These changes include growth in lower-emitting stoves and a reduction in higher emitting stoves. Projection factors for the remaining nonpoint sources such as stationary source fuel combustion, industrial processes, solvent utilization, and waste disposal, implement comments received on the projection of these sources as a result of recent rulemakings and outreach to states on emission inventories, and they also include emission reductions due to control programs. Portable fuel container (PFC) projection factors reflect the impact of the final Mobile Source Air Toxics (MSAT2) rule. Upstream impacts from EISA, including post-2011 cellulosic ethanol plants are also reflected.

For onroad, nonroad, and commercial marine vessel mobile sources, all national measures for which data were available at the time of modeling have been included. The Tier 3 standards finalized in March, 2014 (see http://www.epa.gov/otaq/tier3.htm) are represented in the onroad and nonroad emissions. The 2011 and 2025 onroad mobile source emissions were developed using emissions factors derived from the Tier 3 FRM version of the MOtor Vehicle Emission Simulator (MOVES; http://www.epa.gov/otaq/models/moves/). The emissions factors for year 2025 were developed using the same meteorology and procedures used to produce the 2011 emission factors. The onroad mobile source emissions were computed by using SMOKE to combine the county-, vehicle type-, and temperature-specific emission factors with vehicle miles traveled and vehicle population activity data, while taking into account hourly gridded temperature data.

The MOVES-based 2025 onroad emissions account for changes in activity data and the impact of on-the-books national rules including: the Tier 3 Vehicle Emission and Fuel Standards Program, the Light-Duty Vehicle Tier 2 Rule, the Heavy Duty Diesel Rule, the Mobile Source Air Toxics Rule, the Renewable Fuel Standard (RFS2), the Light Duty Green House Gas/Corporate Average Fuel Efficiency (CAFÉ) standards for 2012-2016, the Heavy-Duty Vehicle Greenhouse Gas Rule, the 2017 and the Later Model Year Light-Duty Vehicle Greenhouse Gas Emissions and Corporate Average Fuel Economy Standards; Final Rule (LD GHG). The MOVES-based 2025 emissions also include state rules related to the adoption of LEV standards, inspection and maintenance programs, Stage II refueling controls, and local fuel restrictions. For California, the base case emissions included most of this state's on-the-books regulations, such as those for idling of heavy-duty vehicles, chip reflash, public fleets, track trucks, drayage trucks, and heavy duty trucks and buses. The California emissions do not reflect the impacts of the GHG/Smartway regulation, nor do they reflect state GHG regulations for the projection of other emission sectors because that information was not included in the provided inventories.

The nonroad mobile 2025 emissions, including railroads and commercial marine vessel emissions also include all national control programs. These control programs include the Locomotive-Marine Engine rule, the Nonroad Spark Ignition rule and the Class 3 commercial marine vessel "ECA-IMO" program. For California, the 2025 emissions for these categories reflect the state's Off-Road Construction Rule for "In-Use Diesel", cargo handling equipment rules in place as of 2011 (see http://www.arb.ca.gov/ports/cargo/cargo.htm), and state rules through 2011 related to Transportation Refrigeration Units, the Spark-Ignition Marine Engine and Boat Regulations adopted on July 24, 2008 for pleasure craft, and the 2007 and 2010 regulations to reduce emissions from commercial harbor craft. For ocean-going vessels, the emissions data reflect the 2005 voluntary Vessel Speed Reduction (VSR) within 20 nautical miles, the 2007 and 2008 auxiliary engine rules, the 40 nautical mile VSR program, the 2009 Low Sulfur Fuel regulation, the 2009-2018 cold ironing regulation, the use of 1% sulfur fuel in the Emissions Control Area (ECA) zone, the 2012-2015 Tier 2 NO_x controls, the 2016 0.1% sulfur fuel regulation in ECA zone, and the 2016 International Marine Organization (IMO) Tier 3 NO_x controls. Control and growth-related assumptions for 2025 came from the Emissions Modeling Platform and are described in more detail in EPA (2014b). Non-U.S. and U.S. category 3 commercial marine emissions were projected to 2025 using consistent methods that incorporated controls based on ECA and IMO global NO_x and SO_2 controls.

All modeled 2011 and 2025 emissions cases use the 2006 Canada emissions data. Note that 2006 is the latest year for which Canada had provided data at the time the modeling was performed, and no accompanying future-year projected base case inventories were provided in a form suitable for this analysis. For Mexico, 2012 and 2018 projections of the 1999 Mexico National Emissions Inventory were used as described in the Development of Mexico National Emissions Inventory Projections for 2008, 2012, and 2030 (ERG, 2009) and the associated technical memorandum titled Mexico 2018 Emissions Projections for Point, Area, On-Road Motor Vehicle and Nonroad Mobile Sources (ERG, 2009). Mexico emissions were held at 2018 levels because no 2025 projected emissions were available. Offshore oil platform emissions for the United States represent the year 2008 because 2011 emissions were not available as of the time of the modeling. Biogenic and fire emissions were held constant for all emissions cases and were based on 2011-specific data. Table 3-1 shows the modeled 2011 and 2025 NO_X and VOC emissions by sector. Additional details on the emissions by state are given in the Emissions Modeling TSD.

Table 3-1. 2011 and 2025 Base Case NO_x and VOC Emissions by Sector (thousand tons)

Sector	2011 NO_x	2025 NO_x	2011 VOC	2025 VOC
EGU-point	1,948	1,508	33	42
NonEGU-point	1,768	1,803	872	881
Point oil and gas	17	22	88	107
Wild and Prescribed Fires	347	347	5,175	5,175
Nonpoint oil and gas	653	874	2,273	2,551
Residential wood combustion	36	42	447	489
Other nonpoint	832	856	3,793	3,605
Nonroad	1,630	796	2,025	1,188
Onroad	5,592	1,492	2,738	1,060
C3 Commercial marine vessel (CMV)	125	105	5	8
Locomotive and C1/C2 CMV	1,046	666	48	24
Biogenics	1,018	1,018	40,696	40,696
TOTAL	**15,012**	**9,530**	**58,192**	**55,826**

3.1.4 Emissions Sensitivity Simulations

A total of 12 emissions sensitivity runs were conducted to determine ozone response to emissions reductions of NO_x and VOC in different locations (Table 3-2). We determined that this was an efficient and flexible approach that allowed us to evaluate impacts from multiple source regions and levels of emission reductions simultaneously. All emissions sensitivity simulations were incremental to the 2025 base case emissions described in section 3.1.3. There were three types of emissions cases that were modeled in these sensitivity runs:

1) Explicit emissions control cases
2) Across-the-board reductions in anthropogenic emissions for different pollutants and locations
3) Combination runs that included both explicit emissions controls and across the board reductions.

Explicit Emissions Controls: Four explicit emissions control sensitivity cases were created. First, we modeled a case that represented one possible implementation of the EPA's proposed carbon pollution guidelines under section 111(d) of the Clean Air Act (CAA) (i.e., option 1 state;

hereafter referred to as the 111(d) sensitivity). Emissions for this simulation are described in the regulatory impact analysis for that proposed rule (EPA, 2014c). Second, we modeled three additional emissions cases that included NO_x emissions controls applied to specific sources centered around the three regions of the country projected to have nonattainment monitors above 70 ppb in the 2025 base case: California, Texas, and the Northeastern U.S. Figure 3-2 shows the three areas for which the explicit emissions controls were identified and modeled. CoST was used to determine potential controls in these areas. NO_x controls were identified for all nonpoint, non-EGU point, and nonroad sources that emitted more than 50 tons of NO_x per year and which had available known controls that could be applied for less than $15,000/ton (see chapter 7 for additional discussion). These emissions cases are referred to as "explicit control cases" because they represent the impact of specific controls rather than sensitivities to all emissions within a region. The extent of the area was determined by creating 200 km buffers around all monitors projected above 70 ppb in 2025. All counties that fell completely within the buffer were targeted for controls. In Texas and California these buffers were restricted to state boundaries. In the Northeast, buffers were restricted to states/counties that are currently under the jurisdiction of the Ozone Transport Commission (OTC). More details on the specific emissions controls identified for these emissions cases are provided in Chapter 4.

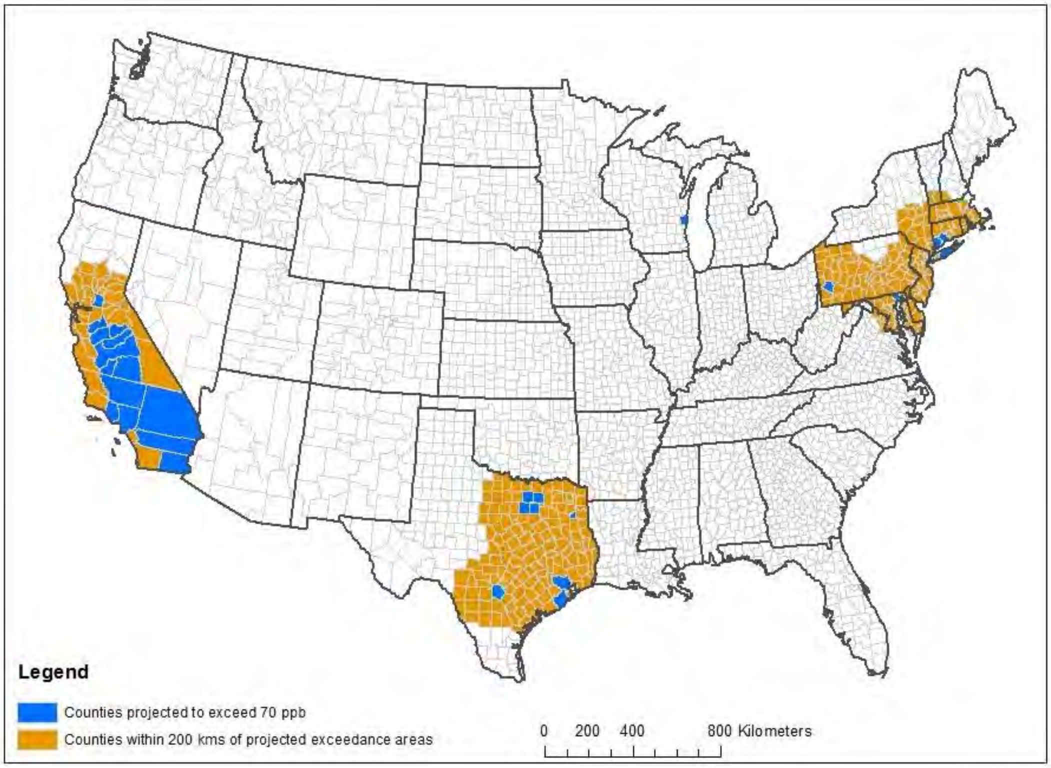

Figure 3-2. Map of Counties for Which Explicit Emissions Controls Were Identified and Modeled in CAMx (shaded in orange) and Counties that Contained One or More Monitor Projected above 70 ppb in the 2025 Base Case Modeling (shaded in blue).[17]

Across-the-board Emissions Reductions: Areas of the U.S. projected to contain monitors with ozone design values greater than 60 ppb were split into 5 regions for the purpose of determining ozone response to emissions reductions (Figure 3-3). Three emissions sensitivity cases with across-the-board cuts in emissions from the 2025 base case were created and modeled:

1) 50% cut in all anthropogenic NO_x in the Southwest region,
2) 50% cut in all anthropogenic NO_x in the Midwest region, and

[17] Note that no buffer was created for the Sheboygan, WI area because emissions reductions from the proposed carbon pollution guidelines under section 111(d) of the CAA are projected to be sufficient to bring that location down to 70 ppb in 2025.

3) 50% cut in all U.S. anthropogenic VOC emissions across the 48 contiguous states.

Combination Emissions Sensitivities: Five additional emissions sensitivity cases were created and modeled that combined the explicit emissions controls with across-the-board reductions. For all combination emissions sensitivity cases, the area over which emissions reductions were applied in the explicit emissions control runs was a subset of the full area for which across-the-board NO_x reductions were applied. These runs included two cases for California: explicit emissions controls in California + additional 50% cut in all California anthropogenic NO_x emissions and explicit emissions controls in California + additional 90% cut in all California anthropogenic NO_x emissions. Two more emission sensitivities were investigated for the Northeast region: explicit emissions controls in the Northeast + additional 50% cut in all Northeast region anthropogenic NO_x emissions and explicit emissions controls in the Northeast + additional 90% cut in all Northeast region anthropogenic NO_x emissions. We identified California and the Northeast as the two regions most likely to need NO_x reductions beyond 50% to reach one or more of the alternative standard levels considered based on a previous EPA analysis (EPA, 2014d). Therefore both a 50% and a 90% NO_x cut were performed for each of these regions to better capture nonlinearities in ozone response to large NOx emissions changes. Finally, a single emissions sensitivity was created for the Central region: explicit emissions controls in Texas + additional 50% cut in all Central region anthropogenic NO_x emissions. A summary of Anthropogenic NO_x and VOC emissions that were considered for controls in this analysis is given in Table 3-3 by region from the 2025 base case and explicit emissions control cases. In other words, the emissions summarized in Table 3-3 only include sectors for which emissions reductions were considered to meet various levels of the ozone standard. Conversely, Table 3-1 summarizes all U.S. emissions that were included in the modeling simulation including sources which contribute to background ozone.

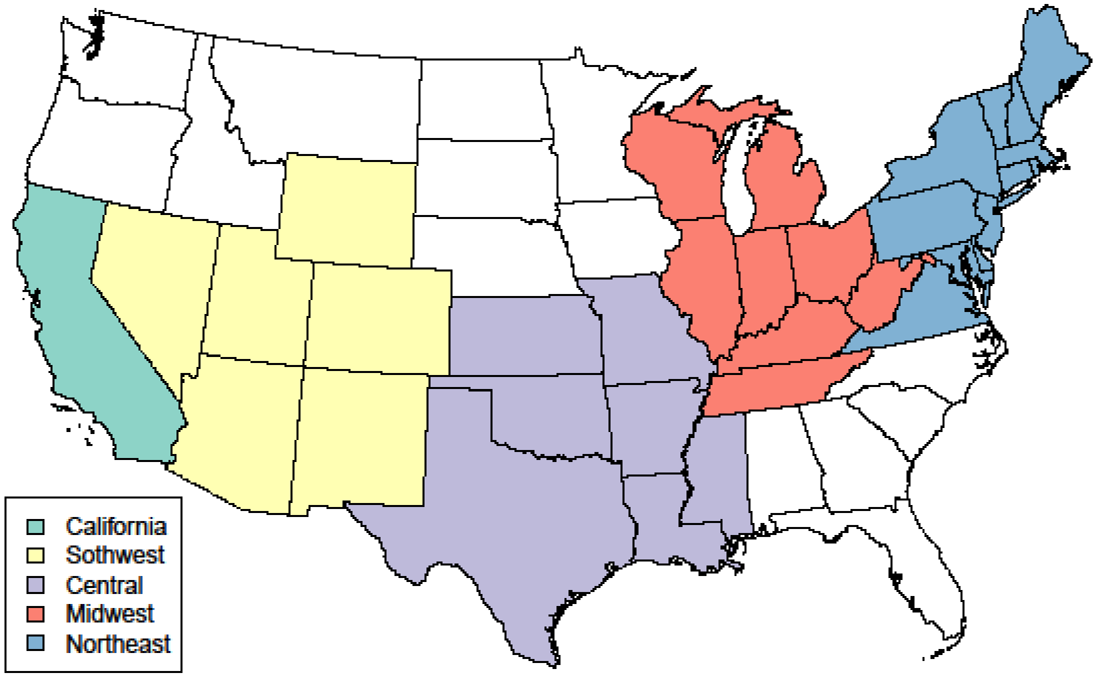

Figure 3-3. Five U.S. Regions Used to Create Across-the-Board Emissions Reduction and Combination Cases

Table 3-2. List of Emissions Sensitivity Cases that Were Modeled in CAMx to Determine Ozone Response Factors

Emissions Sensitivity Case	Region	Pollutant	Emissions Change
1	National	All	111(d) option 1 state
2	National	VOC	50% VOC cut
3	California	NOx	CA explicit emissions control case
4	California	NOx	CA explicit emissions control case + 50% NOx cut
5	California	NOx	CA explicit emissions control case + 90% NOx cut
6	Southwest	NOx	50% NOx cut
7	Texas	NOx	TX explicit emissions control case
8	Central	NOx	TX explicit emissions control case + 50% NOx cut (central)
9	Midwest	NOx	50% NOx cut
10	Northeast	NOx	Northeast explicit emissions control case
11	Northeast	NOx	Northeast explicit emissions control case + 50% NOx cut
12	Northeast	NOx	Northeast explicit emissions control case + 90% NOx cut

Table 3-3. Anthropogenic NO_x and VOC Emissions from the 2025 Base and Explicit Control Cases*

Region	NO_x emissions in 2025 base case (thousand tons)	NO_x emissions in 2025 explicit control cases (thousand tons)	VOC emissions in 2025 base case (thousand tons)
Northeast	1,185	1,074	1,345
Midwest	1,771	----	1,803
Central	2,176	2,073	3,066
Southwest	713	----	1,016
California	446	416	478
Other states	1,840	----	2,264
Total contiguous US	8,130	----	9,971

*Note that unlike Table 3-1, these numbers do not include tribal, biogenic or fire emissions.

3.2 Methods for Calculating Current and Future Year Ozone Design Values

3.2.1 Current Year Ozone Design Value Calculations

As described in chapter 2, hourly ozone concentrations are used to calculate a statistic referred to as a "design value" (DV) which is then compared to the standard level to determine whether a monitor is above or below the NAAQS level in question. For ozone, the DV is calculated as the 3-year average of the annual 4th highest daily maximum 8-hour ozone concentration in parts per billion (ppb), with decimal digits truncated. For the purpose of this analysis, the data handling and data completeness criteria used are those being proposed for the new NAAQS in the proposed appendix U to 40 CFR Part 50 – Interpretation of the Primary and Secondary National Ambient Air Quality Standards for ozone. A standard level of 60 ppb was used when determining data completeness criteria as this results in the most inclusive set of monitoring site DVs for analysis. For the purpose of this analysis, ozone DVs came from data reported in EPA's air quality system (AQS) for the years 2009-2013. The current-year DVs were calculated as the average of 3 consecutive DVs (2009-2011, 2010-2012, and 2011-2013) which creates a 5-year weighted average DV. The 5-year weighted average DV is used as the base from which to project a future year DV as is recommended by the EPA in its SIP modeling guidance (US EPA, 2014e) because it stabilizes year-to-year meteorologically driven variability in ozone DVs given that the future year meteorology is unknown. For cases in which there are fewer than five years of valid monitoring data at a site, the current year DV was calculated only when there was at least three years of consecutive valid data (i.e., at least one complete DV). If a monitor had less than three consecutive years of data, then no current year DV was calculated for that site and the monitor was not used in this analysis.

3.2.2 Future Year Ozone Design Value Projections

Future year ozone design values were calculated at monitor locations using the Model Attainment Test Software (MATS) program (Abt Associates, 2014**)**. MATS calculates the 5-year weighted average DV based on observed data and projects future year values using the relative response predicted by the model as described below. Equation (3-1) describes the recommended model attainment test in its simplest form, as applied for monitoring site *i*:

$$(\mathrm{DVF})_{\mathrm{i}} = (RRF)_i \times (DVB)_i \qquad \text{Equation 3-1}$$

$\mathrm{DVF_i}$ is the estimated design value for the future year in which attainment is required at monitoring site *i*; $\mathrm{RRF_i}$ is the relative response factor at monitoring site *i*; and $\mathrm{DVB_i}$ is the base design value monitored at site i. The relative response factor for each monitoring site $(RRF)_i$ is the fractional change of the DV in the vicinity of the monitor that is simulated on high ozone days due to emissions changes between the base and future years. The recently released draft version of EPA's ozone and $PM_{2.5}$ photochemical modeling guidance (US EPA, 2014e) includes updates to the recommended ozone attainment test used to calculate future year design values for attainment demonstrations. The guidance recommends calculating RRFs based on the highest 10 modeled ozone days in the ozone season near each monitor location. Given the similar goal of this analysis relative to an attainment demonstration, we are using the recommended modeling guidance attainment test approach for the analyses. Specifically, the RRF was only calculated based on the 10 highest days in the base year modeling at the monitor location when the base 8-hr daily maximum ozone values were greater than or equal to 60 ppb for that day. In cases for which the base model simulation did not have 10 days with ozone values greater than or equal to 60 ppb at a site, we used all days where ozone >= 60 ppb, as long as there are at least 5 days that meet that criteria. At monitor locations with less than 5 days with ozone >= 60 ppb, no RRF or DVF was calculated for the site and the monitor in question was not included in this analysis.

In determining the ozone RRF we considered model response in grid cells immediately surrounding the monitoring site along with the grid cell in which the monitor is located, as is currently recommended by the EPA in its SIP modeling guidance (US EPA, 2014e). The RRF was based on a 3 x 3 array of 12 km grid cells centered on the location of the grid cell containing the monitor. The grid cell with the highest base ozone value in the 3 x 3 array was used for both the base and future components of the RRF calculation.

3.3 Determining Tons of Emissions Reductions to Meet Various NAAQS Levels

The following section describes how projected ozone DVs from the 2025 base case and 12 emissions sensitivity cases were used to determine the expected emissions reductions needed to attain the current and potential alternative ozone NAAQS. The scenario for which all U.S. ozone monitors are projected to meet the current ozone NAAQS of 75 ppb is referred to as the "2025 baseline" scenario. The costs and benefits for meeting 70, 65, and 60 ppb standards will be determined incrementally from this baseline. Note that the 2025 baseline is different from the 2025 base case, which is the emissions scenario described in section 3.1.3 and represents the ozone concentrations that are projected to occur in 2025 if there were no distinct reductions made for the purpose of meeting the current or alternative ozone NAAQS.

3.3.1 Determining Ozone Response from Each Emissions Sensitivity

Section 3.2.2 describes, in general terms, how the 2025 projections for ozone DVs were computed. This procedure was followed for the 2025 base case modeling and for each of the 12 emissions sensitivity cases. Using the projected DVs and corresponding emissions changes, a unique ppb per ton response factor was calculated for each ozone monitor and for each emissions sensitivity case based on equation 3-2:

$$R_{i,j} = \frac{DV_{i,j} - DV_{2025base,j}}{\Delta E_i} \qquad \text{Equation 3-2}$$

In equation 3-2, $R_{i,j}$ represents the response at monitor j to emissions changes in emissions sensitivity case i, $DV_{i,j}$ represents the DV at monitor j in emissions sensitivity i, $DV_{2025base,j}$ represents the DV at monitor j in the 2025 base case and ΔE_i represents the difference in NO_x or VOC emissions (tons) between the 2025 base case and emissions sensitivity case i. In cases for which emissions reductions in sensitivity i, were incremental to emissions reductions in another case (k), the following equation was used:

$$R_{i,j} = \frac{DV_{i,j} - DV_{k,j}}{\Delta E_{ik}} \qquad \text{Equation 3-3}$$

in which ΔE_{ik} represents the difference in NO_x or VOC emissions (tons) between the emissions case k and emission case i. Thus at each monitoring site in regions with multiple emissions sensitivity cases, we determined a set of incremental DV responses per ton of emissions reductions. The modeled impacts from the individual cases were then combined in a linear manner to estimate the net impacts from multiple cases. For example, in the Northeast, we would use the following equation to determine the DVs that would result from a 75% reduction in Northeast emissions beyond the explicit emissions control case:

$$DV_{75\%NE,j} = DV_{2025,j} + \left(R_{NE\_explicitcontrol,j} \times \Delta E_{NE\_explicitcontrol}\right) + \left(R_{NE50NOx,j} \times \Delta E_{50NOx}\right) + \left(R_{NE90NOX,j} \times \left(\frac{25}{40}\right) \Delta E_{90NOX}\right) \qquad \text{Equation 3-4}$$

In equation 3-4, $\Delta E_{NE\_explicitcontrol}$ represents the difference in NO_x emissions between the 2025 base case and the 2025 Northeast explicit emissions control case, ΔE_{50NOx} represents the difference in NO_x emissions between the 2025 Northeast explicit emissions control case and the combined Northeast explicit control case with 50% Northeast NOx cuts and ΔE_{90NOX} represents the difference in NO_x emissions between the combined Northeast explicit control case with 50% Northeast NOx cuts and the combined Northeast explicit control case with 90% Northeast NO_x cuts. The $\frac{25}{40}$ multiplier represents the ratio of required to modeled emissions (i.e., the difference between the 50% and 90% NO_x cut emissions sensitivities represent emissions equivalent to 40% of the Northeast explicit emissions control case, while in the example above, we only require an addition 25% emission reduction beyond the 50% NO_x cut simulation).

In two cases, we determined it was appropriate to compute response factors for smaller geographic areas than were modeled in the emissions sensitivity simulations described in section 3.1.4. One of the cases pertains to splitting the responses between different air basins in California and the other case involves the geographic scale of ozone impacts associated with emissions reductions of VOC. Both of these case are described below.

In California, 2025 base case ozone DVs were substantially higher in the South Coast Air Basin located in the southern portion of the state than in the San Joaquin Valley and in areas further north. Additionally, the Transverse Mountain Ranges in Southern California generally isolate the air masses in the South Coast Air Basin from those in the San Joaquin Valley.

Consequently, it is unrealistic to force emissions reductions in locations in Northern California to bring Southern California ozone DVs into attainment with the current or alternative levels of the NAAQS. Therefore, when applying the results of the 50% and 90% California NO_x emissions reduction sensitivities, we made a simplifying assumption that the ozone responses predicted in the San Joaquin Valley and areas of California further north are solely due to emissions changes in those areas. We made a similar assumption about the response of ozone to emissions changes for the southern portion of California. Using this approach we created distinct response factors based on changes in ozone DVs and emissions from each of the two California sub-regions. In general, this assumption seems reasonable based on the topography and wind patterns in these areas and the mountain ranges that separate Los Angeles from the San Joaquin Valley. This approach may lead to either some underestimation or overestimation of the air quality impacts of emissions reductions in the Central and Northern California locations depending on the extent to which emissions reductions in Southern California actually impact ozone in the San Joaquin Valley or vice versa. A more complete description of how these areas were delineated and the rational is provided in Appendix 3A.

We followed a conceptually similar approach for geographically allocating the response of ozone DVs to the 50% reduction in US anthropogenic VOC emissions. Past work has shown that impacts of anthropogenic VOC emissions on ozone DVs in the U.S. tend to be localized (Jin et al., 2008; Nopmongcol et al., 2014) and so consistent with past analyses (US EPA, 2008) we have made the assumption that VOC reductions do not impact ozone at distances more than 100km from the emissions source. Consequently, we created a series of VOC impact regions for urban areas in which ozone is responsive to VOC emissions reductions. These VOC impact regions were only created for the urban areas with the highest projected 2025 base case ozone DVs in each region: New York City, Pittsburgh, and Baltimore in the Northeast; Detroit, Chicago, and Louisville in the Midwest; Houston and Dallas in the Central region; Denver in the Southwest; and Northern and Southern California. VOC impact regions were delineated by creating a 100km buffer around counties containing violating monitors. Since the counties with violating monitors differed at each standard level, a separate set of VOC impact regions was developed for standard levels of 70, 65, and 60 ppb.[18]

In addition, VOC impact regions were constrained by state boundaries except in cases where a current nonattainment area straddled multiple states (for instance New Jersey and Connecticut counties that are included in the New York City nonattainment area were also included in the New York City VOC impact region). The in-state constraint was also waived for the Chicago area since it is well established that emissions from Chicago and Milwaukee are often advected over Lake Michigan where they photochemically react and then impact locations in Wisconsin, Illinois, Indiana, and Michigan that border the lake (Dye et al., 1995). In cases where a county fell within two adjacent overlapping buffer areas (i.e. Easton County which lies in both the greater Chicago and Detroit buffers) the county was assigned to the VOC impact area that is most likely to be upwind based on prevailing wind patterns (i.e. Detroit). Finally, for California, the VOC impact areas were delineated identically to the Northern and Southern NO_x sub-regions described above but were restricted to counties included in the explicit NO_x control cases. For each monitoring site within a VOC impact area, an ozone DV response factor ($R_{i,j}$) was calculated using the VOC emissions reductions that occurred within that area based on the U.S. 50% VOC sensitivity simulation. Figure 3-4 shows the VOC impact areas that were developed for the 60 ppb standard. Maps for the 65 and 70 ppb VOC impact areas look very similar to the map in Figure 3-4, but in some cases they include fewer counties around the outside of the region.

[18] The 70 ppb VOC impact areas were also used to construct the baseline (75 ppb) scenario.

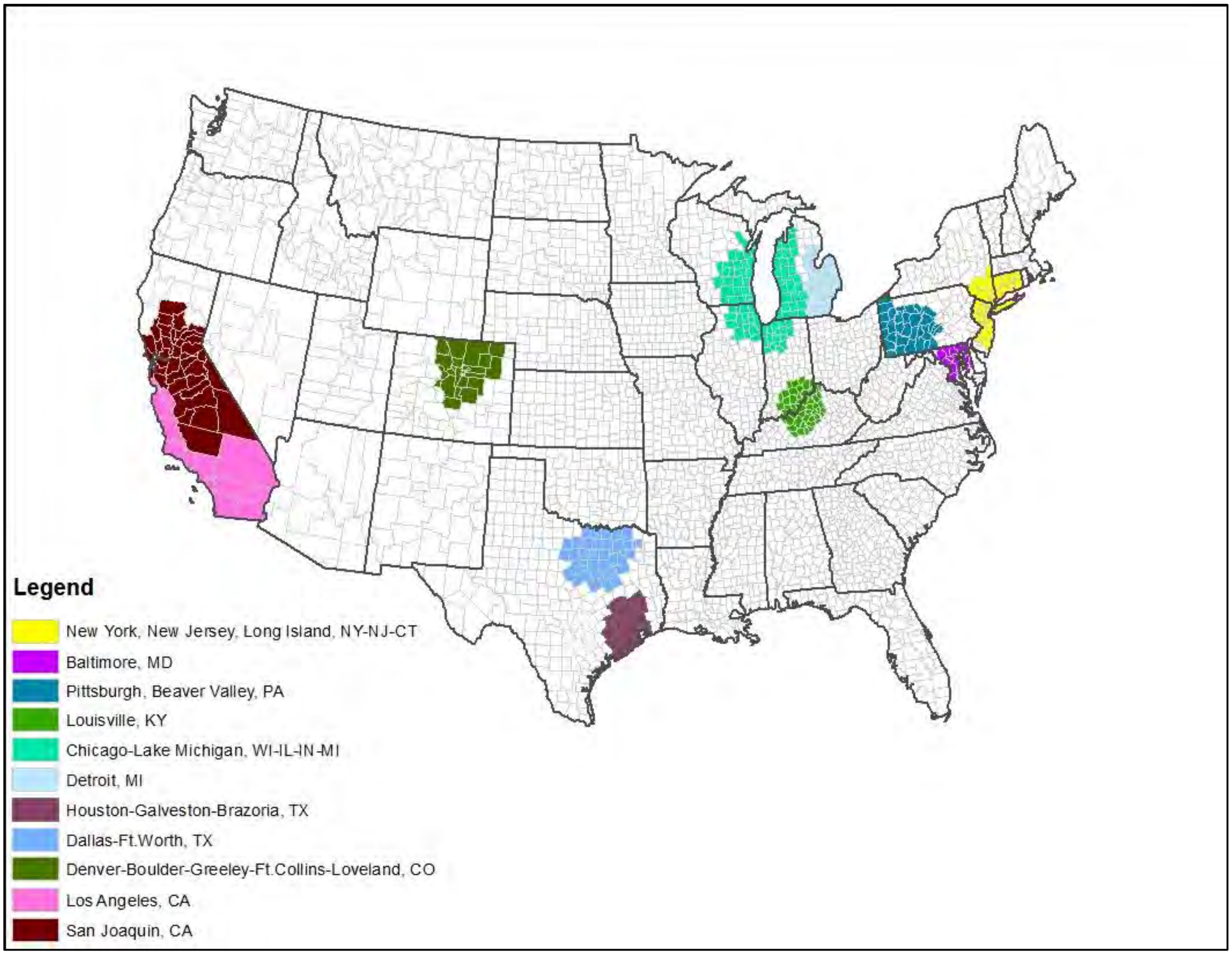

Figure 3-4. Map of VOC Impact Areas Applied in the Evaluation of a 60 ppb Alternative Standard Level

3.3.2 Combining Response from Multiple Sensitivity Runs To Construct Baseline And Alternative Standard Scenarios

Ozone DVs were calculated for the baseline scenario as well as the proposed range of 65-70ppb and a more stringent alternative standard of 60ppb by applying response factors described in section 3.3.1 and the emissions reductions from multiple modeled sensitivity scenarios using Equation 3-5:.

$$DV_j = DV_{2025,j} + (R_{1,j} \times \Delta E_1) + (R_{2,j} \times \Delta E_2) + (R_{3,j} \times \Delta E_3) + \cdots \qquad \text{Equation 3-5}$$

For the baseline as well as the three alternative standards analyzed, we determine the least amount of emissions reductions (tons) needed to bring the ozone DVs at all monitors down to the particular NAAQS level. Note that the emissions reductions were applied on a region-specific basis for most monitors. That is, given the construct of the analytic approach, we did not account for the co-benefits of inter-regional transport except for monitors that are located near the border to two regions. For instance, to determine the requisite tons of NO_x and VOC reductions necessary to bring monitors in the Central region down to 65 ppb, only emissions in the central region were considered (i.e., emissions from the Texas explicit emissions control case, the 50% Central region NO_x reductions sensitivity case, and the VOC emissions from the Houston and Dallas VOC impact areas). This constraint was applied with the assumption that most states would not take into account emissions reductions from upwind states when designing their State Implementation Plans but also acknowledging that there has been a history of some states cooperating with neighboring states in the same region to determine multi-state pollution reduction plans.[19] This approach avoids the complexity of determining the order in which emissions reductions should be considered across the multiple regions modeled for this analysis. The result of this constraint is that the amount of emissions reductions estimated for attainment in this analysis may be larger than necessary because downwind states may benefit from upwind reductions that are not accounted for when determining the regional emissions reduction needed to attain. There were two cases where emissions reductions from two regions were applied to given monitors in a border area: monitors in the Illinois suburbs of St. Louis (Midwest Region) that are clearly affected by emissions in neighboring Missouri (Central Region), and monitors in Pittsburgh, PA and in Buffalo, NY (Northeast Region) that are substantially impacted by emissions in Ohio (Midwest Region).

There are several assumptions inherent in this methodology. First, when applying responses from the across-the-board emissions reduction sensitivities we do not have any information about how ozone DVs respond differently to emissions from different locations within the region. Therefore, for the most part, we assume that every ton of NO_x or VOC

[19] For instance the Ozone Transport Commission (OTC) is an organization made up of states in the eastern U.S. which is responsible for "developing and implementing regional solutions to the ground-level ozone problem in the Northeast and Mid-Atlantic regions" (www.otcair.org)

reduced within the region (or VOC impact area) results in the same ozone response regardless of where the emissions reductions are identified. In locations for which we have both explicit emissions control case reductions and regional across-the-board reductions, it is possible to make some more distinctions as described below, but we are still not able to fully account for variable response to emissions from different locations within the region. Where possible, we try to locate emissions reductions closer to the highest DV monitors with the understanding that emissions reductions are likely to have lower impact when they occur further from the monitor location. A second assumption is that NO_x and VOC responses are additive. In the case of monitors impacted by emissions from multiple regions, we also assume that the responses from multiple regions are additive. Again, we do not have any more refined information that would allow us to account for nonlinear interactions of these emissions. Third, we assume that ozone response within each of these sensitivity simulations is linear (i.e., the first ton of NO_x reduced results in the same ozone response as the last ton of NO_x reduced). In cases for which we have multiple levels of emissions reductions (i.e., California, the Central region, and the Northeast) we assume linearity within each simulation but are able to capture discrete shifts in ozone response for each sensitivity simulation (i.e., one response for explicit emissions control case reductions, another response level up to 50% NO_x reductions beyond the explicit emissions control case emissions, and a third level of response between 50% and 90% NO_x reductions beyond the explicit emissions control case). For the Central states there are only two discrete response levels, while in California and the Northeast there are three. Finally, for regions without 90% NOx cut emissions sensitivity scenarios (Southwest, Central, and Midwest), response to NO_x reductions greater than 50% must be extrapolated beyond the modeled emissions reductions.

3.3.3 Creation of the Baseline Scenario

Computing the response of DVs to emissions reductions from each emissions sensitivity simulation allowed us to determine what emissions reductions would be needed in each region to create the baseline scenario (i.e. to reach 75 ppb at every monitor location). We determine how those emissions reductions could be achieved by applying controls in the following order: (1) emissions changes from the 111(d) sensitivity, (2) known controls of NO_x emissions from nonpoint, non-EGU point, and nonroad sources greater than 50 tons per year (explicit control cases), (3) mobile source emissions changes between 2025 and 2030 (California only), (4) known controls of VOC emissions, and (5) additional NO_x controls. All reductions identified from these sources were above and beyond reductions from on-the-books regulations that were included in the 2025 base case modeling. The emissions changes from the 111(d) sensitivity were applied throughout the entire U.S. in creating the baseline scenario. Other emissions changes were only applied in the areas projected to have DVs greater than 75 ppb in the 2025 base case scenario: California and Texas. In California, all five types of emissions reductions were needed to meet the current 75 ppb standard while in Texas only the 111(d) emissions changes and a portion of the explicit modeled controls were needed. The 2025 to 2030 mobile source changes were applied in California because, as discussed at the beginning of this chapter, many locations in California will likely have attainment dates substantially further out than 2025. Although emissions projections for those years were not generally available, the state of California did provide emissions projections for mobile sources in the year 2030. Emissions of both VOC and NO_x were available for onroad, nonroad, locomotive, and C1/C2 commercial marine vessel sectors by county. There were both increases and decreases between 2025 and 2030 depending on the county and sector, but overall these mobile source changes resulted in VOC emissions that were 1% less than those modeled in the California explicit emissions control case and NO_x emissions that were 4% less than those modeled in the California explicit emissions control case in the Northern California sub-region and 3% less than those modeled in the Southern California sub-region. The NO_x and VOC mobile source emissions changes were applied to create the baseline scenario in California using the response ratios developed from the 50% California NOx cut and the 50% U.S. VOC cut sensitivity simulations. Summaries of the emissions reductions are presented by region in Appendix 3A. In addition, resulting ozone DVs at all evaluated monitors are provided in Appendix 3A.

3.3.4 Creation of the 70, 65, and 60 ppb Alternative Standard Level Scenarios

To create the scenarios for the three alternative standard levels (i.e. 70, 65, and 60 ppb), we started with the baseline and then identified additional controls for each region from the five categories listed in section 3.3.3. Not all types of emissions reductions were required in each region for each scenario. For regions that contained a NO_x explicit emissions control case buffer, only the known controls within the buffer were applied before the known VOC controls. In those regions, after explicit emissions control case reductions and the VOC known controls were applied, then additional NO_x controls were considered. In those regions, an additional constraint was also applied that forced the tons of additional NO_x reductions applied within and outside the explicit emissions control case area to be applied proportionally to the starting NO_x emissions within and outside the explicit emissions control case buffer areas (e.g., if 40% of the starting emissions in the explicit control scenario simulation were located within the buffer, then 40% of the emissions reductions also had to come from within the buffer). This constraint was applied because the monitors with the highest DVs were located within the buffers and the response factors were based on an average ppb/ton across the region. Thus, the constraint ensured some measure of spatial equivalence between the location of the modeled emissions reductions and those applied to construct the scenario. In some cases, this constraint also resulted in including unknown controls to create the scenario even when known controls were still available within the region but outside of the explicit control case buffer. In regions without an explicit emissions control case buffer area, all known controls of NO_x emissions from nonpoint, non-EGU point, and nonroad sources greater than 50 tons per year were applied throughout the entire region before any VOC emissions reductions were applied. A numeric example of the calculation methodology is provided in Appendix 3A. Summaries of the emissions reductions are presented by region in Appendix 3A and by source category in Chapter 4. In addition, ozone DVs at all evaluated monitors are provided for each scenario in Appendix 3A.

3.3.5 Monitoring Sites Excluded from Quantitative Analysis

There were 1219 ozone monitors with complete ozone data for at least one DV period covering the years 2009-2013. Of those sites, we quantitatively analyzed 1150 (94%) in this analysis. In determining the necessary tons of emissions reductions for each of the four scenarios, there were three types of sites that were not treated quantitatively, i.e. emissions reductions necessary to reach the alternative standard levels at these sites were not quantified. First, tons of emissions reductions were not determined for 36 sites that did not have a valid projected 2025 base case DV due to less than 5 modeled days above 60 ppb in the 2011 CAMx simulation as required to project a DV in the EPA SIP modeling guidance (US EPA, 2014e). It is unlikely that these sites would have any substantial impact on resulting costs and benefits, since the reason that projections could not be made is that they have no more than 4 modeled days above 60 ppb, in which case they would likely already be meeting all standard levels evaluated in this analysis using the current year data. These sites are listed in appendix 3A.

Second, 7 sites for which the DVs were influenced by wintertime ozone episodes were not included because the modeling tools are not currently sufficient to properly characterize ozone formation during wintertime ozone episodes. It is not appropriate to apply the model-based response (RRF) developed based on summertime conditions to a wintertime ozone event, which is driven by different types of chemistry and meteorology. Since there was no technically feasible method for projecting DVs at these sites, these sites were not included in determining required reductions in NO_x and VOCs to meet current or alternative standard levels. Wintertime ozone events tend to be very localized phenomena driven by local emissions from oil and gas operations (Schnell et al, 2009; Rappengluck et al., 2014; Helmig et al., 2014). Consequently, the emissions reductions needed to lower wintertime ozone levels would likely be different from those targeted for summertime ozone events. It follows that there could be additional emissions reductions required to lower ozone at these locations and thus potential additional costs and benefits that are not quantified in this analysis. Appendix 3A includes a list of sites influenced by wintertime ozone and the methodology used to identify those sites.

Finally, while the majority of the sites had projected ozone exceedances primarily caused by local and regional emissions, there were a set of 26 relatively remote, rural sites in the Western U.S. with projected 2025 base case DVs between 62 and 69 ppb[20] that showed limited response to the regional NO_x emission and national VOC emission sensitivities in our modeling. Air agencies responsible for these locations may choose to pursue one or more of the Clean Air Act provisions that offer varying degrees of regulatory relief. Regulatory relief may include:

- Relief from designation as a nonattainment area (through exclusion of data affected by exceptional events)
- Relief from the more stringent requirements of higher nonattainment area classifications (through treatment as a rural transport area; through exclusion of data affected by exceptional events; or through international transport provisions)

[20] Except in California where sites had projected 2025 base case DVs up to 75 ppb. The California sites all had estimated ozone DVs below 70 ppb in the post-2025 baseline scenario.

- Relief from adopting more than reasonable controls to demonstrate attainment (through international transport provisions)

In addition, some of these sites could potentially benefit from the CAA's interstate transport provisions found in sections 110(a)(2)(D) and 126. Appendix 3A provides additional detail on the treatment of these sites.

3.4 Creating Spatial Surfaces

The emissions reductions for attainment of the alternative NAAQS levels were used to create spatial fields of ozone concentrations (i.e., spatial surfaces) for input to the calculation of the benefits associated with attainment of each NAAQS level, incremental to the baseline. The spatial surfaces used to calculate health-related benefits with the BenMap tool (Chapter 5) are described below in section 3.4.1. Spatial surfaces used to calculate welfare-related benefits with the FASOMGHG model (Chapter 6) are described below in section 3.4.2.

3.4.1 BenMap Surfaces

Two ozone metrics are used to evaluate health benefits associated with meeting different ozone standard levels. These metrics and the studies that they are derived from are described in more detail in Chapter 5**.** Briefly, ozone surfaces for the baseline and each alternative NAAQS level were created for the following metrics: May-Sep seasonal mean of 8-hr daily maximum ozone and Apr-Sep seasonal mean of 1-hr daily maximum ozone. For each metric, surfaces were created for a total of 11 scenarios. These scenarios include:

- 2025 baseline
- post-2025 baseline
- 2025 70 ppb partial attainment
- 2025 70 ppb full attainment
- post-2025 70 ppb full attainment
- 2025 65 ppb partial attainment
- 2025 65 ppb full attainment
- post-2025 65 ppb full attainment
- 2025 60 ppb partial attainment
- 2025 60 ppb full attainment

- post-2025 60 ppb full attainment

The surfaces created for the 2025 scenarios represent all continental U.S. monitors outside of California attaining the standard being evaluated while the surfaces for the post-2025 scenarios represent all continental U.S. monitors including California meeting the standard being evaluated. The effects due only to California meeting the standard are isolated in Chapter 5 through a series of BenMap simulations using these surfaces and varying assumptions about population demographics. In addition, for the 2025 scenarios we include "partial" and "full" attainment in which the partial attainment scenarios only include emissions reductions identified from known control measures while the full attainment scenarios include emissions reductions necessary to attain the standard from both known and unknown controls.

The ozone surfaces were created using the following steps. Each step is described in more detail below:

- Step 1: Aggregate gridded hourly modeled concentrations into relevant seasonal ozone metrics
    - Inputs: Hourly gridded model concentrations for 2011, 2025 base case, and 12 2025 emissions sensitivity simulations detailed in Section 3.1.4
    - Outputs: Seasonal ozone metrics for 2011, 2025 base case, and 12 2025 emissions sensitivity simulations
- Step 2: Calculate response factors for each seasonal ozone metric from each emissions sensitivity simulation
    - Inputs: Seasonal ozone metrics for 2011, 2025 base case, and 12 2025 emissions sensitivity simulations; Amount of emissions reductions (tons) modeled in each emissions case
    - Outputs: Gridded ppb/ton response factor for each seasonal ozone metric from each emissions sensitivity simulation
- Step 3: Create gridded field for each attainment scenario and each seasonal ozone metric
    - Inputs: Gridded ppb/ton response factor for each seasonal ozone metric from each emissions sensitivity simulation; Amount of emissions reductions from each region described in Appendix 3A.
    - Outputs: Gridded seasonal ozone metrics for each attainment scenario
- Step 4: Create 2011 enhanced Voronoi Neighbor Averaging (eVNA) fused surface of modeled and observed values for each seasonal ozone metric

- Inputs: 2010-2012 observed ozone values (seasonal ozone metrics at each monitor location); 2011 modeled ozone (seasonal ozone metrics at each grid cell)
  - Outputs: 2011 fused modeled/monitored surfaces for each seasonal ozone metric
- Step 5: Create eVNA fused modeled/monitored surface for each attainment scenario and each seasonal ozone metric
  - Inputs: 2011 fused model/obs surfaces for each seasonal ozone metric; modeled seasonal ozone metrics (gridded fields) for 2011 and each attainment scenario
  - Outputs: Fused modeled/monitored surface for each attainment scenario and each seasonal ozone metric

Step 1:

Gridded hourly ozone modeled concentrations were aggregated to the relevant metric for the 2011, 2025 base case, and each of the 12 emissions sensitivity simulations. This step resulted in 15 ozone fields for each of the two metrics.

Step 2:

A gridded ppb/ton response factor was determined for each metric and for each emissions sensitivity simulation.

Step 3:

Based on the emissions reductions provided in appendix 3A, the response factors were multiplied by the relevant tons of emissions reductions for each sensitivity and then summed to create a gridded field representing the scenario in question (Equation 3-6)[21]

$$O3_{xy,s,m} = O3_{xy,2025,m} + (R_{xy,1,m} \times \Delta E_{1,s}) + (R_{xy,2,m} \times \Delta E_{2,s}) + (R_{xy,2,m} \times \Delta E_{3,s}) + \cdots$$

Equation 3-6

[21] An extra 3,500 tons of VOC reductions available in Northern California outside of the N California sub-region was mistakenly applied in creating all surfaces. This lead to absolute changes in gridded ozone concentrations of less than 0.01 ppb. Since this error was carried through all surfaces, the incremental changes in ozone between surfaces were not impacted.

In equation 3-6, $ozone_{xy,s,m}$ represents the ozone concentrations at grid cell *x,y*, for scenario *s*, and using metric, *m*. Similarly $ozone_{xy,2025,m}$ represents the modeled ozone from the 2025 base simulation at grid cell *x,y* aggregated to metric *m*. $R_{xy,1,m}$ represents the response factor (ppb/ton) in grid cell *x,y* using metric *m,* for the sensitivity simulation #1. Finally $\Delta E_{1,s}$ represents the amount of emissions reductions from sources modeled in sensitivity #1 determined necessary for scenario *s*. Partial attainment surfaces at each standard level were created by first starting with the full attainment surface and then subtracting off impacts from emissions reductions that were identified from unknown controls. For the 70 ppb scenario in which all unknown controls were located in the explicit emissions control case buffer areas, the ppb/ton response ratios from the explicit emissions control case sensitivity simulations were applied to back out impacts from unknown controls. For the 65 and 60 ppb scenarios, unknown controls were located both within and outside explicit emissions control case buffer areas. We did not have any response ratios that represent emissions from outside the explicit emissions control case buffer areas alone. Therefore, for the 65 and 60 ppb scenarios, we applied the response ratios from the regional 50% NO_x reduction sensitivity simulations. This leads to additional uncertainty in the 65 and 60 pp partial attainment surfaces since the relative proportions of unknown emissions reductions within and outside the buffer areas were different from the relative proportions of emissions reductions within and outside of the buffer areas that were applied in the 50% regional NO_x reduction simulations.

Step 4:

The MATS tool was used to create a fused gridded 2011 field using both ambient and modeled data using the eVNA technique (Abt, 2014). This method essentially takes an interpolated field of observed data and adjusts it up or down based on the modeled spatial gradients. For this purpose, the 2010-2012 ambient data was interpolated and fused with the 2011 model data. One “fused” eVNA surface was created for each of the two seasonal ozone metrics.

Step 5:

The 22 model-based surfaces (i.e., 11 scenarios and 2 metrics) were used as inputs in the MATS tool along with the gridded 2011 eVNA surfaces. For each metric and each scenario a gridded RRF field was created by dividing the gridded ozone field for scenario *s* by the gridded 2011 model field. This RRF field was then multiplied by the 2011 eVNA field to create a gridded eVNA field for each scenario.

Results of this process for the May-September 8-hr daily maximum ozone metric are shown in Figures 3-5, 3-6, 3-7, 3-8. These figures show the post-2025 baseline and the changes in ozone between the post-2025 baseline and each of the post-2025 scenarios for lower standard levels: 70, 65, and 60 ppb. The post-2025 baseline represents the case where all continental US monitors meet the current 75 ppb standard and similarly the post-2025 alternative standard scenarios represent the case where all continental US monitors meet the standard level being evaluated.

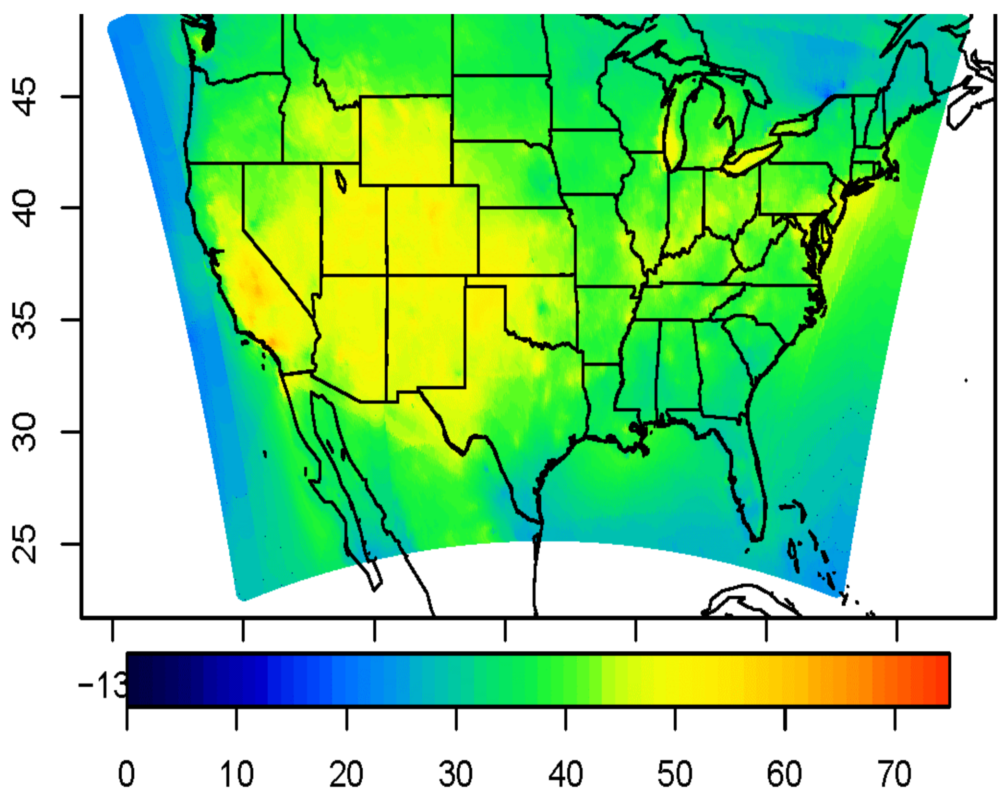

Figure 3-5. Projected post-2025 Baseline Scenario May-September Mean of 8-hr Daily Maximum Ozone (ppb)

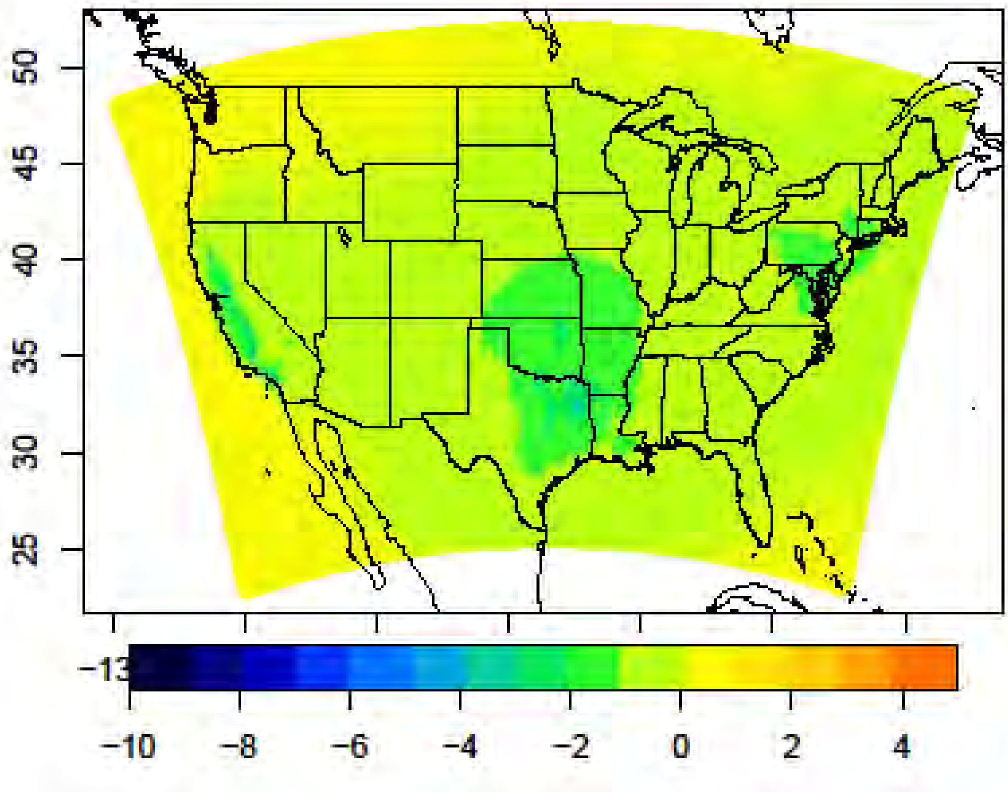

Figure 3-6. Change in May-September Mean of 8-hr daily Maximum Ozone (ppb) between the post-2025 Baseline Scenario and the post-2025 70 ppb Scenario

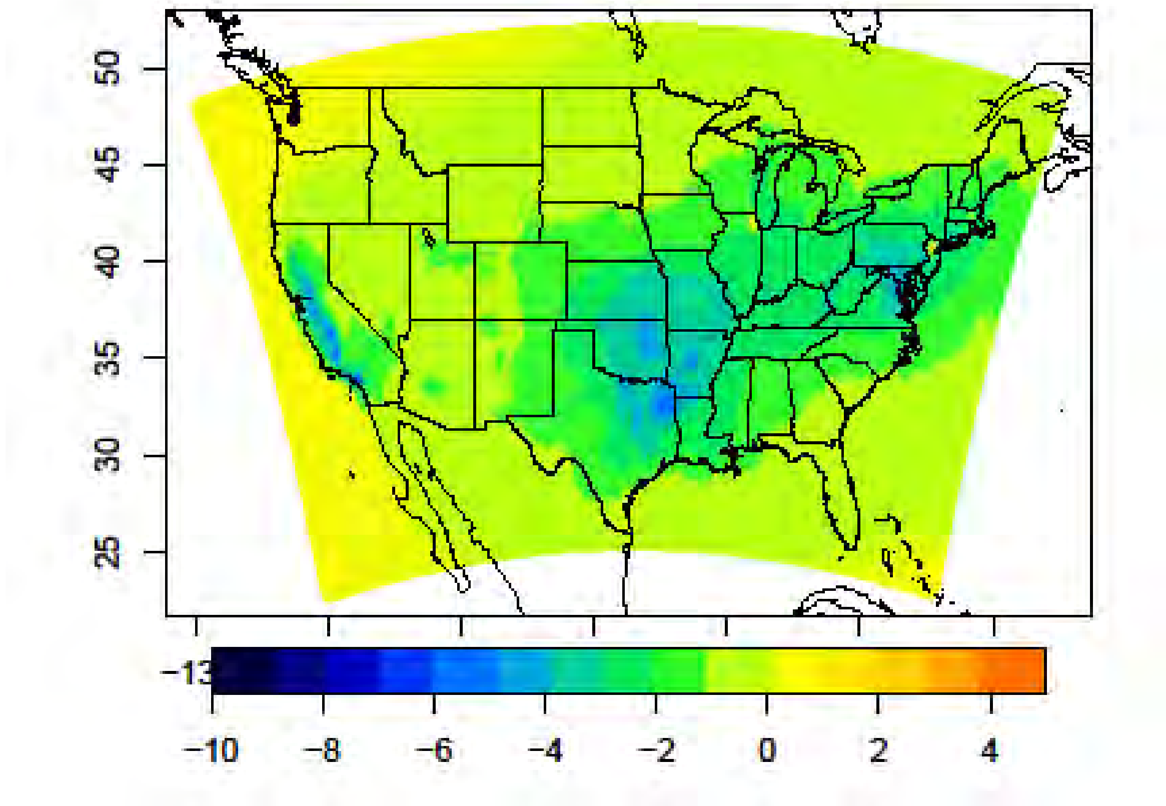

Figure 3-7. Change in May-September Mean of 8-hr daily Maximum Ozone (ppb) between the post-2025 Baseline Scenario and the post-2025 65 ppb Scenario

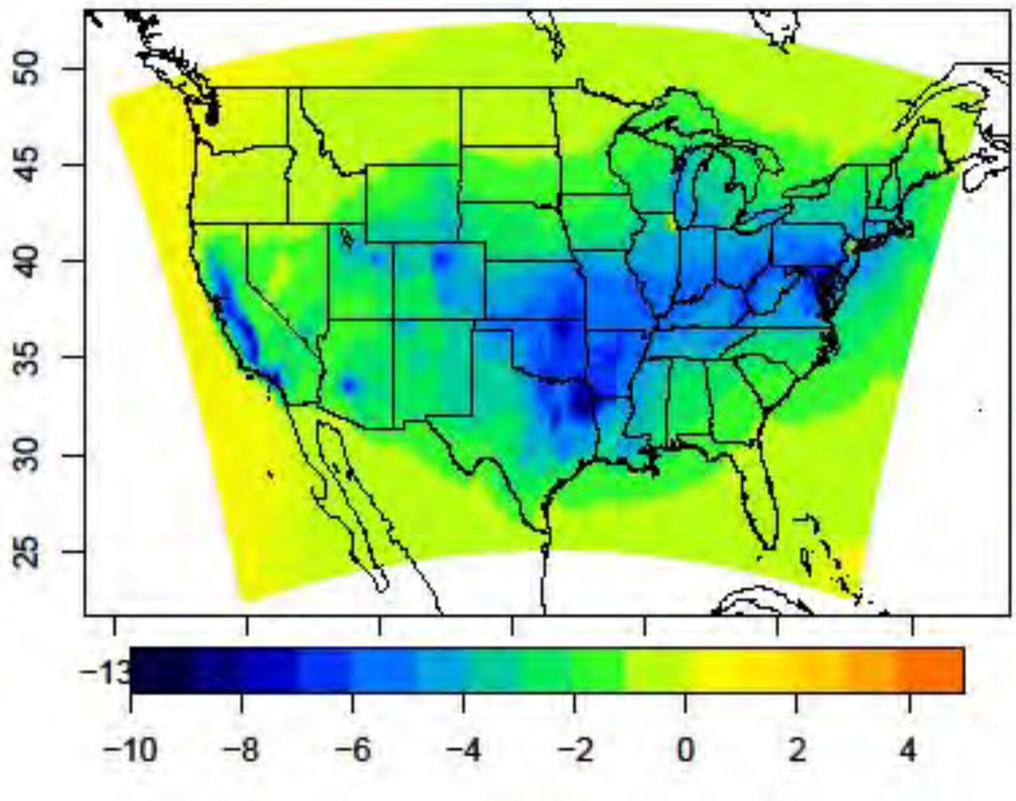

Figure 3-8. Change in May-September Mean of 8-hr Daily Maximum Ozone (ppb) between the post-2025 Baseline Scenario and the post-2025 60 ppb Scenario

3.4.2 W126 surfaces

This section describes the creation of ozone surfaces aggregated using the W126 metric. The general methodology for calculating the W126 metric is provided in appendix 3A. Ozone surfaces aggregated to the W126 metric were created for eight scenarios:

- 2025 baseline
- post-2025 baseline
- 2025 70 ppb full attainment
- post-2025 70 ppb

- 2025 65 ppb full attainment
- post-2025 65 ppb
- 2025 60 ppb full attainment
- post-2025 60 ppb

Several steps were followed to create these surfaces. First, as was done with the projected ozone DVs and the ozone surfaces for health benefits, ppb/ton response factors was determined for each sensitivity simulation. In this case, the response factors were created based on hourly ozone data in the 2025 base case and 12 emissions sensitivity simulations. Therefore, six months of gridded hourly response factors were created for each emission sensitivity simulation. Then, based on the emissions reductions described in Appendix 3A, the hourly response factors were multiplied by the relevant tons of emissions reductions from each emissions sensitivity and then summed to create a gridded field representing the scenario in question (Equation 3-6). For the W126 calculations, the metric in equation 3-6 is hourly ozone. At the end of this step, there were eight sets of hourly gridded ozone fields, one for each scenario. These gridded hourly ozone fields, along with the 2011 modeled hourly ozone field, were then aggregated into the W126 metric, which is described in more detail in Chapter 2. The MATS tool was used to project W126 values at each monitor location. The set-up was similar to the approach for projecting DVs. In essence the 2011 and eight W126 scenarios gridded fields were used to create an RRF for each monitor location using the 3x3 matrix of grid cells surrounding the monitor location as described in section 3.2.2. These model-based RRFs were multiplied by the three year average (2010-2012) of the measured W126 at each monitor. At the end of this step, there were a set of W126 values at all monitor locations for each scenario. Finally, a gridded field of W126 values was created for each scenario by spatially interpreting the projected monitor values using an inverse distance weighted Voronoi Neighbor Averaging (VNA) technique (Gold, 1997; Chen et al, 2004). This is similar to how W126 gridded fields have been created for previous EPA analyses (EPA, 2014f). Figure 3-9 shows the W126 gridded field for the post-2025 baseline scenarios. Figures 3-10, 3-11, and 3-12 show the W126 gridded field of the post-2025 scenarios for 70, 65, and 60.

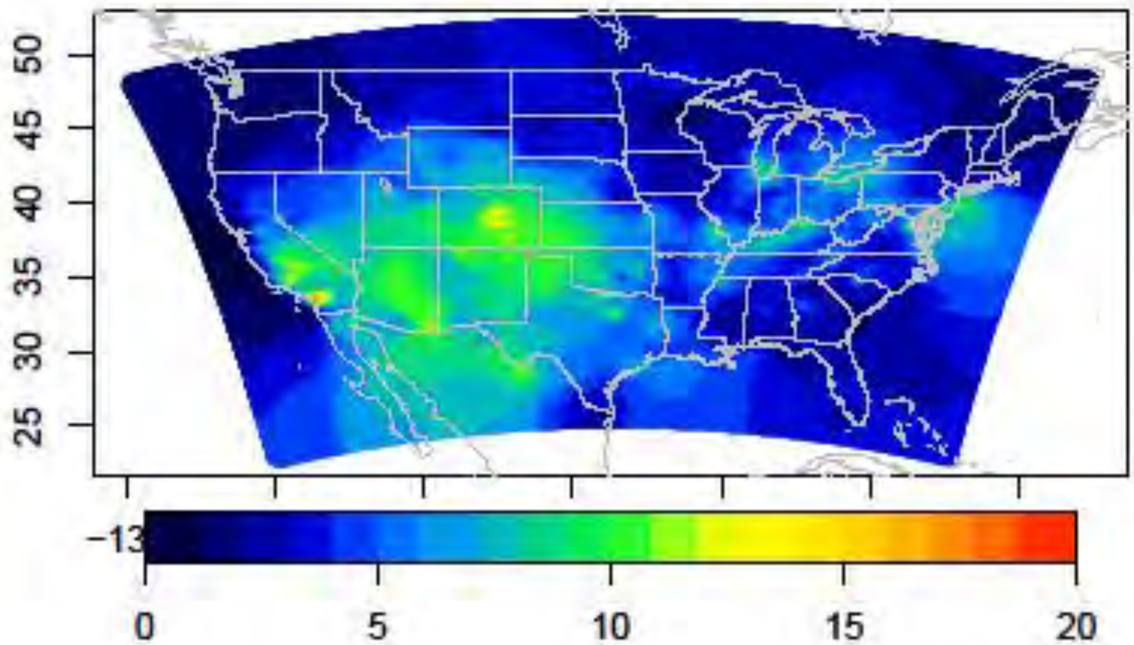

Figure 3-9. Projected post-2025 Baseline Scenario W126 Values (ppm-hrs)

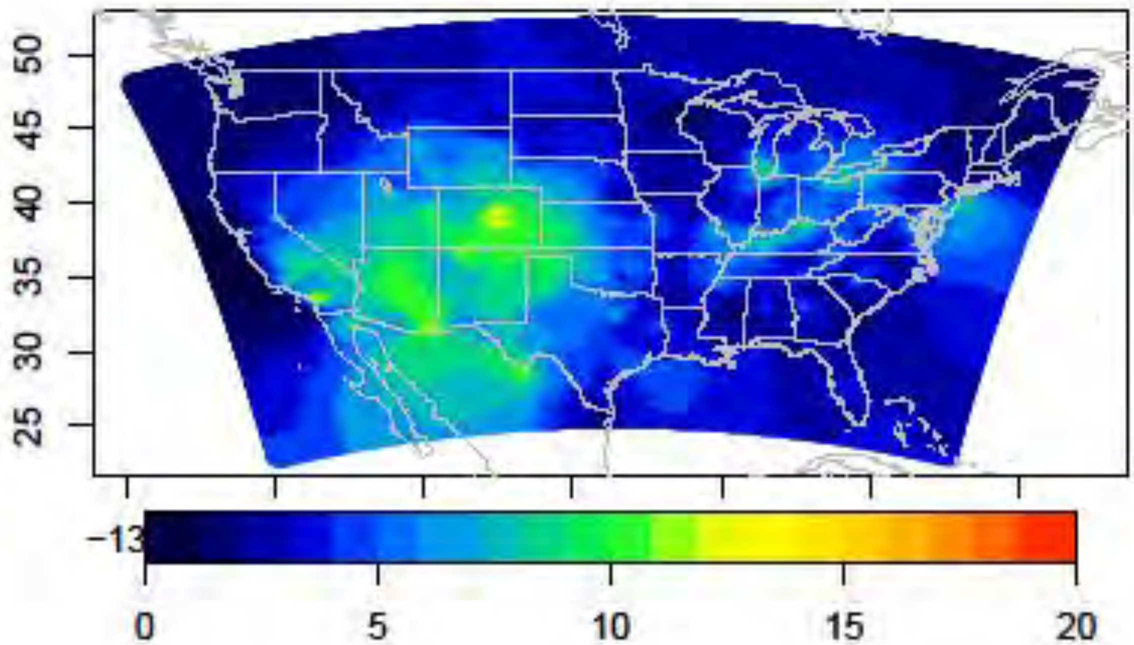

Figure 3-10. Projected post-2025 70 ppb Scenario W126 Values (ppm-hrs)

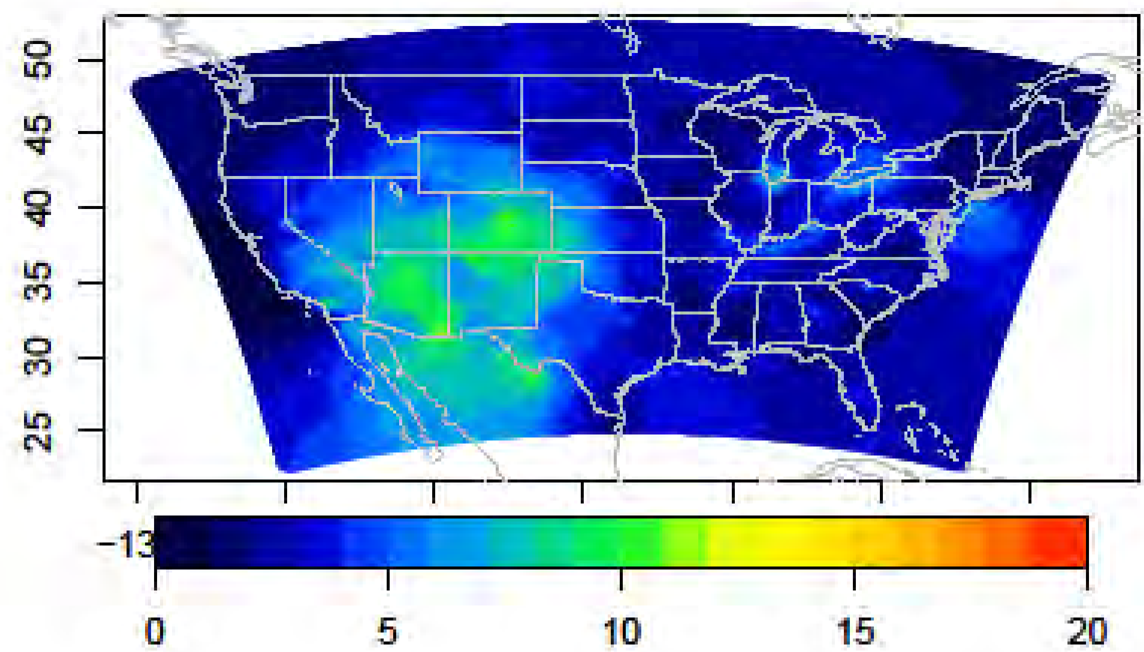

Figure 3-11. Projected post-2025 65 ppb Scenario W126 Values (ppm-hrs)

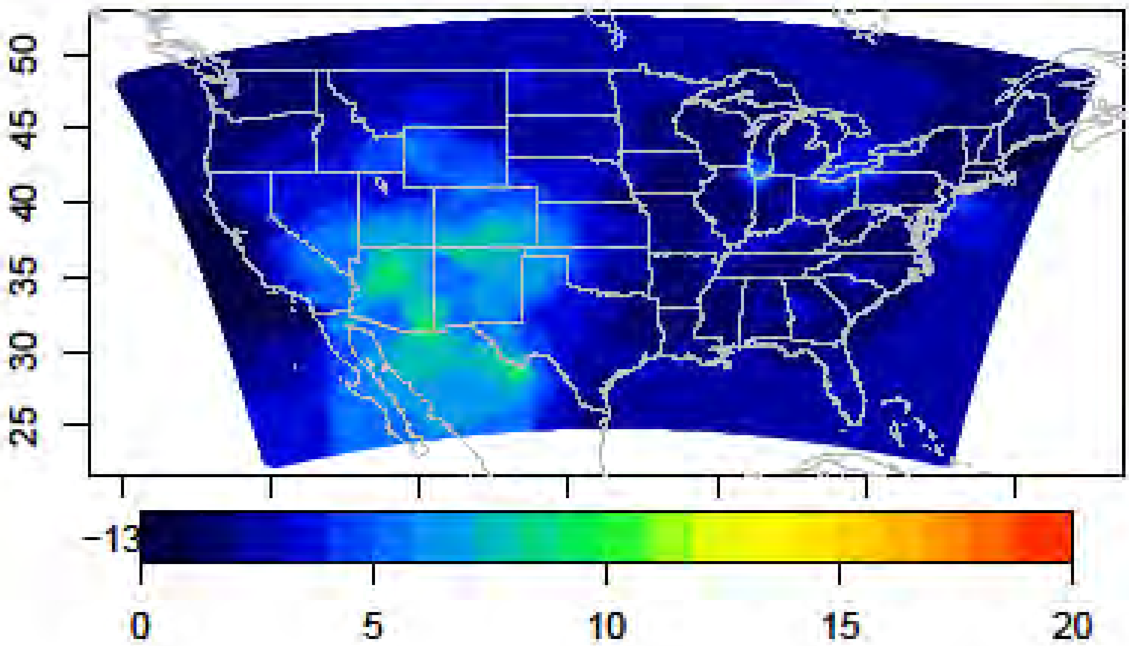

Figure 3-12. Projected post-2025 60 ppb Scenario W126 Values (ppm-hrs)

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APPENDIX 3: ADDITIONAL AIR QUALITY ANALYSIS AND RESULTS

3A.1 2011 Model Evaluation for Ozone

An operational model evaluation was conducted for the 2011 base year CAMx annual model simulation performed for the 12-km U.S. modeling domain.[22] The purpose of this evaluation was to examine the ability of the Ozone NAAQS RIA air quality modeling platform to replicate the magnitude and spatial and temporal variability of measured (i.e., observed) ozone concentrations within the modeling domain. The model evaluation for ozone was based upon comparisons of model predicted 8-hour daily maximum concentrations to the corresponding observed data at monitoring sites in the EPA Air Quality System (AQS) and the Clean Air Status and Trends Network (CASTNet). Included in the evaluation are statistical measures of model performance based upon model-predicted versus observed concentrations that were paired in space and time on an hourly basis.

Model performance statistics were calculated for several spatial scales and temporal periods. Statistics were calculated for individual monitoring sites and for each of five regions of the 12-km U.S. modeling domain. The regions include the Northeast, Midwest, Southeast, and Central and Western states which are defined based upon the states contained within the Regional Planning Organizations (RPOs)[23]. For maximum daily average 8-hour (MDA8) ozone, the statistics for each site and region were calculated for the May through September ozone season.[24] In addition to the performance statistics, we prepared several graphical presentations of

[22] See Chapter 3, section 3.1.2 of the RIA document for a description of the 12-km U.S. modeling domain.

[23] The subregions are defined by States where: Midwest is IL, IN, MI, OH, and WI; Northeast is CT, DE, MA, MD, ME, NH, NJ, NY, PA, RI, and VT; Southeast is AL, FL, GA, KY, MS, NC, SC, TN, VA, and WV; Central is AR, IA, KS, LA, MN, MO, NE, OK, and TX; West is AK, CA, OR, WA, AZ, NM, CO, UT, WY, SD, ND, MT, ID, and NV.

[24] In calculating the ozone season statistics we limited the data to those observed and predicted pairs with observations that exceeded 60 ppb in order to focus on concentrations at the upper portion of the distribution of values.

model performance for MDA8 ozone which is the key pollutant for the Ozone NAAQS Rule. These graphical presentations include:

(1) regional maps which show the mean bias and error as well as normalized mean bias and error calculated for MDA8 ≥ 60 ppb for May through September at individual monitoring sites,

(2) bar and whisker plots which show the distribution of the predicted and observed data by month (May through September) and by region, and

(3) time series plots (May through September) of observed and predicted concentrations for 13 representative high ozone sites in the urban areas with the highest projected ozone levels in each region from the 2025 base case CAMx simulation.

The Atmospheric Model Evaluation Tool (AMET) was used to calculate the model performance statistics used in this document (Gilliam et al., 2005). For this analysis and summary of the 2011 model evaluation for ozone, we have selected the mean bias, mean error, normalized mean bias, and normalized mean error to characterize model performance which are consistent with the recommendations in Simon et al. (2012) and the draft SIP modeling guidance (US EPA 2014). As noted above, we calculated the performance statistics by the May through September ozone season.

Mean bias (MB) is used as average of the difference (predicted – observed) divided by the total number of replicates (*n*). Mean bias is given in units of ppb and is defined as:

MB = $\frac{1}{n}\sum_1^n(P-O)$, where P = predicted and O = observed concentrations.

Mean error (ME) calculates the absolute value of the difference (predicted - observed) divided by the total number of replicates (*n*). Mean error is given in units of ppb and is defined as:

ME = $\frac{1}{n}\sum_1^n|P-O|$

Normalized mean bias (NMB) is used as a normalization to facilitate a range of concentration magnitudes. This statistic averages the difference (predicted - observed) over the sum of observed values. NMB is a useful model performance indicator because it avoids over inflating the observed range of values, especially at low concentrations. Normalized mean bias is given in units of % and is defined as:

$$\text{NMB} = \frac{\sum_1^n (P-O)}{\sum_1^n (O)} * 100$$

Normalized mean error (NME) is also similar to NMB, where the performance statistic is used as a normalization of the mean error. NME calculates the absolute value of the difference (predicted - observed) over the sum of observed values. Normalized mean error is given in units of % and is defined as:

$$\text{NME} = \frac{\sum_1^n |P-O|}{\sum_1^n (O)} * 100$$

In general, the model performance statistics indicate that the 8-hour daily maximum ozone concentrations predicted by the 2011 CAMx modeling platform closely reflect the corresponding 8-hour observed ozone concentrations in space and time in each region of the 12-km U.S. modeling domain. The acceptability of model performance was judged by considering the 2011 CAMx performance results in light of the range of performance found in recent regional ozone model applications (NRC, 2002; Phillips et al., 2007; Simon et al., 2011; US EPA, 2005; US EPA, 2009; US EPA, 2011). These other modeling studies represent a wide range of modeling analyses which cover various models, model configurations, domains, years and/or episodes, chemical mechanisms, and aerosol modules. Overall, the ozone model performance results for the 2011 CAMx simulations performed for the Ozone NAAQS are within the range found in other recent applications. The model performance results, as described in this document, demonstrate that the predictions from the Ozone NAAQS modeling platform closely replicate the corresponding observed concentrations in terms of the magnitude, temporal fluctuations, and spatial differences for 8-hour daily maximum ozone.

Consistent with EPA's guidance for attainment demonstration modeling, we have applied the model predictions performed as part of the Ozone NAAQS in a relative manner for projecting future concentrations of ozone. The National Research Council (NRC, 2002) states that using air quality modeling in a relative manner "may help reduce the bias introduced by modeling errors and, therefore, may be more accurate than using model results directly (absolute values) to estimate future pollutant levels". Thus, the results of this evaluation together with the manner in which we are applying model predictions gives us confidence that our air quality model applications using the CAMx 2011 modeling platform provides a scientifically credible approach for assessing ozone for the Ozone NAAQS Rule.

The 8-hour ozone model performance bias and error statistics by network for the ozone season (May-September average) for each region are provided in Table 3A-1. The statistics shown were calculated using data pairs on days with observed 8-hour ozone of ≥ 60 ppb. The distributions of observed and predicted 8-hour ozone by month in the 5-month ozone season for each region are shown in Figures 3A-1 through 3A-5. Spatial plots of the mean bias and error as well as the normalized mean bias and error for individual monitors are shown in Figures 3A-6 and 3A-9. The statistics shown in these two figures were calculated over the ozone season using data pairs on days with observed 8-hour ozone of ≥ 60 ppb. Time series plots of observed and predicted 8-hour ozone during the ozone season at the 13 representative high ozone monitoring sites are provided in Figure 3A-10a-m. These sites are listed in Table 3A-2.

As indicated by the statistics in Table 3A-1, bias and error for 8-hour daily maximum ozone are relatively low in each region. Generally, MB for 8-hour ozone ≥ 60 ppb during the ozone season is less than 5 ppb except in the Western region and at rural (CASTNET) sites in the central region for which ozone is somewhat under-predicted. The monthly distribution of 8-hour daily maximum ozone during the ozone season generally corresponds well with that of the observed concentrations, as indicated by the graphics in Figures 3A-1 through 3A-5. The predicted concentrations tend to be close to the observed 25th percentile, median and 75th percentile values for each region, although there is a small persistent overestimation bias for these metrics. The CAMx model also has a tendency to under-predict the highest observational concentrations at both the AQS and CASTNet network sites.

Figures 3A-6 through 3A-9 show the spatial variability in bias and error at monitor locations. Mean bias, as seen from Figure 3A-6, is less than 6 ppb at most of the sites across the modeling domain. Figure 3A-7 indicates that the normalized mean bias for days with observed 8-hour daily maximum ozone greater than or equal to 60 ppb is within ± 10 percent at the vast majority of monitoring sites across the modeling domain. There are regional differences in model performance, where the model tends to over-predict from the Southeast into the Mid-Atlantic States and generally under predict in the Central and Western U.S. Model performance in the Midwest states shows both under and over predictions.

Model error, as seen from Figure 3A-8, is 10 ppb or less at most of the sites across the modeling domain. Figure 3A-9 indicates that the normalized mean error for days with observed 8-hour daily maximum ozone greater than or equal to 60 ppb is within 10 percent at the vast majority of monitoring sites across the modeling domain. Somewhat greater error is evident at sites in several areas most notably along portions of the Northeast Corridor and in portions of Florida, North Dakota, Illinois, Ohio, North Carolina, and the western most part of the modeling domain.

In addition to the above analysis of overall model performance, we also examine how well the modeling platform replicates day to day fluctuations in observed 8-hour daily maximum concentrations at 13 high ozone monitoring sites. For this site specific analysis we present the time series of observed and predicted 8-hour daily maximum concentrations by site over the ozone season, May through September. These monitors were chosen as representative high ozone sites in urban areas with the highest projected ozone levels in the 2025 base case simulation. The results, as shown in Figures A-10a through m, indicate that the modeling platform replicates the day-to-day variability in ozone during this time period. For example, several of the sites not only have minimal bias but also accurately capture both the seasonal and day-to-day variability in the observations: Alleghany County, PA; Frederick County, MD; Wayne County, MI; Jefferson County, KY. Many additional sites generally track well and capture day-to-day variability but underestimate some of the peak ozone days: Tarrant County, TX; Brazoria County, TX; Harford County, MD; Queens County, NY; Suffolk County, NY; Sheboygan County, WI; Douglas County, CO. Finally, the daily modeled ozone at the two California sites evaluated correlates well with observations but has a persistent low bias. Looking across all 13 sites indicates that the modeling platform is able to capture the site to site differences in the short-term variability of ozone concentrations.

Table 3A-1. Daily Maximum 8-hour Ozone Performance Statistics ≥ 60 ppb by Region, by Network

Network	Subregion	No. of Obs	MB	ME	NMB (%)	NME (%)
AQS	Northeast	3,746	0.6	7.3	0.9	10.7
	Mid-West	4,240	-0.7	7.8	-1.0	11.5
	Central	6,087	-4.4	8.2	-6.4	11.9
	South	6,736	2.2	7.1	3.3	10.6
	West	13,568	-6.6	9.2	-9.6	13.4
CASTNet	Northeast	264	1.1	5.9	1.7	8.7
	Mid-West	240	-4.2	6.6	-6.3	9.8
	Central	216	-8.2	8.7	-12.4	13.1
	South	443	-0.7	5.7	-1.1	8.8
	West	905	-11.0	11.5	-16.0	16.7

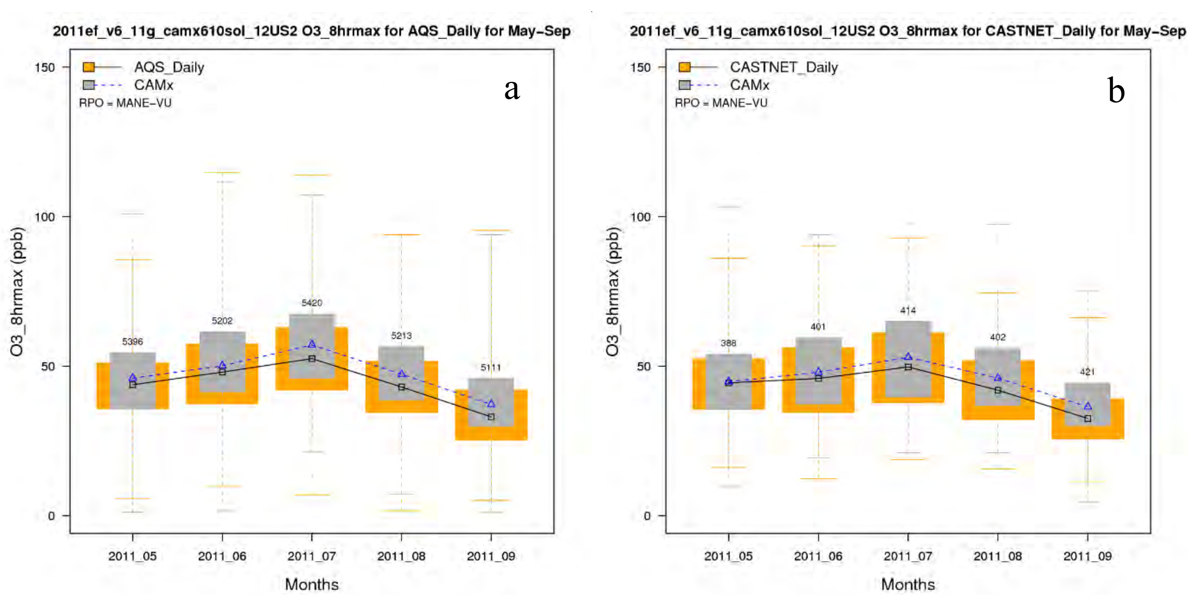

Figure 3A-1. Distribution of observed and predicted MDA8 ozone by month for the period May through September for the Northeast subregion, (a) AQS network and (b) CASTNet network. [symbol = median; top/bottom of box = 75th/25th percentiles; top/bottom line = max/min values]

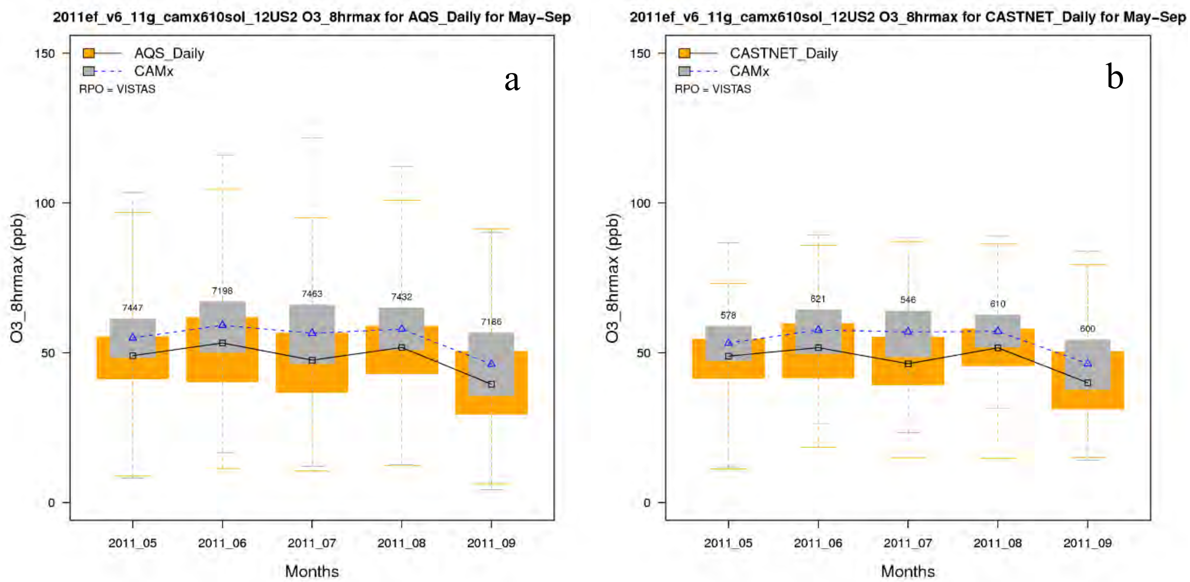

Figure 3A-2. Distribution of observed and predicted MDA8 ozone by month for the period May through September for the Southeast subregion, (a) AQS network and (b) CASTNet network

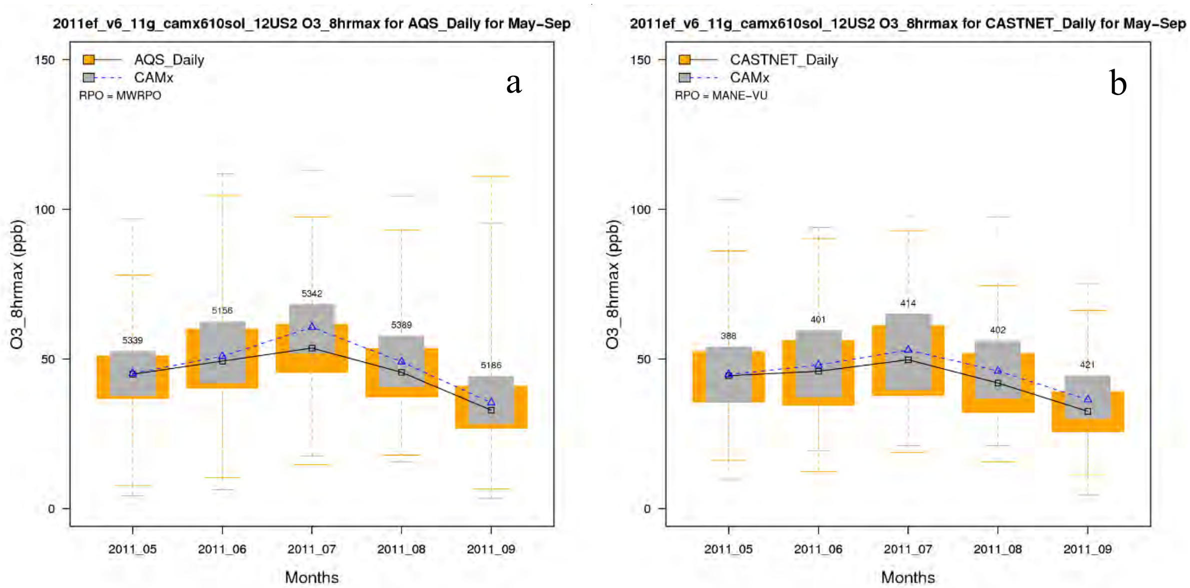

Figure 3A-3. Distribution of observed and predicted MDA8 ozone by month for the period May through September for the Midwest subregion, (a) AQS network and (b) CASTNet network

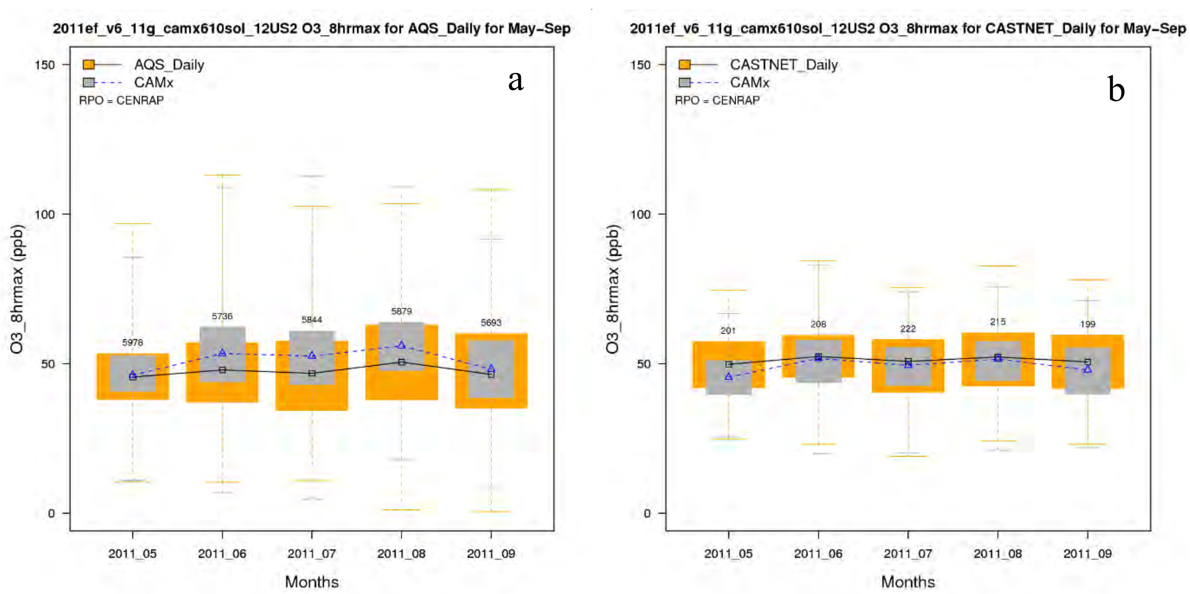

Figure 3A-4. Distribution of observed and predicted MDA8 ozone by month for the period May through September for the Central states, (a) AQS network and (b) CASTNet network

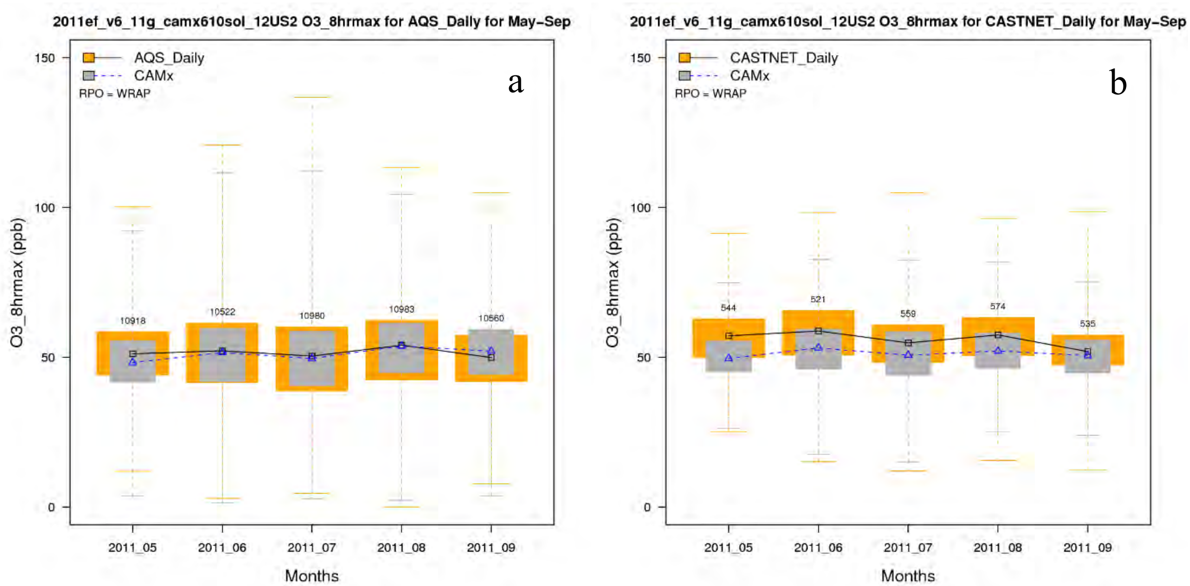

Figure 3A-5. Distribution of observed and predicted MDA8 ozone by month for the period May through September for the West, (a) AQS network and (b) CASTNet network

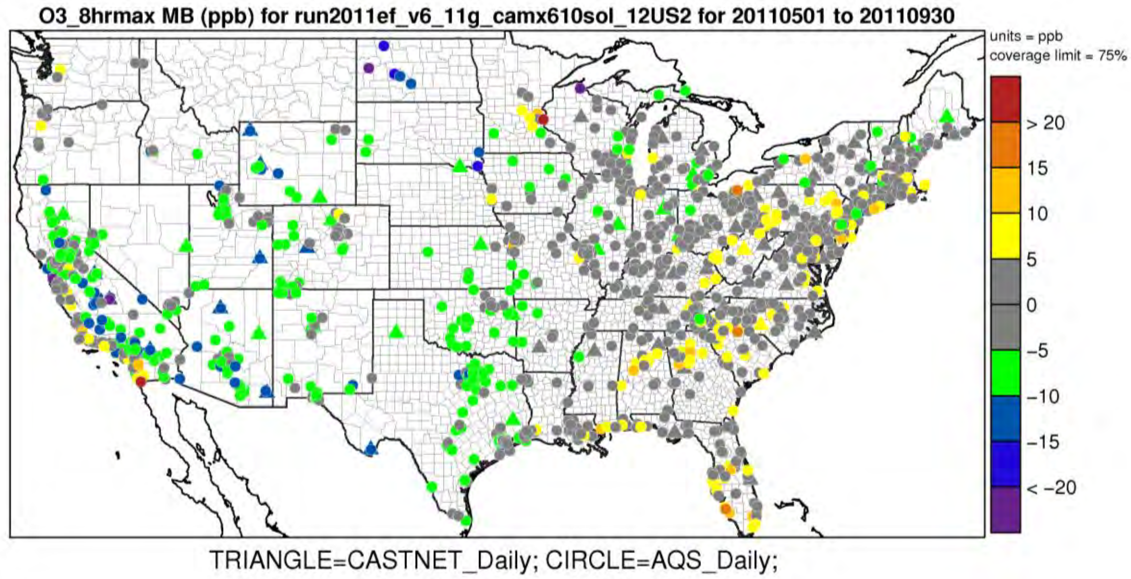

Figure 3A-6. Mean Bias (ppb) of MDA8 ozone greater than 60 ppb over the period May-September 2011 at AQS and CASTNet monitoring sites in 12-km U.S. modeling domain

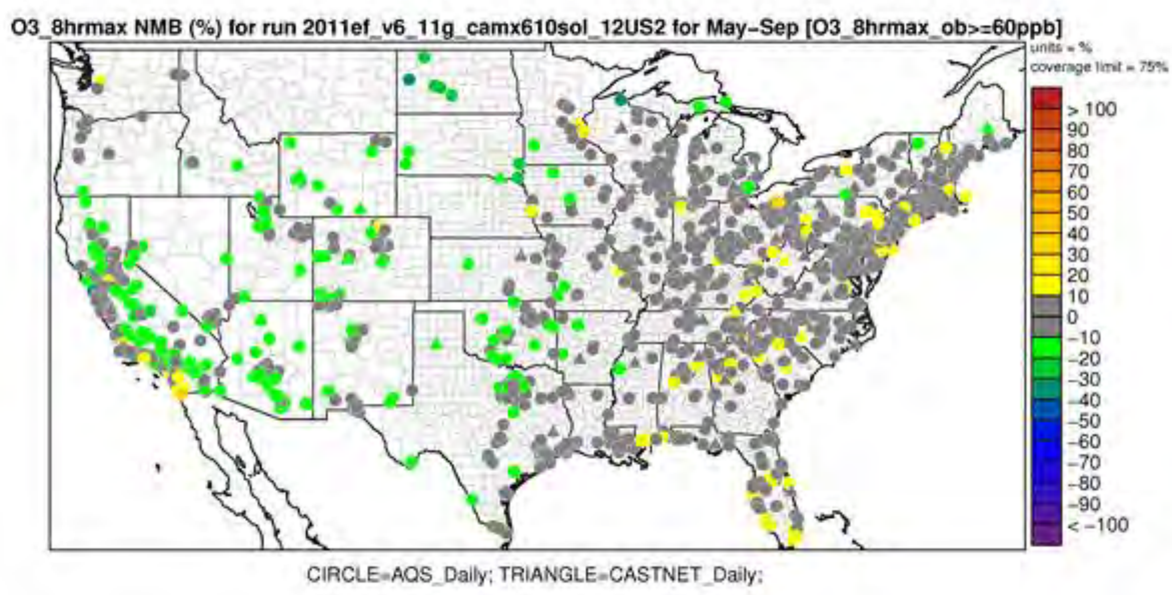

Figure 3A-7. Normalized Mean Bias (%) of MDA8 ozone greater than 60 ppb over the period May-September 2011 at AQS and CASTNet monitoring sites in 12-km U.S. modeling domain

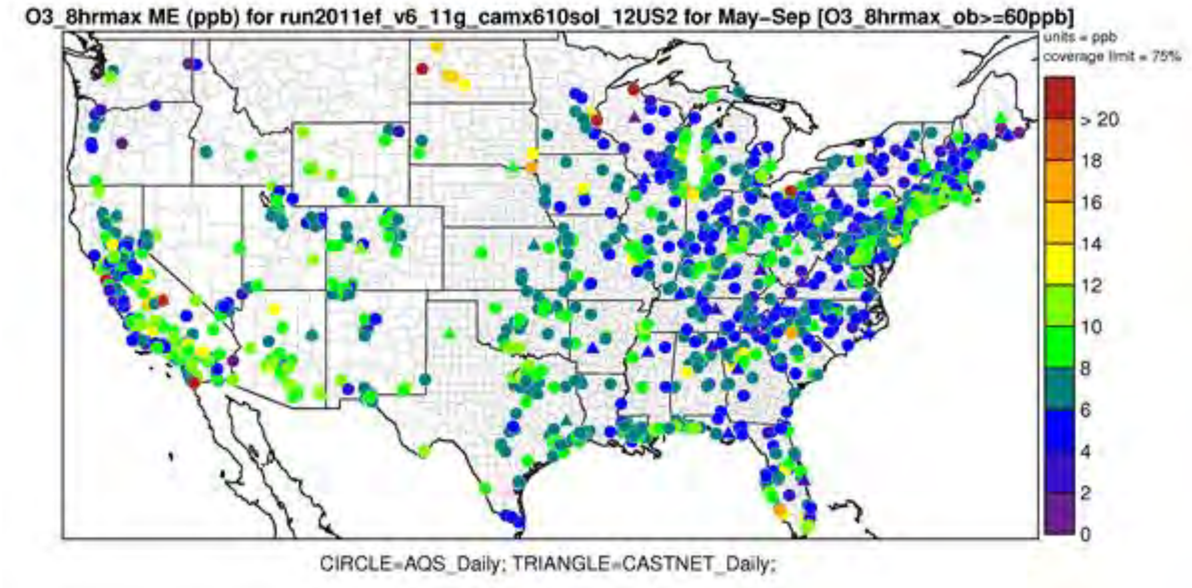

Figure 3A-8. Mean Error (ppb) of MDA8 ozone greater than 60 ppb over the period May-September 2011 at AQS and CASTNet monitoring sites in 12-km U.S. modeling domain

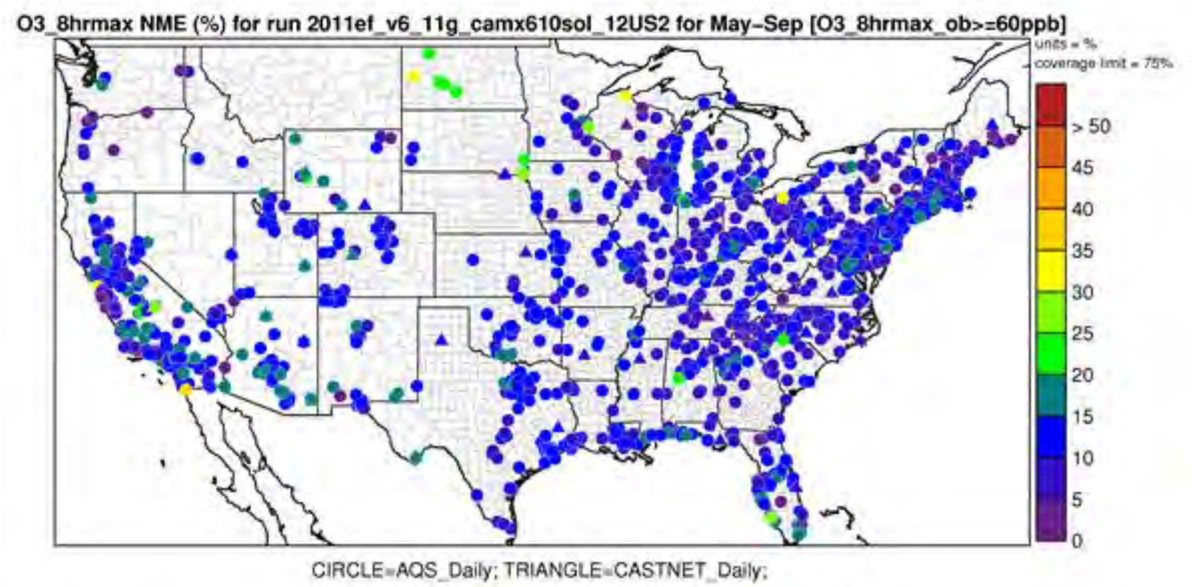

Figure 3A-9. Normalized Mean Error (%) of MDA8 ozone greater than 60 ppb over the period May-September 2011 at AQS and CASTNet monitoring sites in 12-km U.S. modeling domain

Table 3A-2. Key Monitoring Sites Used for the Ozone Time Series Analysis

County	State	Monitoring Site ID
Fresno	California	60195001
San Bernardino	California	60710005
Tarrant	Texas	484392003
Brazoria	Texas	480391004
Allegheny	Pennsylvania	420031005
Frederick	Maryland	240210037
Harford	Maryland	240251001
Queens	New York	360810124
Suffolk	New York	361030002
Sheboygan	Wisconsin	551170006
Wayne	Michigan	261630019
Jefferson	Kentucky	211110067
Douglas	Colorado	80350004

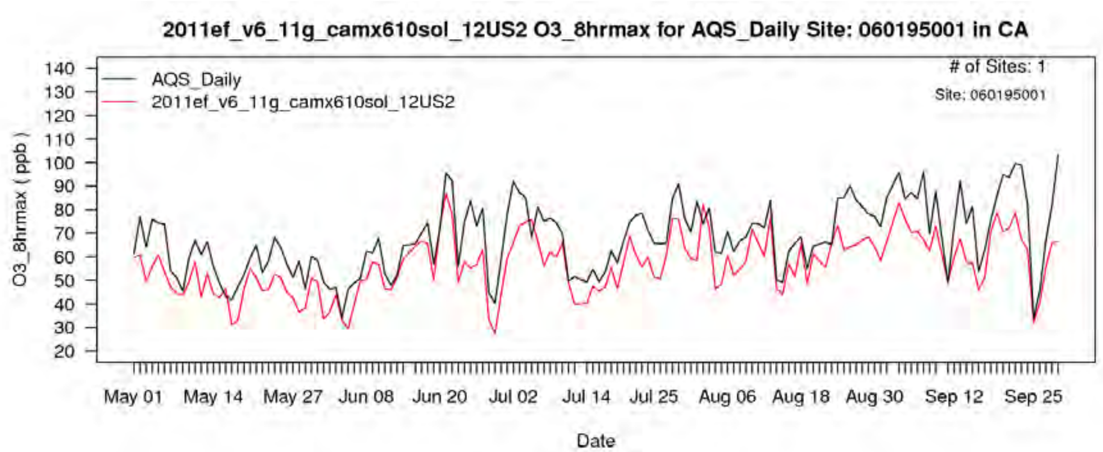

Figure 3A-10a. Time series of observed (black) and predicted (red) MDA8 ozone for May through September 2011 at site 60195001 in Fresno Co., California

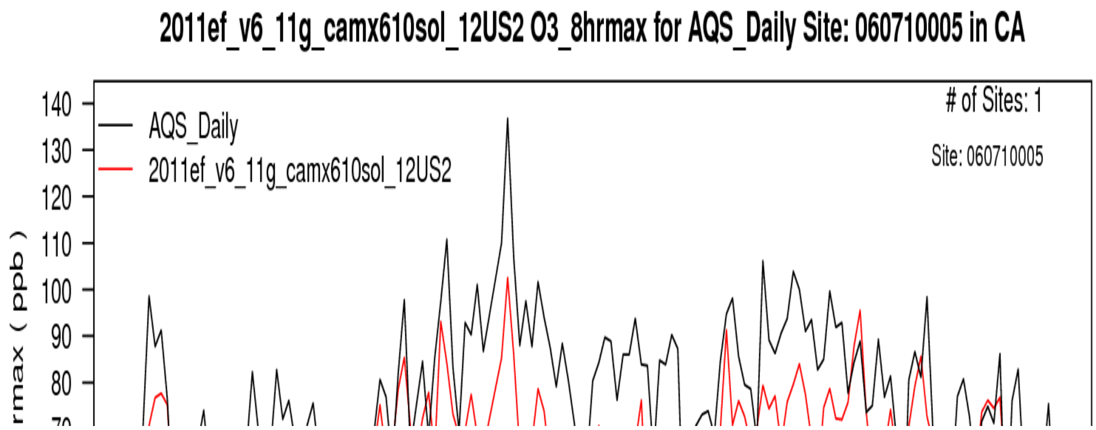

Figure 3A-10b. **Time series of observed (black) and predicted (red) MDA8 ozone for May through September 2011 at site 60710005 in San Bernardino Co., California**

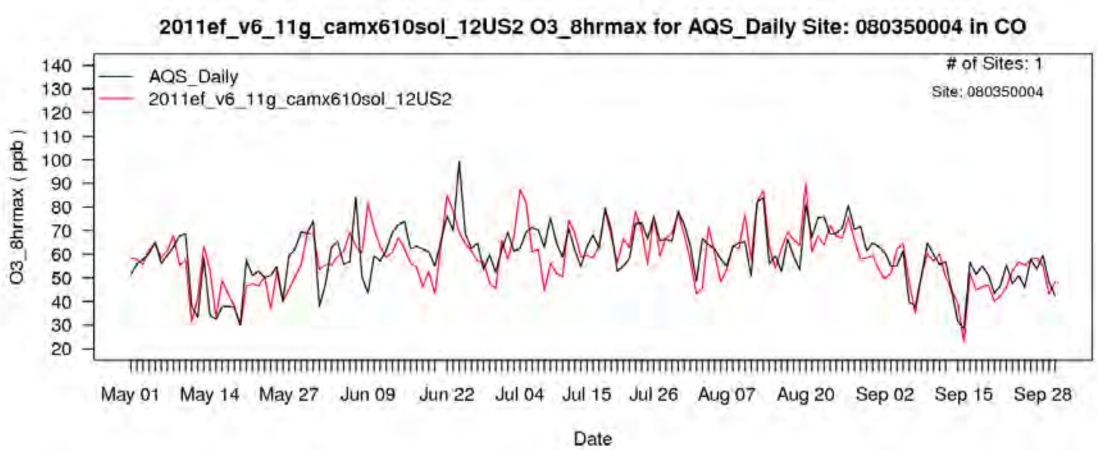

Figure 3A-10c. **Time series of observed (black) and predicted (red) MDA8 ozone for May through September 2011 at site 80350004 in Douglas Co., Colorado**

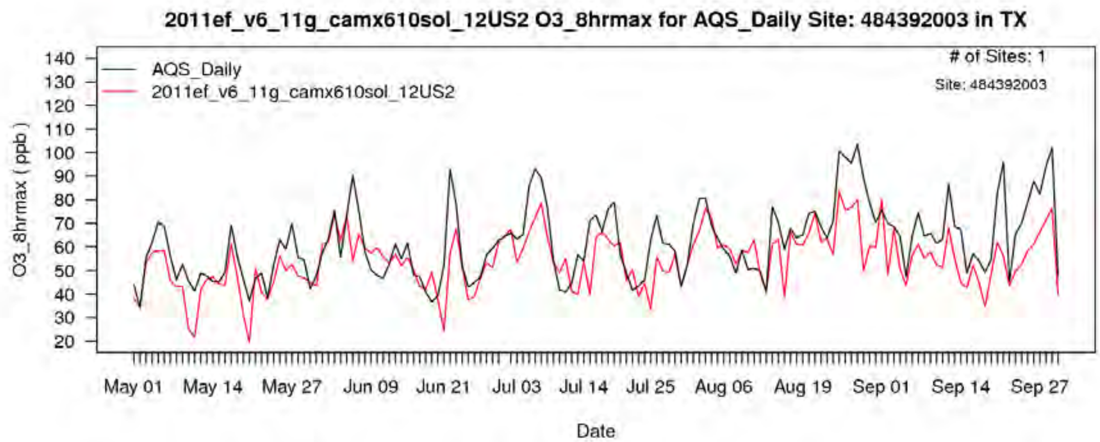

Figure 3A-10d. Time series of observed (black) and predicted (red) MDA8 ozone for May through September 2011 at site 484392003 in Tarrant Co., Texas

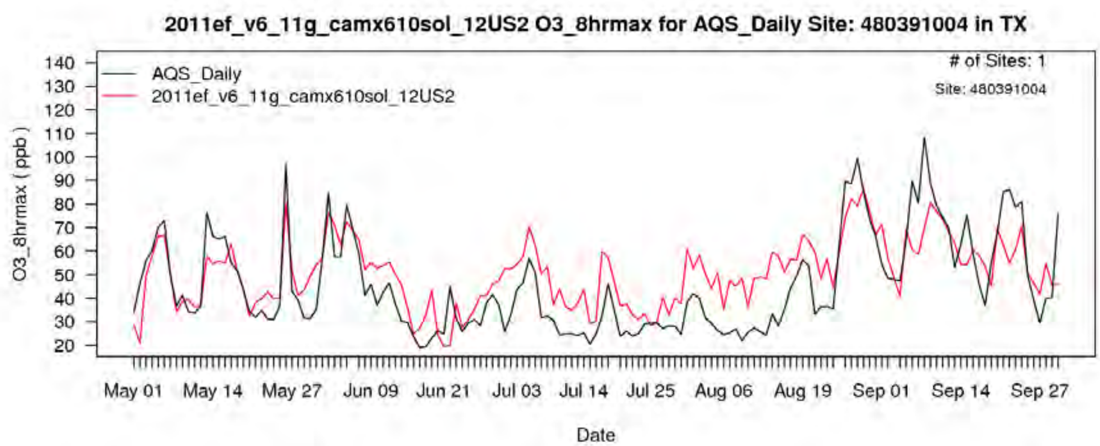

Figure 3A-10e. Time series of observed (black) and predicted (red) MDA8 ozone for May through September 2011 at site 480391004 in Brazoria Co., Texas

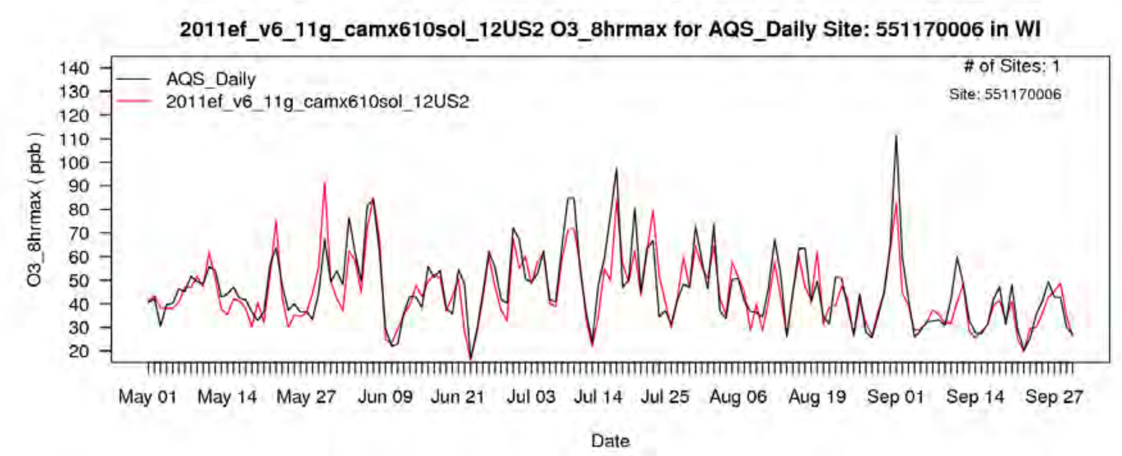

Figure 3A-10f. **Time series of observed (black) and predicted (red) MDA8 ozone for May through September 2011 at site 551170006 in Sheboygan Co., Wisconsin**

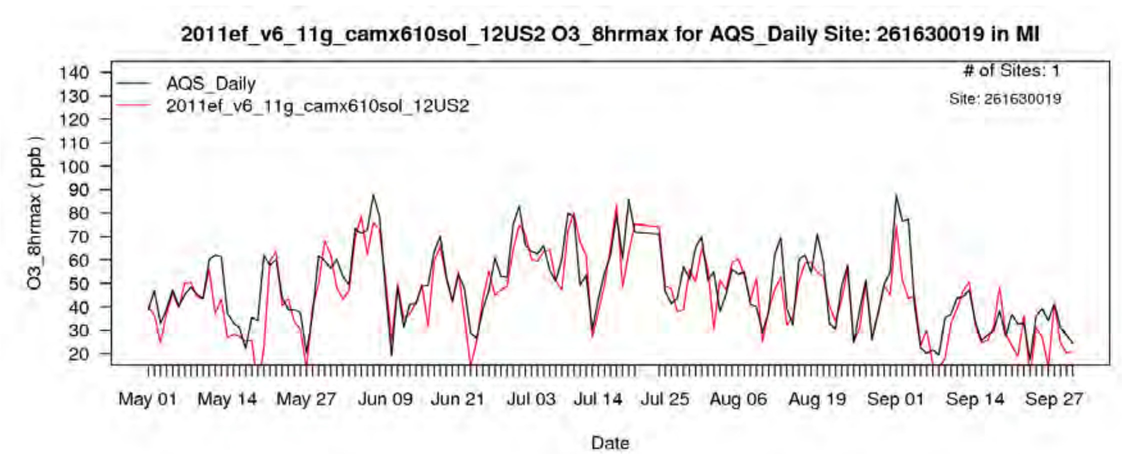

Figure 3A-10g. **Time series of observed (black) and predicted (red) MDA8 ozone for May through September 2011 at site 261630019 in Wayne Co., Michigan**

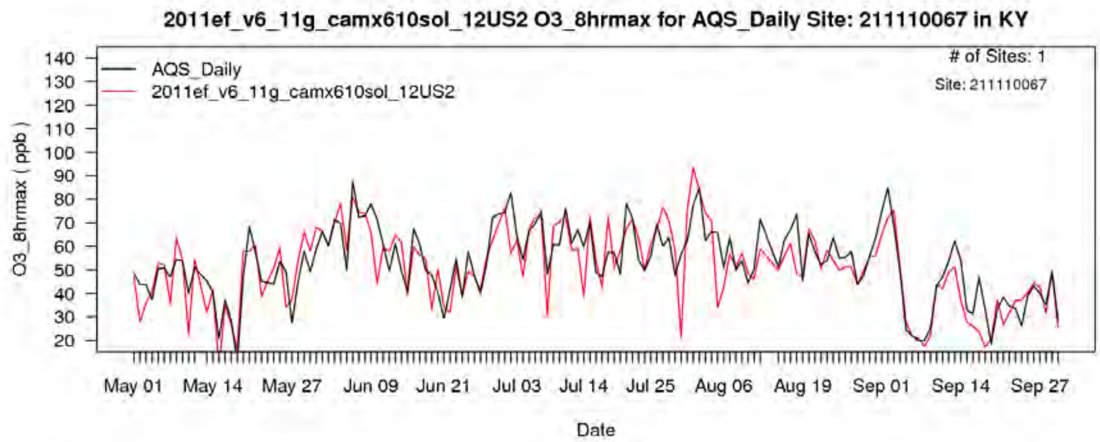

Figure 3A-10h. **Time series of observed (black) and predicted (red) MDA8 ozone for May through September 2011 at site 211110067 in Jefferson Co., Kentucky**

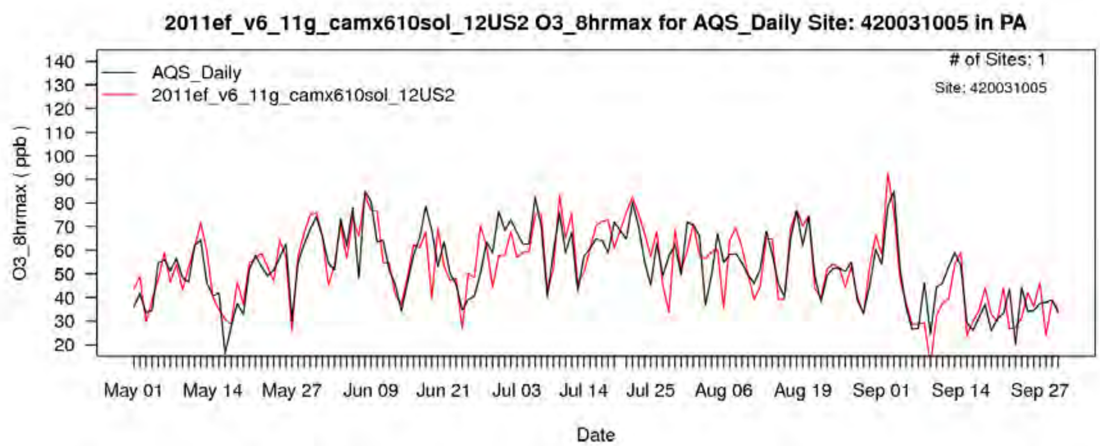

Figure 3A-10i. **Time series of observed (black) and predicted (red) MDA8 ozone for May through September 2011 at site 420031005 in Allegheny Co., Pennsylvania**

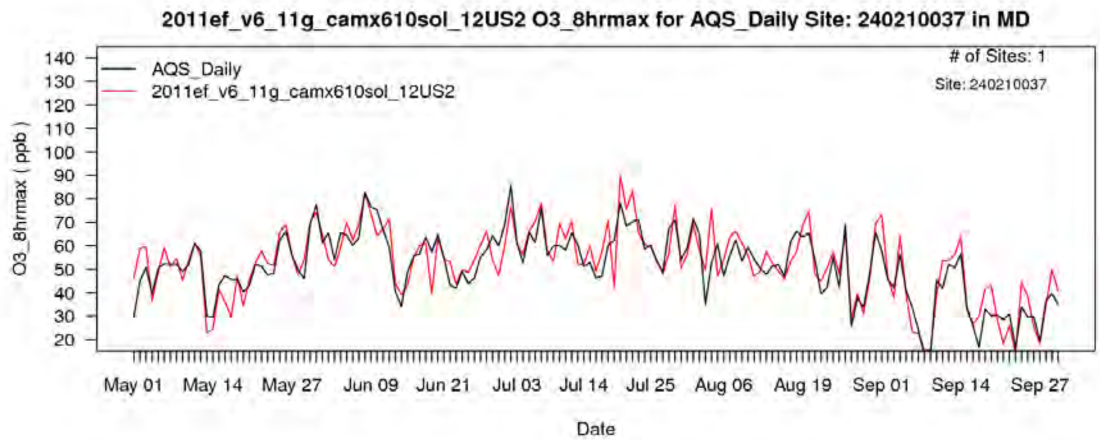

Figure 3A-10j. Time series of observed (black) and predicted (red) MDA8 ozone for May through September 2011 at site 240210037 in Frederick Co., Maryland

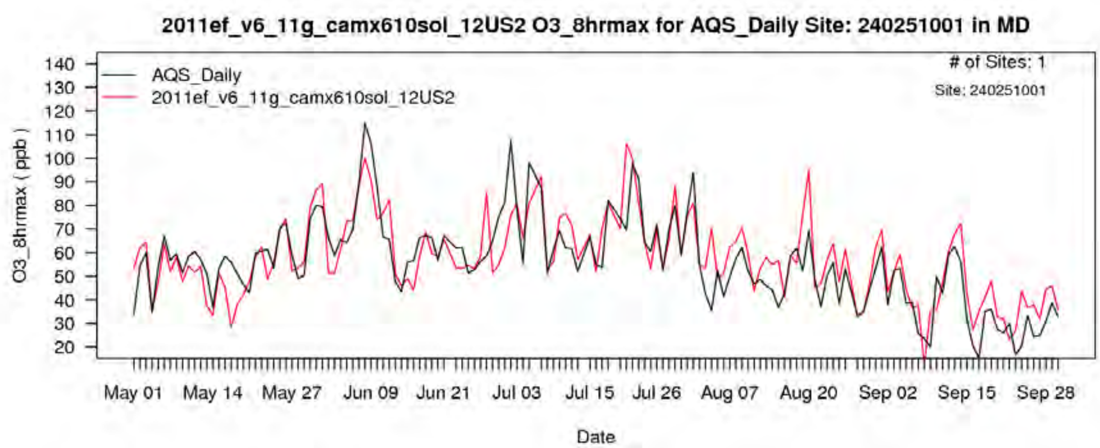

Figure 3A-10k. Time series of observed (black) and predicted (red) MDA8 ozone for May through September 2011 at site 240251001 in Harford Co., Maryland

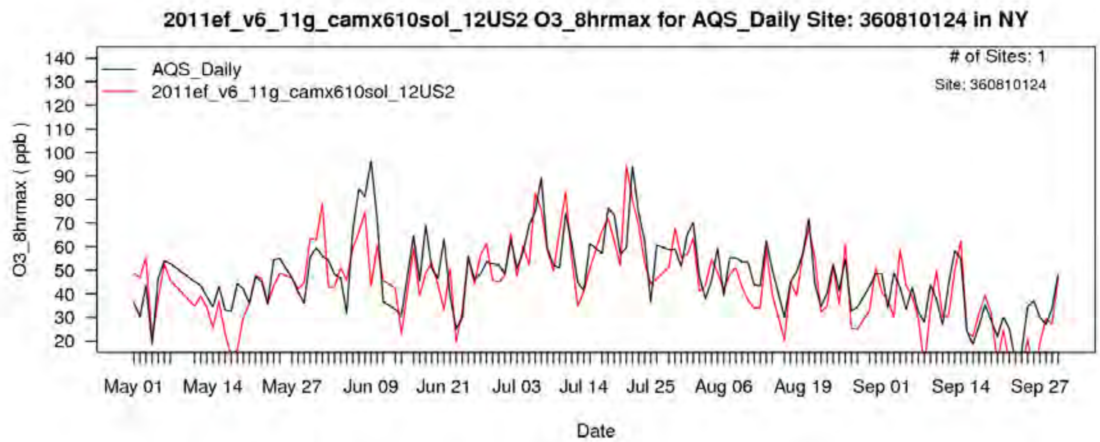

Figure 3A-10l. Time series of observed (black) and predicted (red) MDA8 ozone for May through September 2011 at site 360810124 in Queens, New York

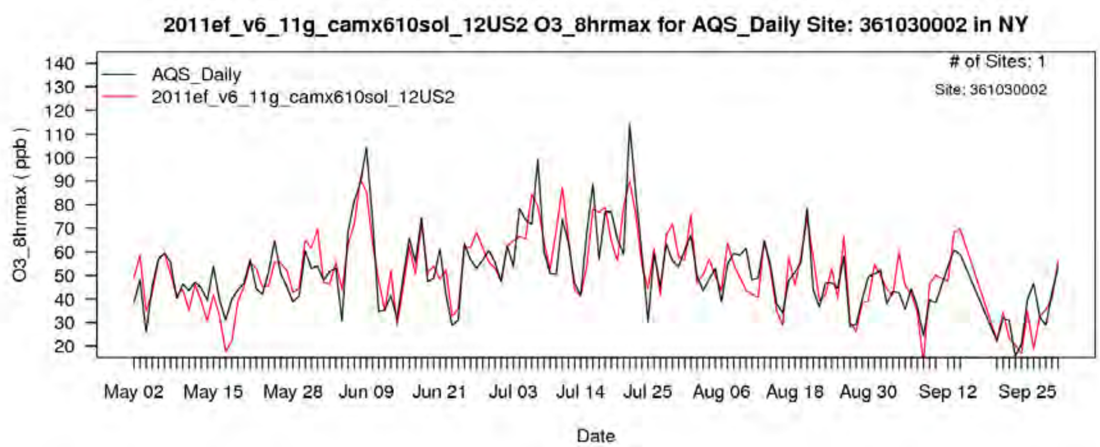

Figure 3A-10m. Time series of observed (black) and predicted (red) MDA8 ozone for May through September 2011 at site 361030002 in Suffolk County, New York.

3A.2 California Sub-Regions and Areas of Influence

As discussed in chapter 3 of the ozone RIA, we performed air quality modeling to gauge the sensitivity of ozone to 50% and 90% cuts in NOx emissions statewide in California. When applying the results of these model simulations to estimate emissions reductions to attain the current and alternative NAAQS, we made a simplifying assumption that the model-predicted ozone response at locations in the San Joaquin Valley and areas of California further north are solely due to emissions changes in these areas. That is, we associated the predicted ozone changes in northern California to emissions changes in this portion of the state even though the air quality model simulation included reductions statewide. We made a similar assumption about the response of ozone to emissions changes for the southern portion of California. The northern and southern source regions were identified for the purpose of determining which emissions reductions would be considered to impact design values at specific monitors. In calculating the impact of emissions changes on design values only monitors within each source regions were assumed to be impacted from emissions from within that source region. For VOC emissions reductions from available known controls, the source regions were defined to include only those portions of the sub-regions that also fell within the NOx buffer area for which known NOx controls were applied (see figure 3-4 from chapter 3 of the RIA). Therefore, when determining the tons of emissions reductions needed to meet various standard levels, the impacts of emissions reductions within each source region were applied to just those monitors located within that regions. In addition, we determined the areas outside of California that are expected to be most affected by emissions reductions from each of the two California sub-regions (i.e., downwind impact areas) (Figure 3A-14). When creating the BenMap and FASOMGHG surfaces described in section 3.4 of the main chapter, we determined the impact on ozone in these downwind areas due to emissions reductions from within each of the two California sub-regions.

Several considerations were considered when delineating these two California sub-regions for the various steps in this analysis. The spatial extent of the two California NOx source regions were based on the geographic boundaries of California air basins as defined by the California Air Resources Board (CARB). CARB designates Air Basins for the "purpose of managing air resources" in areas with "generally . . . similar meteorology and geographic conditions throughout" (http://www.arb.ca.gov/ei/maps/statemap/abmap.htm). The various Californian Air Basins were then combined into two larger sub-regions for this analysis. The geographic groupings were based on general air flow patterns which are governed by mountain topography and onshore/offshore wind flows. The Air Basins, sub-regions, and predominant California wind patterns are shown in Figure 3A-11. One county, Kern County, was split between Air Basins that were assigned to different sub-regions: San Joaquin Valley Air Basin (Northern sub-region) and Mojave Desert Air Basin (Southern sub-region). Since emissions inventories are categorized by county, we assigned all Kern County emissions to the Northern sub-region because Bakersfield, the most populated area of Kern County, is located in the Northern sub-region.

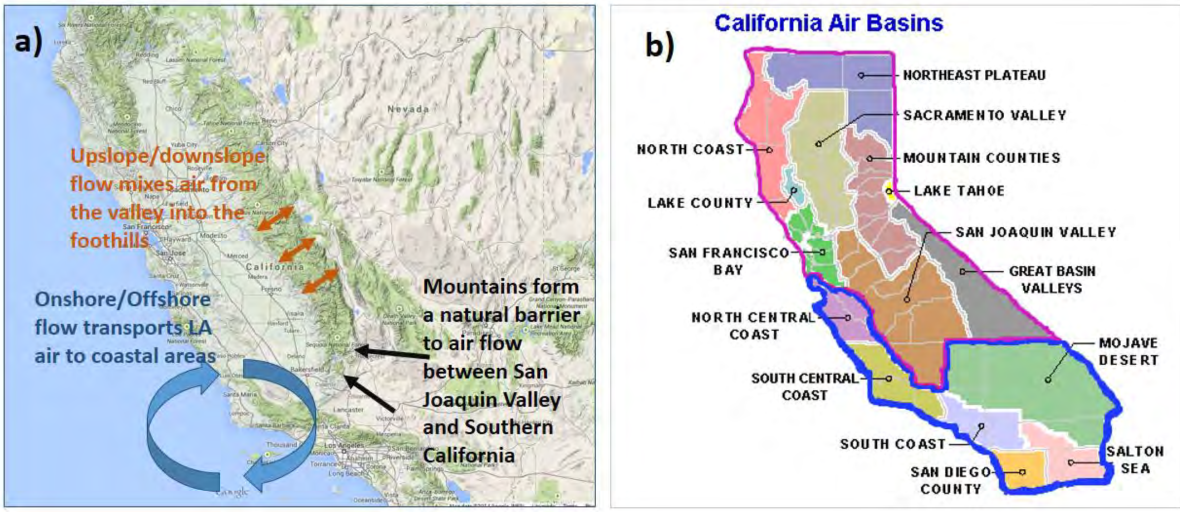

Figure 3A-11. a) Depiction of governing wind patterns and topography. b) California Air Basins and sub-regions used for this analysis. Northern sub-region is outlined in pink and southern sub-region is outlined in blue.

The downwind receptor regions outside of California were determined by examining the spatial patterns of ozone impacted by the the 50% cut California in NOx. Ozone changes due to the state-wide emissions reductions appear to follow fairly distinct widespread plumes. From this analysis it was determined that impacts of emissions reductions from the Northern sub-region were generally limited to Oregon, Washington, and a few Nevada counties near Carson City. Conversely, emissions reductions from the Southern sub-region appear to have widespread impacts across much of the Southwestern and Central US. One caveat is that the geographic extent of these downwind regions are based on general transport patterns, as determined by examining model outputs from a single year of meteorology (i.e., 2011) and are not intended to fully represent the downwind transport of ozone from California emissions on all days and at all times. However, this approach was necessary in order to match the Northern and Southern California sub-region emissions reductions to ozone impacts in downwind areas outside this state. Figure 3A-12 shows examples of the impact on 8-hr daily maximum ozone from 50% California NOx cuts on a few representative days. Figure 3A-13 shows the downwind receptor regions that were applied in this analysis. Note that VOC impacts were only applied within the source regions consistent with how VOC reductions were treated for other areas of the country. Figure 3A-14 shows that ozone impacts from a 50% cut in US anthropogenic VOC emissions were localized within the two California sub-regions and do not appear to impact downwind states.

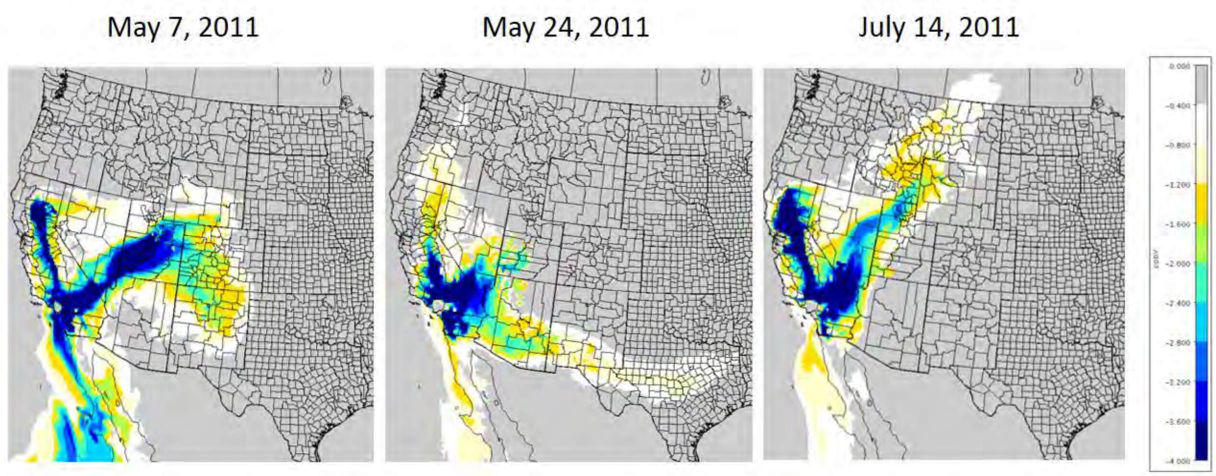

Figure 3A-12. Impact of 50% anthropogenic California NOx cuts (ppb) on 8-hr daily average ozone concentrations on three days in 2011

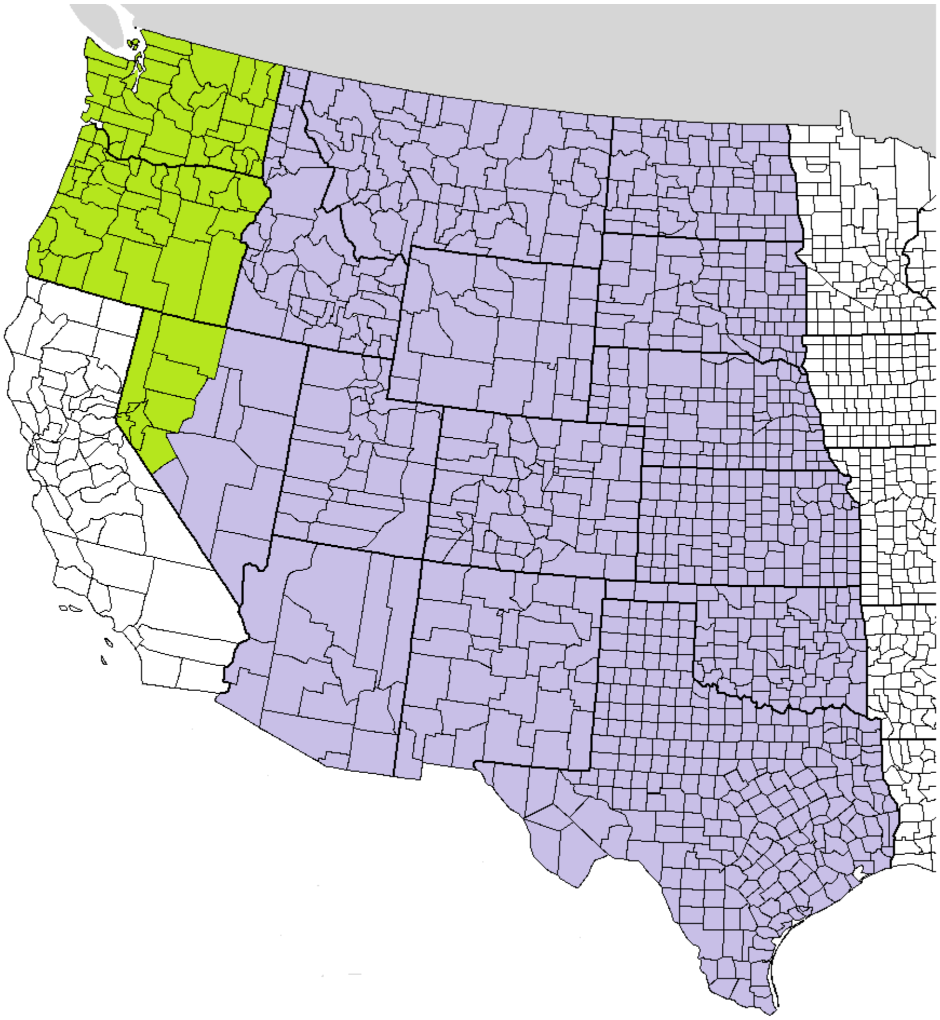

Figure 3A-13. Downwind California receptor regions for Northern California (green) and Southern California (purple)

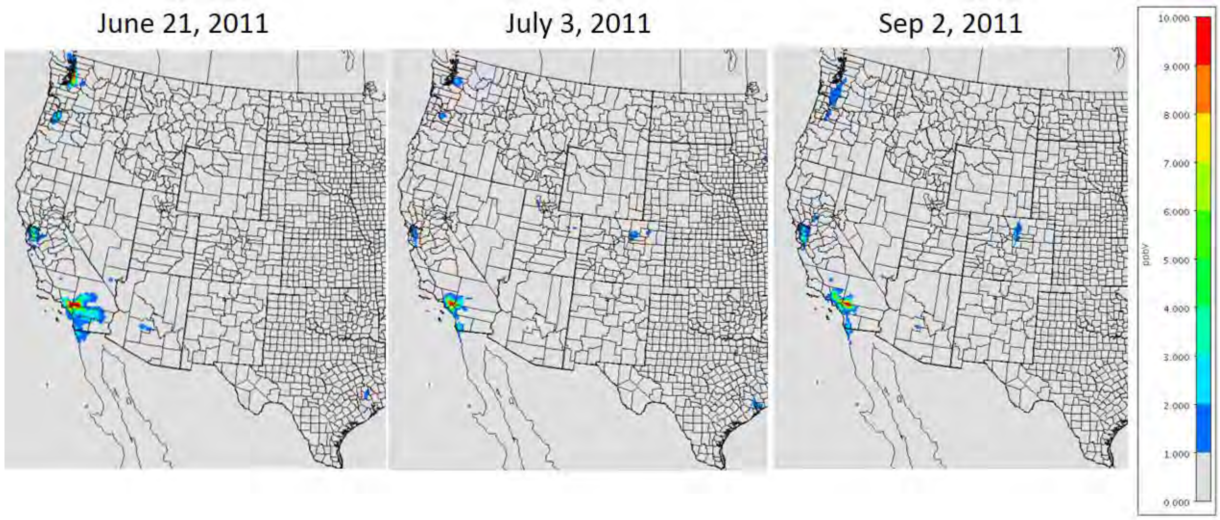

Figure 3A-14. Impact of 50% US anthropogenic VOC cuts (ppb) on 8-hr daily average ozone concentrations on three days in 2011

3A.3 VOC Impact Areas

As described in chapter 3, we defined VOC impact regions for the following urban areas: New York City, Pittsburgh, Baltimore, Detroit, Chicago, Louisville, Houston, Dallas, Denver, Northern California and Southern California. Not only did these areas have the highest design values in each region, but ozone in these areas was also sensitive to VOC emissions reductions in our modeling. Figure 3A-15 shows the impact of 50% US anthropogenic VOC cuts on July monthly average 8-hour daily maximum ozone concentrations across the US. Ozone in each of the areas listed above is shown to have at least 0.2 ppb response to VOC emissions cuts.

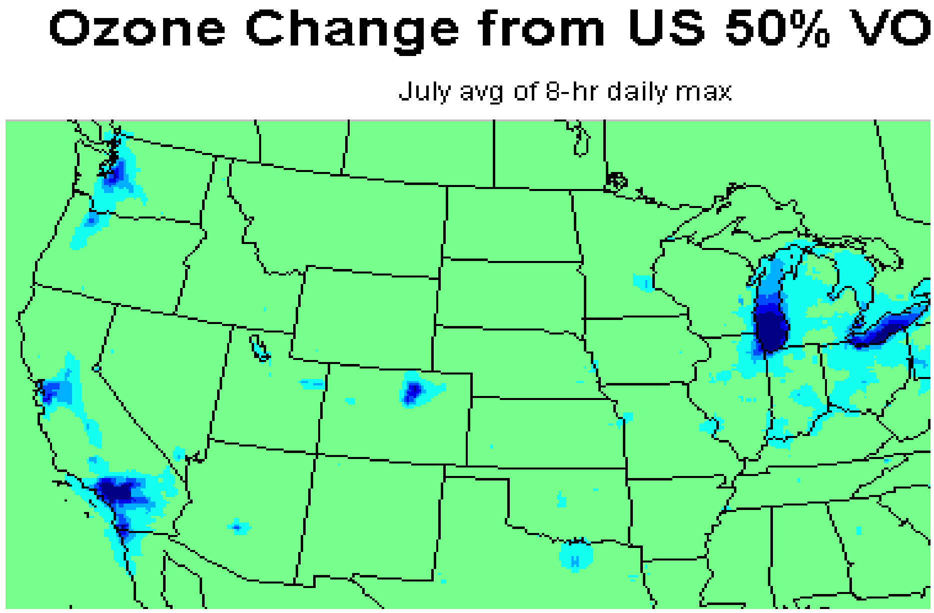

Figure 3A-15. Change in July average of 8-hr daily maximum ozone concentration (ppb) due to 50% cut in US anthropogenic VOC emissions

3A.4 Numeric Examples of Calculation Methodology for Changes in Design Values

In this section we use the data for two monitoring sites to demonstrate how changes in design values were calculated, as described in section 3.3. For each monitor, numerical examples are given for calculating the emissions reductions necessary to attain the current 75 ppb NAAQS (i.e., the baseline scenario) as well as the 65 ppb scenario, which is incremental to the baseline. Note that design values are truncated when they are compared to a standard level, so a calculated design value of 75.9 is truncated to 75 ppb and, therefore, meets the current 75 ppb standard. Similarly, a design value of 65.9 would meet an alternative standard level of 65. For each monitor, we start with the base case design value, then account for ozone changes simulated in the 111(d) sensitivity simulation and then apply equation 3-5 from chapter 3.

$$DV_j = DV_{2025,j} + \left(R_{1,j} \times \Delta E_1\right) + \left(R_{2,j} \times \Delta E_2\right) + \left(R_{3,j} \times \Delta E_3\right) + \cdots \qquad \text{Eq 3-5}$$

<u>Example 1. Fresno California monitor 60195001 (baseline):</u>

$$DV_{60195001,baseline}$$

$$= \underbrace{83.5}_{DV_{60195001,2025}} + \overbrace{\underbrace{-0.7}_{\Delta DV_{60195001,111d}}}^{NOx+VOC} + \overbrace{\left(\underbrace{-9.9 \times 10^{-5}}_{R_{60195001,CAcontrl,NCA}} \times \underbrace{15{,}000}_{\Delta E_{CAcontrl,NCA}}\right)}^{NOx}$$

$$+ \overbrace{\left(\underbrace{-1.3 \times 10^{-4}}_{R_{60195001,CAcontrol+50NOx,NCA}} \times \underbrace{8100 + 32{,}000}_{\Delta E_{mobile,2030}+\Delta E}\right)}^{NOx}$$

$$+ \overbrace{\left(\underbrace{-9.9 \times 10^{-6}}_{R_{60195001,VOC\_50,NCA}} \times \underbrace{3200 + 22{,}000}_{\Delta E_{mobile,2030}+\Delta E}\right)}^{VOC} = 75.9\ ppb$$

<u>Example 2. Fresno California monitor 60195001 (65 ppb scenario):</u>

$$DV_{j,65} = \underbrace{75.9}_{DV_{j,baseline}} + \overbrace{\left(\underbrace{-1.3 \times 10^{-4}}_{R_{60195001,CAcontrol+50NOx,NCA}} \times \underbrace{61{,}000}_{\Delta E}\right)}^{NOx\ up\ to\ 50\%\ of\ CA\ modeled\ control\ sensitivity}$$

$$+ \overbrace{\left(\underbrace{-2.0 \times 10^{-4}}_{R_{60195001,CAcontrol50-90NOx,NCA}} \times \underbrace{12{,}000}_{\Delta E}\right)}^{NOx\ beyond\ 50\%\ of\ CA\ modeled\ control\ sensitivity} = 65.6\ ppb$$

<u>Example 3. Dallas monitor 484392003 (baseline):</u>

$$DV_{484392003,baseline} = \underbrace{77.1}_{DV_{484392003,2025}} + \overbrace{\underbrace{-1.0}_{\Delta DV_{484392003,111d}}}^{NOx+VOC} + \overbrace{\left(\underbrace{-1.6 \times 10^{-5}}_{R_{484392003,TXcontrol}} \times \underbrace{44{,}000}_{\Delta E}\right)}^{NOx}$$

$$= 75.4\ ppb$$

<u>Example 4. Dallas monitor 484392003 (65 ppb scenario):</u>

$$DV_{484392003,65} = \underbrace{75.4}_{DV_{484392003,baseline}} + \overbrace{\left(\underbrace{-1.6 \times 10^{-5}}_{R_{484392003,TXcontrol}} \times \underbrace{56{,}000}_{\Delta E}\right)}^{NOx}$$
$$+ \overbrace{\left(\underbrace{-1.1 \times 10^{-5}}_{R_{484392003,TXcontrol+50centralNOx}} \times \underbrace{770{,}000}_{\Delta E}\right)}^{NOx}$$
$$+ \overbrace{\left(\underbrace{-9.2 \times 10^{-6}}_{R_{484392003,VOC\_50,Dallas}} \times \underbrace{17{,}000}_{\Delta E}\right)}^{VOC} = 65.9\ ppb$$

3A.5 Emissions Reductions Applied to Create Baseline and Alternative Standard Level Scenarios

The following tables present emissions reductions applied in each region to create the baseline and alternative standard level scenarios. These emissions reductions were determined using the methodology described in section 3.3.2 of the main RIA and demonstrated in section 3A.4 of this appendix. Sector-specific controls used for these reductions are discussed in more detail in chapter 4 of the RIA. These emissions reductions were used to create the ozone surfaces described in section 3.4 of the RIA.

Table 3A-3. Emissions Reductions Applied to Create the Baseline Scenario*

	Emissions reductions (thousand tons) applied from			
	2025-2030 California mobile source changes	**NOx reductions from one of the explicit control cases**	**VOC reductions identified from maxcontrol CoST run**	**Additional NOx reductions**
Northeast	N/A	N/A	N/A	N/A
Midwest	N/A	N/A	N/A	N/A
Central	N/A	45 (TX explicit emissions control case)	N/A	N/A
Southwest	N/A	N/A	N/A	N/A
California	14 (NOx) 6 (VOC)	29 (CA explicit emissions control case)	51 (in N and S CA buffer region)	32 (N California); 130 (S California)

*These emission are in addition to changes modeled in the simulation representing option 1(state) of the proposed carbon pollution guidelines under section 111(d) of the CAA.

Table 3A-4. Emissions Reductions Applied Beyond the Baseline Scenario to Create the 70 ppb Scenario

	Emissions reductions (thousand tons) applied from

	NOx reductions from one of the explicit control cases*	VOC reductions identified from maxcontrol CoST run	Additional NOx reductions
Northeast	110	31 (NY area); 6 (Baltimore area)	130 (98 within NE buffer; 31 outside the NE buffer)
Midwest	N/A	N/A	N/A
Central	58 (TX explicit emissions control case)	18 (Houston area)	350 (95 within TX buffer; 260 outside of TX buffer)
Southwest	N/A	N/A	N/A
California	Exhausted in baseline scenario	Exhausted in baseline scenario	38 (N California); 15 (S California)

Table 3A-5. Emissions Reductions Applied Beyond the Baseline Scenario to Create the 65 ppb Scenario

	Emissions reductions (thousand tons) applied from		
	NOx reductions from one of the explicit control cases*	VOC reductions identified from maxcontrol CoST run	Additional NOx reductions
Northeast	110	36 (NY area)	400 (300 within NE buffer; 97outside the NE buffer)
Midwest	250	28 (Chicago area)	180
Central	58 (TX explicit emissions control case)	18 (Houston area); 17 (Dallas area)	770 (210 within TX buffer; 560 outside of TX buffer)
Southwest	77	7 (Denver area)	36
California	Exhausted in baseline scenario	Exhausted in baseline scenario	73 (N California); 32 (S California)

*In regions without a modeled explicit control case (Southwest and Midwest) this represents equivalent emissions reductions that would have been identified in an explicit control case that covered the entire region

Table 3A-6. Emissions Reductions Applied Beyond the Baseline Scenario to Create the 60 ppb Scenario

	Emissions reductions (thousand tons) applied from		
	NOx reductions from one of the explicit control cases*	VOC reductions identified from maxcontrol CoST run	Additional NOx reductions
Northeast	110	36 (NY area); 6 (Baltimore area)	610 (470 within NE buffer; 150 outside the NE buffer)
Midwest	250	32 (Chicago area); 41 (additional Chicago area VOC control)	620
Central	58 (TX explicit emissions control case)	18 (Houston area); 18 (Dallas area)	1,200 (340 within TX buffer; 910 outside of TX buffer)
Southwest	77	7 (Denver area)	240
California	Exhausted in baseline scenario	Exhausted in baseline scenario	97 (N California); 48 (S California)

*In regions without a modeled explicit control case (Southwest and Midwest) this represents equivalent emissions reductions that would have been identified in an explicit control case that covered the entire region

3A.6 Design Values for All Monitors included in the Quantitative Analysis

Table 3A-7. Design Values for California Region Monitors

Site ID	Lat	long	State	County	Base Case	Baseline	70	65	60
60010007	37.69	-121.78	California	Alameda	66	61	57	53	48
60010009	37.74	-122.17	California	Alameda	47	44	42	40	36
60010011	37.81	-122.28	California	Alameda	46	43	41	39	35
60012001	37.65	-122.03	California	Alameda	54	51	48	45	41
60050002	38.34	-120.76	California	Amador	60	54	50	46	43
60070007	39.71	-121.62	California	Butte	63	58	54	50	47
60070008	39.76	-121.84	California	Butte	53	49	46	43	40
60090001	38.20	-120.68	California	Calaveras	63	57	53	49	46
60111002	39.20	-122.02	California	Colusa	53	49	46	43	41
60130002	37.94	-122.03	California	Contra Costa	65	60	57	53	48
60131002	38.01	-121.64	California	Contra Costa	65	59	56	51	47
60131004	37.96	-122.36	California	Contra Costa	51	47	45	42	38
60170010	38.73	-120.82	California	El Dorado	67	60	55	50	46
60170012	38.81	-120.03	California	El Dorado	61	59	57	56	55
60170020	38.89	-121.00	California	El Dorado	69	62	57	52	47
60190007	36.71	-119.74	California	Fresno	82	74	69	64	59
60190011	36.79	-119.77	California	Fresno	81	73	68	63	58
60190242	36.84	-119.87	California	Fresno	81	74	**70**	**65**	**60**
60192009	36.63	-120.38	California	Fresno	64	59	55	52	49
60194001	36.60	-119.50	California	Fresno	76	69	65	60	56
60195001	36.82	-119.72	California	Fresno	83	**75**	**70**	**65**	**60**
60210003	39.53	-122.19	California	Glenn	56	52	49	46	44
60250005	32.68	-115.48	California	Imperial	69	62	61	60	59
60254003	33.03	-115.62	California	Imperial	64	54	53	51	50
60254004	33.21	-115.55	California	Imperial	63	53	52	50	48
60290007	35.35	-118.85	California	Kern	81	74	**70**	**65**	**60**
60290008	35.05	-119.40	California	Kern	72	67	63	59	55
60290011	35.05	-118.15	California	Kern	71	58	57	55	53
60290014	35.36	-119.04	California	Kern	77	71	66	61	56
60290232	35.44	-119.02	California	Kern	77	71	66	61	57
60295002	35.24	-118.79	California	Kern	74	68	64	59	55
60296001	35.50	-119.27	California	Kern	73	67	63	59	55
60311004	36.31	-119.64	California	Kings	74	67	63	59	55
60333001	39.03	-122.92	California	Lake	48	45	43	41	38

Site ID	Lat	long	State	County	Base Case	Baseline	70	65	60
60370002	34.14	-117.92	California	Los Angeles	75	55	52	48	43
60370016	34.14	-117.85	California	Los Angeles	88	65	61	56	51
60370113	34.05	-118.46	California	Los Angeles	62	49	46	43	39
60371002	34.18	-118.32	California	Los Angeles	73	53	49	46	42
60371103	34.07	-118.23	California	Los Angeles	63	48	45	41	37
60371201	34.20	-118.53	California	Los Angeles	83	62	59	55	51
60371302	33.90	-118.21	California	Los Angeles	57	52	51	49	47
60371602	34.01	-118.07	California	Los Angeles	66	55	52	48	44
60371701	34.07	-117.75	California	Los Angeles	80	62	58	54	50
60372005	34.13	-118.13	California	Los Angeles	74	55	51	47	43
60374002	33.82	-118.19	California	Los Angeles	55	50	49	47	45
60376012	34.38	-118.53	California	Los Angeles	88	64	60	56	51
60379033	34.67	-118.13	California	Los Angeles	79	60	57	54	51
60390004	36.87	-120.01	California	Madera	70	64	60	56	52
60392010	36.95	-120.03	California	Madera	73	67	63	59	55
60410001	37.97	-122.52	California	Marin	47	44	41	39	36
60430006	37.55	-119.84	California	Mariposa	65	61	58	55	53
60470003	37.28	-120.43	California	Merced	71	65	61	57	53
60530002	36.50	-121.73	California	Monterey	50	40	39	37	36
60530008	36.21	-121.13	California	Monterey	51	41	40	39	37
60531003	36.70	-121.64	California	Monterey	46	36	35	33	32
60550003	38.31	-122.30	California	Napa	53	49	46	43	40
60570005	39.23	-121.06	California	Nevada	63	58	54	50	47
60570007	39.32	-120.85	California	Nevada	61	56	52	48	45
60590007	33.83	-117.94	California	Orange	60	49	47	44	42
60591003	33.67	-117.93	California	Orange	58	48	46	44	41
60592022	33.63	-117.68	California	Orange	60	45	42	40	37
60595001	33.93	-117.95	California	Orange	66	54	51	48	45
60610003	38.94	-121.10	California	Placer	69	62	57	52	47
60610004	39.10	-120.95	California	Placer	61	55	51	47	44
60610006	38.75	-121.27	California	Placer	70	63	58	53	48
60650004	34.01	-117.52	California	Riverside	77	60	57	53	50
60650008	33.74	-115.82	California	Riverside	56	47	46	44	43
60650009	33.45	-117.09	California	Riverside	60	46	43	41	39
60650012	33.92	-116.86	California	Riverside	85	64	60	56	52
60650016	33.58	-117.08	California	Riverside	64	48	45	43	40
60651016	33.95	-116.83	California	Riverside	87	65	61	58	53
60652002	33.71	-116.22	California	Riverside	74	59	57	55	52
60655001	33.85	-116.54	California	Riverside	81	63	60	57	54
60656001	33.79	-117.23	California	Riverside	78	58	54	51	47

Site ID	Lat	long	State	County	Base Case	Baseline	70	65	60
60658001	34.00	-117.42	California	Riverside	89	68	64	60	55
60658005	34.00	-117.49	California	Riverside	85	65	61	57	53
60659001	33.68	-117.33	California	Riverside	74	55	52	49	45
60659003	33.61	-114.60	California	Riverside	60	54	53	52	51
60670002	38.71	-121.38	California	Sacramento	65	59	54	49	45
60670006	38.61	-121.37	California	Sacramento	66	60	55	50	45
60670010	38.56	-121.49	California	Sacramento	61	55	51	47	42
60670011	38.30	-121.42	California	Sacramento	62	56	52	48	44
60670012	38.68	-121.16	California	Sacramento	77	69	64	58	52
60670014	38.65	-121.51	California	Sacramento	60	54	50	46	42
60675003	38.49	-121.21	California	Sacramento	72	65	60	54	49
60690002	36.84	-121.36	California	San Benito	54	42	40	38	36
60690003	36.49	-121.16	California	San Benito	61	48	46	45	43
60710001	34.90	-117.02	California	San Bernardino	69	57	55	53	51
60710005	34.24	-117.27	California	San Bernardino	**99**	**75**	**70**	**65**	**60**
60710012	34.43	-117.56	California	San Bernardino	85	65	62	58	54
60710306	34.51	-117.33	California	San Bernardino	77	60	57	54	50
60711004	34.10	-117.63	California	San Bernardino	91	71	66	62	56
60711234	35.76	-117.40	California	San Bernardino	65	60	59	59	58
60712002	34.10	-117.49	California	San Bernardino	96	74	69	64	59
60714001	34.42	-117.29	California	San Bernardino	88	68	64	60	55
60714003	34.06	-117.15	California	San Bernardino	97	73	68	64	59
60719002	34.07	-116.39	California	San Bernardino	82	66	63	61	58
60719004	34.11	-117.27	California	San Bernardino	91	69	64	60	55
60730001	32.63	-117.06	California	San Diego	60	54	53	52	51
60730003	32.79	-116.94	California	San Diego	61	49	47	45	43
60730006	32.84	-117.13	California	San Diego	62	50	49	47	45
60731001	32.95	-117.26	California	San Diego	57	48	46	45	44
60731002	33.13	-117.08	California	San Diego	58	44	42	40	38
60731006	32.84	-116.77	California	San Diego	69	55	52	50	47
60731008	33.22	-117.40	California	San Diego	56	44	43	41	39
60731010	32.70	-117.15	California	San Diego	55	49	48	47	46
60731016	32.85	-117.12	California	San Diego	59	47	46	44	42
60731201	33.36	-117.09	California	San Diego	58	45	43	41	39
60732007	32.55	-116.94	California	San Diego	54	48	47	46	45
60771002	37.95	-121.27	California	San Joaquin	59	54	50	46	42
60773005	37.68	-121.44	California	San Joaquin	70	65	61	56	52
60790005	35.63	-120.69	California	San Luis Obispo	55	46	44	43	42
60792006	35.26	-120.67	California	San Luis Obispo	47	38	37	36	35
60793001	35.37	-120.84	California	San Luis Obispo	46	39	38	38	37

Site ID	Lat	long	State	County	Base Case	Baseline	70	65	60
60794002	35.03	-120.50	California	San Luis Obispo	50	41	40	39	37
60798001	35.49	-120.67	California	San Luis Obispo	53	44	43	42	40
60798005	35.64	-120.23	California	San Luis Obispo	67	55	53	52	50
60798006	35.35	-120.04	California	San Luis Obispo	65	53	51	49	47
60811001	37.48	-122.20	California	San Mateo	53	50	49	46	42
60830008	34.46	-120.03	California	Santa Barbara	51	44	43	42	41
60830011	34.43	-119.69	California	Santa Barbara	49	42	41	40	39
60831008	34.95	-120.44	California	Santa Barbara	43	35	34	33	32
60831013	34.73	-120.43	California	Santa Barbara	54	45	44	42	41
60831014	34.54	-119.79	California	Santa Barbara	58	49	48	47	45
60831018	34.53	-120.20	California	Santa Barbara	49	43	43	42	41
60831021	34.40	-119.46	California	Santa Barbara	58	49	48	47	46
60831025	34.49	-120.05	California	Santa Barbara	60	52	50	49	48
60832004	34.64	-120.46	California	Santa Barbara	47	40	39	38	37
60832011	34.45	-119.83	California	Santa Barbara	49	42	41	41	40
60833001	34.61	-120.08	California	Santa Barbara	52	44	43	41	40
60834003	34.60	-120.63	California	Santa Barbara	54	47	46	45	44
60850002	37.00	-121.57	California	Santa Clara	58	53	50	46	42
60850005	37.35	-121.89	California	Santa Clara	56	52	49	45	41
60851001	37.23	-121.98	California	Santa Clara	59	54	51	47	43
60852006	37.08	-121.60	California	Santa Clara	62	57	53	49	45
60852009	37.32	-122.07	California	Santa Clara	56	52	49	45	41
60870007	36.98	-121.99	California	Santa Cruz	47	36	34	32	30
60890004	40.55	-122.38	California	Shasta	52	47	44	41	38
60890007	40.45	-122.30	California	Shasta	58	53	49	46	43
60890009	40.69	-122.40	California	Shasta	60	54	51	47	44
60893003	40.54	-121.57	California	Shasta	58	55	53	51	50
60950004	38.10	-122.24	California	Solano	53	49	46	43	39
60950005	38.23	-122.08	California	Solano	58	53	50	46	43
60953003	38.36	-121.95	California	Solano	58	53	50	46	43
60970003	38.44	-122.71	California	Sonoma	39	36	34	32	31
60990005	37.64	-120.99	California	Stanislaus	66	60	56	52	48
60990006	37.49	-120.84	California	Stanislaus	76	69	64	59	55
61010003	39.14	-121.62	California	Sutter	55	50	47	43	41
61010004	39.21	-121.82	California	Sutter	64	59	55	52	49
61030004	40.26	-122.09	California	Tehama	65	60	56	53	50
61030005	40.18	-122.24	California	Tehama	63	58	55	51	49
61070009	36.49	-118.83	California	Tulare	79	72	68	64	60
61072002	36.33	-119.29	California	Tulare	71	64	60	56	52
61072010	36.03	-119.06	California	Tulare	74	68	64	60	56

Site ID	Lat	long	State	County	Base Case	Baseline	70	65	60
61090005	37.98	-120.38	California	Tuolumne	62	58	54	51	48
61110007	34.21	-118.87	California	Ventura	64	49	47	44	42
61110009	34.40	-118.81	California	Ventura	66	51	49	46	44
61111004	34.45	-119.23	California	Ventura	67	56	55	54	52
61112002	34.28	-118.68	California	Ventura	72	55	52	50	46
61113001	34.25	-119.14	California	Ventura	55	44	43	42	40
61130004	38.53	-121.77	California	Yolo	57	52	49	45	42
61131003	38.66	-121.73	California	Yolo	60	54	51	47	43

*The design value from the monitor(s) with the highest projected ozone in each scenario is shown in bold blue text

Table 3A-8. Design Values for Southwest Monitors

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
40051008	35.21	-111.65	Arizona	Coconino	63	63	63	62	**60**
40070010	33.65	-111.11	Arizona	Gila	62	62	62	60	52
40130019	33.48	-112.14	Arizona	Maricopa	65	65	65	62	51
40131004	33.56	-112.07	Arizona	Maricopa	66	66	66	62	51
40131010	33.45	-111.73	Arizona	Maricopa	58	58	58	55	46
40132001	33.57	-112.19	Arizona	Maricopa	62	62	62	58	48
40132005	33.71	-111.86	Arizona	Maricopa	62	62	62	59	50
40133002	33.46	-112.05	Arizona	Maricopa	62	62	62	59	48
40133003	33.48	-111.92	Arizona	Maricopa	63	63	63	60	50
40134003	33.40	-112.08	Arizona	Maricopa	64	64	64	60	50
40134004	33.30	-111.88	Arizona	Maricopa	61	61	61	57	48
40134005	33.41	-111.93	Arizona	Maricopa	59	59	59	55	46
40134008	33.82	-112.02	Arizona	Maricopa	62	62	62	59	49
40134010	33.64	-112.34	Arizona	Maricopa	58	58	58	55	46
40134011	33.37	-112.62	Arizona	Maricopa	56	56	56	54	47
40137003	33.29	-112.16	Arizona	Maricopa	60	60	60	57	49
40137020	33.49	-111.86	Arizona	Maricopa	62	62	62	58	48
40137021	33.51	-111.76	Arizona	Maricopa	63	63	63	60	50
40137022	33.47	-111.81	Arizona	Maricopa	60	60	60	57	48
40137024	33.51	-111.84	Arizona	Maricopa	61	61	61	58	48
40139508	33.98	-111.80	Arizona	Maricopa	59	59	59	56	48
40139702	33.55	-111.61	Arizona	Maricopa	62	62	62	59	49
40139704	33.61	-111.73	Arizona	Maricopa	62	62	62	59	49
40139706	33.72	-111.67	Arizona	Maricopa	61	61	61	58	49
40139997	33.50	-112.10	Arizona	Maricopa	64	64	64	61	50
40170119	34.82	-109.89	Arizona	Navajo	61	61	61	58	55
40190021	32.17	-110.74	Arizona	Pima	62	62	62	57	51

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
40191011	32.20	-110.88	Arizona	Pima	57	57	57	53	47
40191018	32.43	-111.06	Arizona	Pima	58	58	58	56	50
40191020	32.05	-110.77	Arizona	Pima	60	60	60	55	48
40191028	32.30	-110.98	Arizona	Pima	57	57	57	54	48
40191030	31.88	-111.00	Arizona	Pima	60	60	60	55	50
40191032	32.17	-110.98	Arizona	Pima	56	56	56	52	46
40191034	32.38	-111.13	Arizona	Pima	56	56	56	53	48
40213001	33.42	-111.54	Arizona	Pinal	61	61	61	58	49
40213003	32.95	-111.76	Arizona	Pinal	59	59	59	57	51
40213007	32.51	-111.31	Arizona	Pinal	61	61	61	59	55
40217001	33.08	-111.74	Arizona	Pinal	60	60	60	58	51
40218001	33.29	-111.29	Arizona	Pinal	64	64	64	61	52
40258033	34.55	-112.48	Arizona	Yavapai	63	63	63	62	**60**
80013001	39.84	-104.95	Colorado	Adams	61	61	61	58	48
80050002	39.57	-104.96	Colorado	Arapahoe	66	66	66	63	53
80050006	39.64	-104.57	Colorado	Arapahoe	62	62	62	58	50
80130011	39.96	-105.24	Colorado	Boulder	61	61	61	58	48
80310014	39.75	-105.03	Colorado	Denver	58	58	58	55	46
80310025	39.70	-105.00	Colorado	Denver	58	58	58	55	46
80350004	39.53	-105.07	Colorado	Douglas	68	68	68	**65**	55
80410013	38.96	-104.82	Colorado	El Paso	65	65	65	63	58
80410016	38.85	-104.90	Colorado	El Paso	67	67	67	**65**	**60**
80450012	39.54	-107.78	Colorado	Garfield	62	62	62	60	55
80590002	39.80	-105.10	Colorado	Jefferson	58	58	58	55	46
80590005	39.64	-105.14	Colorado	Jefferson	64	64	64	61	51
80590006	39.91	-105.19	Colorado	Jefferson	67	67	67	63	53
80590011	39.74	-105.18	Colorado	Jefferson	66	66	66	63	52
80590013	39.54	-105.30	Colorado	Jefferson	62	62	62	59	49
80677001	37.14	-107.63	Colorado	La Plata	64	64	64	63	**60**
80677003	37.10	-107.87	Colorado	La Plata	63	63	63	62	59
80690007	40.28	-105.55	Colorado	Larimer	64	64	64	61	53
80690011	40.59	-105.14	Colorado	Larimer	**68**	**68**	**68**	64	54
80690012	40.64	-105.28	Colorado	Larimer	61	61	61	58	49
80691004	40.58	-105.08	Colorado	Larimer	60	60	60	57	47
80770020	39.13	-108.31	Colorado	Mesa	63	63	63	62	57
80810002	40.51	-107.89	Colorado	Moffat	61	61	61	59	55
80830006	37.35	-108.59	Colorado	Montezuma	61	61	61	60	57
80830101	37.20	-108.49	Colorado	Montezuma	60	60	60	59	55
81030005	40.04	-107.85	Colorado	Rio Blanco	60	60	60	58	55
81230009	40.39	-104.74	Colorado	Weld	67	67	67	64	54

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
320010002	39.47	-118.78	Nevada	Churchill	53	53	53	53	52
320030022	36.39	-114.91	Nevada	Clark	61	61	61	59	55
320030023	36.81	-114.06	Nevada	Clark	57	57	57	56	55
320030043	36.11	-115.25	Nevada	Clark	67	67	67	**65**	57
320030071	36.17	-115.26	Nevada	Clark	66	66	66	64	57
320030073	36.17	-115.33	Nevada	Clark	66	66	66	64	57
320030075	36.27	-115.24	Nevada	Clark	65	65	65	63	56
320030538	36.14	-115.06	Nevada	Clark	61	61	61	60	53
320030540	36.14	-115.08	Nevada	Clark	61	61	61	60	53
320030601	35.98	-114.85	Nevada	Clark	65	65	65	64	**60**
320031019	35.79	-115.36	Nevada	Clark	66	66	66	64	**60**
320032002	36.19	-115.12	Nevada	Clark	61	61	61	60	53
320190006	39.60	-119.25	Nevada	Lyon	61	61	61	60	59
320310016	39.53	-119.81	Nevada	Washoe	60	60	60	59	58
320310020	39.47	-119.78	Nevada	Washoe	61	61	61	60	59
320310025	39.40	-119.74	Nevada	Washoe	60	60	60	60	58
320311005	39.54	-119.75	Nevada	Washoe	61	61	61	60	59
320312002	39.25	-119.96	Nevada	Washoe	55	55	55	55	55
320312009	39.65	-119.84	Nevada	Washoe	61	61	61	60	58
325100002	39.17	-119.73	Nevada	Carson City	61	61	61	60	60
350010023	35.13	-106.59	New Mexico	Bernalillo	60	60	60	58	54
350010024	35.06	-106.58	New Mexico	Bernalillo	61	61	61	59	55
350010027	35.15	-106.70	New Mexico	Bernalillo	64	64	64	62	59
350010029	35.02	-106.66	New Mexico	Bernalillo	61	61	61	60	55
350010032	35.06	-106.76	New Mexico	Bernalillo	58	58	58	57	52
350011012	35.19	-106.51	New Mexico	Bernalillo	64	64	64	63	59
350011013	35.19	-106.61	New Mexico	Bernalillo	61	61	61	59	55
350130008	31.93	-106.63	New Mexico	Dona Ana	61	61	61	60	59
350130023	32.32	-106.77	New Mexico	Dona Ana	60	60	60	59	58
350171003	32.69	-108.12	New Mexico	Grant	61	61	61	61	59
350250008	32.73	-103.12	New Mexico	Lea	60	60	60	60	58
350290003	32.26	-107.72	New Mexico	Luna	59	59	59	58	57

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
350431001	35.30	-106.55	New Mexico	Sandoval	56	56	56	55	52
350439004	35.62	-106.72	New Mexico	Sandoval	59	59	59	59	57
350450009	36.74	-107.98	New Mexico	San Juan	58	58	58	57	52
350450018	36.81	-107.65	New Mexico	San Juan	64	64	64	62	57
350451005	36.80	-108.47	New Mexico	San Juan	56	56	56	54	50
350451233	36.81	-108.70	New Mexico	San Juan	55	55	55	53	48
350490021	35.62	-106.08	New Mexico	Santa Fe	60	60	60	60	58
350610008	34.81	-106.74	New Mexico	Valencia	58	58	58	57	52
490030003	41.49	-112.02	Utah	Box Elder	59	59	59	57	49
490037001	41.95	-112.23	Utah	Box Elder	60	60	60	59	55
490050004	41.73	-111.84	Utah	Cache	59	59	59	57	54
490071003	39.61	-110.80	Utah	Carbon	64	64	64	60	56
490110004	40.90	-111.88	Utah	Davis	61	61	61	58	52
490131001	40.21	-110.84	Utah	Duchesne	63	63	63	62	59
490352004	40.74	-112.21	Utah	Salt Lake	65	65	65	62	54
490353006	40.74	-111.87	Utah	Salt Lake	65	65	65	61	53
490450003	40.54	-112.30	Utah	Tooele	64	64	64	61	53
490490002	40.25	-111.66	Utah	Utah	64	64	64	62	57
490495008	40.43	-111.80	Utah	Utah	59	59	59	57	53
490495010	40.14	-111.66	Utah	Utah	63	63	63	61	57
490570002	41.21	-111.98	Utah	Weber	64	64	64	61	54
490571003	41.30	-111.99	Utah	Weber	64	64	64	61	53
560050123	44.65	-105.29	Wyoming	Campbell	60	60	60	58	53
560050456	44.15	-105.53	Wyoming	Campbell	60	60	60	58	53
560070100	41.39	-107.62	Wyoming	Carbon	60	60	60	58	56
560130232	43.08	-107.55	Wyoming	Fremont	61	61	61	60	58
560210100	41.18	-104.78	Wyoming	Laramie	61	61	61	59	54
560350700	42.49	-110.10	Wyoming	Sublette	60	60	60	59	58
560370077	41.16	-108.62	Wyoming	Sweetwater	60	60	60	58	55
560370200	41.68	-108.02	Wyoming	Sweetwater	60	60	60	57	53
560370300	41.75	-109.79	Wyoming	Sweetwater	61	61	61	60	56
560410101	41.37	-111.04	Wyoming	Uinta	58	58	58	57	54

*The design value from the monitor(s) with the highest projected ozone in each scenario is shown in bold blue text

Table 3A-9. Design Values for Central Region Monitors

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
50199991	34.18	-93.10	Arkansas	Clark	56	54	52	48	44
50350005	35.20	-90.19	Arkansas	Crittenden	64	63	63	61	60
51010002	35.83	-93.21	Arkansas	Newton	57	55	54	49	46
51130003	34.45	-94.14	Arkansas	Polk	65	63	60	56	53
51190007	34.76	-92.28	Arkansas	Pulaski	55	52	51	44	39
51191002	34.84	-92.26	Arkansas	Pulaski	58	55	53	47	42
51191008	34.68	-92.33	Arkansas	Pulaski	57	54	52	46	42
51430005	36.18	-94.12	Arkansas	Washington	62	60	58	54	50
200910010	38.84	-94.75	Kansas	Johnson	60	59	55	51	46
201030003	39.33	-94.95	Kansas	Leavenworth	59	58	54	50	44
201070002	38.14	-94.73	Kansas	Linn	60	58	55	51	47
201619991	39.10	-96.61	Kansas	Riley	63	62	60	57	55
201730001	37.78	-97.34	Kansas	Sedgwick	55	54	52	49	46
201730010	37.70	-97.31	Kansas	Sedgwick	64	63	61	57	53
201730018	37.90	-97.49	Kansas	Sedgwick	63	62	59	55	51
201770013	39.02	-95.71	Kansas	Shawnee	63	62	59	57	53
201910002	37.48	-97.37	Kansas	Sumner	66	65	63	59	55
201950001	38.77	-99.76	Kansas	Trego	67	67	65	62	60
202090021	39.12	-94.64	Kansas	Wyandotte	56	55	51	47	42
220050004	30.23	-90.97	Louisiana	Ascension	63	62	58	54	49
220150008	32.54	-93.75	Louisiana	Bossier	69	67	61	55	49
220170001	32.68	-93.86	Louisiana	Caddo	67	64	59	53	47
220190002	30.14	-93.37	Louisiana	Calcasieu	67	67	64	61	57
220190008	30.26	-93.28	Louisiana	Calcasieu	61	61	57	55	51
220190009	30.23	-93.58	Louisiana	Calcasieu	66	64	61	57	53
220330003	30.42	-91.18	Louisiana	East Baton Rouge	67	67	63	58	53
220330009	30.46	-91.18	Louisiana	East Baton Rouge	64	63	59	56	51
220330013	30.70	-91.06	Louisiana	East Baton Rouge	59	59	55	52	48
220470009	30.22	-91.32	Louisiana	Iberville	63	62	58	54	49
220470012	30.21	-91.13	Louisiana	Iberville	65	65	61	57	52
220511001	30.04	-90.28	Louisiana	Jefferson	63	62	59	55	50
220550007	30.22	-92.05	Louisiana	Lafayette	60	59	57	54	50
220570004	29.76	-90.77	Louisiana	Lafourche	61	61	57	52	47
220630002	30.31	-90.81	Louisiana	Livingston	62	61	57	53	49
220710012	29.99	-90.10	Louisiana	Orleans	60	58	55	51	47
220730004	32.51	-92.05	Louisiana	Ouachita	59	59	54	51	46
220770001	30.68	-91.37	Louisiana	Pointe Coupee	62	61	58	54	50
220870004	29.94	-89.92	Louisiana	St. Bernard	61	59	57	53	50
220890003	29.98	-90.41	Louisiana	St. Charles	59	58	55	51	47

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
220930002	29.99	-90.82	Louisiana	St. James	58	57	53	49	45
220950002	30.06	-90.61	Louisiana	St. John the Baptist	62	61	57	53	48
221030002	30.43	-90.20	Louisiana	St. Tammany	63	62	59	56	52
221210001	30.50	-91.21	Louisiana	West Baton Rouge	60	59	55	52	47
280010004	31.56	-91.39	Mississippi	Adams	56	55	53	51	49
280110001	33.75	-90.72	Mississippi	Bolivar	63	62	60	57	53
280330002	34.82	-89.99	Mississippi	DeSoto	58	57	57	54	52
280450003	30.30	-89.40	Mississippi	Hancock	54	51	51	47	44
280470008	30.39	-89.05	Mississippi	Harrison	58	53	55	47	44
280490010	32.39	-90.14	Mississippi	Hinds	50	49	47	44	41
280590006	30.38	-88.53	Mississippi	Jackson	57	55	55	51	48
280750003	32.36	-88.73	Mississippi	Lauderdale	52	51	51	48	47
280810005	34.26	-88.77	Mississippi	Lee	51	51	50	49	48
281619991	34.00	-89.80	Mississippi	Yalobusha	53	52	51	49	48
290030001	39.95	-94.85	Missouri	Andrew	60	59	55	51	46
290190011	39.08	-92.32	Missouri	Boone	57	56	53	50	46
290270002	38.71	-92.09	Missouri	Callaway	55	55	52	49	46
290370003	38.76	-94.58	Missouri	Cass	58	57	53	50	45
290390001	37.69	-94.04	Missouri	Cedar	63	61	58	54	50
290470003	39.41	-94.27	Missouri	Clay	63	62	58	53	48
290470005	39.30	-94.38	Missouri	Clay	62	61	57	52	47
290470006	39.33	-94.58	Missouri	Clay	64	63	58	54	48
290490001	39.53	-94.56	Missouri	Clinton	64	63	58	54	48
290770036	37.26	-93.30	Missouri	Greene	57	56	53	49	46
290770042	37.32	-93.20	Missouri	Greene	59	58	55	51	47
290970004	37.24	-94.42	Missouri	Jasper	66	62	60	54	49
290990019	38.45	-90.40	Missouri	Jefferson	64	64	60	57	52
291130003	39.04	-90.86	Missouri	Lincoln	63	62	59	56	53
291370001	39.48	-91.79	Missouri	Monroe	58	58	55	53	50
291570001	37.70	-89.70	Missouri	Perry	61	62	59	58	56
291831002	38.87	-90.23	Missouri	Saint Charles	68	67	63	58	53
291831004	38.90	-90.45	Missouri	Saint Charles	65	64	61	57	53
291860005	37.90	-90.42	Missouri	Sainte Genevieve	60	60	57	54	50
291890005	38.49	-90.71	Missouri	Saint Louis	59	58	55	51	47
291890014	38.71	-90.48	Missouri	Saint Louis	66	65	61	58	53
292130004	36.71	-93.22	Missouri	Taney	58	56	54	50	46
295100085	38.66	-90.20	Missouri	St. Louis City	64	63	59	55	49
400019009	35.75	-94.67	Oklahoma	Adair	66	62	61	55	51
400159008	35.11	-98.25	Oklahoma	Caddo	64	62	60	55	50
400170101	35.48	-97.75	Oklahoma	Canadian	62	62	58	54	49

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
400219002	35.85	-94.99	Oklahoma	Cherokee	66	61	61	54	49
400270049	35.32	-97.48	Oklahoma	Cleveland	64	62	59	55	50
400310651	34.63	-98.43	Oklahoma	Comanche	66	65	62	59	55
400370144	36.11	-96.36	Oklahoma	Creek	65	62	60	54	48
400430860	36.16	-98.93	Oklahoma	Dewey	66	65	63	60	56
400719010	36.96	-97.03	Oklahoma	Kay	64	62	60	56	53
400871073	35.16	-97.47	Oklahoma	McClain	63	61	58	54	50
400892001	34.48	-94.66	Oklahoma	McCurtain	62	60	59	55	52
400979014	36.23	-95.25	Oklahoma	Mayes	68	63	63	55	50
401090033	35.48	-97.49	Oklahoma	Oklahoma	66	65	61	58	53
401090096	35.48	-97.30	Oklahoma	Oklahoma	65	64	60	57	52
401091037	35.61	-97.48	Oklahoma	Oklahoma	67	66	62	59	54
401159004	36.92	-94.84	Oklahoma	Ottawa	64	61	60	54	50
401210415	34.90	-95.78	Oklahoma	Pittsburg	65	63	60	55	50
401359021	35.41	-94.52	Oklahoma	Sequoyah	63	61	59	54	50
401430137	36.36	-96.00	Oklahoma	Tulsa	67	64	62	55	50
401430174	35.95	-96.00	Oklahoma	Tulsa	65	61	60	52	46
401430178	36.13	-95.76	Oklahoma	Tulsa	66	63	60	54	48
401431127	36.20	-95.98	Oklahoma	Tulsa	67	64	62	55	49
480271047	31.09	-97.68	Texas	Bell	64	62	60	57	54
480290032	29.52	-98.62	Texas	Bexar	69	68	65	62	58
480290052	29.63	-98.56	Texas	Bexar	71	69	66	63	59
480290059	29.28	-98.31	Texas	Bexar	62	60	58	54	51
480391004	29.52	-95.39	Texas	Brazoria	**77**	**75**	**70**	**65**	**60**
480391016	29.04	-95.47	Texas	Brazoria	65	63	61	57	54
480610006	25.89	-97.49	Texas	Cameron	58	57	56	54	52
480850005	33.13	-96.79	Texas	Collin	72	70	65	61	56
481130069	32.82	-96.86	Texas	Dallas	71	70	65	61	56
481130075	32.92	-96.81	Texas	Dallas	72	71	66	62	57
481130087	32.68	-96.87	Texas	Dallas	71	69	65	60	55
481210034	33.22	-97.20	Texas	Denton	74	72	68	63	58
481211032	33.41	-96.94	Texas	Denton	72	70	66	61	56
481390016	32.48	-97.03	Texas	Ellis	67	66	62	58	54
481391044	32.18	-96.87	Texas	Ellis	63	60	57	53	50
481410029	31.79	-106.32	Texas	El Paso	58	58	58	57	56
481410055	31.75	-106.40	Texas	El Paso	63	63	62	61	**60**
481410057	31.67	-106.29	Texas	El Paso	63	62	62	61	**60**
481671034	29.25	-94.86	Texas	Galveston	70	69	66	62	59
481830001	32.38	-94.71	Texas	Gregg	73	67	62	54	48
482010024	29.90	-95.33	Texas	Harris	74	73	68	64	59

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
482010026	29.80	-95.13	Texas	Harris	71	69	65	61	56
482010029	30.04	-95.67	Texas	Harris	72	71	67	63	59
482010046	29.83	-95.28	Texas	Harris	70	69	64	60	56
482010047	29.83	-95.49	Texas	Harris	70	69	64	60	55
482010051	29.62	-95.47	Texas	Harris	71	70	65	60	55
482010055	29.70	-95.50	Texas	Harris	72	71	66	61	56
482010062	29.63	-95.27	Texas	Harris	70	69	64	59	54
482010066	29.72	-95.50	Texas	Harris	69	68	63	59	54
482010070	29.74	-95.32	Texas	Harris	69	68	63	59	54
482010075	29.75	-95.35	Texas	Harris	71	69	64	60	55
482010416	29.69	-95.29	Texas	Harris	70	69	64	59	54
482011015	29.76	-95.08	Texas	Harris	68	67	63	59	54
482011034	29.77	-95.22	Texas	Harris	75	73	68	63	58
482011035	29.73	-95.26	Texas	Harris	72	71	66	61	56
482011039	29.67	-95.13	Texas	Harris	76	74	**70**	**65**	**60**
482011050	29.58	-95.02	Texas	Harris	72	71	67	63	59
482030002	32.67	-94.17	Texas	Harrison	67	63	58	52	47
482150043	26.23	-98.29	Texas	Hidalgo	56	55	54	52	51
482151048	26.13	-97.94	Texas	Hidalgo	55	55	53	52	50
482210001	32.44	-97.80	Texas	Hood	67	66	62	58	54
482311006	33.15	-96.12	Texas	Hunt	62	61	57	54	50
482450009	30.04	-94.07	Texas	Jefferson	66	64	61	56	52
482450011	29.90	-93.99	Texas	Jefferson	66	65	61	57	53
482450022	29.86	-94.32	Texas	Jefferson	64	62	59	55	51
482450101	29.73	-93.89	Texas	Jefferson	70	69	66	62	58
482450102	29.94	-94.00	Texas	Jefferson	63	62	58	54	50
482450628	29.87	-93.96	Texas	Jefferson	65	63	60	56	52
482451035	29.98	-94.01	Texas	Jefferson	65	63	59	55	51
482510003	32.35	-97.44	Texas	Johnson	70	68	64	60	56
482570005	32.56	-96.32	Texas	Kaufman	64	61	58	53	50
483091037	31.65	-97.07	Texas	McLennan	65	63	60	55	52
483390078	30.35	-95.43	Texas	Montgomery	68	67	63	59	55
483491051	32.03	-96.40	Texas	Navarro	64	61	58	53	50
483550025	27.77	-97.43	Texas	Nueces	66	64	62	59	56
483550026	27.83	-97.56	Texas	Nueces	66	64	62	58	55
483611001	30.09	-93.76	Texas	Orange	66	64	61	57	52
483611100	30.19	-93.87	Texas	Orange	63	60	58	53	49
483670081	32.87	-97.91	Texas	Parker	70	68	64	61	57
483739991	30.70	-94.67	Texas	Polk	62	61	59	56	54
483970001	32.94	-96.46	Texas	Rockwall	68	66	62	59	54

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
484230007	32.34	-95.42	Texas	Smith	67	64	61	56	52
484390075	32.99	-97.48	Texas	Tarrant	73	71	66	62	57
484391002	32.81	-97.36	Texas	Tarrant	71	70	65	61	56
484392003	32.92	-97.28	Texas	Tarrant	**77**	**75**	**70**	**65**	**60**
484393009	32.98	-97.06	Texas	Tarrant	75	73	69	64	59
484393011	32.66	-97.09	Texas	Tarrant	72	70	66	61	57
484530014	30.35	-97.76	Texas	Travis	65	64	61	57	54
484530020	30.48	-97.87	Texas	Travis	63	61	59	55	52
484690003	28.84	-97.01	Texas	Victoria	63	60	57	53	50
484790016	27.51	-99.52	Texas	Webb	60	59	57	56	54

*The design value from the monitor(s) with the highest projected ozone in each scenario is shown in bold blue text

Table 3A-10. Design Values for Midwest Monitors

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
170010007	39.92	-91.34	Illinois	Adams	57	56	56	55	54
170190007	40.24	-88.19	Illinois	Champaign	58	58	58	55	53
170191001	40.05	-88.37	Illinois	Champaign	60	59	59	57	54
170230001	39.21	-87.67	Illinois	Clark	58	58	58	54	50
170310001	41.67	-87.73	Illinois	Cook	63	62	62	58	54
170310032	41.76	-87.55	Illinois	Cook	60	59	59	60	60
170310064	41.79	-87.60	Illinois	Cook	55	54	54	55	55
170310076	41.75	-87.71	Illinois	Cook	63	62	62	58	54
170311003	41.98	-87.79	Illinois	Cook	52	51	51	53	54
170311601	41.67	-87.99	Illinois	Cook	63	63	63	59	54
170314002	41.86	-87.75	Illinois	Cook	57	56	56	56	56
170314007	42.06	-87.86	Illinois	Cook	49	49	49	50	51
170314201	42.14	-87.80	Illinois	Cook	56	56	56	58	59
170317002	42.06	-87.67	Illinois	Cook	54	54	54	57	59
170436001	41.81	-88.07	Illinois	DuPage	59	58	58	54	50
170491001	39.07	-88.55	Illinois	Effingham	57	57	57	54	51
170650002	38.08	-88.62	Illinois	Hamilton	62	63	63	60	56
170831001	39.11	-90.32	Illinois	Jersey	62	62	62	61	60
170859991	42.29	-90.00	Illinois	Jo Daviess	58	57	57	56	55
170890005	42.05	-88.27	Illinois	Kane	61	60	60	56	52
170971007	42.47	-87.81	Illinois	Lake	58	58	58	59	**60**
171110001	42.22	-88.24	Illinois	McHenry	60	60	60	55	51
171132003	40.52	-89.00	Illinois	McLean	58	56	56	53	51
171150013	39.87	-88.93	Illinois	Macon	59	58	58	56	53

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
171170002	39.40	-89.81	Illinois	Macoupin	57	56	56	55	54
171190008	38.89	-90.15	Illinois	Madison	64	63	63	62	49
171191009	38.73	-89.96	Illinois	Madison	62	61	61	60	58
171193007	38.86	-90.11	Illinois	Madison	63	62	62	61	49
171199991	38.87	-89.62	Illinois	Madison	60	59	59	58	57
171430024	40.69	-89.61	Illinois	Peoria	54	51	51	48	45
171431001	40.75	-89.59	Illinois	Peoria	61	58	58	55	52
171570001	38.18	-89.79	Illinois	Randolph	58	57	57	55	53
171613002	41.51	-90.52	Illinois	Rock Island	49	48	48	47	45
171630010	38.61	-90.16	Illinois	Saint Clair	62	61	61	60	59
171670014	39.83	-89.64	Illinois	Sangamon	58	57	57	56	55
171971011	41.22	-88.19	Illinois	Will	55	54	54	50	46
172012001	42.33	-89.04	Illinois	Winnebago	57	56	56	53	50
180030002	41.22	-85.02	Indiana	Allen	56	56	56	53	49
180030004	41.09	-85.10	Indiana	Allen	57	56	56	53	50
180110001	40.00	-86.40	Indiana	Boone	60	60	60	56	51
180150002	40.54	-86.55	Indiana	Carroll	58	57	57	54	50
180190008	38.39	-85.66	Indiana	Clark	65	65	65	59	53
180350010	40.30	-85.25	Indiana	Delaware	55	54	54	51	47
180390007	41.72	-85.83	Indiana	Elkhart	56	55	55	51	48
180431004	38.31	-85.83	Indiana	Floyd	63	63	63	58	52
180550001	38.99	-86.99	Indiana	Greene	**68**	67	67	62	57
180570006	40.07	-85.99	Indiana	Hamilton	57	57	57	53	49
180590003	39.94	-85.84	Indiana	Hancock	53	53	53	49	45
180630004	39.76	-86.40	Indiana	Hendricks	56	55	55	52	48
180690002	40.96	-85.38	Indiana	Huntington	54	54	54	51	48
180710001	38.92	-86.08	Indiana	Jackson	57	57	57	52	47
180810002	39.42	-86.15	Indiana	Johnson	57	57	57	53	48
180839991	38.74	-87.49	Indiana	Knox	65	65	65	60	55
180890022	41.61	-87.30	Indiana	Lake	54	54	54	52	50
180890030	41.68	-87.49	Indiana	Lake	58	58	58	56	54
180892008	41.64	-87.49	Indiana	Lake	58	58	58	56	54
180910005	41.72	-86.91	Indiana	LaPorte	66	66	66	62	59
180910010	41.63	-86.68	Indiana	LaPorte	59	59	59	56	52
180950010	40.00	-85.66	Indiana	Madison	54	54	54	50	46
180970050	39.86	-86.02	Indiana	Marion	60	60	60	56	51
180970057	39.75	-86.19	Indiana	Marion	59	58	58	54	50
180970073	39.79	-86.06	Indiana	Marion	59	59	59	55	50
180970078	39.81	-86.11	Indiana	Marion	59	59	59	55	50
181090005	39.58	-86.48	Indiana	Morgan	56	56	56	52	47

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
181230009	38.11	-86.60	Indiana	Perry	65	65	65	60	54
181270024	41.62	-87.20	Indiana	Porter	56	56	56	54	52
181270026	41.51	-87.04	Indiana	Porter	54	54	54	51	47
181290003	38.01	-87.72	Indiana	Posey	62	62	62	58	53
181410010	41.55	-86.37	Indiana	St. Joseph	52	51	51	48	45
181410015	41.70	-86.21	Indiana	St. Joseph	58	57	57	53	49
181411007	41.74	-86.11	Indiana	St. Joseph	53	53	53	49	45
181450001	39.61	-85.87	Indiana	Shelby	60	60	60	56	51
181630013	38.11	-87.54	Indiana	Vanderburgh	64	63	63	59	54
181630021	38.01	-87.58	Indiana	Vanderburgh	63	63	63	59	54
181670018	39.49	-87.40	Indiana	Vigo	55	55	55	51	47
181670024	39.56	-87.31	Indiana	Vigo	55	55	55	51	47
181699991	40.82	-85.66	Indiana	Wabash	61	60	60	57	53
181730008	38.05	-87.28	Indiana	Warrick	63	63	63	59	54
181730009	38.19	-87.34	Indiana	Warrick	63	62	62	58	53
181730011	37.95	-87.32	Indiana	Warrick	64	64	64	59	54
210130002	36.61	-83.74	Kentucky	Bell	52	52	52	49	46
210150003	38.92	-84.85	Kentucky	Boone	56	56	56	51	46
210190017	38.46	-82.64	Kentucky	Boyd	60	60	60	54	49
210290006	37.99	-85.71	Kentucky	Bullitt	60	60	60	56	51
210373002	39.02	-84.47	Kentucky	Campbell	64	64	64	58	52
210430500	38.24	-82.99	Kentucky	Carter	57	56	56	52	47
210470006	36.91	-87.32	Kentucky	Christian	62	61	61	58	55
210590005	37.78	-87.08	Kentucky	Daviess	69	68	68	64	58
210610501	37.13	-86.15	Kentucky	Edmonson	57	57	57	54	50
210670012	38.07	-84.50	Kentucky	Fayette	58	58	58	54	50
210890007	38.55	-82.73	Kentucky	Greenup	60	60	60	54	49
210910012	37.94	-86.90	Kentucky	Hancock	65	65	65	60	54
210930006	37.71	-85.85	Kentucky	Hardin	58	58	58	53	49
211010014	37.87	-87.46	Kentucky	Henderson	68	68	68	63	58
211110027	38.14	-85.58	Kentucky	Jefferson	65	65	65	60	54
211110051	38.06	-85.90	Kentucky	Jefferson	68	68	68	62	57
211110067	38.23	-85.65	Kentucky	Jefferson	70	**70**	**70**	64	58
211130001	37.89	-84.59	Kentucky	Jessamine	57	59	59	55	51
211390003	37.16	-88.39	Kentucky	Livingston	58	62	62	58	54
211451024	37.06	-88.57	Kentucky	McCracken	58	63	63	60	57
211759991	37.92	-83.07	Kentucky	Morgan	60	60	60	55	50
211850004	38.40	-85.44	Kentucky	Oldham	66	66	66	61	55
211930003	37.28	-83.21	Kentucky	Perry	62	62	62	57	52
211950002	37.48	-82.54	Kentucky	Pike	63	63	63	57	52

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
211990003	37.10	-84.61	Kentucky	Pulaski	53	53	53	49	45
212130004	36.71	-86.57	Kentucky	Simpson	54	53	53	50	46
212218001	36.78	-87.85	Kentucky	Trigg	57	57	57	53	49
212219991	36.78	-87.85	Kentucky	Trigg	58	58	58	54	50
212270008	37.04	-86.25	Kentucky	Warren	50	50	50	47	44
212299991	37.70	-85.05	Kentucky	Washington	56	57	57	53	48
260050003	42.77	-86.15	Michigan	Allegan	69	69	69	64	58
260190003	44.62	-86.11	Michigan	Benzie	61	61	61	56	52
260210014	42.20	-86.31	Michigan	Berrien	68	68	68	63	58
260270003	41.90	-86.00	Michigan	Cass	63	62	62	58	54
260370001	42.80	-84.39	Michigan	Clinton	57	56	56	52	49
260490021	43.05	-83.67	Michigan	Genesee	61	60	60	57	53
260492001	43.17	-83.46	Michigan	Genesee	60	59	59	55	52
260630007	43.84	-82.64	Michigan	Huron	61	61	61	57	54
260650012	42.74	-84.53	Michigan	Ingham	57	56	56	53	49
260770008	42.28	-85.54	Michigan	Kalamazoo	60	59	59	56	52
260810020	42.98	-85.67	Michigan	Kent	60	60	60	56	51
260810022	43.18	-85.42	Michigan	Kent	59	58	58	54	50
260910007	42.00	-83.95	Michigan	Lenawee	61	60	60	57	53
260990009	42.73	-82.79	Michigan	Macomb	66	65	65	61	57
260991003	42.51	-83.01	Michigan	Macomb	68	68	68	64	**60**
261010922	44.31	-86.24	Michigan	Manistee	60	60	60	55	51
261050007	43.95	-86.29	Michigan	Mason	61	60	60	56	52
261130001	44.31	-84.89	Michigan	Missaukee	58	57	57	54	51
261210039	43.28	-86.31	Michigan	Muskegon	66	65	65	60	55
261250001	42.46	-83.18	Michigan	Oakland	66	66	66	62	57
261390005	42.89	-85.85	Michigan	Ottawa	63	62	62	58	53
261470005	42.95	-82.46	Michigan	St. Clair	64	63	63	60	56
261530001	46.29	-85.95	Michigan	Schoolcraft	60	60	60	56	52
261579991	43.61	-83.36	Michigan	Tuscola	58	57	57	54	50
261610008	42.24	-83.60	Michigan	Washtenaw	62	62	62	58	54
261619991	42.42	-83.90	Michigan	Washtenaw	61	60	60	56	53
261630001	42.23	-83.21	Michigan	Wayne	61	61	61	58	55
261630019	42.43	-83.00	Michigan	Wayne	69	68	68	64	60
261659991	44.18	-85.74	Michigan	Wexford	56	55	55	52	49
390030009	40.77	-84.05	Ohio	Allen	61	61	61	57	53
390071001	41.96	-80.57	Ohio	Ashtabula	62	62	62	57	52
390090004	39.31	-82.12	Ohio	Athens	57	57	57	53	48
390170004	39.38	-84.54	Ohio	Butler	66	65	65	60	55
390170018	39.53	-84.39	Ohio	Butler	66	65	65	60	54

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
390179991	39.53	-84.73	Ohio	Butler	64	63	63	59	54
390230001	40.00	-83.80	Ohio	Clark	61	60	60	56	51
390230003	39.86	-84.00	Ohio	Clark	60	60	60	55	50
390250022	39.08	-84.14	Ohio	Clermont	62	62	62	57	51
390271002	39.43	-83.79	Ohio	Clinton	62	61	61	56	51
390350034	41.56	-81.58	Ohio	Cuyahoga	59	58	58	58	57
390350060	41.49	-81.68	Ohio	Cuyahoga	52	51	51	51	51
390350064	41.36	-81.86	Ohio	Cuyahoga	56	56	56	55	54
390355002	41.54	-81.46	Ohio	Cuyahoga	58	58	58	57	57
390410002	40.36	-83.06	Ohio	Delaware	59	59	59	55	51
390479991	39.64	-83.26	Ohio	Fayette	57	57	57	52	48
390490029	40.08	-82.82	Ohio	Franklin	66	66	66	61	56
390490037	39.97	-82.96	Ohio	Franklin	61	61	61	56	52
390490081	40.09	-82.96	Ohio	Franklin	58	58	58	54	49
390550004	41.52	-81.25	Ohio	Geauga	60	60	60	56	52
390570006	39.67	-83.94	Ohio	Greene	58	57	57	53	48
390610006	39.28	-84.37	Ohio	Hamilton	68	68	68	62	56
390610010	39.21	-84.69	Ohio	Hamilton	64	64	64	58	53
390610040	39.13	-84.50	Ohio	Hamilton	66	66	66	60	54
390810017	40.37	-80.62	Ohio	Jefferson	61	60	60	57	53
390830002	40.31	-82.69	Ohio	Knox	59	59	59	55	51
390850003	41.67	-81.42	Ohio	Lake	59	59	59	59	58
390850007	41.73	-81.24	Ohio	Lake	53	53	53	52	52
390870011	38.63	-82.46	Ohio	Lawrence	55	55	55	50	45
390870012	38.51	-82.66	Ohio	Lawrence	60	60	60	54	49
390890005	40.03	-82.43	Ohio	Licking	59	58	58	54	49
390930018	41.42	-82.10	Ohio	Lorain	54	53	53	53	53
390950024	41.64	-83.55	Ohio	Lucas	56	55	55	54	53
390950027	41.49	-83.72	Ohio	Lucas	58	58	58	54	51
390950034	41.68	-83.31	Ohio	Lucas	61	61	61	59	56
390970007	39.79	-83.48	Ohio	Madison	59	58	58	54	49
390990013	41.10	-80.66	Ohio	Mahoning	57	57	57	53	50
391030004	41.06	-81.92	Ohio	Medina	57	57	57	53	49
391090005	40.08	-84.11	Ohio	Miami	59	59	59	54	50
391130037	39.79	-84.13	Ohio	Montgomery	62	62	62	57	52
391219991	39.94	-81.34	Ohio	Noble	52	51	51	48	44
391331001	41.18	-81.33	Ohio	Portage	56	56	56	52	48
391351001	39.84	-84.72	Ohio	Preble	59	58	58	55	51
391510016	40.83	-81.38	Ohio	Stark	61	61	61	57	52
391510022	40.71	-81.60	Ohio	Stark	58	58	58	53	49

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
391514005	40.93	-81.12	Ohio	Stark	59	58	58	54	50
391530020	41.11	-81.50	Ohio	Summit	59	59	59	55	50
391550009	41.45	-80.59	Ohio	Trumbull	57	56	56	53	49
391550011	41.24	-80.66	Ohio	Trumbull	61	61	61	57	53
391650007	39.43	-84.20	Ohio	Warren	62	62	62	57	51
391670004	39.43	-81.46	Ohio	Washington	60	59	59	55	50
391730003	41.38	-83.61	Ohio	Wood	60	60	60	56	52
470010101	35.97	-84.22	Tennessee	Anderson	56	56	56	52	47
470090101	35.63	-83.94	Tennessee	Blount	61	61	61	56	51
470090102	35.60	-83.78	Tennessee	Blount	53	53	53	49	44
470259991	36.47	-83.83	Tennessee	Claiborne	49	48	48	45	42
470370011	36.21	-86.74	Tennessee	Davidson	51	51	51	47	43
470370026	36.15	-86.62	Tennessee	Davidson	54	54	54	50	46
470419991	36.04	-85.73	Tennessee	DeKalb	55	55	55	51	48
470651011	35.23	-85.18	Tennessee	Hamilton	57	56	56	53	49
470654003	35.10	-85.16	Tennessee	Hamilton	57	56	56	53	49
470890002	36.11	-83.60	Tennessee	Jefferson	59	59	59	54	50
470930021	36.09	-83.76	Tennessee	Knox	54	54	54	50	45
470931020	36.02	-83.87	Tennessee	Knox	56	56	56	52	47
471050109	35.72	-84.34	Tennessee	Loudon	59	58	58	54	49
471210104	35.29	-84.95	Tennessee	Meigs	56	56	56	53	50
471490101	35.73	-86.60	Tennessee	Rutherford	52	52	52	48	44
471550101	35.70	-83.61	Tennessee	Sevier	59	58	58	55	51
471550102	35.56	-83.50	Tennessee	Sevier	58	58	58	56	53
471570021	35.22	-90.02	Tennessee	Shelby	63	62	62	57	53
471570075	35.15	-89.85	Tennessee	Shelby	63	63	63	58	54
471571004	35.38	-89.83	Tennessee	Shelby	61	59	59	56	52
471632002	36.54	-82.42	Tennessee	Sullivan	62	62	62	57	52
471632003	36.58	-82.49	Tennessee	Sullivan	61	61	61	56	52
471650007	36.30	-86.65	Tennessee	Sumner	59	58	58	54	50
471650101	36.45	-86.56	Tennessee	Sumner	55	54	54	51	47
471870106	35.95	-87.14	Tennessee	Williamson	54	54	54	49	45
471890103	36.06	-86.29	Tennessee	Wilson	55	55	55	51	47
540030003	39.45	-77.96	West Virginia	Berkeley	56	55	55	53	50
540110006	38.42	-82.43	West Virginia	Cabell	60	59	59	54	49
540219991	38.88	-80.85	West Virginia	Gilmer	53	53	53	48	43
540250003	37.91	-80.63	West Virginia	Greenbrier	55	55	55	51	47
540291004	40.42	-80.58	West Virginia	Hancock	63	63	63	59	56
540390010	38.35	-81.63	West Virginia	Kanawha	63	63	63	57	51
540610003	39.65	-79.92	West Virginia	Monongalia	63	62	62	57	52

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
540690010	40.11	-80.70	West Virginia	Ohio	61	60	60	56	52
540939991	39.09	-79.66	West Virginia	Tucker	56	56	56	52	48
541071002	39.32	-81.55	West Virginia	Wood	57	56	56	52	47
550090026	44.53	-87.91	Wisconsin	Brown	59	58	58	55	51
550210015	43.32	-89.11	Wisconsin	Columbia	56	55	55	52	50
550250041	43.10	-89.36	Wisconsin	Dane	55	54	54	52	49
550270001	43.47	-88.62	Wisconsin	Dodge	62	61	61	58	54
550290004	45.24	-86.99	Wisconsin	Door	63	63	63	59	54
550350014	44.76	-91.14	Wisconsin	Eau Claire	52	51	51	50	49
550390006	43.69	-88.42	Wisconsin	Fond du Lac	61	60	60	57	53
550410007	45.56	-88.81	Wisconsin	Forest	54	53	53	51	48
550550002	43.00	-88.82	Wisconsin	Jefferson	58	57	57	55	52
550590019	42.50	-87.81	Wisconsin	Kenosha	60	59	59	60	**60**
550610002	44.44	-87.51	Wisconsin	Kewaunee	63	63	63	59	54
550630012	43.78	-91.23	Wisconsin	La Crosse	54	53	53	52	51
550710007	44.14	-87.62	Wisconsin	Manitowoc	66	65	65	60	55
550730012	44.71	-89.77	Wisconsin	Marathon	54	53	53	51	49
550790010	43.02	-87.93	Wisconsin	Milwaukee	56	55	55	53	51
550790026	43.06	-87.91	Wisconsin	Milwaukee	60	60	60	57	54
550790085	43.18	-87.90	Wisconsin	Milwaukee	65	64	64	61	57
550870009	44.31	-88.40	Wisconsin	Outagamie	59	58	58	56	53
550890008	43.34	-87.92	Wisconsin	Ozaukee	66	66	66	62	58
550890009	43.50	-87.81	Wisconsin	Ozaukee	62	62	62	58	53
551010017	42.71	-87.80	Wisconsin	Racine	57	57	57	57	56
551050024	42.51	-89.06	Wisconsin	Rock	59	58	58	55	52
551110007	43.44	-89.68	Wisconsin	Sauk	54	53	53	51	49
551170006	43.68	-87.72	Wisconsin	Sheboygan	**71**	**70**	**70**	**65**	**60**
551199991	45.21	-90.60	Wisconsin	Taylor	54	53	53	52	50
551270005	42.58	-88.50	Wisconsin	Walworth	59	58	58	55	52
551330027	43.02	-88.22	Wisconsin	Waukesha	57	57	57	53	50

*The design value from the monitor(s) with the highest projected ozone in each scenario is shown in bold blue text

Table 3A-11. Design Values for Northeast Monitors

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
90010017	41.00	-73.59	Connecticut	Fairfield	70	70	66	61	56
90011123	41.40	-73.44	Connecticut	Fairfield	67	67	63	58	54
90013007	41.15	-73.10	Connecticut	Fairfield	73	73	68	63	58
90019003	41.12	-73.34	Connecticut	Fairfield	74	74	69	**65**	59
90031003	41.78	-72.63	Connecticut	Hartford	63	63	59	54	50

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
90050005	41.82	-73.30	Connecticut	Litchfield	58	58	54	50	46
90070007	41.55	-72.63	Connecticut	Middlesex	65	65	61	56	52
90090027	41.30	-72.90	Connecticut	New Haven	64	64	60	56	51
90099002	41.26	-72.55	Connecticut	New Haven	72	72	67	63	57
90110124	41.35	-72.08	Connecticut	New London	67	67	63	59	54
90131001	41.98	-72.39	Connecticut	Tolland	63	63	59	55	50
90159991	41.84	-72.01	Connecticut	Windham	58	58	55	51	47
100010002	38.98	-75.56	Delaware	Kent	59	59	55	51	48
100031007	39.55	-75.73	Delaware	New Castle	59	59	55	51	48
100031010	39.82	-75.56	Delaware	New Castle	61	61	56	52	48
100031013	39.77	-75.50	Delaware	New Castle	62	62	57	53	49
100032004	39.74	-75.56	Delaware	New Castle	60	60	55	51	48
100051002	38.64	-75.61	Delaware	Sussex	61	61	58	54	51
100051003	38.78	-75.16	Delaware	Sussex	64	64	60	57	53
110010041	38.90	-76.95	D.C	D.C.	59	59	55	50	46
110010043	38.92	-77.01	D.C.	D.C.	62	62	58	53	49
230010014	43.97	-70.12	Maine	Androscoggin	51	51	48	44	40
230052003	43.56	-70.21	Maine	Cumberland	58	58	55	50	46
230090102	44.35	-68.23	Maine	Hancock	59	59	56	53	50
230090103	44.38	-68.26	Maine	Hancock	56	56	53	49	46
230112005	44.23	-69.79	Maine	Kennebec	52	52	49	45	42
230130004	43.92	-69.26	Maine	Knox	56	56	53	49	46
230173001	44.25	-70.86	Maine	Oxford	46	46	45	43	42
230194008	44.74	-68.67	Maine	Penobscot	48	48	45	42	39
230230006	44.01	-69.83	Maine	Sagadahoc	50	50	47	44	40
230290019	44.53	-67.60	Maine	Washington	50	50	48	45	43
230290032	44.96	-67.06	Maine	Washington	47	47	45	43	41
230310038	43.66	-70.63	Maine	York	50	50	47	44	41
230310040	43.59	-70.88	Maine	York	53	53	50	47	44
230312002	43.34	-70.47	Maine	York	60	60	57	52	48
240030014	38.90	-76.65	Maryland	Anne Arundel	64	64	60	55	51
240051007	39.46	-76.63	Maryland	Baltimore	65	65	60	55	51
240053001	39.31	-76.47	Maryland	Baltimore	66	66	61	56	51
240090011	38.54	-76.62	Maryland	Calvert	63	63	59	54	50
240130001	39.44	-77.04	Maryland	Carroll	60	60	57	54	51
240150003	39.70	-75.86	Maryland	Cecil	65	65	61	57	53
240170010	38.50	-76.81	Maryland	Charles	62	62	58	54	51
240199991	38.45	-76.11	Maryland	Dorchester	60	60	56	51	47
240210037	39.42	-77.38	Maryland	Frederick	63	63	59	57	54

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
240230002	39.71	-79.01	Maryland	Garrett	60	60	59	58	58
240251001	39.41	-76.30	Maryland	Harford	73	73	68	62	57
240259001	39.56	-76.20	Maryland	Harford	63	63	58	53	49
240290002	39.31	-75.80	Maryland	Kent	62	62	57	53	49
240313001	39.11	-77.11	Maryland	Montgomery	61	61	56	52	48
240330030	39.06	-76.88	Maryland	Prince George's	61	61	57	52	48
240338003	38.81	-76.74	Maryland	Prince George's	64	64	59	55	50
240339991	39.03	-76.82	Maryland	Prince George's	62	62	58	53	49
240430009	39.57	-77.72	Maryland	Washington	59	59	57	55	54
245100054	39.33	-76.55	Maryland	Baltimore (City)	62	62	57	52	48
250010002	41.98	-70.02	Massachusetts	Barnstable	60	60	57	53	48
250034002	42.64	-73.17	Massachusetts	Berkshire	58	58	55	52	49
250051002	41.63	-70.88	Massachusetts	Bristol	60	60	57	53	49
250070001	41.33	-70.79	Massachusetts	Dukes	65	65	61	57	53
250092006	42.47	-70.97	Massachusetts	Essex	58	58	56	53	50
250094005	42.81	-70.82	Massachusetts	Essex	57	57	54	51	47
250095005	42.77	-71.10	Massachusetts	Essex	57	57	54	50	46
250130008	42.19	-72.56	Massachusetts	Hampden	60	60	57	52	48
250150103	42.40	-72.52	Massachusetts	Hampshire	53	53	50	46	43
250154002	42.30	-72.33	Massachusetts	Hampshire	58	58	54	50	47
250170009	42.63	-71.36	Massachusetts	Middlesex	55	55	52	49	45
250171102	42.41	-71.48	Massachusetts	Middlesex	55	55	52	48	45
250213003	42.21	-71.11	Massachusetts	Norfolk	59	59	56	53	50
250250041	42.32	-70.97	Massachusetts	Suffolk	57	57	55	52	48
250250042	42.33	-71.08	Massachusetts	Suffolk	50	50	48	46	43
250270015	42.27	-71.88	Massachusetts	Worcester	56	56	53	50	46
250270024	42.10	-71.62	Massachusetts	Worcester	56	56	53	49	46
330012004	43.57	-71.50	New Hampshire	Belknap	52	52	50	47	45
330050007	42.93	-72.27	New Hampshire	Cheshire	51	51	48	46	43
330074001	44.27	-71.30	New Hampshire	Coos	58	58	57	56	55
330074002	44.31	-71.22	New Hampshire	Coos	51	51	50	49	48
330090010	43.63	-72.31	New Hampshire	Grafton	50	50	48	46	44
330111011	42.72	-71.52	New Hampshire	Hillsborough	54	54	51	48	45
330115001	42.86	-71.88	New Hampshire	Hillsborough	58	58	55	51	49
330131007	43.22	-71.51	New Hampshire	Merrimack	53	53	50	47	45
330150014	43.08	-70.75	New Hampshire	Rockingham	55	55	52	48	44
330150016	43.05	-70.71	New Hampshire	Rockingham	55	55	52	48	45

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
330150018	42.86	-71.38	New Hampshire	Rockingham	56	56	53	49	46
340010006	39.46	-74.45	New Jersey	Atlantic	61	61	57	54	50
340030006	40.87	-73.99	New Jersey	Bergen	64	64	59	56	52
340071001	39.68	-74.86	New Jersey	Camden	67	67	62	58	53
340110007	39.42	-75.03	New Jersey	Cumberland	58	58	54	50	46
340130003	40.72	-74.19	New Jersey	Essex	64	64	60	55	52
340150002	39.80	-75.21	New Jersey	Gloucester	68	68	63	58	54
340170006	40.67	-74.13	New Jersey	Hudson	64	64	59	55	51
340190001	40.52	-74.81	New Jersey	Hunterdon	63	63	58	55	51
340210005	40.28	-74.74	New Jersey	Mercer	65	65	60	56	52
340219991	40.31	-74.87	New Jersey	Mercer	62	62	58	54	50
340230011	40.46	-74.43	New Jersey	Middlesex	66	66	61	56	52
340250005	40.28	-74.01	New Jersey	Monmouth	66	66	61	56	51
340273001	40.79	-74.68	New Jersey	Morris	60	60	56	53	50
340290006	40.06	-74.44	New Jersey	Ocean	67	67	62	57	53
340315001	41.06	-74.26	New Jersey	Passaic	61	61	56	53	49
340410007	40.92	-75.07	New Jersey	Warren	52	52	48	45	43
360010012	42.68	-73.76	New York	Albany	57	57	52	49	46
360050133	40.87	-73.88	New York	Bronx	64	64	60	56	52
360130006	42.50	-79.32	New York	Chautauqua	62	62	61	59	58
360130011	42.29	-79.59	New York	Chautauqua	62	62	60	59	57
360150003	42.11	-76.80	New York	Chemung	57	57	55	53	52
360270007	41.79	-73.74	New York	Dutchess	58	58	54	50	46
360290002	42.99	-78.77	New York	Erie	61	61	60	60	**60**
360310002	44.37	-73.90	New York	Essex	57	57	56	55	54
360310003	44.39	-73.86	New York	Essex	57	57	55	55	54
360410005	43.45	-74.52	New York	Hamilton	57	57	55	54	52
360430005	43.69	-74.99	New York	Herkimer	55	55	54	53	52
360450002	44.09	-75.97	New York	Jefferson	62	62	60	59	58
360530006	42.73	-75.78	New York	Madison	55	55	53	51	49
360551007	43.15	-77.55	New York	Monroe	59	59	58	57	56
360610135	40.82	-73.95	New York	New York	64	64	61	58	55
360631006	43.22	-78.48	New York	Niagara	64	64	63	62	58
360650004	43.30	-75.72	New York	Oneida	53	53	52	50	49
360671015	43.05	-76.06	New York	Onondaga	60	60	58	56	54
360715001	41.52	-74.22	New York	Orange	56	56	52	48	44
360750003	43.28	-76.46	New York	Oswego	58	58	56	55	53
360790005	41.46	-73.71	New York	Putnam	58	58	54	50	46
360810124	40.74	-73.82	New York	Queens	71	71	67	**65**	**60**

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
360830004	42.78	-73.46	New York	Rensselaer	57	57	53	50	47
360850067	40.60	-74.13	New York	Richmond	71	71	67	63	58
360870005	41.18	-74.03	New York	Rockland	62	62	58	53	49
360910004	43.01	-73.65	New York	Saratoga	56	56	52	49	47
360930003	42.80	-73.94	New York	Schenectady	54	54	51	48	45
361010003	42.09	-77.21	New York	Steuben	57	57	55	54	53
361030002	40.75	-73.42	New York	Suffolk	**75**	**75**	**70**	**65**	**60**
361030004	40.96	-72.71	New York	Suffolk	67	67	63	58	52
361030009	40.83	-73.06	New York	Suffolk	72	72	67	62	57
361099991	42.40	-76.65	New York	Tompkins	58	58	56	55	54
361111005	42.14	-74.49	New York	Ulster	59	59	56	53	50
361173001	43.23	-77.17	New York	Wayne	56	56	55	53	52
361192004	41.05	-73.76	New York	Westchester	63	63	58	54	50
420010002	39.93	-77.25	Pennsylvania	Adams	57	57	53	51	49
420019991	39.92	-77.31	Pennsylvania	Adams	59	59	55	52	50
420030008	40.47	-79.96	Pennsylvania	Allegheny	67	67	65	62	**60**
420030010	40.45	-80.02	Pennsylvania	Allegheny	65	65	63	60	58
420030067	40.38	-80.17	Pennsylvania	Allegheny	65	65	63	62	**60**
420031005	40.61	-79.73	Pennsylvania	Allegheny	71	71	68	65	58
420050001	40.81	-79.56	Pennsylvania	Armstrong	64	64	61	59	57
420070002	40.56	-80.50	Pennsylvania	Beaver	63	63	62	62	56
420070005	40.68	-80.36	Pennsylvania	Beaver	67	67	65	63	57
420070014	40.75	-80.32	Pennsylvania	Beaver	64	64	63	61	**60**
420110006	40.51	-75.79	Pennsylvania	Berks	59	59	54	51	48
420110011	40.38	-75.97	Pennsylvania	Berks	62	62	57	54	51
420130801	40.54	-78.37	Pennsylvania	Blair	66	66	61	58	55
420170012	40.11	-74.88	Pennsylvania	Bucks	66	66	61	57	53
420210011	40.31	-78.92	Pennsylvania	Cambria	62	62	58	55	52
420270100	40.81	-77.88	Pennsylvania	Centre	64	64	60	57	55
420279991	40.72	-77.93	Pennsylvania	Centre	65	65	61	58	55
420290100	39.83	-75.77	Pennsylvania	Chester	61	61	55	51	47
420334000	41.12	-78.53	Pennsylvania	Clearfield	65	65	61	58	56
420430401	40.25	-76.85	Pennsylvania	Dauphin	59	59	54	51	48
420431100	40.27	-76.68	Pennsylvania	Dauphin	63	63	57	53	50
420450002	39.84	-75.37	Pennsylvania	Delaware	61	61	57	52	49
420479991	41.60	-78.77	Pennsylvania	Elk	56	56	54	52	51
420490003	42.14	-80.04	Pennsylvania	Erie	60	60	60	59	59
420550001	39.96	-77.48	Pennsylvania	Franklin	56	56	53	51	49
420590002	39.81	-80.27	Pennsylvania	Greene	59	59	58	57	57

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
420630004	40.56	-78.92	Pennsylvania	Indiana	67	67	63	60	57
420690101	41.48	-75.58	Pennsylvania	Lackawanna	60	60	56	53	50
420692006	41.44	-75.62	Pennsylvania	Lackawanna	58	58	54	51	49
420710007	40.05	-76.28	Pennsylvania	Lancaster	66	66	57	53	50
420710012	40.04	-76.11	Pennsylvania	Lancaster	64	64	57	53	50
420730015	41.00	-80.35	Pennsylvania	Lawrence	61	61	59	57	55
420750100	40.34	-76.38	Pennsylvania	Lebanon	63	63	58	54	51
420770004	40.61	-75.43	Pennsylvania	Lehigh	62	62	57	53	50
420791100	41.21	-76.00	Pennsylvania	Luzerne	55	55	51	47	44
420791101	41.27	-75.85	Pennsylvania	Luzerne	54	54	51	47	44
420810100	41.25	-76.92	Pennsylvania	Lycoming	57	57	53	51	48
420850100	41.22	-80.48	Pennsylvania	Mercer	62	62	61	60	59
420859991	41.43	-80.15	Pennsylvania	Mercer	55	55	54	53	52
420890002	41.08	-75.32	Pennsylvania	Monroe	54	54	50	47	45
420910013	40.11	-75.31	Pennsylvania	Montgomery	63	63	58	55	51
420950025	40.63	-75.34	Pennsylvania	Northampton	61	61	56	53	49
420958000	40.69	-75.24	Pennsylvania	Northampton	57	57	52	49	46
420990301	40.46	-77.17	Pennsylvania	Perry	59	59	55	53	51
421010004	40.01	-75.10	Pennsylvania	Philadelphia	55	55	51	48	44
421010024	40.08	-75.01	Pennsylvania	Philadelphia	69	69	65	60	56
421011002	40.04	-75.00	Pennsylvania	Philadelphia	67	67	62	58	54
421119991	39.99	-79.25	Pennsylvania	Somerset	55	55	53	51	50
421174000	41.64	-76.94	Pennsylvania	Tioga	60	60	57	54	52
421250005	40.15	-79.90	Pennsylvania	Washington	61	61	59	57	56
421250200	40.17	-80.26	Pennsylvania	Washington	60	60	59	57	56
421255001	40.45	-80.42	Pennsylvania	Washington	61	61	60	59	58
421290006	40.43	-79.69	Pennsylvania	Westmoreland	63	63	60	57	55
421290008	40.30	-79.51	Pennsylvania	Westmoreland	61	61	58	55	53
421330008	39.97	-76.70	Pennsylvania	York	61	61	53	50	46
421330011	39.86	-76.46	Pennsylvania	York	62	62	55	51	47
440030002	41.62	-71.72	Rhode Island	Kent	61	61	57	52	48
440071010	41.84	-71.36	Rhode Island	Providence	61	61	58	54	50
440090007	41.50	-71.42	Rhode Island	Washington	64	64	60	56	51
500030004	42.89	-73.25	Vermont	Bennington	54	54	51	48	46
500070007	44.53	-72.87	Vermont	Chittenden	52	52	51	50	49
510030001	38.08	-78.50	Virginia	Albemarle	54	54	52	50	49
510130020	38.86	-77.06	Virginia	Arlington	65	65	60	55	51
510330001	38.20	-77.38	Virginia	Caroline	57	57	53	49	45
510360002	37.34	-77.26	Virginia	Charles	61	61	56	51	46

Site ID	lat	long	State	County	Base Case	Baseline	70	65	60
510410004	37.36	-77.59	Virginia	Chesterfield	58	58	52	48	44
510590030	38.77	-77.10	Virginia	Fairfax	65	65	60	55	51
510610002	38.47	-77.77	Virginia	Fauquier	50	50	48	45	43
510690010	39.28	-78.08	Virginia	Frederick	54	54	52	51	50
510719991	37.33	-80.56	Virginia	Giles	51	51	50	49	49
510850003	37.61	-77.22	Virginia	Hanover	58	58	53	49	45
510870014	37.56	-77.40	Virginia	Henrico	61	61	55	50	46
511071005	39.02	-77.49	Virginia	Loudoun	60	60	56	53	49
511130003	38.52	-78.44	Virginia	Madison	59	59	57	56	56
511390004	38.66	-78.50	Virginia	Page	55	55	54	53	52
511479991	37.17	-78.31	Virginia	Prince Edward	52	52	48	46	44
511530009	38.85	-77.63	Virginia	Prince William	58	58	55	52	49
511611004	37.28	-79.88	Virginia	Roanoke	53	53	52	49	47
511630003	37.63	-79.51	Virginia	Rockbridge	51	51	50	48	47
511650003	38.48	-78.82	Virginia	Rockingham	55	55	53	53	52
511790001	38.48	-77.37	Virginia	Stafford	57	57	53	48	44
511970002	36.89	-81.25	Virginia	Wythe	57	57	56	55	55
515100009	38.81	-77.04	Virginia	Alexandria City	63	63	59	54	49
516500008	37.10	-76.39	Virginia	Hampton City	58	58	55	51	47
518000004	36.90	-76.44	Virginia	Suffolk City	59	59	57	53	49
518000005	36.67	-76.73	Virginia	Suffolk City	56	56	54	52	50

*The design value from the monitor(s) with the highest projected ozone in each scenario is shown in bold blue text

3A.7 Monitors Excluded from the Quantitative Analysis

There were 1219 ozone monitors with complete ozone data for at least one DV period covering the years 2009-2013. Of those sites, we quantitatively analyzed 1150 in this analysis. As discussed in chapter 3, 69 sites were excluded from the quantitative analysis of emissions reductions needed to reach alternative standard levels. These sites fall into one of three categories, as discussed in more detail in the following three subsections.

3A.7.1 Sites without Projections Due to Insufficient Days

Some monitors were excluded from the analysis because no future design value could be projected at the site. This occurred when there were not enough modeled high ozone days (4 or fewer) at the site to compute a design value according to EPA SIP modeling guidance. A list of the 36 sites falling into this category is given in Table 3A-12.

Table 3A-12. Monitors Without Projections due to Insufficient High Modeling Days to Meet EPA Guidance for Projecting Design Values

Site ID	lat	long	State	County
60231004	40.78	-124.18	California	Humboldt
60450008	39.15	-123.20	California	Mendocino
60750005	37.77	-122.40	California	San Francisco
60932001	41.73	-122.63	California	Siskiyou
160230101	43.46	-113.56	Idaho	Butte
230031100	46.70	-68.03	Maine	Aroostook
260330901	46.49	-84.36	Michigan	Chippewa
270052013	46.85	-95.85	Minnesota	Becker
270177416	46.71	-92.52	Minnesota	Carlton
270750005	47.95	-91.50	Minnesota	Lake
270834210	44.44	-95.82	Minnesota	Lyon
271370034	48.41	-92.83	Minnesota	Saint Louis
300298001	48.51	-114.00	Montana	Flathead
300490004	46.85	-111.99	Montana	Lewis and Clark
311079991	42.83	-97.85	Nebraska	Knox
380070002	46.89	-103.38	North Dakota	Billings
380130004	48.64	-102.40	North Dakota	Burke
380150003	46.83	-100.77	North Dakota	Burleigh
380171004	46.93	-96.86	North Dakota	Cass
380250003	47.31	-102.53	North Dakota	Dunn
380530002	47.58	-103.30	North Dakota	McKenzie
380570004	47.30	-101.77	North Dakota	Mercer
380650002	47.19	-101.43	North Dakota	Oliver
410170122	44.02	-121.26	Oregon	Deschutes
410290201	42.23	-122.79	Oregon	Jackson
410591003	45.83	-119.26	Oregon	Umatilla
460110003	44.35	-96.81	South Dakota	Brookings
530090013	48.30	-124.62	Washington	Clallam
530330080	47.57	-122.31	Washington	King
530530012	46.78	-121.74	Washington	Pierce
530570020	48.40	-122.50	Washington	Skagit
530730005	48.95	-122.55	Washington	Whatcom
550030010	46.60	-90.66	Wisconsin	Ashland
551250001	46.05	-89.65	Wisconsin	Vilas
560390008	43.67	-110.60	Wyoming	Teton

Site ID	lat	long	State	County
560391011	44.56	-110.40	Wyoming	Teton

3A.7.2 Winter Ozone

As discussed in Chapter 2 of the RIA, high winter ozone concentrations that have been observed in mountain valleys in the Western U.S. are believed to result from the combination of strong wintertime inversions, large NOx and VOC emissions from nearby oil and gas operations, increased UV intensity due to reflection off of snow surfaces and potentially still uncharacterized sources of free radicals. Current modeling tools are not sufficient to properly characterize ozone formation for these winter ozone episodes due to 1) the challenging task of capturing complex local "cold pool" meteorology using a model resolution that is optimized to capture regional and synoptic scale process, 2) uncertainties in quantifying the local emissions from oil and gas operations and 3) uncertainties in the chemistry that occurs both in the atmosphere and on snow surfaces during these episodes. Therefore, it was not appropriate to project ozone design values at monitors impacted by winter events. To identify sites impacted by winter events, we examined the ambient data that went into creating the 2009-2013 5-year weighted design value in locations known to have conditions favorable for winter ozone formation (i.e. all sites in Wyoming, Utah, and Colorado). At these sites, we evaluated the four highest 8-hr daily maximum ozone values in each year from 2009-2013 to identify wintertime ozone episodes. A site was categorized as having a design value impacted by wintertime ozone if at least 20% of the days examined (4 out 20) had ozone values greater than or equal to 75 ppb and occurred during a "winter" month (November-March). The seven sites identified as being affected by wintertime ozone events are listed in Table 3A-13.

Table 3A-13. Monitors Determined to Have Design Values Affected by Winter Ozone Events

Site ID	lat	long	State	County	# of summer DV days* >= 75	# of winter DV days* >= 75	highest winter 8-hr daily max	2009-2013 DV
081030006	40.09	-108.76	Colorado	Rio Blanco	0	7	106	71
560130099	42.53	-108.72	Wyoming	Fremont	1	4	93	67
560350097	42.98	-110.35	Wyoming	Sublette	0	3	83	64

560350099	42.72	-109.75	Wyoming	Sublette	0	4	123	77
560350100	42.79	-110.06	Wyoming	Sublette	0	4	84	67
560350101	42.87	-109.87	Wyoming	Sublette	0	4	89	66
560351002	42.37	-109.56	Wyoming	Sublette	0	4	94	68

*DV days defined here are the days with the 4 highest 8-hr daily maximum ozone values in each year from 2009-2013 (20 days).

3A.7.3 Monitoring Sites in Rural/Remote Areas of the West and Southwest

As mentioned in chapter 3 of the RIA, model-predicted ozone concentrations at 26 sites in rural/remote areas in the West and Southwest were excluded from the quantitative analysis (see list of sites in Table 3A-14). All of these 26 monitoring sites have 2025 baseline concentrations below 70 ppb, which is the upper end of the NAAQS range being proposed by the EPA. Therefore, no emissions reductions would be required for these sites for a primary standard of 70 ppb. Furthermore, only 15 of these sites would exceed a standard of 65 ppb. The remaining 11 would only exceed a standard of 60 ppb, which is not within the EPA's proposed range.

These 26 sites have two common characteristics. First, they have small modeled response to large regional NOx and VOC reductions in 2025 compared to other sites in the region. Second, these monitors would have DVs that remain above the standard after applying reductions needed to bring large urban areas in the region into attainment. Figure 3A-16 shows the response of design values at all sites in the Southwest region to a 75% NOx reduction in this region. Although design values at many urban sites drop by more than 10 ppb beyond the 2025 base case values, the response at the more remote and rural sites to modeled NOx reductions is relatively small. Many of these sites do show response between 2009-2013 DVs and base case 2025 DVs (up to 15 ppb decreases) suggesting that national on-the-books controls and proposed EPA rules could lower ozone DVs in these areas. However, modeling of additional NOx reductions within the region provide little incremental benefit suggesting that most of the regional anthropogenic sources impacting ozone at these locations have already been accounted for in the 2025 base case scenario.

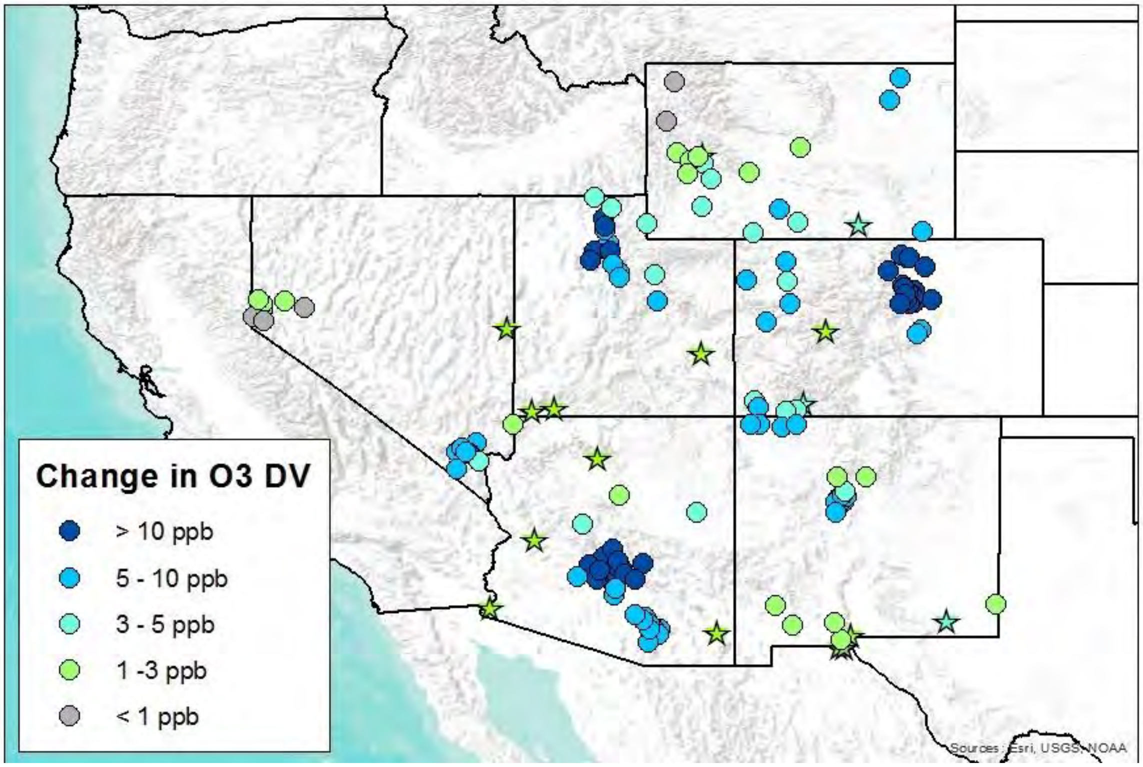

Figure 3A-16. Projected change in 2025 ozone design values with an additional 75% regional NOx control (Southwest region; stars represent sites identified in Table 3A-14)

A variety of influences including transport from California and other regional transport, cross-border pollution from Mexico, and exceptional events (e.g., wildfires and stratospheric intrusions) could contribute to ozone concentrations at the 26 sites. Each of these contributors is described further below, along with the Clean Air Act provisions that offer varying degrees of regulatory relief.

We have qualitatively characterized the predominant ozone influence for each site in Table 3A-14. These qualitative characterizations are based on the modeled response to large regional NOx reductions in 2025, proximity to the Mexican border (i.e., potential influence from trans-border pollution) and altitude (e.g., potential influence of ozone transported from the free troposphere: stratospheric intrusions or long range transport of international anthropogenic ozone). Figure 3A-17 shows the location of all sites listed in Table 3A-14 and for demonstrative purposes assigns each site to a category based on the predominant source of ozone in that location. As the table and figure indicate, all 26 sites have 2025 baseline design values below 70 ppb, 15 sites have design values between 65-70 ppb, and 11 sites have design values between 60-65 ppb. Of the 26 sites, 12 sites are characterized as border sites, 8 sites are characterized as being strongly influenced by California emissions, and 6 sites are influenced by other ozone sources.

Table 3A-14. Monitors with Limited Response to Regional NOx and National VOC Emissions Reductions in the 2025 Baseline

Name	Site ID	State	County	Altitude (m)	Monitor Type	Predominant O3 Sources	2009-2013 DV	Baseline DV
Chiricahua NM	40038001	Arizona	Cochise	1570	CASTNET	Mexican border	72	67
Grand Canyon NP	40058001	Arizona	Coconino	2152	CASTNET	California + Other sources	71	66
Alamo Lake	40128000	Arizona	La Paz	376	SLAMS	California	71	65
Yuma Supersite	40278011	Arizona	Yuma	51	SLAMS	Mexican border + California	75	67
El Centro-9th st	60251003	California	Imperial	-	SLAMS	California + Mexican Border	81	66
Death Valley NM	60270101	California	Inyo	125	Non-EPA Federal (NPS)	California + Other sources	71	66
Yosemite NP	60430003	California	Mariposa	5265	CASTNET	California + Other sources	77	68
Sequoia and Kings Canyon NP	61070006	California	Tulare	1890	Non-EPA Federal (NPS)	California + Other sources	81	67
Gothic	80519991	Colorado	Gunnison	2926	CASTNET	Other sources	66	64
Weminuche Wilderness Area	80671004	Colorado	La Plata	2367	Non-EPA Federal (USFS)	Southwest region + Other sources	72	68
Great Basin NP	320330101	Nevada	White Pine	2060	CASTNET	California + Other sources	72	66
Sunland Park City Yard	350130017	New Mexico	Dona Ana	-	SLAMS	Central region + Mexican border	66	63
3 Miles N of El Paso	350130020	New Mexico	Dona Ana	1250	SLAMS	Central region + Mexican border + Other sources	67	62

Name	Site ID	State	County	Altitude (m)	Monitor Type	Predominant O3 Sources	2009-2013 DV	Baseline DV
2MI from MT Cristo Rey	350130021	New Mexico	Dona Ana	1219	SLAMS	Central region + Mexican border	71	67
US-Mexico Border Crossing	350130022	New Mexico	Dona Ana	1280	SLAMS	Central region + Mexican border	70	66
BLM land near Carlsbad	350151005	New Mexico	Eddy	780	SLAMS	Central region + Southwest region + Mexican border	70	66
Big Bend NP	480430101	Texas	Brewster	1052	CASTNET	Mexican border	70	69
El Paso UTEP	481410037	Texas	El Paso	1158	SLAMS	Central region + Mexican border	71	67
Skyline Park	481410044	Texas	El Paso	1158	SLAMS	Central region + Mexican border	69	65
El Paso Chamizal	481410058	Texas	El Paso	1201	SLAMS	Central region + Mexican border	69	65
BLM Land/Carlsbad	483819991	Texas	Randall	780	SLAMS	Central region + Mexican border + Other sources	73	66
Canyonlands NP	490370101	Utah	San Juan	1814	CASTNET	Other sources	68	64
North Lava Flow Dr	490530006	Utah	Washington	846	SLAMS	California	68	62
Zion NP	490530130	Utah	Washington	1213	Non-EPA Federal (NPS)	California + Other sources	71	65
Centennial	560019991	Wyoming	Albany	3178	CASTNET	Other sources	69	65
Pinedale	560359991	Wyoming	Sublette	2388	CASTNET	Southwest region + Other sources	65	62

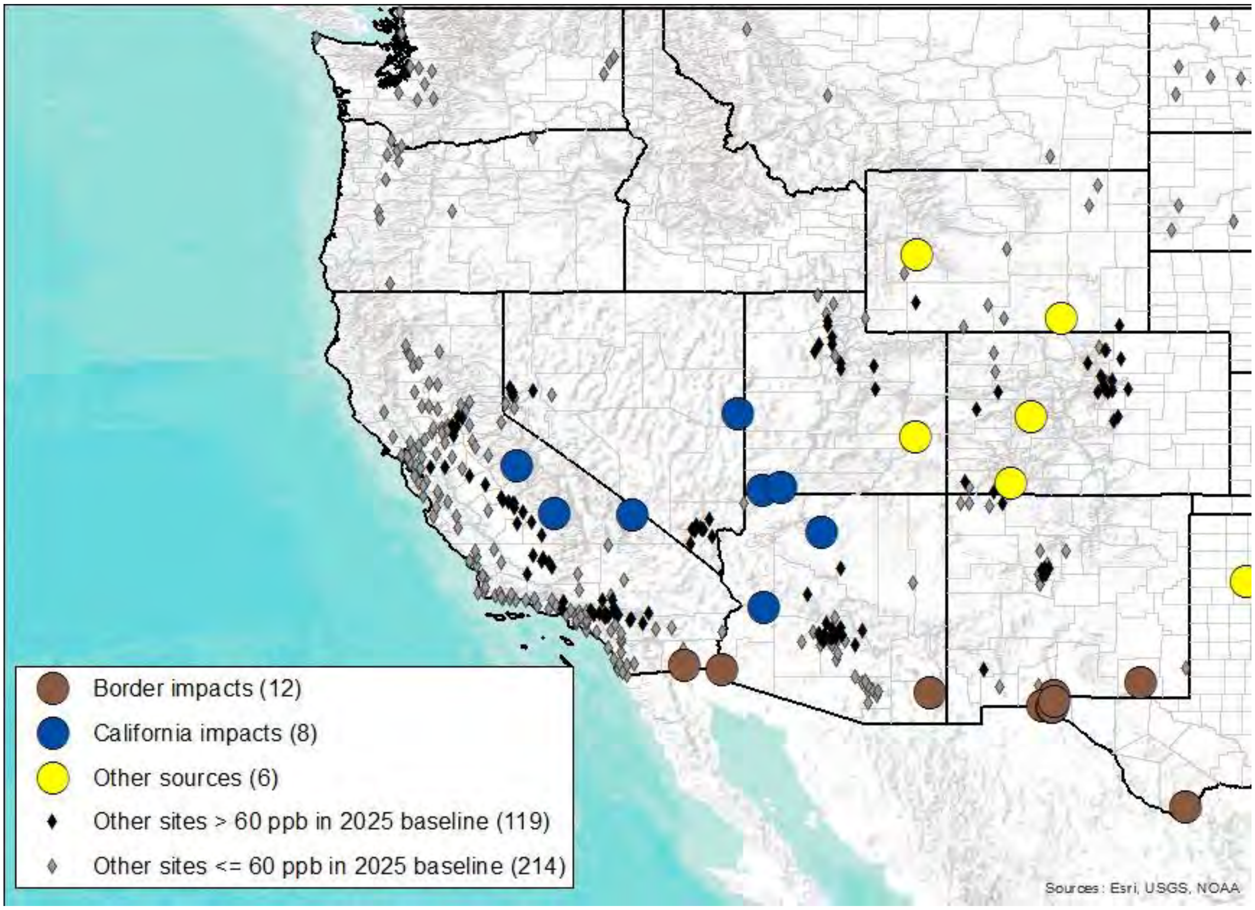

Figure 3A-17. Location of sites identified in Table 3A-14

In Figure 3A-17, the colored dots categorize sites by the predominant source of ozone. Many sites may be influenced by more than one source but are placed in a single category for illustrative purposes in the Figure. All ozone monitoring sites categorized as not substantially affected by natural or transported influences in Table 3A-14 are shown as small diamonds. Gray diamonds represent sites that had DVs less than or equal to 60 ppb in the 2025 baseline (or post-2025 baseline for California sites). Black diamonds represent sites that had DVs greater than 60 ppb in the 2025 baseline (or post-2025 baseline for California sites).

In this section we look at examples of how these sites might leverage various Clean Air Act provisions to comply with requirements for lower alternative standard levels including: interstate transport provisions, exceptional events demonstrations, rural transport designations, and requirements for nonattainment areas in international border areas (i.e., section 179B of the Clean Air Act).

Clean Air Act sections 110 and 126 have provisions designed to reduce significant transport contributions from upwind areas to downwind nonattainment areas. Although the RIA accounted for the impacts of regional NOx reductions it was not able to account for the impacts of emissions reductions in California on ozone at downwind sites in other states since California is likely to have an attainment date that is later than that of other western states. As discussed in the main RIA, many areas of California will not be required to meet the current ozone standard until after 2025 (i.e., 2027 or later) and may not be required to meet a new ozone standard until sometime between 2032 and 203.[25] Although California will likely implement some of the emissions reductions necessary to meet these standards prior to their attainment date, there is a considerable uncertainty about how to quantify the portion of the emissions reductions that may occur by 2025. Therefore, we did not account for the benefits of California emissions reductions on design values in downwind states.

However, it is very likely that reductions in California emissions would lead to substantial reductions in DVs at some monitoring locations in downwind states. For example, as shown in Figure 3A-18, a 90% NOx reduction in California has the potential to substantially improve ozone concentrations at downwind receptors in Nevada, Utah, and Arizona. Given the number of monitors in California projected to violate a revised ozone standard in 2025, it is quite likely that the state will adopt substantial local and regional controls to reach attainment. These controls would benefit areas outside of California as well. In addition, California and other states may have obligations to reduce emissions if those emissions are contributing substantially to interstate transport, as required under sections 110 and 126 of the Clean Air Act. Although no assessment of state-receptor linkages has been completed for alternative levels of the NAAQS (70, 65, and 60 ppb) this figure indicates that it is possible that California (and other Western states) could potentially have significant benefit toward attainment at downwind receptors which might then be subject to interstate transport provisions of the Clean Air Act.

[25] The EPA will likely finalize designations for a revised ozone NAAQS in late 2017. Depending on the precise timing of the effective date of those designations, nonattainment areas classified as Severe 15 will likely have to attain sometime between late 2032 and early 2033 and nonattainment areas classified as Extreme will likely have to attain sometime between late 2037 and early 2038.

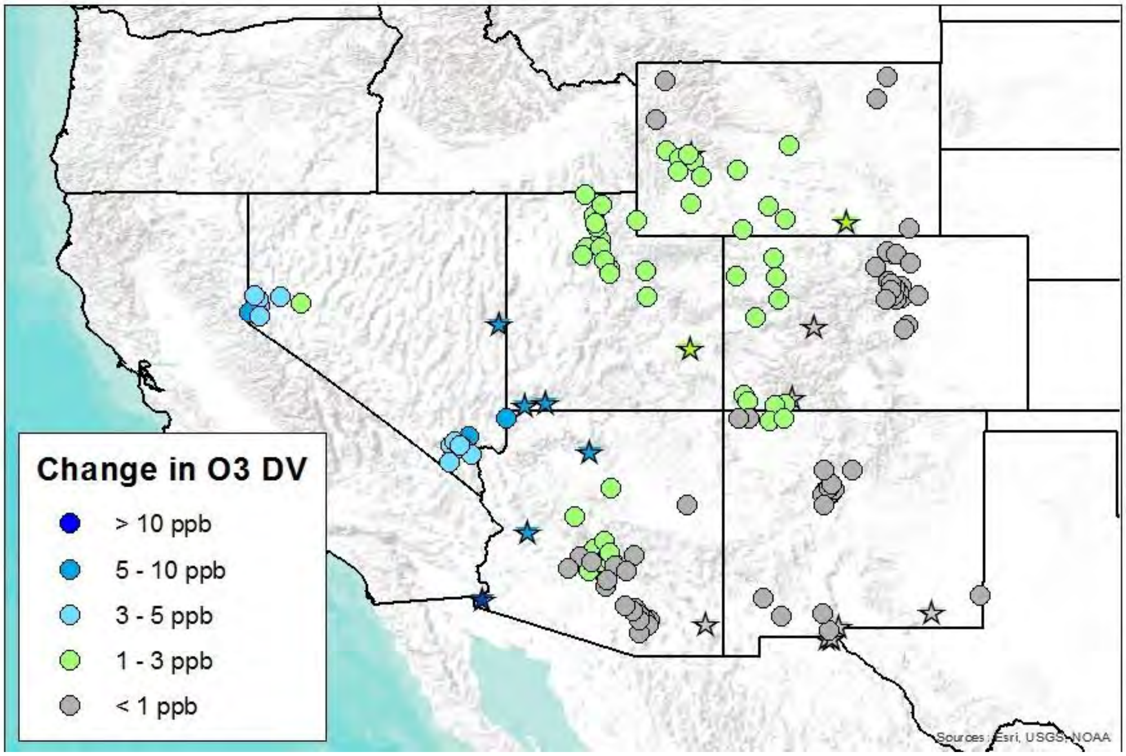

Figure 3A-18. Projected change in 2025 ozone design values with an additional 90% California NOx control (Southwest region; stars represent sites identified in Table 3A-14)

An air agency can request and the EPA can agree to exclude data associated with event-influenced exceedances or violations of a NAAQS provided the event meets the statutory requirements in section 319 of the CAA:

- The event "affects air quality."
- The event "is not reasonably controllable or preventable."
- The event is "caused by human activity that is unlikely to recur at a particular location or [is] a natural event."26

The EPA's implementing regulations, the 2007 Exceptional Events Rule, further specify that states must provide evidence that: [27]

26 A natural event is further described in 40 CFR 50.1(k) as "an event in which human activity plays little or no direct causal role."

[27] See 72 Federal Register 13560 (March 22, 2007), 40 CFR Part 50.1, 40 CFR Part 50.14 and 40 CFR Part 51.930.

- “There is a clear causal relationship between the measurement under consideration and the event that is claimed to have affected the air quality in the area;”
- “The event is associated with a measured concentration in excess of normal historical fluctuations, including background;” and
- “There would have been no exceedance or violation but for the event.”

Once an air agency requests data exclusion by flagging the subject data and submitting supporting documentation showing that the data have been affected by exceptional events (e.g., stratospheric intrusions or wildfires) and the EPA concurs with this request, the event-influenced data would be excluded from the data set used in regulatory decisions, including determining whether or not an area is attaining or violating a NAAQS. As an example, Figure 3A-18 shows five years of daily ozone values at Weminuche Wilderness area in La Plata County, CO. This figure shows evidence of both episodic and persistent high ozone at this monitoring site between 2009 and 2013. Several short periods of elevated ozone in springtime (March and April) could potentially be due to stratospheric intrusion, although more analysis would be required to definitively determine the source(s) contributing to these high concentrations. This figure also shows more prolonged periods of ozone values above 65 ppb which are unlikely to qualify for exclusion as exceptional events. In cases where design values are only 1-2 ppb above an alternative NAAQS level, excluding data from one or two days may be enough to show compliance with the standard even if there are still some periods with high ozone values that do not qualify. This is especially true because the standard is based on a three-year average. As can be seen in Figure 3A-19, some years (2009, 2012 and 2013) have substantially fewer days above 65 ppb at this site than others (2010 and 2011). Eliminating a few high ozone days in 2010 and 2011 could potentially bring this site’s projected design value below a threshold level when averaged with ozone concentrations from one or more low-ozone years.

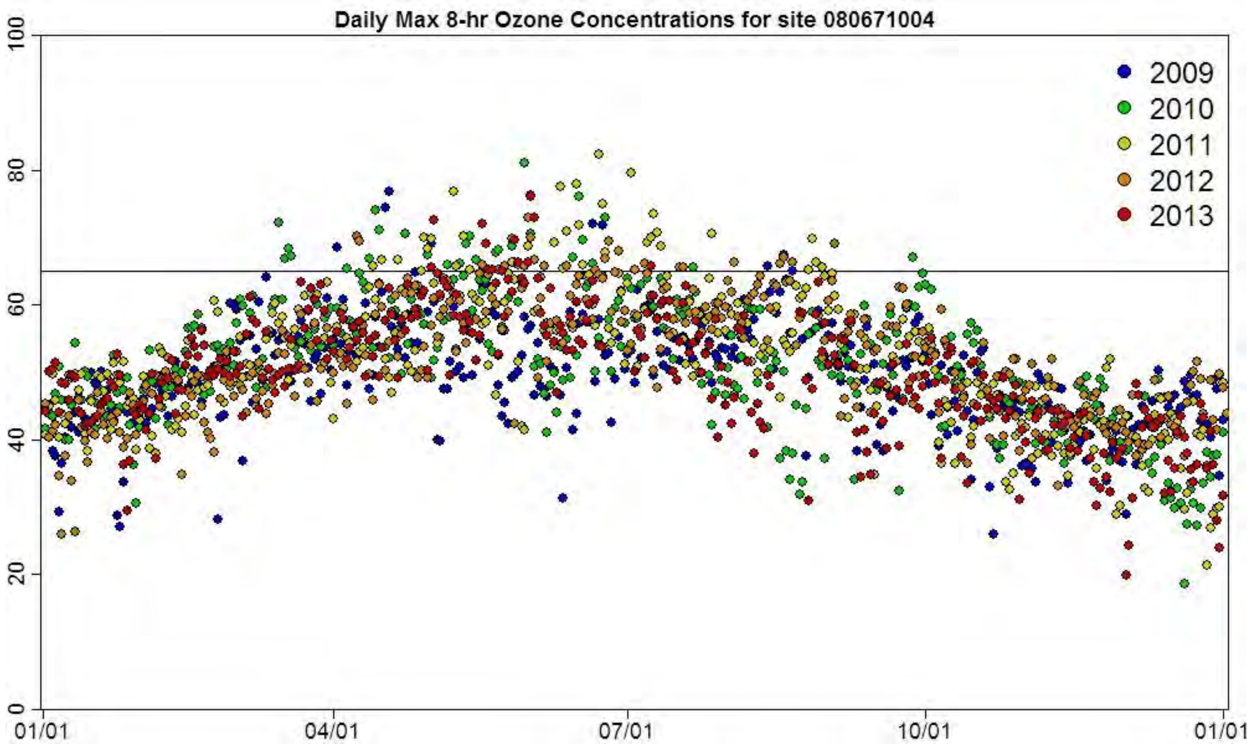

Figure 3A-19. Daily 8-hr maximum ozone values at ozone monitor in Weminuche Wilderness area in La Plata County Colorado from 2009-2013. Horizontal line provided at 65 ppb.

Other CAA provisions that could provide regulatory relief for air agencies and potential regulated entities include designation as a rural transport area (182(h)) or a determination that the area would have attained but for the contribution of international emissions (179B). Rural transport areas must show that the area does not contain emissions sources that substantially impact monitored ozone concentrations in the area or in other areas and that they are not in or adjacent to a Metropolitan Statistical Area (MSA). Section 179B demonstrations are used for areas that can demonstrate they would have attained the standard but for emissions emanating from outside the U.S. The EPA has used section 179B authority previously to approve attainment plans for Mexican border areas in El Paso, TX (O_3, PM10, and CO plans); Nogales, AZ (PM10 plan); and Imperial Valley, CA (PM10 plan). The 1-hour O_3 attainment plan for El Paso, TX was approved by EPA as sufficient to demonstrate attainment of the NAAQS by the Moderate classification deadline of November 15, 1996, taking into account "but for" international emissions sources in Ciudad Juárez, Mexico (69 FR 32450, June 10, 2004). The state's demonstration included airshed modeling using only the U.S. emissions data because emissions data from Ciudad Juárez were not available.

3A.8 Calculation Methodology for W126 Metric

Calculation of the W126 metric occurs in several steps. The first step is to sum the weighted hourly ozone concentrations within each calendar month, resulting in monthly index values. Since plant and tree species are not photosynthetically active during nighttime hours, only ozone concentrations observed during daytime hours (defined as 8:00 AM to 8:00 PM local time) are included in the summations. The monthly W126 index values are calculated from the hourly ozone concentration data as follows:

$$Monthly\ W126 = \sum_{d=1}^{N} \sum_{h=8}^{19} \frac{C_{dh}}{1+4403*exp(-126*C_{dh})} \qquad \text{Equation (3A-1)}$$

where N is the number of days in the month, d is the day of the month (d = 1, 2, …, N), h is the hour of the day (h = 0, 1, …, 23), and C_{dh} is the hourly ozone concentration observed on day d, hour h, in parts per million.

Next, the monthly W126 index values are adjusted for missing data. If N_m is defined as the number of daytime ozone concentrations observed during month m (i.e. the number of terms in the monthly index summation), then the monthly data completeness rate is $V_m = N_m / 12 * N$. The monthly index values are adjusted by dividing them by their respective V_m. Monthly index values are not computed if the monthly data completeness rate is less than 75 percent ($V_m < 0.75$).

Finally, the annual W126 index values are computed as the maximum sum of their respective adjusted monthly index values occurring in three consecutive months (i.e., January–March, February–April, etc.). Three-month periods spanning across two years (i.e., November–January, December–February) are not considered, because the seasonal nature of ozone makes it unlikely for the maximum values to occur at that time of year. The annual W126 concentrations are considered valid if the data meet the annual data completeness requirements for the existing standard. Three-year W126 index values are calculated by taking the average of annual W126 index values in the same three-month period in three consecutive years.

3A.9 References

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CHAPTER 4: CONTROL STRATEGIES AND EMISSIONS REDUCTIONS

Overview

In order to estimate the costs and benefits of alternative ozone standards, the EPA has analyzed hypothetical control strategies that areas across the country might employ to attain alternative revised primary ozone standards of 70, 65, and 60 ppb. This chapter documents the emission control measures EPA applied to simulate attainment with these alternative ozone standards and the projected emission reductions associated with the measures.

This chapter is organized into four sections. Section 4.1 provides a summary of the steps used to conduct the control strategy analysis. Section 4.2 describes the emission reductions by sector in analyzing the baseline controls to meet the current (75 ppb) ozone standard. Section 4.3 discusses control measures and emission reductions applied as part of the alternative standard analyses. Section 4.4 lists the key limitations and uncertainties associated with the control strategy analysis. And finally, Section 4.5 lists the references for the chapter.

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 2025 reflecting current state and federal programs[28]. This does not include control programs specifically for the purpose of attaining the current ozone standard (75 ppb).
- Baseline - For all areas of the U.S. except California, the base case plus additional emissions reductions needed to reach attainment of the current ozone standard (75 ppb) as well as emissions resulting from the Clean Power Plan (U.S. EPA, 2014b). Several areas in California are not required to meet the existing standard by 2025 and may not be required to meet a revised standard until 2037, thus we conducted analyses of attainment in California for post-2025. We explain the baseline treatment for California in more detail later in this chapter.
- Alternative Standard Analysis - Emissions 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

[28] A complete list of programs included in the 2025 base case emissions is included in the Technical Support Document: Preparation of Emissions Inventories for the Version 6.1, 2011 Emissions Modeling Platform (U.S. EPA, 2014d).

> used to classify nonattainment areas, assess progress towards meeting the NAAQS, and develop control strategies. The design value for the 8-hour ozone standard is calculated as the 3-year average of the 4th highest 8-hour daily maximum concentration recorded at each monitoring site.

The EPA analyzed the impact that additional emissions control measures, across numerous sectors, would have on predicted ambient ozone concentrations incremental to the baseline. These control measures are based on information available at the time of this analysis, and include primarily end of pipe controls. Additional emission abatement strategies such as fuel switching, energy efficiency, and process changes may also be employed to reach emission reduction targets. The potential impact of some of these types of strategies is discussed in Chapter 7. Note that we did not conduct this analysis incremental to controls applied as part of previous NAAQS analyses (e.g., NO_x or $PM_{2.5}$ because the data and modeling on which these previous analyses were based are now considered outdated and are not compatible with the current ozone NAAQS analysis. In addition, there were no incremental NOx controls applied in the PM2.5 NAAQS and therefore there would be little to no impact on the controls selected to meet the alternative ozone standards analyzed.[29] Thus, the analysis for the alternative standards focuses specifically on incremental improvements beyond the current standard and other existing and proposed rules. The selection of control strategies is based on a least cost approach selecting from those controls for which we have adequate information on costs, effectiveness, and applicability. The hypothetical control strategies presented in this RIA represent illustrative options for emissions reductions that achieve national attainment of the alternative standards. The hypothetical control strategies are not recommendations or requirements for how a revised ozone standard should be implemented, and states will make all final decisions regarding implementation strategies for a revised ozone NAAQS.

[29] There were no additional NOx controls applied in the $PM_{2.5}$ NAAQS RIA, and therefore there would be little to no impact on the controls selected as part of this analysis. In addition, the only geographic areas that exceed the alternative ozone standard levels analyzed in this RIA and in the 2012 $PM_{2.5}$ NAAQS RIA are in California. The attainment dates for a new $PM_{2.5}$ NAAQS would likely precede attainment dates for a revised ozone NAAQS. While the 2012 $PM_{2.5}$ NAAQS RIA concluded that controls on directly emitted $PM_{2.5}$ were the most cost-effective on a$/ug basis, states may choose to adopt different control options. These options could include NOx controls. It is difficult to determine the impact on costs and benefits for this RIA because it is highly dependent upon the control measures that would be chosen and the costs of these measures.

The NO_x control measures dataset for non-electric generating unit (EGU) point sources applied in these control strategies reflects a number of revisions that EPA made since the completion of the previous ozone NAAQS RIA. These changes include:

- Removal of incorrect links between control measures and SCCs,
- Updates to cost equations where more recent data was available to improve their accuracy,
- Inclusion of information that has recently become available concerning known and emerging technologies for reducing NO_x emissions, and
- Revising costs and control efficiencies from control measures in the dataset based on recently obtained information from industry and multi-jurisdictional organizations (e.g., Ozone Transport Commissions and Lake Michigan Air Directors Consortium).

These revisions made the NO_x control measures dataset for non-EGUs more accurate, defensible, and up to date. These improvements in this dataset will improve our control strategy and cost analyses not only for this RIA, but also for other potential rulemakings where control of NO_x from non-EGUs is an important concern.

4.1 Control Strategy Analysis Steps

The primary year of analysis for analyzing the incremental costs and benefits of meeting a revised ozone standard is 2025. The analysis year was chosen because most areas of the U.S. will be required to meet a revised ozone standard by 2025. In estimating the incremental costs and benefits of potential alternative standards, we recognize that there are several areas that are not required to meet the existing ozone standard by 2025. The Clean Air Act allows areas with more significant air quality problems to take additional time to reach the existing standard. Several areas in California are not required to meet the existing standard by 2025 and may not be required to meet a revised standard until December 31, 2037. Depending on how areas in California are eventually designated, some areas may have attainment dates earlier than 2037, but for simplicity we are not distinguishing unique attainment years for different locations within California. To reflect these differences in required attainment dates, we conducted analyses of attainment in California for the period post-2025.

While our goal for the California analysis was to reflect 2038, we were not able to project emissions and air quality beyond 2025 for California. However, we were able to adjust baseline air quality to reflect mobile source emissions reductions for California that would occur between 2025 and 2030; these emissions reductions were the result of state and federal mobile source regulations expected to be fully implemented by 2030. For ease of discussion throughout the analyses we refer to the time periods for potential attainment in California and in other areas of the U.S. as post-2025 and 2025, respectively. Because we estimate incremental emissions reductions, costs, and benefits for these two distinct time periods, it is not appropriate to add the estimates together or to directly compare the estimates.

To conduct the control strategy analyses, we require information on (i) control costs, (ii) control effectiveness in terms of NOx or VOC emissions reduced, (iii) the sensitivity of ozone design values to the NOx and VOC emissions reductions, and (iv) design value targets for each area. For the air quality modeling, the EPA prepared one control scenario for an alternative standard level of 70 ppb (Step 2 below) because we did not expect to have sufficient known controls for all locations to reach attainment for all of the alternative standards analyzed. The control scenario is not really designed to model how areas reach an alternative standard level of 70 ppb, instead it sets up a process for developing, and applying, a list of potentially available known controls ordered by cost. To develop a sufficient amount of controls for the 65 and 60 ppb alternative standard levels, we also worked to identify known controls in areas or regions expected to contribute to nonattainment for these alternative standards.

The following steps were taken by the EPA to analyze the impacts and costs of the control scenario incremental to the base case air quality modeling:

1. Identify geographic areas in the U.S. projected to exceed the alternative standard of 70 ppb in the year 2025 in the base case air quality modeling.
2. Develop a hypothetical control scenario for these areas and generate a control case 2025 emissions inventory for all areas except California; for California develop a control case post-2025 emissions inventory.
3. Perform air quality modeling to assess the air quality impacts of the hypothetical control scenario. Additionally, perform a series of emissions sensitivity simulations to develop average ozone response to across-the-board NOx and VOC emissions reductions in different areas (see Chapter 3).

4. Calculate the portion of the hypothetical control scenario emission reductions that are attributed to meeting the baseline. Estimate any additional emissions reductions beyond the known controls that are needed to meet the current standard based on average ozone response factors. These are the baseline emission reductions.
5. Estimate the additional emissions reductions incremental to the baseline that are needed to meet the alternative standards of 70, 65, and 60 ppb. Costs of controls incremental to (i.e., over and above) the baseline reductions are attributed to the costs of meeting the alternative standards. These emissions reductions can come from specific known controls or emission reductions needed beyond known controls, also referred to as unknown controls. Potential controls may be categorized as unknown because these needed emissions reductions come from sectors for which we have not sufficiently explored emissions abatement opportunities or sectors that might require non-traditional abatement through measures like energy efficiency or process changes.

The following sections will discuss in more detail the analysis steps presented above.

4.2 Baseline Control Strategy

Establishing the baseline allows us to estimate the incremental costs and benefits of attaining the alternative standards. Three steps were used to develop the baseline. First, we estimated 2025 base case emissions and air quality, reflecting "on the books" regulations (see Section 3.1.3 Emissions Inventories for a discussion of the rules included in the Base Case for this analysis).[30] Second, we accounted for changes in ozone predicted to occur due to one potential approach for implementing the Clean Power Plan. Third, we identified additional controls that could be applied to demonstrate attainment of the current ozone standard of 75 ppb.

Additional control measures were used in three sectors to meet the current ozone standard in establishing the baseline:[31] Non-Electric Generating Unit Point Sources (Non-EGUs), Non-Point (Area) Sources, and Nonroad Mobile Sources. See Table 4-1 for a summary of controls

[30] Among others factors, the baseline for this analysis is also affected by the choice of the future year -- a year farther into the future allows for more time for federal measures to work and to attain. This baseline is also affected by the air quality starting point, potentially reducing the amount of emissions reductions required for attainment – this analysis started from a standard of 75 ppb, where the analysis in 2008 started from a standard of 84 ppb. In addition, we have identified additional "known" controls to apply in this analysis, controls which are less expensive per ton than unknown controls.

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

applied in the baseline analysis. There were several areas in California that did not reach attainment of the current standard with known controls. For these geographic areas, we estimated the additional emissions reductions needed beyond those achieved by identified known controls for NO_x and VOC to attain the current standard.

Table 4-1. Controls Applied for the Alternative Standard Analyses Control Strategy

Sector	NO_x	VOC
Non-EGU Point	LEC (Low Emission Combustion)	Solvent Recovery System
	SCR (Selective Catalytic Reduction)	Work Practices, and Material Reformulation/Substitution
	SNCR (Selective Non-Catalytic Reduction)	Low-VOC materials Coatings and Add-On Controls
	NSCR (Non-Selective Catalytic Reduction)	Low VOC Adhesives and Improved Application Methods
	LNB (Low NO_x Burner Technology)	Permanent Total Enclosure (PTE)
	LNB + SCR	Solvent Substitution, Non-Atomized Resin Application Methods
	LNB + SNCR	Petroleum Wastewater Treatment Controls
	OXY-Firing	Incineration (Thermal, Catalytic, etc) to Reduce VOC Emissions
	Biosolid Injection Technology	
	LNB + Flue Gas Recirculation	
	LNB + Over Fire Air	
	Ignition Retard	
	Natural Gas Reburn	
	Ultra LNB	
NonPoint	NSCR (Non-Selective Catalytic Reduction)	Process Modification to Reduce Fugitive VOC Emissions
	LEC (Low Emission Combustion)	Reformulation to Reduce VOC Content
	LNB (Low NO_x Burner Technology)	Incineration (Thermal, Catalytic, etc) to Reduce VOC Emissions
	LNB Water Heaters	Low Pressure/Vacuum (LPV) Relief Valves in Gasoline Storage Tanks
		Reduced Solvent Utilization
		Gas Recovery in Landfills
Nonroad	Diesel Retrofits & Engine Rebuilds	

A map of the country is presented in Figure 41, which shows the counties projected to exceed the current ozone standard of 75 ppb in the 2025 base case scenario. This includes 8 projected exceeding counties in California and 3 exceeding counties in Texas. NOx control measures were applied in these 11 counties in the baseline analysis to meet the current ozone standard. In addition, NOx control measures were applied to 40 California counties and 52 Texas counties adjacent to exceeding counties in order to address transport coming from these adjacent counties. A map of the areas where control measures were applied to demonstrate attainment of the current standard and establish the baseline is presented in Figure 4-2.

To construct the post-2025 baseline, we included mobile source NO_x and VOC emissions changes that are projected to occur in California from 2025 - 2030 as a result of California's current mobile source control programs and projected changes in vehicle miles traveled and nonroad activity levels. These changes were included because they would result in emission reductions that would contribute toward attainment of the current ozone standard. No emission projections were available for other sectors for this time period, and no mobile source emissions projections were available beyond 2030. Additionally, VOC controls were applied in California counties highlighted in Figure 4-2. Even with the above mentioned controls, some areas in California did not reach attainment with known controls. For these areas, we estimated the additional NO_x emission reductions needed beyond identified known controls to attain the standard.

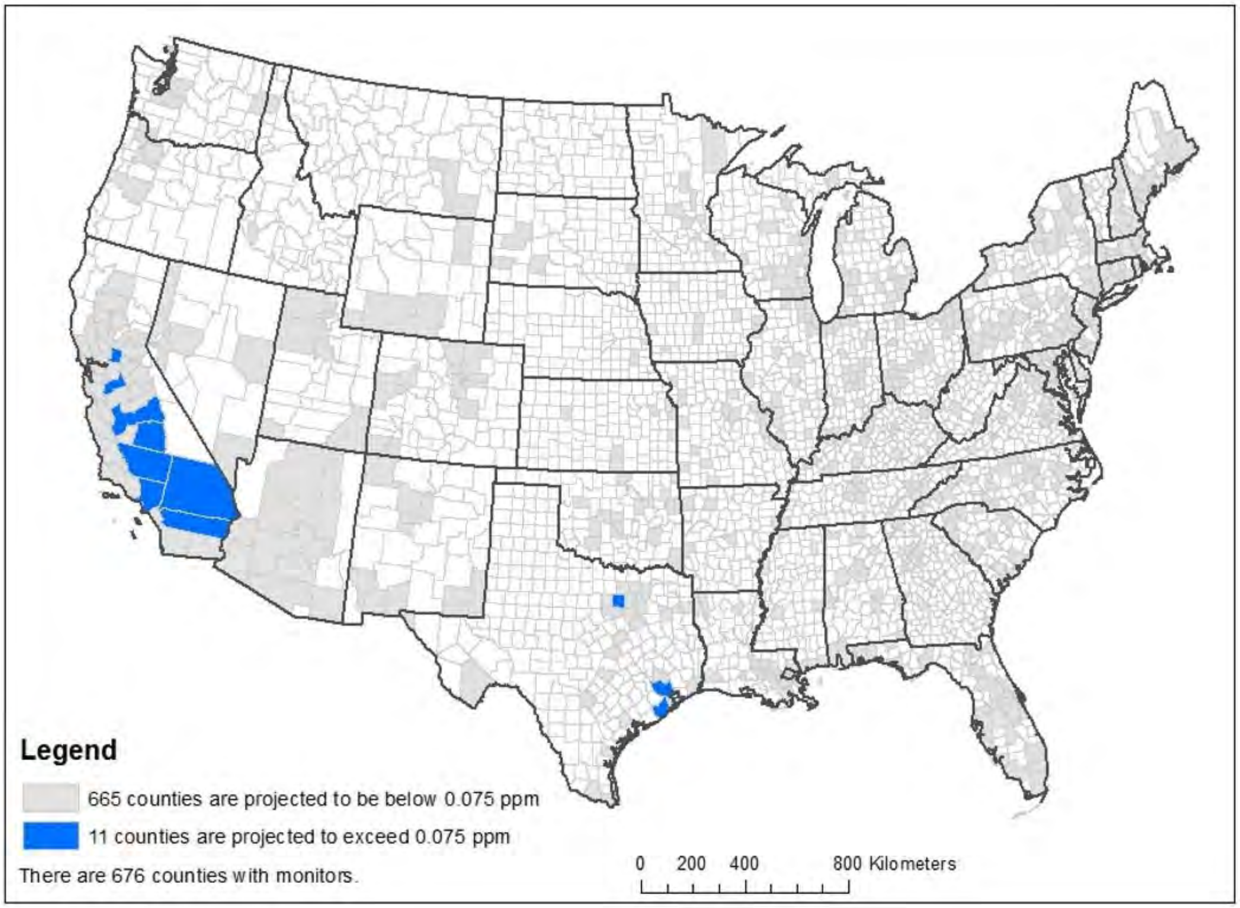


Figure 4-1. Counties Projected to Exceed the Baseline Level of the Current Ozone Standard (75 ppb) in 2025 Base Case

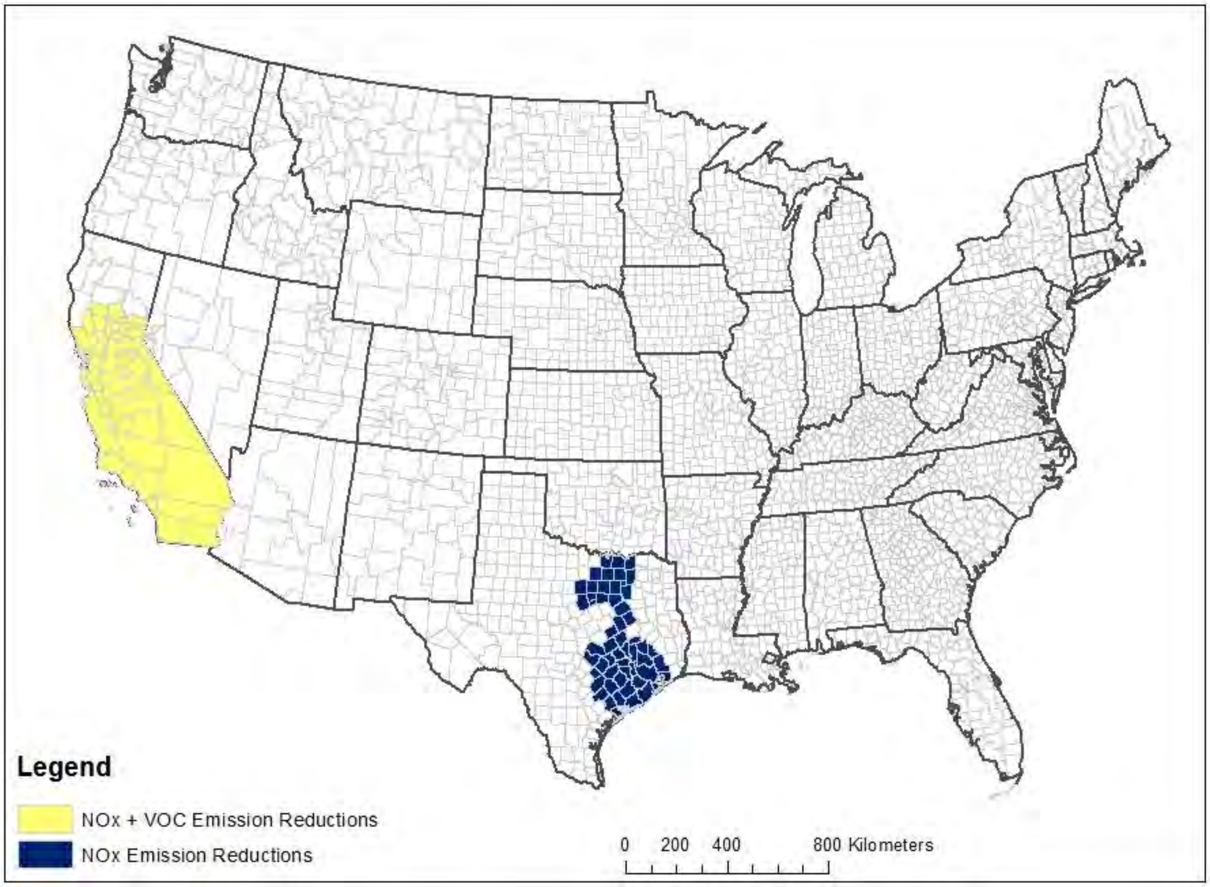


Figure 4-2. Counties Where Emissions Reductions Were Applied to Demonstrate Attainment of the Current Standard for the Baseline Analysis

Tables 4-2 and 4-3 summarize the NO_x and VOC emission reductions needed to demonstrate attainment of the current ozone standard (75 ppb).

Table 4-2. Summary of Emission Reductions by Sector for Known Controls Applied to Demonstrate Attainment of the Current Standard for the 2025 Baseline - U.S., except California (1,000 tons/year)[a]

Geographic Area [b]	Emissions Sector	NO_x	VOC
East	EGU	-	-
	Non-EGU Point	18	-
	Nonpoint	26	-
	Nonroad	0.83	-
	Onroad	-	-
	Total	**45**	**-**
West	EGU	-	-
	Non-EGU Point	-	-
	Nonpoint	-	-
	Nonroad	-	-
	Onroad	-	-
	Total	**-**	**-**

[a] Estimates are rounded to two significant figures.

[b] For the control strategy and cost analysis, "East" includes the Northeast, Midwest, and Central regions, and "West" includes the Southwest region. See Chapter 3 for a description of these regions.

Table 4-3. Summary of Emission Reductions (Known and Unknown Controls) Applied to Demonstrate Attainment in California for the post-2025 Baseline (1,000 tons/year)[a]

	Emissions Sector	NO_x	VOC
Known Controls	EGU	-	-
	Non-EGU Point	14	0.61
	Nonpoint	14	47
	Nonroad	4.2	-
	Onroad	-	-
	Total	**31**	**48**
Unknown Controls	All	160	-
	Total	**190**	**48**

[a] Emission reduction estimates are rounded to two significant figures.

The 2025 baseline for this analysis presents one scenario of future year air quality based upon specific control measures, additional emission reductions beyond known controls, promulgated federal rules such as Tier 3, and specific years of initial values for 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 with the current standard.

4.3 Alternative Standard Analyses

After identifying the controls in the baseline scenario, additional controls needed to meet the alternative standards were identified in four sectors: Electric Generating Units (EGUs), Non-Electric Generating Unit Point Sources (Non-EGUs), Non-Point (Area) Sources, and Nonroad Mobile Sources. Onroad mobile source controls were not applied because they are largely addressed in existing rules such as the recent Tier 3 rule. Controls applied for the alternative standard analyses were the same as were applied for the baseline analysis (see Table 4-1 for a summary of controls applied in the baseline) with the addition of Selective Catalytic Reduction applied to EGUs. Other than the addition of EGU controls, the primary difference between the controls applied for the alternative standards versus the baseline was the geographic areas to which they were applied.

The EPA performed a national scale air quality modeling analysis to estimate ozone concentrations for the future base case year of 2025. To accomplish this, we modeled multiple emissions cases for 2025, including the 2025 base case and twelve 2025 emissions sensitivity simulations. The twelve emissions sensitivity simulations were used to develop ozone sensitivity factors (ppb/ton) from the modeled response of ozone to changes in NOx and VOC emissions from various sources and locations. These ozone sensitivity factors were then used to determine the amount of emissions reductions needed to reach the 2025 baseline and evaluate potential alternative standard levels of 70, 65, and 60 ppb incremental to the baseline. As mentioned previously, only a subset of known controls were included in the modeled control scenarios. Therefore, any additional emissions reductions may include both known and unknown controls. In areas of the country outside of California, Texas, and the northeast, total emissions reductions needed beyond the baseline were based entirely on response factors, i.e., not based on the hypothetical control scenario used in the air quality modeling.

Figure 4-3 shows the counties projected to exceed the alternative standards analyzed for the 2025 baseline for areas other than California. For the 70 ppb scenario, emissions reductions were required for monitors in the Central and Northeast regions (see Chapter 3, Figure 3-3 for a depiction of the regions). For the 65 and 60 ppb scenarios, emissions reductions were applied in all regions with projected baseline DVs above these levels. For the 60 ppb scenario, additional VOC tons were identified in Chicago because some sites in that area experienced NO_x disbenefits meaning that the regional NO_x reductions resulted in ozone DV increases from below 60 ppb to above 60 ppb. Therefore it was not possible to identify a scenario in which regional NO_x reductions and maximum known VOC controls alone resulted in all Midwest monitors meeting a 60 ppb standard. An iterative approach was used determine a combination of NO_x and VOC emissions reductions that would not lead to over-control at either the NO_x-limited monitor with the highest design value or the VOC-limited monitor with the highest design value. Because of the regional approach we used for locations other than Texas, we were not able to geographically fine tune the control strategies, and thus there is some uncertainty in the actual amounts of emissions reductions estimated for attaining the alternative standards.

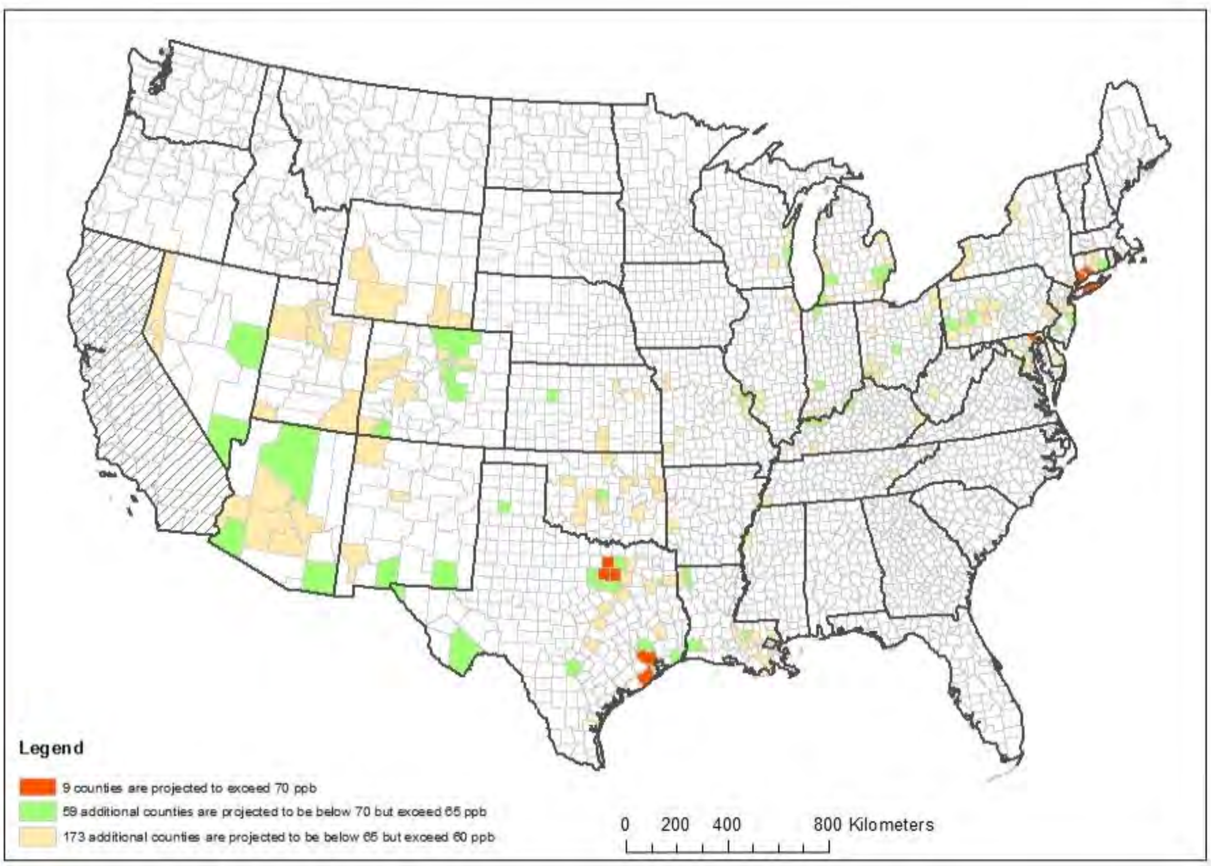


Figure 4-3. Projected Ozone Design Values in the 2025 Baseline Scenario

Figure 4-4 shows the counties projected to exceed the alternative standards analyzed for the post-2025 baseline analysis for California. For the California post-2025 alternative standard analyses, all known controls were applied in the baseline so incremental reductions are from unknown controls.

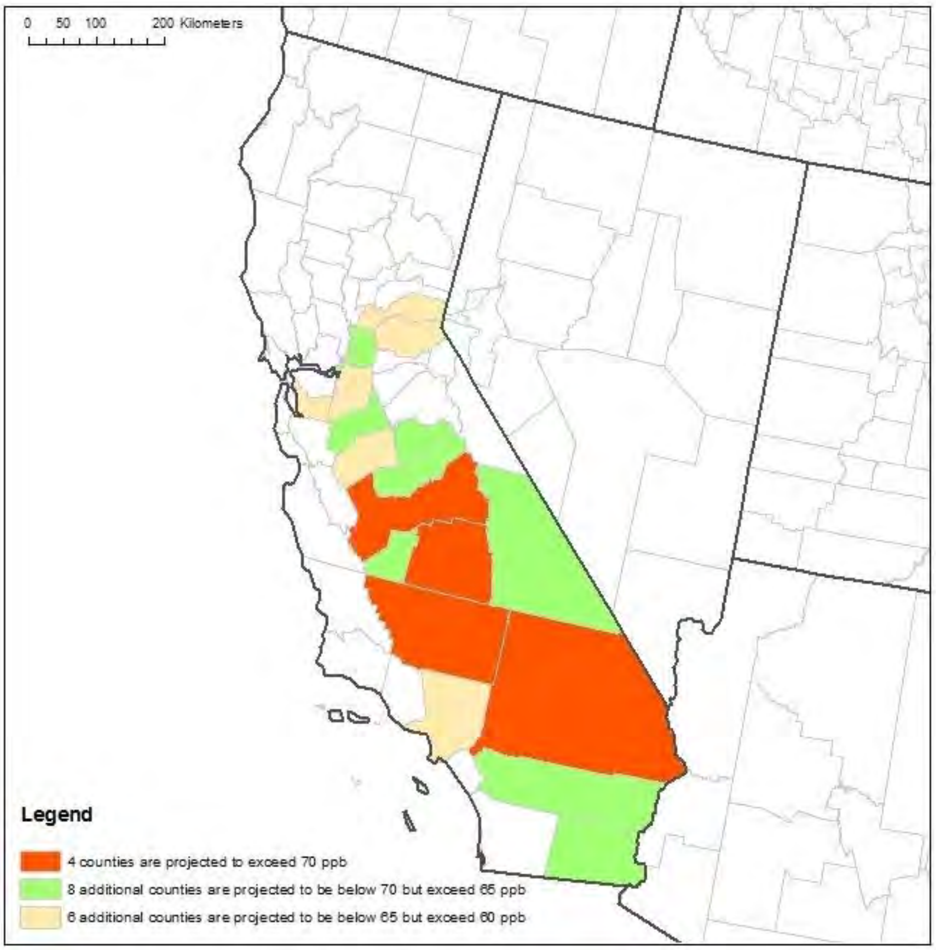


Figure 4-4. Projected Ozone Design Values in the post-2025 Baseline Scenario

4.3.1 Identifying Known Controls Needed to Meet the Alternative Standards

For the 2025 alternative control strategy analyses of 70, 65 and 60 ppb, known NO_x controls for four sectors were used: EGUs, non-EGU point, nonpoint, and nonroad mobile sources. In a smaller number of geographic areas, VOC controls were applied to non-EGU point and nonpoint sources.

For Texas, reductions were first applied to NO_x sources in the areas surrounding Dallas and Houston. Additional reductions then were applied to VOC sources in the area surrounding Houston, to NO_x sources in other parts of Texas, and to sources in the surrounding states. For the Northeast, Midwest, and Southwest areas of the country, reductions were applied to NO_x sources within the regions. For regions where additional reductions were needed, controls were applied to VOC sources in urban areas with the highest ozone design values in the region.[32]

For California, all known controls were applied in the baseline analysis so there were no known controls available to apply toward the incremental emissions reductions needed for the alternative analysis levels for post-2025. Maps of the areas where control measures were applied to demonstrate attainment of the alternative analysis levels are presented in Figures 4-5 and 4-6. Note that we do not account for between region transport of ozone, and therefore, especially for the 65 and 60 ppb alternative standard levels, we may be overstating the amount of emissions reductions needed for attainment.

[32] Texas, California, and the northeast were included in the hypothetical control scenario for an alternative standard of 70 ppb. Other regions were modeled as part of the 2025 emissions sensitivity simulations.

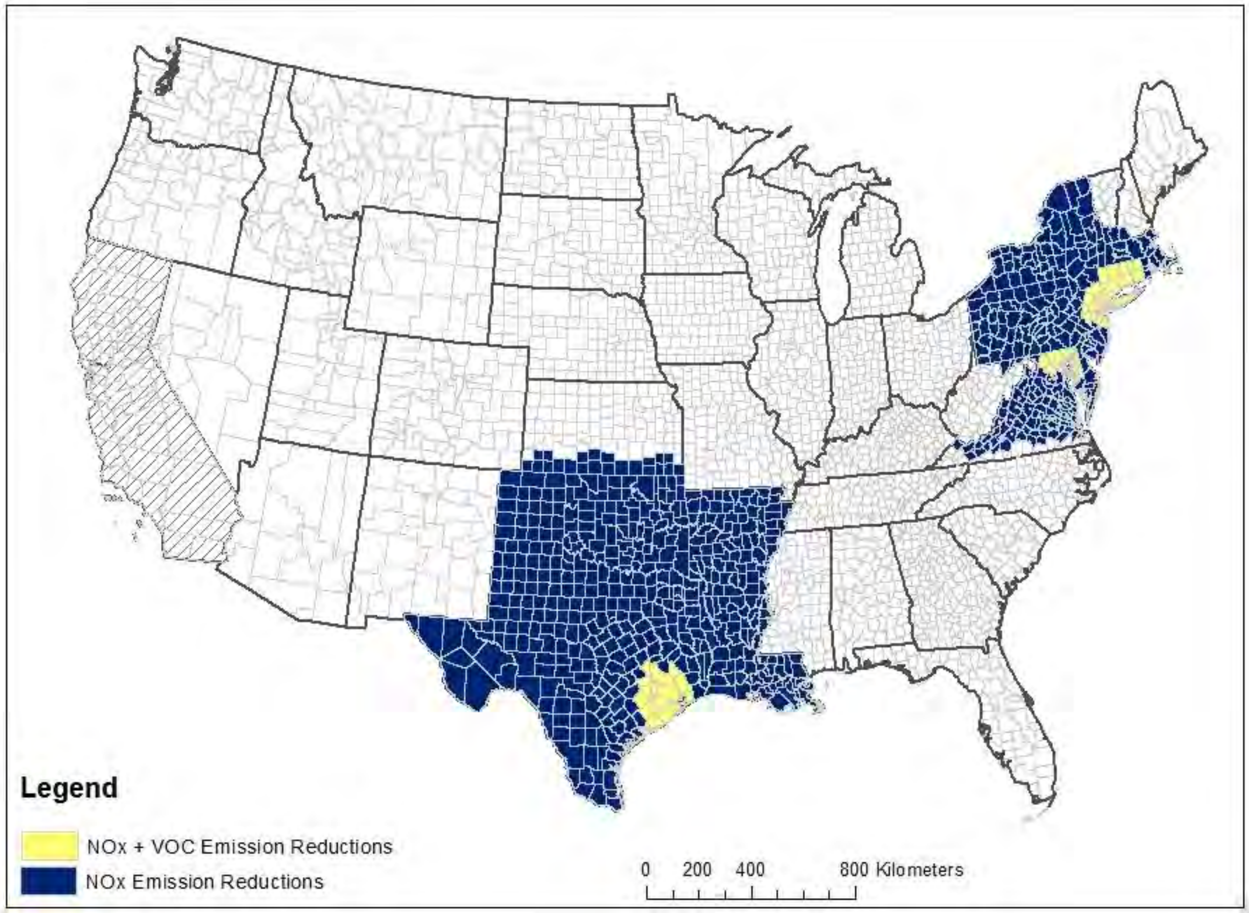


Figure 4-5. Counties Where Emissions Reductions Were Applied to Demonstrate Attainment with a 70 ppb Ozone Standard in the 2025 Analysis

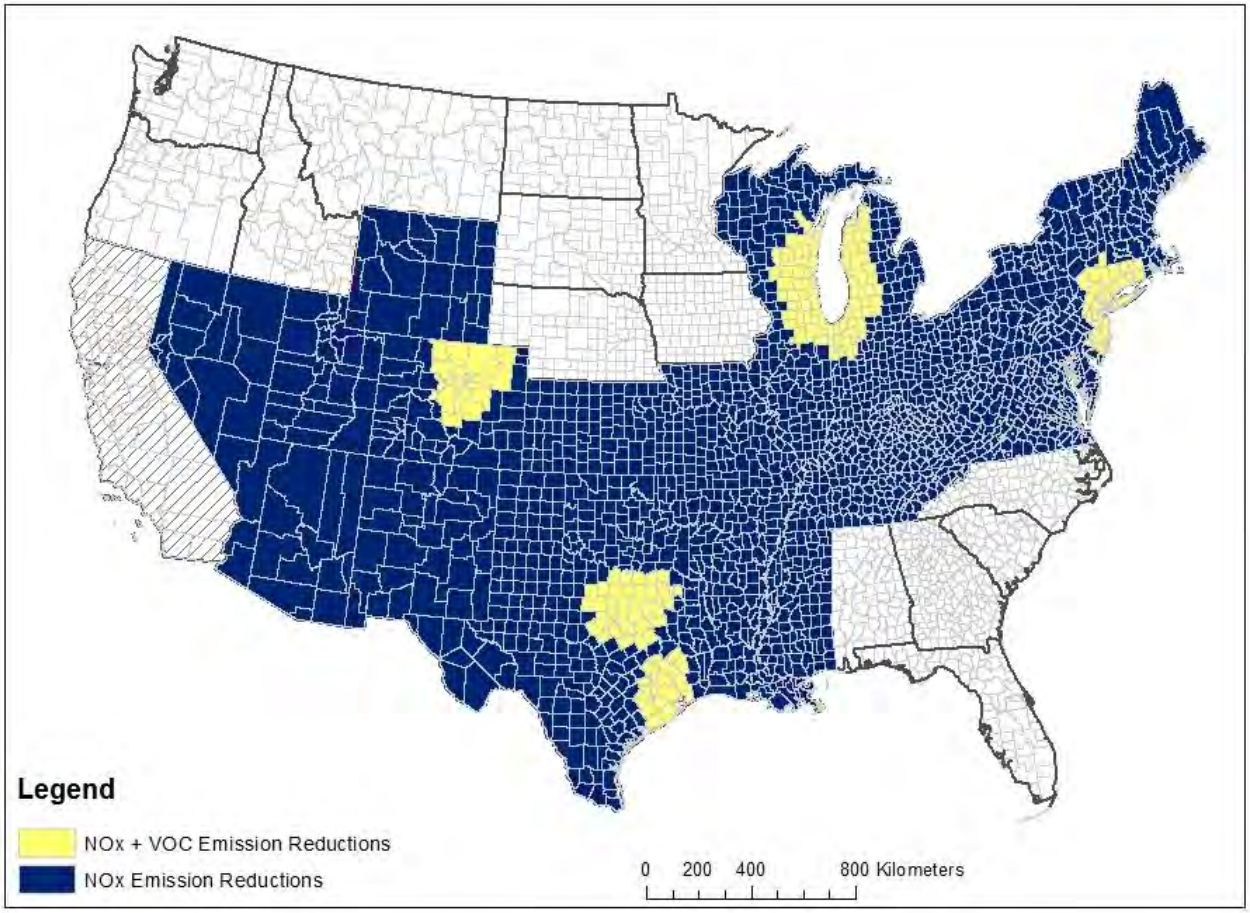


Figure 4-6. Counties Where Emissions Reductions Were Applied to Demonstrate Attainment with 65 and 60 ppb Ozone Standards in the 2025 Analyses

Table 4-4 shows the number of exceeding counties and the number of adjacent counties to which controls were applied for the alternative standards. For a complete list of geographic areas for the alternative standards see Appendix 4.A.

Table 4-4. Number of Counties with Exceedances and Number of Additional Counties Where Reductions Were Applied for the 2025 Alternative Standards Analyses - U.S., except California

Alternative Standard	Number of Counties with Exceedances	Number of Additional Counties Where Reductions Were Applied
70 ppb	15	479
65 ppb	115	1,925
60 ppb	289	1,791[a]

[a] Number of additional counties where reductions are applied declined for 60 ppb analysis because the number of overall counties in the analysis remained the same while the number of exceeding counties increased.

Tables 4-5 through 4-7 show the emissions reductions from known controls for the alternative standards analyzed. No exceedances were projected for the West region, outside of California, for the 70 ppb alternative standard. For the lower alternative standards of 65 and 60 ppb, similar controls were applied as were used in the 70 ppb analysis, but the geographic area in which they were applied increased. The largest emission reductions were in the non-EGU point source and nonpoint sectors. For details regarding emission reductions by control measure see Appendix 4.A

Table 4-5. Summary of Emission Reductions by Sector for Known Controls Applied to Demonstrate Nationwide Attainment with a 70 ppb Ozone Standard in 2025, except California (1,000 tons/year)[a]

Geographic Area	Emissions Sector	Baseline Emissions		Emission Reductions	
		NO_x	VOC	NO_x	VOC
East	EGU	884		25	-
	Non-EGU Point	1,485		210	0.98
	Nonpoint	1,487		260	54
	Nonroad	1,235		5	-
	Onroad	1,135		-	-
	Total	**6,226**		**490**	**55**
West	EGU	159		-	-
	Non-EGU Point	226		-	-
	Nonpoint	193		-	-
	Nonroad	219		-	-
	Onroad	197		-	-
	Total	**1,025**		**-**	**-**

[a] Emission reduction estimates are rounded to two significant figures.

Table 4-6. Summary of Emission Reductions by Sector for Known Controls Applied to Demonstrate Nationwide Attainment with a 65 ppb Ozone Standard in 2025 - except California (1,000 tons/year)[a]

Geographic Area	Emissions Sector	NO_x	VOC
East	EGU	170	-
	Non-EGU Point	410	3.6
	Nonpoint	420	95
	Nonroad	12	-
	Total	**1,000**	**99**
West	EGU	36	-
	Non-EGU Point	38	0.47
	Nonpoint	37	6.6
	Nonroad	1.3	-
	Total	**110**	**7**

[a] Emission reduction estimates are rounded to two significant figures.

Table 4-7. Summary of Emission Reductions by Sector for Known Controls Applied to Demonstrate Nationwide Attainment with a 60 ppb Ozone Standard in 2025 - except California (1,000 tons/year)[a]

Geographic Area	Emissions Sector	NO_x	VOC
East	EGU	170	-
	Non-EGU Point	410	4.2
	Nonpoint	420	99
	Nonroad	12	-
	Total	**1,000**	**100**
West	EGU	62	-
	Non-EGU Point	48	0.47
	Nonpoint	39	6.6
	Nonroad	1.3	-
	Total	**150**	**7**

[a] Emission reduction estimates are rounded to two significant figures.

4.3.2 Known Control Measures Analyzed

Known control measures were applied to electric generating units (EGU), non-EGU point, nonpoint (area), and nonroad mobile sources for demonstration of attainment with the current and alternative standards. The applied control measures were identified using the EPA's Control Strategy Tool (CoST) (U.S. EPA, 2014c), Integrated Planning Model (IPM), and NONROAD Model. CoST models emissions reductions and engineering costs associated with control strategies applied to point, area, and mobile sources of air pollutant emissions by matching control measures to emissions sources using algorithms such as "maximum emissions reduction", "least cost", and "apply measures in series". For this analysis, we applied the maximum emissions reduction algorithm. These controls are described further in Appendix 4.A. Specific controls were applied in the air quality modeling only for a portion of the analysis (for California, Texas, and the northeast, areas projected to exceed an alternative standard level of 70 ppb). A majority of the emission reductions needed were identified using the ozone sensitivity factors developed from the twelve emissions sensitivity simulations (see Chapter 3, Section 3.1 for a discussion of the development of the ozone sensitivity factors).

Nonpoint and nonroad mobile source emissions data are generated at the county level, and therefore controls for these emissions sectors were applied at the county level. EGU and non-EGU point source controls are applied to individual point sources. Control measures were applied to point and nonpoint sources of NO_x, including: industrial boilers, commercial and institutional boilers, reciprocating internal combustion engines in the oil and gas industry and other industries, glass manufacturing furnaces, and cement kilns. The analysis for nonroad mobile sources applied NO_x controls to diesel engines.

In a portion of the geographic areas where NOx controls were applied, the EPA also applied control measures to sources of VOC including surface coating, solvents, and fuel storage tanks. VOC reductions were analyzed in the urban areas with the highest ozone design values in each region: northern and southern California; Denver in the Southwest; Houston and Dallas in the Central region; Chicago, Detroit, and Louisville in the Midwest; New York, Baltimore, and Pittsburgh in the Northeast. Even among these areas, in some cases NO_x reductions necessary to bring the very highest urban areas into attainment made the VOC reductions unnecessary in other high ozone urban areas within the region.

To more accurately depict available controls, the EPA employed a decision rule in which controls were not applied to any non-EGU or nonpoint sources with less than 25 tons/year of emissions per pollutant. This decision rule is more inclusive of sources than the rule we employed in the previous O_3 and $PM_{2.5}$ NAAQS RIAs where we applied a minimum of 50 tons/year for each pollutant. We modified the decision rule for this NAAQS analysis to recognize the potential for emissions reductions in the large number of sources emitting in the 25-50 tons/year range in order to devise control strategies for full, or closer to full, attainment.

Historically, the reason for not applying controls to sources emitting less than 25 tons/year has been that many point sources with emissions below this level already have controls in place. For the analysis, we applied best engineering judgement to apply controls to select sources.

4.3.3 Emissions Reductions beyond Known Controls Needed to Meet the Alternative Standards

There were several areas where known controls did not achieve enough emissions reductions to attain the alternative standards of 70, 65, and 60 ppb. To complete the analysis, the EPA then estimated the additional emissions reductions beyond known controls needed to reach attainment, also referred to as unknown controls. For information on the methodology used to develop the emission reductions estimates, see Chapter 3. Table 4-8 shows the emissions reductions needed from unknown controls in 2025 for the U.S., except California, for the alternative standards analyzed. Table 4-9 shows the reductions needed from unknown controls for California for the post-2025 analysis.

Table 4-8. Summary of Emissions Reductions by Alternative Standard for Unknown Controls for 2025 - except California (1,000 tons/year)[a]

Alternative Standard	Region	NO_x	VOC
70 ppb[b]	East	150	-
	West	-	-
65 ppb[c]	East	750	-
	West	-	-
60 ppb[d]	East	1,900	41
	West	350	-

[a] Estimates are rounded to two significant figures.

[b] Unknown controls for the 70 ppb alternative standard are needed in the Northeast and Central regions (see Chapter 3 for a description of these regions).

[c] Unknown controls for the 65 ppb alternative standard are needed in the Northeast, Central, and Midwest regions (see Chapter 3 for a description of these regions).

[d] Unknown controls for the 60 ppb alternative standard are needed in the Northeast, Central, Midwest, and Southwest regions (see Chapter 3 for a description of these regions).

Table 4-9. Summary of Emissions Reductions by Alternative Level for Unknown Controls for post-2025 - California (1,000 tons/year)[a]

Alternative Standard	Region	NO_x	VOC
70 ppb	CA	53	-
65 ppb	CA	110	-

60 ppb	CA	140	-

[a] Estimates are rounded to two significant figures.

4.3.4 Summary of Emissions Reductions Needed to Meet the Alternative Standards

Table 4-10 summarizes the known and unknown emissions reductions needed to meet the alternative standard levels in 2025 for the East and West, except California. In the East for 2025, the unknown NOx reductions needed as percentage of the total rises from 23 percent to 66 percent as the alternative standard level decreases from 70 ppb to 60 ppb. Meanwhile, no unknown VOC reductions are needed in the East for the 70 ppb and 65 ppb levels. In the West (except California) for 2025, unknown NOx reductions are not needed until the 60 ppb level, when the unknown tons constitute about 70 percent of the total reductions needed. No unknown VOC reductions are needed in the West (except California) for 2025 for any of the alternative standard levels.

Table 4-10. Summary of Known and Unknown Emissions Reductions by Alternative Standard Levels in 2025, Except California (1,000 tons/year)[a]

		Alternative Standard		
Geographic Area	**Emissions Reductions**	**70 ppb**	**65 ppb**	**60 ppb**
East	NOx Known	490	1,000	1,000
	NOx Unknown	150	750	1,900
	% NOx Unknown	**23%**	**43%**	**66%**
	VOC Known	55	99	100
	VOC Unknown	0	0	41
	% VOC Unknown	**0%**	**0%**	**29%**
West	NOx Known	0	110	150
	NOx Unknown	0	0	350
	% NOx Unknown	**N/A**	**0%**	**70%**
	VOC Known	0	7	7
	VOC Unknown	0	0	0
	% VOC Unknown	**N/A**	**0%**	**0%**

[a] Estimates are rounded to two significant figures.

Table 4-11 shows again that there were no known NOx emissions reductions identified for meeting the alternative standard levels for post-2025 California and that 100 percent of the NOx tons needed were unknown. Meanwhile, no unknown VOC reductions are needed for the any of the alternative standard levels for post-025 California.

Table 4-11. Summary of Known and Unknown Emissions Reductions by Alternative Standard Levels for post-2025 - California (1,000 tons/year)[a]

		Alternative Standard		
Geographic Area	**Emissions Reductions**	**70 ppb**	**65 ppb**	**60 ppb**
California	NOx Known	0	0	0
	NOx Unknown	53	110	140
	% NOx Unknown	**100%**	**100%**	**100%**
	VOC Known	0	0	0
	VOC Unknown	0	0	0
	% VOC Unknown	**N/A**	**N/A**	**N/A**

[a] Estimates are rounded to two significant figures.

4.4 Limitations and Uncertainties

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

- **Illustrative control strategy:** A control strategy is the set of actions that States may take to meet a standard, such as which industries should be required to install end-of-pipe controls or certain types of equipment and technology. The illustrative control strategy analysis in this RIA presents only one potential pathway to attainment. The control strategies are not recommendations for how a revised ozone standard should be implemented, and States will make all final decisions regarding implementation strategies for the revised NAAQS. We do not presume that the control strategies presented in this RIA are an exhaustive list of possibilities for emissions reductions.
- **Emissions Inventories and Air Quality Modeling:** These serve as a foundation for the projected ozone values, control strategies and costs in this analysis and thus limitations and uncertainties for these inputs impact the results, especially for

issues such as future year emissions projections and information on controls currently in place at sources. Limitations and uncertainties for these inputs are discussed in previous chapters devoted to these subject areas. In addition, there are factors that affect emissions, such as economic growth and the makeup of the economy (e.g., growth in the oil and natural gas sector), that introduce additional uncertainty.

- **Projecting level and geographic scope of exceedances:** Estimates of the geographic areas that would exceed revised alternative levels of the standard in a future year, and the level to which those areas would exceed, are approximations based on a number of factors. The actual nonattainment determinations that would result from a revised standard will likely depend on the consideration of local issues, changes in source operations between the time of this analysis and implementation of a new standard, and changes in control technology over time.
- **Assumptions about the baseline:** There is significant uncertainty about the illustration of the impact of rules, especially the Clean Power Plan because it is a proposal and because it contains significant flexibility for states to determine how to choose measures to comply with the standard.
- **Applicability of control measures:** The applicability of a control measure to a specific source varies depending on a number of process equipment factors such as age, design, capacity, fuel, and operating parameters. These can vary considerably from source to source and over time. This analysis makes assumptions across broad categories of sources nationwide.
- **Control measure advances over time:** The control measures applied do not reflect potential effects of technological change that may be available in future years and the effects of "learning by doing" or "learning by researching" are not accounted for in the emissions reduction estimates. Thus, all estimates of impacts associated with control measures applied reflect our current knowledge, and not projections, of the measures' effectiveness. In our analysis, we do not have the necessary data for cumulative output, fuel sales, or emissions reductions for all sectors included in order to properly generate control costs that reflect learning-curve impacts or the

impacts of technological change. We believe the effect of including these impacts would be to lower our estimates of costs for our projected year control strategies.

- **Pollutants to be targeted:** Local knowledge of atmospheric chemistry in each geographic area may result in a different prioritization of pollutants (VOC and NO_x) for control. For the baseline in this analysis, we included only promulgated or proposed rules, but that there may be additional regulations promulgated in the future that reduce NOx or VOC emissions. These regulations could reduce the current baseline levels of emissions.

4.5 References

U.S. Environmental Protection Agency (U.S. EPA). 2014a. Control of Air Pollution from Motor Vehicles: Tier 3 Motor Vehicle Emission and Fuel Standards. Office of Transportation and Air Quality. Available at http://www.epa.gov/otaq/tier3.htm.

U.S. Environmental Protection Agency (U.S. EPA). 2014b. Proposed Carbon Pollution Guidelines for Existing Power Plants and Emission Standards for Modified and Reconstructed Power Plants. Available at http://www2.epa.gov/sites/production/files/2014-06/documents/20140602ria-clean-power-plan.pdf

U.S. Environmental Protection Agency (U.S. EPA). 2014c. Control Strategy Tool (CoST) Documentation Report. Office of Air Quality Planning and Standards, Research Triangle Park, NC. Available at http://www.epa.gov/ttnecas1/cost.htm.

U.S. Environmental Protection Agency (2014d) Preparation of Emissions Inventories for the Version 6.1, 2011 Emissions Modeling Platform (http://www.epa.gov/ttn/chief/emc

APPENDIX 4: CONTROL STRATEGIES AND EMISSIONS REDUCTIONS

Overview

Chapter 4 describes the approach that EPA used in applying control measures to demonstrate attainment of alternative ozone standard levels of 70, 65, and 60 ppb and estimating the resulting emissions reductions. This Appendix contains more detailed information about the control strategy analyses, including the control measures that were applied and the geographic areas in which they were applied.

4A.1 Types of Control Measures

Several types of control measures were applied in the analyses for the baseline and alternative standard levels. These can be grouped into the following classes:

- Max NO_x Reductions – NO_x control measures for nonEGU point, nonpoint, and nonroad sources. For each of these sources, we identified the most effective control (i.e., control with the highest percent reduction) that could be applied to the source, given the following constraints:
  - the source must emit at least 50 tons/yr of NO_x (see description of controls on smaller sources below);
  - any control for nonEGU point sources must result in a reduction of NO_x emissions of at least 5 tons/yr; and
  - any replacement control (i.e., a more effective control replacing an existing control) must achieve at least 10% more reduction than the existing control (e.g., we would not replace a 60% control with a 65% control).
- NO_x Reductions from EGU SCRs – SCRs applied to coal-fired EGUs where no SCR is currently in place.
- NO_x 25-50 TPY Source Reductions – Similar to the Max NO_x Reductions above, except for smaller sources in the 25-50 ton/year NO_x emissions range.
- Max VOC Reductions – Similar to Max NO_x Reductions described above, except this includes only VOC controls.

4A.2 Application of Control Measures in Geographic Areas

Control measures were applied to geographic areas including or adjacent to areas that were projected to exceed the baseline and alternative standards. See Tables 4A-1 to 4A-4 for a listing of the NO_x and VOC control groups and geographic areas to which they were applied.

Table 4A-1. Geographic Areas for Application of NOx Controls in the Baseline and Alternative Standard Analyses - U.S., except California[a]

Geographic Areas and Control Groups	Baseline	70 ppb	65 ppb	60 ppb
EAST				
Central Region				
Max NOx Reductions within TX buffer	x	x	x	x
NOx EGU SCR within TX buffer		x	x	x
NOx 25-50 tpy Source controls within TX buffer		x	x	x
Unknown control NOx Reductions within TX buffer		U	U	U
Max NOx Reductions outside of TX buffer		x	x	x
NOx EGU SCR outside of TX buffer			x	x
NOx 25-50 tpy Source controls outside of TX buffer			x	x
Unknown control NOx Reductions outside of TX buffer			U	U
Northeast Region				
Max NOx Reductions within Northeast buffer		x	x	x
NOx EGU SCR within Northeast buffer		x	x	x
NOx 25-50 tpy Source controls within Northeast buffer		x	x	x
Unknown control NOx Reductions within Northeast buffer		U	U	U
Max NOx Reductions outside of Northeast buffer		x	x	x
NOx EGU SCR outside of Northeast buffer			x	x
NOx 25-50 tpy Source controls outside of Northeast buffer			x	x
Unknown control NOx Reductions outside NE buffer			U	U
Midwest Region				
Max NOx Reductions in Midwest Region			x	x
NOx EGU SCR in Midwest Region			x	x
NOx 25-50 tpy Source controls in Midwest Region			x	x
Unknown control NOx Reductions in MW Region			U	U
WEST				
Southwest Region				
Max NOx Reductions in Southwest Region			x	x
NOx EGU SCR in Southwest Region			x	x
NOx 25-50 tpy Source controls in Southwest Region				x
Unknown control NOx Reductions in SW Region				U

[a] “x” indicates known controls were applied; “U” indicates unknown control reductions.

Table 4A-2. Geographic Areas for Application of VOC Controls in the Baseline and Alternative Standard Analyses - U.S., except California[a]

Geographic Areas and Control Groups	Baseline	70 ppb	65 ppb	60 ppb
EAST				
Central Region				
Max VOC Reductions within Houston buffer		x	x	x
Max VOC Reductions within Dallas buffer			x	x
Northeast Region				
Max VOC Reductions within CT-NJ-NY buffer		x	x	x
Max VOC Reductions within Baltimore buffer		x		
Midwest Region				
Max VOC Reductions in Chicago buffer			x	x
Unknown control VOC Reductions in Chicago				U
WEST				
Southwest Region				
Max VOC Reductions in Denver buffer			x	x

[a] "x" indicates known controls were applied; "U" indicates unknown control reductions.

Table 4A-3. Geographic Areas for Application of NO_x Controls in the Baseline and Alternative Standard Analyses - California[a]

Geographic Areas and Control Groups	Baseline	70 ppb	65 ppb	60 ppb
California				
Max NOx Reductions within California (CA) buffer	x			
NOx 25-50 tpy Source controls within N. CA buffer	x			
Unknown Control NOx Reductions within N. CA buffer	U	U	U	U
NOx 25-50 tpy Source controls within S. CA buffer	x			
Unknown Control NOx Reductions within S. CA buffer	U	U	U	U

[a] "x" indicates known controls were applied; "U" indicates unknown control reductions.

Table 4A-4. Geographic Areas for Application of VOC Controls in the Baseline and Alternative Standard Analyses - California[a]

Geographic Areas and Control Groups	Baseline	70 ppb	65 ppb	60 ppb
California				
Max VOC Reductions within N. California buffer	x			
Max VOC Reductions within S. California buffer	x			

[a] "x" indicates known controls were applied.

4A.3 NO_x Control Measures for NonEGU Point Sources

Several types of NO_x control technologies exist for non-EGU point sources: selective catalytic reduction (SCR), selective noncatalytic reduction (SNCR), natural gas reburn (NGR), coal reburn, and low-NO_x burners (LNB). 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 do not make sense 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 point 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. NO_x control measures available for cement kilns include those available to industrial boilers, namely LNB, SCR, and SNCR. 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 at such plants. LNB, SCR, and SCR plus steam injection (SI) are available measures for combustion turbines. Finally, SNCR is an available control technology at incinerators.

Tables 4A-5 through 4A-12 contain lists of the NO_x and VOC control measures applied in these analyses for non-EGU point sources,EGUs, nonpoint sources, and nonroad sources. The table also presents the associated emission reductions for the baseline and alternative standard analyses. The number of geographic areas in which they were applied expanded as the level of the alternative standard analyzed became more stringent.

Table 4A-5. NO_x Control Measures Applied in the Baseline Analysis

NO_x Control Measure	Reductions (tons/yr)
Adjust Air to Fuel Ratio and Ignition Retard - Gas Fired IC Engines	694
Biosolid Injection Technology - Cement Kilns	1,021
Episodic Ban - Open Burning	570
Ignition Retard - IC Engines	30
Low Emission Combustion - Gas Fired Lean Burn IC Engines	1,552
Low NOx Burner - Commercial/Institutional Boilers & IC Engines	6,699
Low NOx Burner - Industr/Commercial/Institutional (ICI) Boilers	165
Low NOx Burner - Industrial Combustion	270
Low NOx Burner - Lime Kilns	309
Low NOx Burner - Natural Gas-Fired Turbines	4,735
Low NOx Burner - Residential Furnaces	5,381
Low NOx Burner - Residential Water Heaters & Space Heaters	6,605
Low NOx Burner and Flue Gas Recirculation - Coke Oven/Blast Furnace	36
Low NOx Burner and Flue Gas Recirculation - Iron & Steel Mills - Reheating	42
Low NOx Burner and SCR - Coal-Fired ICI Boilers	1,174
Low NOx Burner and SCR - Industr/Commercial/Institutional Boilers	5,286
Natural Gas Reburn - Natural Gas-Fired EGU Boilers	79
Nonroad Diesel Retrofits & Engine Rebuilds - e.g., Construction Equipment	4,998
Non-Selective Catalytic Reduction (NSCR) - 4 Cycle Rich Burn IC Engines	20,008
OXY-Firing - Glass Manufacturing	5,429
Selective Catalytic Reduction (SCR) - Cement Kilns	2,982
Selective Catalytic Reduction (SCR) - Coal Fired EGU Boilers	76
Selective Catalytic Reduction (SCR) - Fluid Catalytic Cracking Units	945
Selective Catalytic Reduction (SCR) - IC Engines, Diesel	2,041
Selective Catalytic Reduction (SCR) - ICI Boilers	3,218
Selective Catalytic Reduction (SCR) - Industrial Incinerators	1,062
Selective Catalytic Reduction (SCR) - Petroleum Refinery Gas-Fired Process Heaters	456
Selective Catalytic Reduction (SCR) - Sludge Incineration	366
Selective Catalytic Reduction (SCR) - Utility Boilers	55
Selective Non-Catalytic Reduction (SNCR) - Comm./Inst. Incinerators	16
Selective Non-Catalytic Reduction (SNCR) - Industrial Incinerators	130
Selective Non-Catalytic Reduction (SNCR) - Utility Boilers	158

Table 4A-6. VOC Control Measures Applied in the Baseline Analysis

VOC Control Measure	Reductions (tons/yr)
Control of Fugitive Releases - Oil & Natural Gas Production	39
Control Technology Guidelines - Wood Furniture Surface Coating	777
Gas Recovery - Municipal Solid Waste Landfill	2,064
Improved Work Practices, Material Substitution, Add-On Controls - Cleaning Solvents	155
Improved Work Practices, Material Substitution, Add-On Controls - Printing	77
Incineration - Other	181
Incineration - Surface Coating	6,362
Low-VOC Coatings and Add-On Controls - Surface Coating	945
LPV Relief Valve - Underground Tanks	353
Permanent Total Enclosure (PTE) - Surface Coating	100
Process Modification - Oil and Natural Gas Production	4,311
RACT - Graphic Arts	3,423
Reduced Solvent Utilization - Surface Coating	342
Reformulation - Aerosol Paints	12
Reformulation - Architectural Coatings	1,912
Reformulation - Industrial Adhesives	5,076
Reformulation-Process Modification - Automobile Refinishing	4,665
Reformulation-Process Modification - Cold Cleaning	5,877
Reformulation-Process Modification - Cutback Asphalt	6,478
Reformulation-Process Modification - Open Top Degreasing	47
Reformulation-Process Modification - Surface Coating	4,225
Wastewater Treatment Controls- POTWs	237

Table 4A-7. NOx Control Measures Applied in the 70 ppb Alternative Standard Analysis

NO_x Control Measure	Reductions (tons/yr)
Adjust Air to Fuel Ratio and Ignition Retard - Gas Fired IC Engines	8,091
Biosolid Injection Technology - Cement Kilns	1,315
Episodic Ban - Open Burning	2,086
Ignition Retard - IC Engines	486
Low Emission Combustion - Gas Fired Lean Burn IC Engines	83,046
Low NOx Burner - Coal Cleaning	255
Low NOx Burner - Commercial/Institutional Boilers & IC Engines	20,004
Low NOx Burner - Industr/Commercial/Institutional (ICI) Boilers	2,290
Low NOx Burner - Industrial Combustion	495
Low NOx Burner - Lime Kilns	1,870
Low NOx Burner - Natural Gas-Fired Turbines	14,408
Low NOx Burner - Residential Furnaces	12,056
Low NOx Burner - Residential Water Heaters & Space Heaters	18,843
Low NOx Burner and Flue Gas Recirculation - Coke Oven/Blast Furnace	350

NO_x Control Measure	Reductions (tons/yr)
Low NOx Burner and Flue Gas Recirculation - Iron & Steel Mills - Reheating	388
Low NOx Burner and Over Fire Air - Utility Boilers	178
Low NOx Burner and SCR - Coal-Fired ICI Boilers	7,197
Low NOx Burner and SCR - Industr/Commercial/Institutional Boilers	22,632
Low NOx Burner and SNCR - Industr/Commercial/Institutional Boilers	153
Low Sulfur Fuel - Miscellaneous	2,892
Natural Gas Reburn - Natural Gas-Fired EGU Boilers	262
Nonroad Diesel Retrofits & Engine Rebuilds - e.g., Construction Equipment	4,984
Non-Selective Catalytic Reduction (NSCR) - 4 Cycle Rich Burn IC Engines	189,563
Non-Selective Catalytic Reduction (NSCR) - Nitric Acid Mfg	659
OXY-Firing - Glass Manufacturing	17,155
SCR and Flue Gas Recirculation - Fluid Catalytic Cracking Units	163
SCR and Flue Gas Recirculation - ICI Boilers	307
Selective Catalytic Reduction (SCR) - Ammonia Mfg	4,476
Selective Catalytic Reduction (SCR) - Cement Kilns	18,260
Selective Catalytic Reduction (SCR) - Fluid Catalytic Cracking Units	2,995
Selective Catalytic Reduction (SCR) - IC Engines, Diesel	1,675
Selective Catalytic Reduction (SCR) - ICI Boilers	6,049
Selective Catalytic Reduction (SCR) - Industrial Combustion	1,111
Selective Catalytic Reduction (SCR) - Industrial Incinerators	717
Selective Catalytic Reduction (SCR) - Petroleum Refinery Gas-Fired Process Heaters	4,757
Selective Catalytic Reduction (SCR) - Process Heaters	19
Selective Catalytic Reduction (SCR) - Sludge Incineration	7,991
Selective Catalytic Reduction (SCR) - Utility Boilers	30,482
Selective Non-Catalytic Reduction (SNCR) - Coke Mfg	1,543
Selective Non-Catalytic Reduction (SNCR) - Comm./Inst. Incinerators	1,082
Selective Non-Catalytic Reduction (SNCR) - ICI Boilers	154
Selective Non-Catalytic Reduction (SNCR) - Industrial Incinerators	482
Selective Non-Catalytic Reduction (SNCR) - Miscellaneous	31
Selective Non-Catalytic Reduction (SNCR) - Municipal Waste Combustors	304
Ultra-Low NOx Burner - Process Heaters	229

Table 4A-8. VOC Control Measures Applied in the 70 ppb Alternative Standard Analysis

VOC Control Measure	Reductions (tons/yr)
Control of Fugitive Releases - Oil & Natural Gas Production	16
Control Technology Guidelines - Wood Furniture Surface Coating	1,184
Flare - Petroleum Flare	110
Gas Recovery - Municipal Solid Waste Landfill	242
Improved Work Practices, Material Substitution, Add-On Controls - Cleaning Solvents	10
Improved Work Practices, Material Substitution, Add-On Controls - Printing	148
Incineration - Other	350
Incineration - Surface Coating	14,071
Low-VOC Coatings and Add-On Controls - Surface Coating	209
LPV Relief Valve - Underground Tanks	4,011
Permanent Total Enclosure (PTE) - Surface Coating	446
RACT - Graphic Arts	3,054
Reduced Solvent Utilization - Surface Coating	2,541
Reformulation - Architectural Coatings	17,678
Reformulation - Industrial Adhesives	1,793
Reformulation-Process Modification - Automobile Refinishing	3,209
Reformulation-Process Modification - Cold Cleaning	1,600
Reformulation-Process Modification - Cutback Asphalt	817
Reformulation-Process Modification - Surface Coating	3,571
Solvent Recovery System - Printing/Publishing	31
Wastewater Treatment Controls- POTWs	217

Table 4A-9. NOx Control Measures Applied in the 65 ppb Alternative Standard Analysis

NO_x Control Measure	Reductions (tons/yr)
Adjust Air to Fuel Ratio and Ignition Retard - Gas Fired IC Engines	27,057
Biosolid Injection Technology - Cement Kilns	6,423
Episodic Ban - Open Burning	4,423
Ignition Retard - IC Engines	761
Low Emission Combustion - Gas Fired Lean Burn IC Engines	174,033
Low NOx Burner - Coal Cleaning	518
Low NOx Burner - Commercial/Institutional Boilers & IC Engines	40,691
Low NOx Burner - Industr/Commercial/Institutional (ICI) Boilers	4,226
Low NOx Burner - Industrial Combustion	1,578
Low NOx Burner - Lime Kilns	5,273
Low NOx Burner - Miscellaneous Sources	21
Low NOx Burner - Natural Gas-Fired Turbines	26,982
Low NOx Burner - Residential Furnaces	16,660
Low NOx Burner - Residential Water Heaters & Space Heaters	57,314
Low NOx Burner - Steel Foundry Furnaces	294
Low NOx Burner - Surface Coating Ovens	26
Low NOx Burner and Flue Gas Recirculation - Coke Oven/Blast Furnace	420

NO_x Control Measure	Reductions (tons/yr)
Low NOx Burner and Flue Gas Recirculation - Iron & Steel Mills - Reheating	892
Low NOx Burner and Over Fire Air - Utility Boilers	333
Low NOx Burner and SCR - Coal-Fired ICI Boilers	30,790
Low NOx Burner and SCR - Industr/Commercial/Institutional Boilers	37,948
Low NOx Burner and SNCR - Industr/Commercial/Institutional Boilers	263
Low Sulfur Fuel - Miscellaneous	3,194
Natural Gas Reburn - Natural Gas-Fired EGU Boilers	480
Nonroad Diesel Retrofits & Engine Rebuilds - e.g., Construction Equipment	12,863
Non-Selective Catalytic Reduction (NSCR) - 4 Cycle Rich Burn IC Engines	323,763
Non-Selective Catalytic Reduction (NSCR) - Nitric Acid Mfg	927
OXY-Firing - Glass Manufacturing	29,546
SCR and Flue Gas Recirculation - Fluid Catalytic Cracking Units	185
SCR and Flue Gas Recirculation - ICI Boilers	318
SCR and Flue Gas Recirculation - Process Heaters	548
Selective Catalytic Reduction (SCR) - Ammonia Mfg	5,151
Selective Catalytic Reduction (SCR) - Cement Kilns	36,013
Selective Catalytic Reduction (SCR) - Fluid Catalytic Cracking Units	4,108
Selective Catalytic Reduction (SCR) - IC Engines, Diesel	8,905
Selective Catalytic Reduction (SCR) - ICI Boilers	18,284
Selective Catalytic Reduction (SCR) - Industrial Combustion	4,428
Selective Catalytic Reduction (SCR) - Industrial Incinerators	1,006
Selective Catalytic Reduction (SCR) - Iron Ore Processing	1,195
Selective Catalytic Reduction (SCR) - Petroleum Refinery Gas-Fired Process Heaters	7,691
Selective Catalytic Reduction (SCR) - Process Heaters	19
Selective Catalytic Reduction (SCR) - Sludge Incineration	9,007
Selective Catalytic Reduction (SCR) - Space Heaters	272
Selective Catalytic Reduction (SCR) - Utility Boilers	211,200
Selective Non-Catalytic Reduction (SNCR) - Coke Mfg	2,399
Selective Non-Catalytic Reduction (SNCR) - Comm./Inst. Incinerators	1,260
Selective Non-Catalytic Reduction (SNCR) - ICI Boilers	170
Selective Non-Catalytic Reduction (SNCR) - Industrial Combustion	69
Selective Non-Catalytic Reduction (SNCR) - Industrial Incinerators	1,502
Selective Non-Catalytic Reduction (SNCR) - Miscellaneous	132
Selective Non-Catalytic Reduction (SNCR) - Municipal Waste Combustors	1,351
Selective Non-Catalytic Reduction (SNCR) - Utility Boilers	329
Ultra-Low NOx Burner - Process Heaters	300

Table 4A-10. VOC Control Measures Applied in the 65 ppb Alternative Standard Analysis

VOC Control Measure	Reductions (tons/yr)
Control of Fugitive Releases - Oil & Natural Gas Production	31
Control Technology Guidelines - Wood Furniture Surface Coating	1,928
Flare - Petroleum Flare	110
Gas Recovery - Municipal Solid Waste Landfill	332
Improved Work Practices, Material Substitution, Add-On Controls - Cleaning Solvents	245
Improved Work Practices, Material Substitution, Add-On Controls - Printing	564
Incineration - Other	379
Incineration - Surface Coating	25,785
Low VOC Adhesives and Improved Application Methods - Industrial Adhesives	223
Low-VOC Coatings and Add-On Controls - Surface Coating	1,267
LPV Relief Valve - Underground Tanks	7,317
Permanent Total Enclosure (PTE) - Surface Coating	1,554
Petroleum and Solvent Evaporation - Surface Coating Operations	159
RACT - Graphic Arts	5,988
Reduced Solvent Utilization - Surface Coating	2,796
Reformulation - Architectural Coatings	39,057
Reformulation - Industrial Adhesives	1,698
Reformulation-Process Modification - Automobile Refinishing	5,264
Reformulation-Process Modification - Cutback Asphalt	3,058
Reformulation-Process Modification - Surface Coating	6,913
Solvent Recovery System - Printing/Publishing	842
Solvent Substitution and Improved Application Methods - Fiberglass Boat Mfg	14
Wastewater Treatment Controls- POTWs	242

Table 4A-11. NOx Control Measures Applied in the 60 ppb Alternative Standard Analysis

NO_x Control Measure	Reductions (tons/yr)
Adjust Air to Fuel Ratio and Ignition Retard - Gas Fired IC Engines	27,547
Biosolid Injection Technology - Cement Kilns	6,423
Episodic Ban - Open Burning	4,561
Ignition Retard - IC Engines	821
Low Emission Combustion - Gas Fired Lean Burn IC Engines	178,146
Low NOx Burner - Coal Cleaning	518
Low NOx Burner - Commercial/Institutional Boilers & IC Engines	40,876
Low NOx Burner - Industr/Commercial/Institutional (ICI) Boilers	4,319
Low NOx Burner - Industrial Combustion	1,578
Low NOx Burner - Lime Kilns	5,273
Low NOx Burner - Miscellaneous Sources	35
Low NOx Burner - Natural Gas-Fired Turbines	29,155
Low NOx Burner - Residential Furnaces	17,103
Low NOx Burner - Residential Water Heaters & Space Heaters	57,726
Low NOx Burner - Steel Foundry Furnaces	294

NO_x Control Measure	Reductions (tons/yr)
Low NOx Burner - Surface Coating Ovens	26
Low NOx Burner and Flue Gas Recirculation - Coke Oven/Blast Furnace	420
Low NOx Burner and Flue Gas Recirculation - Iron & Steel Mills - Reheating	892
Low NOx Burner and Over Fire Air - Utility Boilers	333
Low NOx Burner and SCR - Coal-Fired ICI Boilers	30,817
Low NOx Burner and SCR - Industr/Commercial/Institutional Boilers	38,350
Low NOx Burner and SNCR - Industr/Commercial/Institutional Boilers	263
Low Sulfur Fuel - Miscellaneous	3,194
Natural Gas Reburn - Natural Gas-Fired EGU Boilers	502
Nonroad Diesel Retrofits & Engine Rebuilds - e.g., Construction Equipment	12,863
Non-Selective Catalytic Reduction (NSCR) - 4 Cycle Rich Burn IC Engines	325,380
Non-Selective Catalytic Reduction (NSCR) - Nitric Acid Mfg	958
OXY-Firing - Glass Manufacturing	29,546
SCR and Flue Gas Recirculation - Fluid Catalytic Cracking Units	185
SCR and Flue Gas Recirculation - ICI Boilers	318
SCR and Flue Gas Recirculation - Process Heaters	548
Selective Catalytic Reduction (SCR) - Ammonia Mfg	5,151
Selective Catalytic Reduction (SCR) - Cement Kilns	36,013
Selective Catalytic Reduction (SCR) - Fluid Catalytic Cracking Units	4,135
Selective Catalytic Reduction (SCR) - IC Engines, Diesel	9,288
Selective Catalytic Reduction (SCR) - ICI Boilers	18,284
Selective Catalytic Reduction (SCR) - Industrial Combustion	4,428
Selective Catalytic Reduction (SCR) - Industrial Incinerators	1,006
Selective Catalytic Reduction (SCR) - Iron Ore Processing	1,195
Selective Catalytic Reduction (SCR) - Petroleum Refinery Gas-Fired Process Heaters	7,691
Selective Catalytic Reduction (SCR) - Process Heaters	19
Selective Catalytic Reduction (SCR) - Sludge Incineration	9,007
Selective Catalytic Reduction (SCR) - Space Heaters	335
Selective Catalytic Reduction (SCR) - Utility Boilers	236,736
Selective Non-Catalytic Reduction (SNCR) - Coke Mfg	2,399
Selective Non-Catalytic Reduction (SNCR) - Comm./Inst. Incinerators	1,260
Selective Non-Catalytic Reduction (SNCR) - ICI Boilers	170
Selective Non-Catalytic Reduction (SNCR) - Industrial Combustion	92
Selective Non-Catalytic Reduction (SNCR) - Industrial Incinerators	1,502
Selective Non-Catalytic Reduction (SNCR) - Miscellaneous	132
Selective Non-Catalytic Reduction (SNCR) - Municipal Waste Combustors	1,351
Selective Non-Catalytic Reduction (SNCR) - Utility Boilers	329
Ultra-Low NOx Burner - Process Heaters	300

Table 4A-12. VOC Control Measures Applied in the 60 ppb Alternative Standard Analysis

VOC Control Measure	Reductions (tons/yr)
Control of Fugitive Releases - Oil & Natural Gas Production	33
Control Technology Guidelines - Wood Furniture Surface Coating	2,063
Flare - Petroleum Flare	110
Gas Recovery - Municipal Solid Waste Landfill	372
Improved Work Practices, Material Substitution, Add-On Controls - Cleaning Solvents	265
Improved Work Practices, Material Substitution, Add-On Controls - Printing	564
Incineration - Other	379
Incineration - Surface Coating	26,109
Low VOC Adhesives and Improved Application Methods - Industrial Adhesives	237
Low-VOC Coatings and Add-On Controls - Surface Coating	1,523
LPV Relief Valve - Underground Tanks	7,610
Permanent Total Enclosure (PTE) - Surface Coating	1,857
Petroleum and Solvent Evaporation - Surface Coating Operations	237
RACT - Graphic Arts	6,273
Reduced Solvent Utilization - Surface Coating	2,886
Reformulation - Architectural Coatings	40,866
Reformulation - Industrial Adhesives	1,698
Reformulation-Process Modification - Automobile Refinishing	5,633
Reformulation-Process Modification - Cutback Asphalt	3,571
Reformulation-Process Modification - Surface Coating	7,072
Solvent Recovery System - Printing/Publishing	888
Solvent Substitution and Improved Application Methods - Fiberglass Boat Mfg	14
Wastewater Treatment Controls- POTWs	242

4A.4 VOC Control Measures for Non-EGU Point Sources

VOC controls were applied to a number of non-EGU point sources. Some examples are permanent total enclosures (PTE) applied to paper and web coating operations and fabric operations, and incinerators or thermal oxidizers applied to wood products and marine surface coating operations. A PTE confines VOC emissions to a particular area where they can be destroyed or used in a way that limits emissions to the outside atmosphere, and an incinerator or thermal oxidizer destroys VOC emissions through exposure to high temperatures (2,000 degrees Fahrenheit or higher). Another control is petroleum and solvent evaporation applied to printing and publishing sources as well as to surface coating operations.

4A.5 NO_x Control Measures for Nonpoint (Area) and Nonroad Sources

The nonpoint source sector of the emissions inventory is composed of sources that are generally too small and/or numerous to estimate emissions on an individual source basis (e.g., dry cleaners, residential furnaces, woodstoves, fireplaces, backyard waste burning, etc). Instead, we estimate their emissions for each county as a whole, often using an emissions factor that is applied to a surrogate of activity such as population or number of houses.

Control measures for nonpoint sources are also applied at the county level, i.e., to the county level emissions as a whole. Several control measures were applied to NO_x emissions from nonpoint sources. One is low NO_x burner technology to reduce NO_x emissions. This control is applied to industrial oil, natural gas, and coal combustion sources. Other nonpoint source controls include the installation of low-NO_x space heaters and water heaters in commercial and institutional sources, and episodic bans on open burning. The open burning control measure applied to yard waste and land clearing debris. It consists of periodic daily bans on burning such waste, as the predicted ozone levels indicate that such burning activities should be postponed. This control measure is not applied to any prescribed burning activities.

Retrofitting diesel nonroad equipment can provide NO_x and HC benefits. The retrofit strategies included in the RIA nonroad retrofit measure are:

• Installation of emissions after-treatment devices called selective catalytic reduction (“SCRs”)

• Rebuilding engines (“rebuild/upgrade kit”)

We chose to focus on these strategies due to their high NO_x emissions reduction potential and widespread application.

4A.6 VOC Control Measures for Nonpoint (Area) Sources

Some VOC controls for nonpoint sources are for the use of low or no VOC materials for graphic art sources. Other controls involve the application of limits for adhesive and sealant VOC content in wood furniture and solvent source categories. The OTC solvent cleaning rule establishes hardware and operating requirements for specified vapor cleaning machines, as well as solvent volatility limits and operating practices for cold cleaners. The Low Pressure/Vacuum Relief Valve control measure is the addition of low pressure/vacuum (LP/V) relief valves to gasoline storage tanks at service stations with Stage II control systems. LP/V relief valves prevent breathing emissions from gasoline storage tank vent pipes. Another control based on a California South Coast Air Quality Management District (SQAQMD) establishes VOC content limits for metal coatings along with application procedures and equipment requirements. Switching to Emulsified Asphalts is a generic control measure replacing VOC-containing cutback asphalt with VOC-free emulsified asphalt. The Reformulation control measures include switching to and/or encouraging the use of low-VOC materials.

CHAPTER 5: HUMAN HEALTH BENEFITS ANALYSIS APPROACH AND RESULTS

5.1 Synopsis

This chapter presents the estimated human health benefits for the proposed range of National Ambient Air Quality Standards (NAAQS) for ozone. In this chapter, we quantify the health-related benefits of the ozone air quality improvements resulting from the illustrative emission control scenarios that reduce emissions of the ozone precursor pollutants nitrogen oxides (NO_x) and volatile organic compounds (VOCs) to reach the set of alternative ozone NAAQS levels being considered. This chapter also estimates the health co-benefits of the fine particulate matter ($PM_{2.5}$)-related air quality improvements that would occur as a result of reducing NOx emissions.[33] The EPA Administrator is proposing to revise the level of the primary ozone standard to within a range of 65 to 70 ppb and is soliciting comment on alternative standard levels below 65 ppb, and as low as 60 ppb. In the Regulatory Impact Analysis (RIA) we analyze the following alternative standard levels: 70, 65 and 60 ppb.

We selected 2025 as the primary year of analysis because the Clean Air Act requires most areas of the U.S. to meet a revised ozone standard by 2025. Benefits are estimated incremental to attainment of the existing standard of 75 ppb. In estimating the incremental costs and benefits of potential alternative standards, we recognize that there are several areas that the Act does not require to meet the existing ozone standard of 75 ppb by 2025. The Clean Air Act provides areas with more significant air quality problems with additional time to reach the existing standard. Several areas in California are not expected to meet the existing standard by 2025 and may not be required to meet a revised standard until December 31, 2037.

We estimated the benefits of California attaining a revised standard in 2038 to account for the fact that many locations in this state must attain a revised standard at a later date than the rest of the U.S. We assume that projected nonattainment areas everywhere in the U.S. excluding California will be designated such that they attain a revised standard by 2025, and we develop our projected baseline emissions, air quality, and population estimates for 2025. We also assume

[33] VOC reductions associated with simulated attainment of alternative ozone standards also have the potential to impact $PM_{2.5}$ concentrations, but we are not able to model those effects at this time.

that the projected nonattainment areas in California will be designated such that they reach attainment by approximately 2038.

We were not able to project baseline emissions and air quality levels beyond 2025 for California for sectors other than mobile sources. We account for changes in mobile source precursor emissions expected to occur between 2025 and 2030. While there is uncertainty about the precise timing of emissions reductions and related costs for California, we assume costs occur through 2038. We also model benefits for California accounting for population growth to 2038 (see section 5.6.1). Because we were unable to account for the change in emissions from other sectors our projected baseline may under- or over-estimate the post-2025 ozone levels in California. In this analysis, we refer to estimates of nationwide benefits of attaining an alternative standard everywhere in the U.S. except California as the *2025 scenario*. The *post-2025 scenario* refers to estimates of nationwide benefits of attaining an alternative standard just in California.

Because we estimate incremental costs and benefits for these two distinct scenarios reflecting attainment in different years, it is not appropriate to either sum, or directly compare, the estimates. Consequently, in presenting both incidence and dollar benefit estimates in this chapter, we present and discuss the 2025 scenario and post-2025 scenario in separate sections (see sections 5.7.1 and 5.7.2, respectively).

Benefits estimated for the 2025 and post-2025 scenarios are relative to an analytical baseline in which the nation attains the current primary ozone standard (i.e., 4^{th} highest daily maximum 8-hour ozone concentration of 75 ppb) and incorporates promulgated national regulations and illustrative emission controls to simulate attainment with 75 ppb. Table 5-1 summarizes the estimated monetized benefits (total and ozone only) of attaining alternative ozone standards of 70 ppb, 65 ppb, and 60 ppb for the 2025 scenario (i.e., nationwide benefits of attaining everywhere in the U.S. but California). Table 5-2 presents the same types of benefit estimates for the post-2025 scenario (i.e., nationwide benefits of attaining just in California). These estimates reflect the sum of the economic value of estimated morbidity and mortality effects related to changes in exposure to ozone and fine particulate matter ($PM_{2.5}$). However, it is important to emphasize that it is not appropriate to compare the ozone-only benefits to total costs. There are additional unquantified benefits which are described in Section 5.2. The estimated benefits for attaining the proposed standards are incremental to the substantial benefits estimated for several recent implementation rules (e.g., U.S. EPA, 2011e, 2014a).

Table 5-1. Estimated Monetized Benefits of Attainment of the Alternative Ozone Standards for the *2025 Scenario* (nationwide benefits of attaining each alternative standard everywhere in the U.S. except California) – Full Attainment (billions of 2011$) [a]

	Discount Rate	70 ppb	65 ppb	60 ppb
Total Benefits	3%	$6.9 to $14 +B	$20 to $41 +B	$37 to $75 +B
	7%	$6.4 to $13 +B	$19 to $38 +B	$34 to $70 +B
Ozone-only Benefits (range reflects Smith et al., 2009 and Zanobetti and Schwartz, 2008)	b	$2.0 to $3.4 +B	$6.4 to $11 +B	$12 to $20 +B
$PM_{2.5}$ Co-benefits (range reflects Krewski et al., 2009 and Lepeule et al., 2012)	3%	$4.8 to $11	$14 to $31	$25 to $56
	7%	$4.3 to $9.7	$12 to $28	$22 to $50

[a] Rounded to two significant figures. It was not possible to quantify all benefits in this analysis due to data limitations. "B" is the sum of all unquantified health and welfare benefits. These estimates reflect the economic value of avoided morbidities and premature deaths using risk coefficients from the studies noted.
[b] Ozone-only benefits reflect short-term exposure impacts and as such are assumed to occur in the same year as ambient ozone reductions. Consequently, social discounting is not applied to the benefits for this category.

Table 5-2. Estimated Monetized Benefits of Attainment of the Alternative Ozone Standards for the *Post-2025 Scenario* (nationwide benefits of attaining each alternative standard just in California) – Full Attainment (billions of 2011$) [a]

	Discount Rate	70 ppb	65 ppb	60 ppb
Total Benefits	3%	$1.1 to $2.0 +B	$2.3 to $4.2 +B	$3.4 to $6.2 +B
	7%	$1.1 to $2.0 +B	$2.2 to $4.1 +B	$3.2 to $5.9 +B
Ozone-only Benefits (range reflects Smith et al., 2009 and Zanobetti and Schwartz, 2008)	b	$0.66 to $1.1	$1.4 to $2.4	$2.1 to $3.6
$PM_{2.5}$ Co-benefits (range reflects Krewski et al., 2009 and Lepeule et al., 2012)	3%	$0.42 to $0.95	$0.83 to $1.9	$1.1 to $2.6
	7%	$0.38 to $0.86	$0.75 to $1.7	$1.0 to $2.3

[a] Rounded to two significant figures. It was not possible to quantify all benefits in this analysis due to data limitations. "B" is the sum of all unquantified health and welfare benefits. These estimates reflect the economic value of avoided morbidities and premature deaths using risk coefficients from the studies noted.
[b] Ozone-only benefits reflect short-term exposure impacts and as such are assumed to occur in the same year as ambient ozone reductions. Consequently, social discounting is not applied to the benefits for this category.

In addition to ozone and $PM_{2.5}$ benefits, implementing emissions controls to reach some of the alternative ozone standards would reduce other ambient pollutants, such as VOCs and

NO_2. However, because the method used in this analysis to simulate attainment does not account for changes in ambient concentrations of other pollutants, we were not able to quantify the co-benefits of reduced exposure to these pollutants. In addition, due to data and methodology limitations, we were unable to estimate some anticipated health benefits associated with exposure to ozone and $PM_{2.5}$.

5.2 Overview

This chapter presents estimated health benefits for three alternative ozone standards (70, 65 and 60ppb) that the EPA could quantify, given the available resources, data and methods. Separate set of benefits are presented for the 2025 scenario, representing nationwide benefits of attaining an alternative standard everywhere in the U.S. but California in 2025 and the post-2025 scenario, representing nationwide benefits of attaining an alternative standard just in California in 2037. This chapter characterizes the benefits of implementing new ozone standards by answering three key questions:

1. What health effects are avoided by reducing ambient ozone levels to attain a revised ozone standard?
2. What is the economic value of these effects?
3. What are the co-benefits of reductions in ambient $PM_{2.5}$ associated with reductions in emissions of ozone precursors (specifically NOx)?

In this analysis, we quantify an array of adverse health impacts attributable to ozone and $PM_{2.5}$. The *Integrated Science Assessment for Ozone and Related Photochemical Oxidants* ("ozone ISA") (U.S. EPA, 2013a) identifies the human health effects associated with ozone exposure, which include premature death and a variety of illnesses associated with acute (days-long) and chronic (months to years-long) exposures. Similarly, the *Integrated Science Assessment for Particulate Matter* ("PM ISA") (U.S. EPA, 2009b) identifies the human health effects associated with ambient particles, which include premature death and a variety of illnesses associated with acute and chronic exposures. Air pollution can affect human health in a variety of ways, and in Table 5-3 we summarize the "categories" of effects and describe those that we could quantify in our "core" benefits estimates and those we were unable to quantify because we lacked the data, time or techniques.

This list of unquantified benefit categories is not exhaustive and we are not always able to quantify each effect completely. Endpoints that the ozone and PM ISAs classified as causal or likely causal we quantified with confidence. We excluded from quantification effects not identified as having at least a causal or likely causal relationship with the affected pollutants. Selecting endpoints in this way should not imply that these pollutants are unrelated to other human health and environmental effects. Following this criterion, we excluded some effects that were identified in previous lists of unquantified benefits in other RIAs (e.g., UVb exposure), but are not identified in the most recent ISA as having a causal or likely causal relationship with ozone. In designing this benefits analysis, including the identification of endpoints to include in the core estimate, we also considered the design of the Health Risk and Exposure Assessment (HREA) completed as part of this ozone NAAQS review (USEPA, 2014b). The design and implementation of the HREA was subjected to rigorous peer review by the Science Advisory Board's (SAB's) Clean Air Scientific Advisory Committee (CASAC) with the results of that review being presented in letter form (Frey, and Samet 2012 for the first draft and Frey, 2014 for the second draft of the HREA). The overall design of the HREA, including the health endpoints selected for modeling was supported by the CASAC.[34]

This benefits analysis relies on an array of data inputs—including emissions estimates, modeled ozone air quality, health impact functions and valuation estimates among others—which are themselves subject to uncertainty and may in turn contribute to the overall uncertainty in this analysis. We employ several techniques to characterize this uncertainty, which are described in detail in sections 5.5 and 5.7.3.

Table 5-3. Human Health Effects of Pollutants Potentially Affected by Strategies to Attain the Primary Ozone Standards (endpoints included in the core analysis are identified with a red checkmark)

Benefits Category	Specific Effect	Effect Has Been Quantified	Effect Has Been Monetized	More Information
Improved Human Health				

[34] The CASAC expressed their support for the overall design of the HREA, including endpoints selected and epidemiological studies used in supplying the effect estimates used to model those endpoints (Samet and Frey, 2012, p. 15 and Frey, 2014, p. 9).

Benefits Category	Specific Effect	Effect Has Been Quantified	Effect Has Been Monetized	More Information
Reduced incidence of premature mortality from exposure to ozone	Premature mortality based on short-term exposure (all ages)	✓	✓	Section 5.6
	Premature respiratory mortality based on long-term exposure (age 30–99)	✓	[a]	
Reduced incidence of morbidity from exposure to ozone	Hospital admissions—respiratory causes (age > 65)	✓	✓	
	Emergency department visits for asthma (all ages)	✓	✓	
	Asthma exacerbation (age 6-18)	✓	✓	
	Minor restricted-activity days (age 18–65)	✓	✓	
	School absence days (age 5–17)	✓	✓	
	Decreased outdoor worker productivity (age 18–65)	[b]	[b]	
	Other respiratory effects (e.g., medication use, pulmonary inflammation, decrements in lung functioning)	—	—	ozone ISA [d]
	Cardiovascular (e.g., hospital admissions, emergency department visits)	—	—	
	Reproductive and developmental effects (e.g., reduced birthweight, restricted fetal growth)	—	—	
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)	✓	✓	See section 5.6 and Appendix 5D
	Infant mortality (age <1)	✓	✓	
Reduced incidence of morbidity from exposure to $PM_{2.5}$	Non-fatal heart attacks (age > 18)	✓	✓	
	Hospital admissions—respiratory (all ages)	✓	✓	
	Hospital admissions—cardiovascular (age >20)	✓	✓	
	Emergency department visits for asthma (all ages)	✓	✓	
	Acute bronchitis (age 8–12)	✓	✓	
	Lower respiratory symptoms (age 7–14)	✓	✓	
	Upper respiratory symptoms (asthmatics age 9–11)	✓	✓	
	Asthma exacerbation (asthmatics age 6–18)	✓	✓	
	Lost work days (age 18–65)	✓	✓	
	Minor restricted-activity days (age 18–65)	✓	✓	
	Chronic Bronchitis (age >26)	—	—	PM ISA [c]
	Emergency department visits for cardiovascular effects (all ages)	—	—	
	Strokes and cerebrovascular disease (age 50–79)	—	—	
	Other cardiovascular effects (e.g., other ages)	—	—	

Benefits Category	Specific Effect	Effect Has Been Quantified	Effect Has Been Monetized	More Information
	Other respiratory effects (e.g., pulmonary function, non-asthma ER visits, non-bronchitis chronic diseases, other ages and populations)	—	—	PM ISA [c,d]
	Reproductive and developmental effects (e.g., low birth weight, pre-term births, etc.)	—	—	
	Cancer, mutagenicity, and genotoxicity effects	—	—	
Reduced incidence of morbidity from exposure to NO_2	Asthma hospital admissions (all ages)	—	—	NO_2 ISA [e]
	Chronic lung disease hospital admissions (age > 65)	—	—	
	Respiratory emergency department visits (all ages)	—	—	
	Asthma exacerbation (asthmatics age 4–18)	—	—	
	Acute respiratory symptoms (age 7–14)	—	—	
	Premature mortality	—	—	NO_2 ISA [c,d]
	Other respiratory effects (e.g., airway hyperresponsiveness and inflammation, lung function, other ages and populations)	—	—	

[a] Due to concerns over translating incidence estimates into dollar benefits, for long-term ozone exposure-related respiratory mortality, we included estimates of reduced incidence as part of the core analysis, but included associated dollar benefits as a sensitivity analysis (see section 5.3).
[b] We are in the process of considering an update to the worker productivity analysis for ozone based on more recent literature (see section 5.6.3.4). As noted in section 5.6.3.4 we are requesting public comment on the approach we present for modeling this endpoint and will consider that input in determining whether to proceed with an updated simulation of this endpoint.
[c] We assess these benefits qualitatively because we do not have sufficient confidence in available data or methods.
[d] 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.
[e] We assess these benefits qualitatively due to time and resource limitations for this analysis.

As described in Chapter 1 of this RIA, there are important differences worth noting in the design and analytical objectives of NAAQS RIAs compared to RIAs for rules that implement technology standards, such as Tier 3 (U.S. EPA, 2014a). The NAAQS RIAs illustrate the potential costs and benefits of attaining a revised air quality standard nationwide. These analyses simulate an array of strategies to reduce emissions at different sources and may model well-established emission control technologies for sectors and emission controls for which the control technology has not yet been developed (i.e., “unknown” controls). This type of RIA accounts for existing regulations and controls needed to attain the current standards and so estimated benefits and costs are incremental to attaining the current standard. In short, NAAQS RIAs hypothesize, but do not predict, the emission reduction strategies that States may enact when implementing a revised NAAQS. Setting a NAAQS does not result directly in costs or benefits. By contrast, the emission reductions from implementation rules are generally for specific, well-characterized sources, such as the recent MATS rule addressing emissions from coal and oil-fired electricity generating units (U.S. EPA, 2011e). In general, the EPA is more confident in the magnitude and location of the emission reductions for implementation rules. As such, emission reductions achieved under promulgated implementation rules such as MATS have been reflected in the baseline of this NAAQS analysis (the full set of rules reflected in baseline are presented in section 3.1.3). Subsequent implementation rules will be reflected in the baseline for the next ozone NAAQS review. For this reason, the benefits estimated provided in this RIA and all other NAAQS RIAs should not be added to the benefits estimated for implementation rules.

5.3 Updated Methodology Presented in this RIA

The benefits analysis presented in this chapter incorporates an array of policy and technical changes that the Agency has adopted since the previous review of the ozone standards in 2008 and the proposed reconsideration in 2010. Below we note the aspects of this analysis that differ from the reconsideration RIA (U.S. EPA, 2010d):

1. The population demographic data in BenMAP-CE (U.S. EPA, 2014d) reflects the 2010 Census and future projections based on economic forecasting models developed by Woods and Poole, Inc. (Woods and Poole, 2012). These data replace the earlier demographic projection data from Woods and Poole (2007). This update was introduced in the final PM NAAQS RIA (U.S. EPA, 2012b).

2. The baseline incidence rates used to quantify air pollution-related hospital admissions and emergency department visits and the asthma prevalence rates were updated to replace the earlier rates. This update was introduced in the final CSAPR (U.S. EPA, 2011d).

3. We updated the median wage data in the cost-of-illness studies. This update was introduced in the final PM NAAQS RIA (U.S. EPA, 2012b).

4. Updates for ozone-related effects:

    a. Incorporated new mortality studies. We include two new multi-city studies to estimate deaths attributable to short-term exposure for the core analysis (Smith et al., 2009 and Zanobetti and Schwartz 2008). We also estimate long-term respiratory deaths using Jerrett et al. (2009). While we believe the evidence supports including long-term respiratory deaths in the core analysis, limitations in our ability to specify a lag between exposure and the onset of death (i.e. the cessation lag that is required for valuing these deaths) prevents us from estimating dollar benefits in the core analysis. Both the new short-term and long-term mortality studies were included in the HREA completed in support of this NAAQS review with the overall design of that HREA (including inclusion of these new studies) being subjected to rigorous review by CASAC (Frey, and Samet 2012; Frey, 2014).

b. Incorporated new morbidity studies. The ozone ISA (U.S. EPA, 2013a) identifies several new epidemiological studies examining the association between short-term ozone exposure and respiratory hospitalizations, respiratory emergency department visits, and exacerbated asthma. Upon carefully evaluating this new literature, we added several new studies to our health impact assessment. Several of these studies were also included in the HREA, which as noted earlier, underwent rigorous review by CASAC.

c. Expanded uncertainty assessment. We added a comprehensive, qualitative assessment of the various uncertain parameters and assumptions within the benefits analysis and expanded the evaluation of air quality benchmarks for ozone-related mortality. We introduce this expanded assessment in this RIA (see sections 5.5 and 5.7.3).

5. Updates for $PM_{2.5}$-related effects

a. Incorporated new mortality studies. We updated the American Cancer Society cohort study to Krewski et al. (2009) and updated the Harvard Six Cities cohort study to Lepeule et al. (2012). The effect coefficient for Krewski et al. (2009) is identical to the previous coefficient, and the Lepeule et al. (2012) is roughly similar to the previous coefficient. Both studies show narrower confidence intervals. The update for the American Cancer Society cohort was introduced in the proposal RIA for the PM NAAQS review (U.S. EPA, 2012b) and the update for the Harvard Six Cities cohort was introduced in the final RIA for the PM NAAQS review (U.S. EPA, 2012b).

b. Incorporated new morbidity studies. The epidemiological literature has produced several recent studies examining the association between short-term $PM_{2.5}$ exposure and respiratory and cardiovascular hospitalizations, respiratory and cardiovascular emergency department visits, and stroke. Upon careful evaluation of new literature in the PM ISA and Provisional Assessment, we added several new studies and health endpoints to our health impact assessment. These updates were introduced in the proposal (U.S. EPA, 2012) and final RIAs for the PM NAAQS review (U.S. EPA, 2012b).

c. Updated the survival rates for non-fatal acute myocardial infarctions. Based on recent data from Agency for Healthcare Research and Quality's Healthcare Utilization Project National Inpatient Sample database (AHRQ, 2009), we identified death rates for adults hospitalized with acute myocardial infarction stratified by age. These rates replaced the survival rates from Rosamond et al. (1999). This update was introduced in the final RIA for the PM NAAQS review (U.S. EPA, 2012b).

d. Expanded uncertainty assessment. We clarified the comprehensive assessment of the various uncertain parameters and assumptions within the benefits analysis and expanded the evaluation of air quality benchmarks. This update was introduced in the proposed CSAPR RIA (U.S. EPA, 2010g) and refined in the final PM NAAQS RIA (U.S. EPA, 2012b).

Although the list above identifies the major changes implemented since the 2010 ozone reconsideration RIA, the EPA has also updated several additional components of the benefits analysis since the 2008 ozone NAAQS RIA (U.S. EPA, 2008a), which were reflected in the reconsideration RIA. In the Portland Cement NESHAP proposal RIA (U.S. EPA, 2009a), the Agency no longer assumed a concentration threshold in the concentration-response function for $PM_{2.5}$-related health effects and began estimating the benefits derived from the two major cohort studies of $PM_{2.5}$ and mortality as the core benefits estimates, while still including a range of sensitivity estimates based on the EPA's $PM_{2.5}$ mortality expert elicitation. In the NO_2 NAAQS proposal RIA (U.S. EPA, 2009a), we revised the estimate used for the value-of-a-statistical life to be consistent with Agency guidance.

5.4 Human Health Benefits Analysis Methods

We follow a "damage-function" approach in calculating total benefits of the modeled changes in environmental quality.[35] This approach estimates changes in individual health endpoints (specific effects that can be associated with changes in air quality) and assigns values to those changes assuming independence of the values for those individual endpoints. Total benefits are calculated simply as the sum of the values for all non-overlapping health 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 values 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, an impact analysis

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

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 ozone and $PM_{2.5}$.

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

Benefits estimates for ozone were generated using the damage function approach outlined above wherein potential changes in ambient ozone levels (associated with future attainment of alternative standard levels) were explicitly modeled and then translated into reductions in the incidence of specific health endpoints. In generating ozone benefits estimates for the two scenarios considered in the RIA (2025 and post-2025), we actually utilized three distinct benefits simulations including one completed for 2025 and two completed for 2038. The way in which these three benefits simulations were used to generate estimates for the two time periods reported in the RIA (2025 and post-2035) is described in section 5.4.3.

In contrast to ozone, we used a benefit-per-ton (reduced form) approach in modeling $PM_{2.5}$ co-benefits (see section 5.4.4 for additional detail). With this approach, we utilize the results of previous benefits analysis simulations focusing on $PM_{2.5}$ to derive benefits-per-ton estimates for NOx.[36] We then combine these dollar-per-ton estimates with projected reductions in NOx associated with meeting a given alternative standard to project co-benefits associated with $PM_{2.5}$. We acknowledge increased uncertainty associated with the dollar-per-ton approach for $PM_{2.5,}$

[36] In addition to dollar-per-ton estimates for NOx, we also utilized incidence-per-ton values (also for NOx) for specific health endpoints in order to generate incidence reduction estimates associated with the dollar benefits.

relative to explicitly modeling benefits using gridded $PM_{2.5}$ surfaces specific to the baseline and alternative scenarios being considered in this review (see sections 5.4.4 and 5.7.3 and Appendix 5A, Table 5A-1 for additional discussion).

Sections 5.4.1 and 5.4.2 describe respectively, the underlying basis for the health and economic valuation estimates. Section 5.4.3 describes the procedure used to combine the three benefits simulations referenced above in order to generate benefits for the two time periods considered in the RIA (2025 and post-2025). Finally, section 5.4.4 provides an overview of the benefit-per-ton estimates used to estimate the $PM_{2.5}$ co-benefits from NOx emission reductions in this RIA.

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 Benefits Mapping and Analysis Program – Community Edition (BenMAP-CE) 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 (U.S. EPA, 2014d). Analysts have applied the HIA approach to estimate human health impacts resulting from hypothetical changes in pollutant levels (Hubbell et al., 2005; Tagaris et al., 2009; Fann et al., 2012a). The EPA and others have relied upon this method to predict future changes in health impacts expected to result from the implementation of regulations affecting air quality (e.g., U.S. EPA, 2014d). For this assessment, the HIA is limited to those health effects that are directly linked to ambient ozone and $PM_{2.5}$ 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 ozone air quality[37] 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 to this change in population exposure (Hubbell et al., 2009).

A typical health impact function might look as follows:

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

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

[37] Projections of ambient ozone concentrations for this analysis were generated by applying emissions reductions described in chapters 3 and 4 to gridded surfaces of recent-year ozone concentrations. The full methodology which incorporates information both from ambient measurements and from photochemical modeling simulations is described in section 3.4.1 of this RIA.

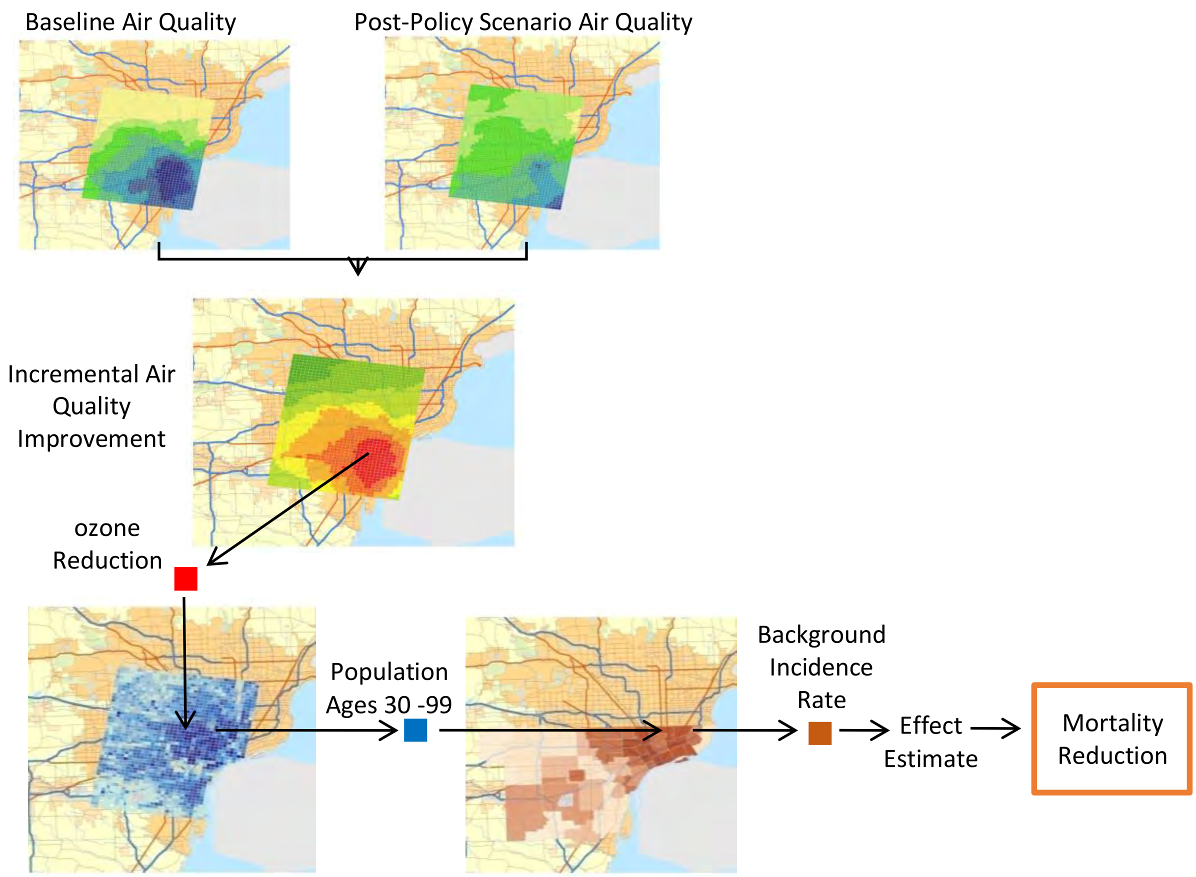


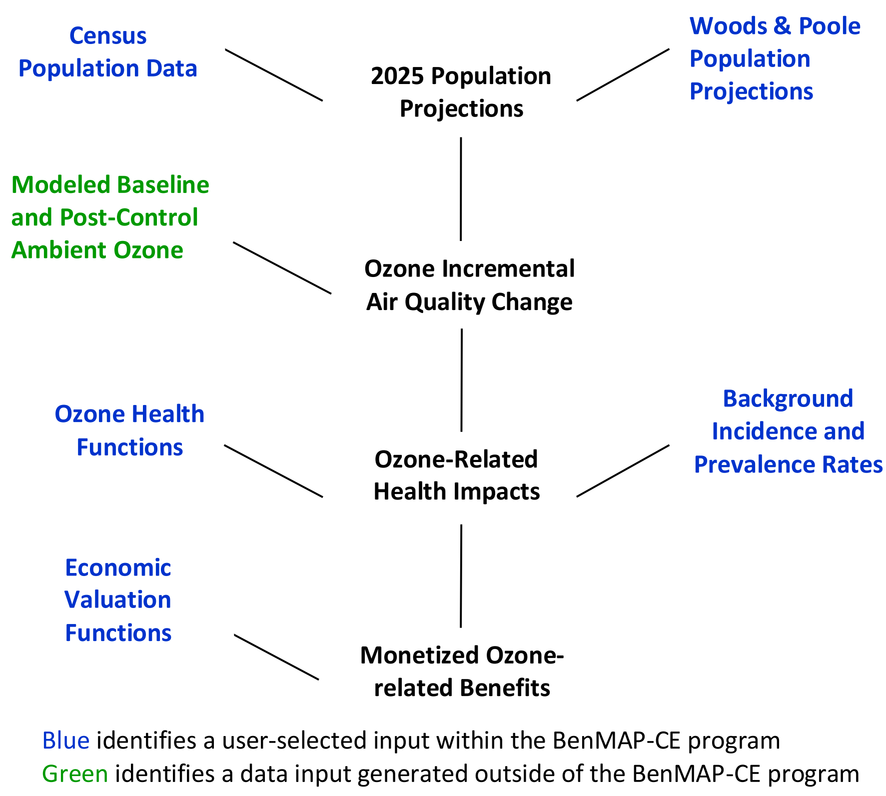
Figure 5-1. Illustration of BenMAP-CE Approach

5.4.2 Economic Valuation of Health Impacts

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

For some health effects, such as hospital admissions, WTP estimates are generally not available. In these cases, we use the costs of treating or mitigating the effect, which generally understate the true value of reductions in risk of a health effect because they exclude the value of avoided pain and suffering from the health effect.

We use the BenMAP-CE version 1.0.8 (U.S. EPA, 2014d) to estimate the health impacts and monetized health benefits for the proposed standard range. Figure 5-2 shows the data inputs and outputs for the BenMAP-CE program.[38]



[38] The environmental Benefits Mapping and Analysis Program—Community Edition (BenMAP-CE) is an open-source PC-based tool that quantifies the number and economic value of air pollution-related deaths and illnesses. As compared to the version that it replaces, BenMAP v4, the BenMAP-CE tool uses the same computational algorithms and input data to calculate incidence counts and dollar values—for a given air quality change, both versions report the same estimates, within rounding. BenMAP-CE differs from the legacy version of BenMAP in two important ways: (1) it is open-source and the uncompiled code is available to the public; (2) it is written in C#, which is both more broadly used and modern than the code it replaces (Delphi). BenMAP-CE was last used to support the Ozone Health Risk and Exposure Assessment completed in support of the current review.

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

5.4.3 Estimating Benefits for the 2025 and Post-2025 Scenarios

As described in section 5.1, we estimated benefit for two scenarios: 2025 and post-2025. The need for these two distinct time periods reflects the fact that, while most of the U.S. will have attained both the current and any alternative standard by 2025, there are portions of the country with more significant air quality problems (including several areas in California) that may not be required to meet an alternative standard until as late as December 31, 2037. Consequently, for each alternative standard we model a 2025 scenario reflecting the nationwide benefits of attaining that standard everywhere in the U.S. except California. We then model a post-2025 scenario, which represents nationwide benefits from attaining that same standard in California. Due to the temporal disconnect between these two scenarios, we do not attempt to sum these two estimates, but instead present each estimate in separate sections of this document (sections 5.7.1 and 5.7.2, respectively).

Our approach for estimating the benefits of attaining alternate ozone standards post-2025 is illustrated in Figure 5-3; in this figure, Simulation A represents our approach for estimating the benefits of attaining alternate ozone standards in every state except California in 2025. We first estimated the benefits occurring in 2038 from all areas (including California) attaining each alternative standard (Simulation C). Next, we simulated the nationwide benefits of attaining each alternate ozone standard in 2038 for every state except California (Simulation B). Subtracting Simulation B from Simulation C calculates the benefits of attaining each alternate ozone standard after 2025— that is, the nationwide benefits from California alone attaining the standard in 2038. There are important caveats associated with this approach mentioned in Section 5.1 and discussed further in section 5.7.3.

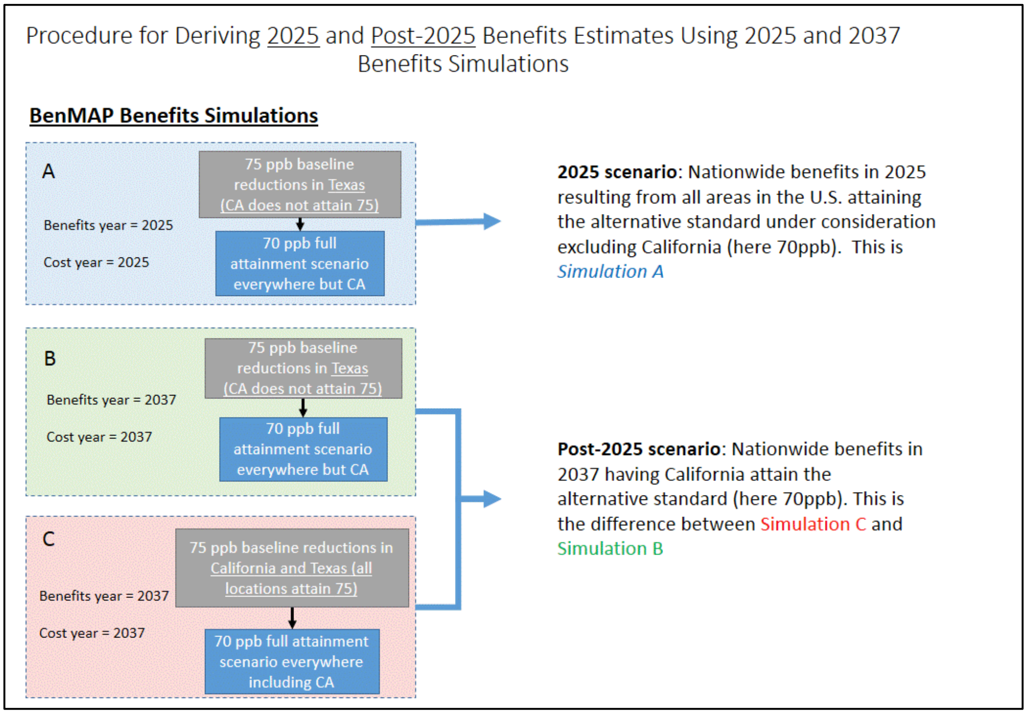


Figure 5-3. Procedure for Generating Benefits Estimates for the 2025 and Post-2025 Scenarios

5.4.4 Benefit-per-ton Estimates for $PM_{2.5}$

We used a "benefit-per-ton" approach to estimate the $PM_{2.5}$ co-benefits in this RIA. EPA has applied this approach in several previous RIAs (e.g., U.S. EPA, 2014a). These benefit-per-ton estimates provide the total monetized human health co-benefits (the sum of premature mortality), of reducing one ton of NOx (as a $PM_{2.5}$ precursor) from a specified source.[39] In general, these estimates apply the same benefits methods (e.g., health impact assessment then economic valuation), which are described further below, for all $PM_{2.5}$ impacts attributable to a sector, and these benefits are then divided by the tons of a $PM_{2.5}$ precursor (e.g., NOx) from that sector. As discussed below, we acknowledge that this approach has greater uncertainty relative to explicitly modeling benefits for $PM_{2.5}$ based on application of gridded surfaces specifically

[39] In generating these estimates, we first use incidence-per-ton values to generate estimates of reductions in morbidity and mortality incidence for core endpoints (see Table 5-3). Then in estimating dollar values associated with these reductions in incidence, we use dollar-per-ton values for mortality only, noting that this is likely to provide coverage for upwards of 97% of the total dollar benefits (i.e., morbidity endpoints provide less than 3% of total benefits – see RIA from last PM review, USEPA, 2012, Table 5-19).

generated for the baseline and alternative standard levels being considered in this ozone NAAQS review. However, resource and time constraints prevented us from completing detailed $PM_{2.5}$ modeling as part of this review.

We used a method to calculate the regional benefit-per-ton estimates that is a slightly modified version of the national benefit-per-ton estimates described in the TSD: *Estimating the Benefit per Ton of Reducing* $PM_{2.5}$ *Precursors from 17 Sectors* (U.S. EPA, 2013b). The national estimates used in this NAAQS review were derived using the approach published in Fann et al. (2012c), but they have since been updated to reflect the epidemiology studies and Census population data first applied in the final PM NAAQS RIA (U.S. EPA, 2012b). The approach in Fann et al. (2012c) is similar to the work previously published by Fann et al. (2009), but the newer study includes improvements that provide more refined estimates of $PM_{2.5}$-related health benefits for emissions reductions in the various sectors. Specifically, the air quality modeling data reflect industrial sectors that are more narrowly defined. In addition, the updated air quality modeling data reflects more recent emissions data -- a 2005 baseline projected to 2016 rather than 2001 baseline projected to 2015 -- and has higher spatial resolution (12 km rather than 36 km grid cells).[40]

In Section 5.6 below, we describe all of the data inputs used in deriving the dollar-per-ton values for each sector, including the demographic data, baseline incidence, and valuation functions. The specification of effect estimates (including selection of epidemiology studies) used in the derivation of the benefit-per-ton values for $PM_{2.5}$ is described in detail in Appendix 5D. The benefit-per-ton estimates (by sector) that resulted from this modeling as well as the NOx reductions for each alternative standard level used in generating the $PM_{2.5}$ cobenefit estimates are presented in Appendix 5E. Additional information on the source apportionment modeling for each of the sectors can be found in Fann et al. (2012c) and the TSD (U.S. EPA, 2013b).

[40] Sector-level estimates of $PM_{2.5}$ are modeled using CAMx version 5.30. Specifically, the particulate source apportionment technology (PSAT) incorporated into CAMx generates estimates of the contribution from specific emission source groups to primary emitted and secondarily formed $PM_{2.5}$, PSAT uses reactive tracers in generating these fractional estimates in order to capture nonlinear formation and removal processes related to $PM_{2.5}$. Contributions from each sector are modeled at the 12km level, while boundary conditions are represented using a 36km grid resolution (additional detail on modeling can be found in Fann et al., 2012)

Specifically for this analysis, we applied the benefit-per-ton estimates for 2025 and 2030 in generating $PM_{2.5}$ co-benefit estimates for the 2025 and post-2025 time periods, respectively (both sets of benefit-per-ton estimates are presented in Appendix 5E).[41]

As discussed in greater detail in section 5.7.3 and Appendix 5A, Table 5A-1, we recognize uncertainty associated with application of the benefit-per-ton approach used in modeling $PM_{2.5}$ cobenefits. The benefit-per-ton estimates used here reflect specific geographic patterns of emissions reductions and specific air quality and benefits modeling assumptions associated with the derivation of those estimates. Consequently, these estimates may not reflect local variability in factors associated with $PM_{2.5}$-realted health impacts (e.g., population density, baseline health incidence rates) since air quality modeling that could have shed light on local conditions was not performed for this RIA. Therefore, use of these benefit-per-ton values to estimate co-benefits may lead to higher or lower benefit estimates than if co-benefits were calculated based on direct air quality modeling.

5.5 Characterizing Uncertainty

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

After reviewing the EPA's approach, the National Research Council (NRC) (2002, 2008), which is part of the National Academies of Science, concluded that the EPA's general methodology for calculating the benefits of reducing air pollution is reasonable and informative in spite of inherent uncertainties. The NRC also highlighted the need to conduct rigorous

[41] We do not have benefit-per-ton estimates for 2038. The last year available is 2030, which is an underestimate of the 2038 benefits because the population grows and ages over time.

quantitative analyses of uncertainty and to present benefits estimates to decision makers in ways that foster an appropriate appreciation of their inherent uncertainty. Since the publication of these reports, the EPA has continued work to improve the characterization of uncertainty in both health incidence and benefits estimates. In response to these recommendations, we have expanded our previous analyses to incorporate additional quantitative and qualitative characterizations of uncertainty. Although we have not yet been able to make as much progress towards a full, probabilistic uncertainty assessment as envisioned by the NAS as we had hoped, we have added a number of additional quantitative and qualitative analyses to highlight the impact that uncertain assumptions may have on the benefits estimates. These additional analyses focus primarily on uncertainty related to the mortality endpoint (for both ozone and $PM_{2.5}$) since mortality is the driver for dollar benefits. In addition, for some inputs into the benefits analysis, such as the air quality data, it is difficult to address uncertainty probabilistically due to the complexity of the underlying air quality models and emission inputs. Therefore, we decline to construct alternative assumptions simply for the purpose of probabilistic uncertainty characterization when there is no scientific literature to support those alternate assumptions.

To characterize uncertainty and variability, we follow an approach that combines elements from two recent analyses by the EPA (U.S. EPA, 2010b; 2011b), 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 5A (results of these assessments are summarized in section 5.7.3). Data limitations prevent us from treating each source of uncertainty quantitatively and from reaching a full-probabilistic simulation of our results, but we were able to consider the influence of uncertainty in the risk coefficients and economic valuation functions by incorporating several 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 for ozone-related health effects. See section 5.5.1 for additional detail on the Monte Carlo assessment.
2. A series of *sensitivity analyses* primarily focused on the mortality endpoint (for both ozone and $PM_{2.5}$). We focus on mortality in conducting sensitivity analyses reflecting the important role that this endpoint plays in driving both ozone-related and $PM_{2.5}$ (co-benefit) related dollar benefits. These sensitivity analyses address factors related to (a) estimating incidence (e.g., multiple epidemiology studies providing alternative effect

estimates, shape of the C-R function including potential for thresholds) and (b) estimating associated dollar benefits (e.g., income elasticity related to willingness to pay functions and uncertainty in specifying lag structures for long-term exposure-related mortality). See section 5.5.2 for additional detail on the set of sensitivity analyses completed for this RIA.

3. *Supplemental analyses* which allow us to consider additional factors related to the benefits analysis. These include an assessment of the age-related differentiation of short-term ozone exposure-related mortality (including life year saved and how estimates of avoided mortality are distributed across age ranges). In addition, we also looked at the relationship between estimates of mortality and the underlying baseline ambient air levels used in their derivation. These analyses allow us to consider which range of baseline levels drive the benefits estimates. Finally, we considered the fraction of benefit estimates (for the 2025 scenario) which are associated with application of (higher-confidence) known emissions controls. See section 5.5.3 for additional detail on the supplemental analyses completed for this RIA.

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-CE 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.

In this RIA, we provide confidence intervals for ozone-related benefits, but we are unable to provide confidence intervals for $PM_{2.5}$-related co-benefits due to the use of benefit-per-ton estimates.

5.5.2 Sensitivity Analysis Addressing Both Incidence and Dollar Benefit Valuation

We assign the greatest economic value to the reduction in mortality risk. Therefore, it is particularly important to characterize to a reasonable extent the uncertainties associated with reductions in premature mortality, including both incidence estimation and the translation of reduced mortality into equivalent dollar benefits. Each of the sensitivity analyses completed for this RIA are briefly described below. The reader is referred to section 5.7.3.1 for discussion of the results and observations stemming from these sensitivity analyses.

- **Alternative C-R functions for short-term ozone exposure-related mortality:** 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 designs. For ozone we have included two multi-city studies (Smith et al., 2009 and Zanobetti and Schwartz 2008) in our core estimate of the range for short-term exposure-related mortality. For the sensitivity analysis addressing this endpoint, we have included additional multi-city and meta-analysis studies utilized in RIAs completed for previous ozone NAAQS reviews (Bell et al., 2004 and 2005, Huang, 2005, Ito et al., 2005 and Levy et al., 2005), as well as alternative model specifications from the Smith et al (2009) study. The selection of studies for the core and sensitivity analyses, reflects consideration for recommendations made both by the NAS in relation to modeling ozone benefits (p. 80, NRC, 2008) and the CASAC in their review of the HREA completed in support of this ozone NAAQS review. We also considered information and recommendations provided in the latest ozone ISA. Additional detail on the selection and use of studies in modeling short-term exposure-related mortality for the core analysis is presented in section 5.6.3.1.

- **Impact of potential thresholds on the modeling of long-term ozone exposure-related respiratory mortality:** Consistent with the HREA, we estimate counts of respiratory deaths from long-term exposure to ozone in our core analysis. As discussed in detail in section 5.6.3.1, the Jerrett et al., 2009 study from which the mortality effect estimate was derived included an exploration of potential thresholds in the concentration-response function. To provide a more comprehensive picture of potential benefits associated with long term ozone exposures, we use the results of the threshold analysis conducted by Jerrett et al, 2009 to conduct a sensitivity analysis evaluating models with a range of potential thresholds in addition to a non-threshold model (see section 5.7.3.1 and Appendix 5B, section 5B.1).

- **Considering alternative C-R functions in estimating long-term $PM_{2.5}$ exposure-related mortality:** In modeling co-benefits related to reductions in long-term exposure to $PM_{2.5}$, our dollar-per-ton approach relies on estimates based on two studies (Krewski et al., 2009 and Lepeule et al., 2012 – see Appendix 5D, section 5D.1). To better understand the concentration-response relationship between $PM_{2.5}$ exposure and premature mortality, the EPA conducted an expert elicitation in 2006 (Roman et al., 2008; IEc, 2006).[42] 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. 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). Application of the expert elicitation-based effect estimates (as part of characterizing uncertainty) is covered in section 5.7.3.1 and Appendix 5B, section 5B.2. In addition to these studies, we have included a discussion of other recent multi-state cohort studies conducted in North America, but we have not estimated benefits using the effect coefficients from these studies (see Appendix 5D, section 5D.1).

- **Specifying the cessation lag for long-term $PM_{2.5}$ exposure-related respiratory mortality:** As discussed in section 5.1 and 5.6.4.1, uncertainty in projecting the cessation lag for long-term $PM_{2.5}$ exposure-related respiratory mortality prevents us from estimating dollar benefits associated with projected reductions in mortality. In the absence of clear evidence pointing to a particular lag structure, we have decided to use two lag structures (the 20 year segmented lag used for $PM_{2.5}$ and an assumption of zero lag – see section 5.6.4.1). The range of dollar benefits that result have been included as sensitivity analyses and not in the core analysis.

- **Income elasticity in the specification of willingness to pay (WTP) functions used for mortality and morbidity endpoints:** There is uncertainty in specifying the degree to which the WTP function used in valuing mortality and some morbidity endpoints tracks projected increase in income over time (i.e., the income elasticity for WTP). We completed a sensitivity analysis to evaluate the potential impact of this factor on dollar estimates generated for mortality (and a subset of morbidity endpoints – see section 5.6.4.1).

Even these multiple estimates (including confidence intervals in the case of estimates generated for ozone) cannot account for the role of other input variables in contributing to overall uncertainty, including emissions and air quality modeling, baseline incidence rates, and population exposure estimates. 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

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

elements. As a result, the reported confidence intervals and range of estimates give an incomplete picture about the overall uncertainty in the estimates. Thus, confidence intervals reported for individual endpoints and for total benefits should be interpreted within the context of the larger uncertainty surrounding the entire analysis.

5.5.3 Supplemental Analyses

We have also conducted a number of supplemental analyses designed to provide additional perspectives on the core mortality estimates generated for this RIA. These analyses (the results of which are described in section 5.7.3.2) include:

- **Age group-differentiated aspects of short-term ozone exposure-related mortality:** We examined several risk metrics intended to characterize how mortality risk reductions are distributed across different age ranges. These include (a) estimated reduction in life years lost, (b) distribution of mortality incidence reductions across age ranges and (c) estimated reductions in baseline mortality incidence rates by age group.

- **Analysis of baseline ozone levels used in modeling short-term ozone exposure-related mortality:** We assess the relationship between short-term exposure-related mortality for ozone and the distribution of baseline (i.e., reflecting attainment of the current standard) 8hr max daily values used in deriving those estimates (see section 5.7.3.2 and Appendix 5C, section 5C.2). This analysis allows us to explore how estimates of ozone- attributable mortality are distributed with regard to projected ambient ozone levels, including the fraction of overall mortality that falls within specific ozone ranges. We note that, while the latest ozone ISA did not provide support for a threshold in relation to short-term exposure-related mortality, it did note that there is reduced confidence in specifying the nature of the concentration-response function at lower ozone levels (in the range of 20ppb and below) (ozone ISA, U.S. EPA, 2013a, section 2.5.4.4). We use the distribution of short-term mortality across ozone levels to determine the fraction of mortality reductions (i.e., benefits) that fall within this lower confidence range.[43]

- **Analysis of baseline $PM_{2.5}$ levels used in modeling short-term ozone exposure-related mortality:** We also include a similar plot of the baseline annual $PM_{2.5}$ levels used in modeling long-term $PM_{2.5}$ exposure-related mortality (see section 5.7.3.2 and Appendix 5C, section 5C.2). However, we are using a reduced form dollar-per-ton

[43] However, care must be taken in interpreting this range of reduced confidence since benefits estimates are based on the average daily 8hr max across the ozone season and not on a true daily time series of 8hr metrics within each grid cell. The use of a seasonal mean 8hr max (rather than the more temporally differentiated daily time series) has been shown to generate nearly identical benefit estimates at the national-level due to underlying linearity in the benefits model being used. However, the use of the seasonal average 8hr metric, rather than a full daily time series, does decrease overall temporal variability in a plot of mortality versus ozone level which introduces uncertainty into the interpretation of these plots.

approach in modeling $PM_{2.5}$ cobenefits, we do not have spatially differentiated $PM_{2.5}$ values (and associated mortality estimates) with which to derive this type of distributional plot specifically for this RIA and consequently, we have reproduced a plot from the earlier analysis used to generate the benefit-per-ton values.

- **Fraction of core ozone and $PM_{2.5}$ (cobenefit) estimates associated with application of known emissions controls for the 2025 scenario:** This analysis estimates the fraction of core incidence and associated dollar benefits that are associated with application of the set of known (higher confidence) emissions control measures. Note that this analysis is only completed for the 2025 scenario since application of controls in California (associated with the post-2025 scenario) exclusively involves application of unknown controls.

5.5.4 Qualitative Assessment of Uncertainty and Other Analysis Limitations

Although we strive to incorporate as many quantitative assessments of uncertainty as possible, there are several aspects we are only able to address qualitatively. These aspects are important factors to consider when evaluating the relative benefits of the emission reduction strategies for the proposed and 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 the EPA's independent SAB (Health Effects Subcommittee of the Advisory Council on Clean Air Compliance Analysis) (SAB-HES) (U.S. EPA- SAB, 2010a) and the National Academies of Science (NAS) (NRC, 2002, 2008). The benefits estimates are subject to a number of assumptions and uncertainties.

To more fully address all these uncertainties including those we cannot quantify, we apply a four-tiered approach using the WHO uncertainty framework (WHO, 2008), which provides a means for systematically linking the characterization of uncertainty to the sophistication of the underlying risk assessment. The EPA has applied similar approaches in previous analyses (U.S. EPA, 2010b, 2011b, U.S. EPA, 2012a – the HREA). 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 section 5.7.3 and Appendix 5A.

As previously described, we strive to monetize as many of the benefits anticipated from the proposed and alternative standards as possible given data and resource limitations, but the monetized benefits estimated in this RIA inevitably only reflect a portion of the total health benefits. Data and methodological limitations prevented the EPA from quantifying or monetizing the benefits from several important health benefit categories from emission reduction strategies to attain the alternative ozone standards analyzed in this RIA, including potential co-benefits from reducing NO_2 exposure (see section 5.6.3.6 for more information) and reductions in VOC exposures.

5.6 Benefits Analysis Data Inputs

In Figure 5-2 above, 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 2008 ozone NAAQS RIA (U.S. EPA, 2008a) and the 2010 ozone NAAQS Reconsideration RIA (U.S. EPA, 2010d).

5.6.1 Demographic Data

Quantified and monetized human health impacts depend on the demographic characteristics of the population, including age, location, and income. We use population projections based on economic forecasting models developed by Woods and Poole, Inc. (Woods and Poole, 2012). The Woods and Poole (WP) database contains county-level projections of population by age, sex, and race out to 2040, relative to a baseline using the 2010 Census data; the 2008 proposal RIA incorporated WP projections relative to a baseline using 2000 Census data. Projections in each county are determined simultaneously with every other county in the United States to take into account patterns of economic growth and migration. The sum of growth in county-level populations is constrained to equal a previously determined national population growth, based on Bureau of Census estimates (Hollman et al., 2000). According to WP, linking county-level growth projections together and constraining to a national-level total growth avoids potential errors introduced by forecasting each county independently. County projections are developed in a four-stage process:

- First, national-level variables such as income, employment, and populations are forecasted.
- Second, employment projections are made for 179 economic areas defined by the Bureau of Economic Analysis (U.S. BEA, 2004), using an "export-base" approach, which relies on linking industrial-sector production of non-locally consumed production items, such as outputs from mining, agriculture, and manufacturing with the national economy. The export-based approach requires estimation of demand equations or calculation of historical growth rates for output and employment by sector.
- Third, population is projected for each economic area based on net migration rates derived from employment opportunities and following a cohort-component method based on fertility and mortality in each area.
- Fourth, employment and population projections are repeated for counties, using the economic region totals as bounds. The age, sex, and race distributions for each region or county are determined by aging the population by single year of age by sex and race for each year through 2040 based on historical rates of mortality, fertility, and migration.

5.6.2 Baseline Incidence and Prevalence Estimates

Epidemiological studies of the association between pollution levels and adverse health effects generally provide a direct estimate of the relationship of air quality changes to the relative risk of a health effect, rather than estimating the absolute number of avoided cases. For example, a typical result might be that a 5 ppb decrease in 8hr max daily ozone levels might be associated with a decrease in hospital admissions of three percent. 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 5-4 summarizes the sources of baseline incidence rates and provides average incidence rates for the endpoints included in the analysis. For both baseline incidence and prevalence data, we used age-specific rates where available. We applied concentration-response functions to individual age groups and then summed over the relevant age range to provide an estimate of total population benefits. In most cases, we used a single national incidence rate, due to a lack of more spatially disaggregated data. Whenever possible, the national rates used are national averages, because these data are most applicable to a national assessment of benefits. For some studies, however, the only available incidence information comes from the studies themselves; in these cases, incidence in the study population is assumed to represent typical incidence at the national level. County, state and regional incidence rates are available for hospital admissions, and county-level data are available for premature mortality.

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

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

For the set of endpoints affecting the asthmatic population, in addition to baseline incidence rates, prevalence rates of asthma in the population are needed to define the applicable population. Table 5-5 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, 2011d).

Table 5-4. Baseline Incidence Rates and Population Prevalence Rates for Use in Impact Functions, General Population

Endpoint	Parameter	Rates	
		Value	Source
Mortality	Daily or annual mortality rate projected to 2025 [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]
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		Ostro et al. (2001)
	daily wheeze	0.173	
	daily cough	0.145	
	daily shortness of breath	0.074	
Acute Bronchitis	Annual bronchitis incidence rate, children	0.043	American Lung Association (2002, 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)		1996 HIS (Adams, Hendershot, and Marano, 1999, Table 41); U.S. Census Bureau (2000)
	Aged 18–24	0.00540	
	Aged 25–44	0.00678	
	Aged 45–64	0.00492	
School Loss Days	Rate per person per year, assuming 180 school days per year	9.9	National Center for Education Statistics (1996) and 1996 HIS (Adams et al., 1999, Table 47);
Minor Restricted-Activity Days	Daily MRAD incidence rate per person	0.02137	Ostro and Rothschild (1989, p. 243)

[a] Mortality rates are only available at 5-year increments.
[b] Healthcare Cost and Utilization Program (HCUP) database contains individual level, state and regional-level hospital and emergency department discharges for a variety of ICD codes (AHRQ, 2007).
[c] Lower respiratory symptoms are defined as two or more of the following: cough, chest pain, phlegm, and wheeze.

Table 5-5. Asthma Prevalence Rates

Population Group	Asthma Prevalence Rates	
	Value	Source
All Ages	0.0780	American Lung Association (2010, 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 (2010, 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 modeling of incidence and benefits for each pollutant employ distinct and separate sets of effect estimates since these are obtained from epidemiological studies specific to a given endpoint/pollutant combination. In the case of $PM_{2.5}$, the dollar-per-ton approach being employed reflects application of effect estimates (for all endpoints) which have been used by the EPA in previous RIA's (e.g., PM NAAQS, U.S. EPA, 2012b). Consequently, while we identify the studies and effect estimates reflected in the dollar-per-ton values used for $PM_{2.5}$ (see Table 5-8), we do not present a detailed discussion of those effect estimates in this section and instead present that more detailed discussion in Appendix 5D. By contrast, with ozone, we conducted detailed benefits modeling using an updated set of effect estimates that reflects a combination of values used in previous RIAs together with updated values. For that reason, we provide a detailed discussion of effect estimates for ozone (including the rationale for their selection) in this section.

The first step in selecting effect coefficients is to identify the health endpoints to be quantified. We base our selection of health endpoints on consistency with the EPA's ISAs (which replace previous "Criteria Documents"), with input and advice from the HES, a scientific review panel specifically established to provide advice on the use of the scientific literature in developing benefits analyses for the *EPA's Report to Congress on The Benefits and Costs of the Clean Air Act 1990 to 2020* (U.S. EPA, 2011a). In addition, we have included more recent epidemiology studies from the ozone ISA (U.S. EPA, 2013a), PM ISA (U.S. EPA, 2009b), and the PM Provisional Assessment (U.S. EPA, 2012c).[44] In selecting health endpoints for ozone, we also considered the suite of endpoints included in core modeling for the HREA, which was supported by CASAC (Frey, and Samet 2012; Frey, 2014). In general, we follow a weight of evidence approach, based on the biological plausibility of effects, availability of concentration-

[44] The peer-reviewed studies in the *Provisional Assessment* have not yet undergone external review by the SAB.

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 ozone and $PM_{2.5}$ and adverse human health effects. We evaluated epidemiological studies using the selection criteria summarized in Table 5-6. 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 (with the exception of mortality)[45] have been selected, they are quantitatively combined or pooled to derive a more robust estimate of the relationship. The BenMAP Manual Technical Appendices for an earlier version of the program provides details of the procedures used to combine multiple impact functions (Abt Associates, 2012). In general, we used fixed or random effects models to pool estimates from different single city studies of the same endpoint. Fixed effect 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 effect 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.[46] Pooled impact functions are used to estimate hospital admissions and asthma exacerbations. When combining evidence across multi-city studies (e.g., cardiovascular hospital admission studies), we use equal weights pooling. The effect estimates drawn from each multi-city study are themselves pooled across a large number of urban areas. For this reason, we elected to give each study an equal weight rather than weighting by the inverse of the variance reported in each study. For more details on methods used to pool incidence estimates, see the BenMAP Manual Appendices (Abt Associates, 2012).

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

[45] In the case of mortality, when we have multiple studies providing effect estimates for the core analysis, we include the range of the resulting incidence and dollar benefit estimates rather than pooling them in order to provide additional characterization of overall confidence associated with this key endpoint. However, for morbidity endpoints we do pool estimates as described here.

[46] EPA recently changed the algorithm BenMAP uses to calculate study variance, which is used in the pooling process. Prior versions of the model calculated population variance, while the version used here calculates sample variance.

Table 5-6. 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	*Prospective vs. cohort*: 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. *Multi-city vs. pooled/meta-analysis:* In recommending approaches for modeling ozone-related mortality, the NAS notes a number of advantages multi-city time series studies as compared to meta-analyses. Multi-city studies utilize a consistent model structure and can include factors that explain differences between effect estimates among the cities. By contrast, with meta-analyses given the aggregation of large sets of studies, the construct definitions can become imprecise and the results difficult to interpret. In addition, meta-analyses can suffer from publication bias which can result in high-biased effect estimates. Ultimately, the NAS recommends that the greatest emphasis be placed on estimates based on systematic new multi-city analyses without excluding consideration of meta-analyses (NRC, 2008). Reflecting these observations by the NAS, in modeling ozone benefits, we have included several newer multi-city studies in the core analysis and a combination of multi-city and meta analyses in the accompanying sensitivity analysis. Placing emphasis on these two multi-city studies was also supported by the CASAC in the context of the HREA (Frey, and Samet 2012; Frey, 2014).
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.
Seasonality	While the measurement of PM is typically collected across the full year, ozone monitoring seasons can vary substantially across different regions of the country. Given modeling constraints, we were not able to consider variation in ozone seasons in modeling benefits and instead, had to select a standard ozone season for the entire country (May 1st through September 31st). Consequently, in selecting effect estimates, we favored those values that reflected ozone seasons close to this fixed ozone season (i.e., we did not include effect estimates based on a full year of ozone monitoring data).
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).

Consideration	Comments
Study Location	U.S. studies are more desirable than non-U.S. studies because of potential differences in pollution characteristics, exposure patterns, medical care system, population behavior, and lifestyle. National estimates are most appropriate when benefits are nationally distributed; the impact of regional differences may be important when benefits only accrue to a single area.
Pollutants Included in Model	An important factor affecting the specification of co-pollutant models for ozone and PM is sampling frequency. While ozone is typically measured every hour of each day during the ozone season for a specific location, PM is typically measured every 3rd or 6th day. For this reason, when modeling the PM effect, epidemiological models specifying co-pollutants are preferred because this approach controls for the potential ozone effect while not diminishing the effective sample size available for specifying the PM effect. However, when modeling the ozone effect, the use of copollutants modeling (with PM) can substantially reduce sample size (by 1/3 to 1/6) since only days with both ozone and PM can be used. While these copollutants models may control for potential PM effects, they also result in a substantially less robust characterization of the ozone effect due to the reduced number of ozone measurements. For this reason, while we favor copollutants models in modeling PM benefits, for ozone we favor single pollutant models for the core estimate and reserve copollutants models for sensitivity analyses.
Measure of PM	For this analysis, impact functions based on $PM_{2.5}$ are preferred to PM_{10} because of the focus on reducing emissions of $PM_{2.5}$ precursors, and because air quality modeling was conducted for this size fraction of PM. Where $PM_{2.5}$ functions are not available, PM_{10} functions are used as surrogates, recognizing that there will be potential downward (upward) biases if the fine fraction of PM_{10} is more (less) toxic than the coarse fraction.
Economically Valuable Health Effects	Some health effects, such as forced expiratory volume and other technical measurements of lung function, are difficult to value in monetary terms. These health effects are not quantified in this analysis.
Non-overlapping Endpoints	Although the benefits associated with each individual health endpoint may be analyzed separately, care must be exercised in selecting health endpoints to include in the overall benefits analysis because of the possibility of double-counting of benefits.

The specific studies from which effect estimates for the core analysis related to ozone exposure are drawn are included in Table 5-7. Table 5-8 identifies studies reflected in the dollar-per-ton analysis for $PM_{2.5}$. We highlight in red those studies that have been added since the benefits analysis conducted for the ozone reconsideration (U.S. EPA, 2010d) or the ozone NAAQS RIA (U.S. EPA, 2008a). 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 5-9 summarizes those health endpoints and studies we have included as sensitivity analyses for ozone.

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

Endpoint	Study	Study Population	Relative Risk or Effect Estimate (β) (with 95th Percentile Confidence Interval or SE, respectively)
	Premature Mortality		
Premature mortality—short-term	Smith et al. (2009)	All ages	β = 0.00032 (0.00008)
	Zanobetti and Schwartz (2008)		β = 0.00051 (0.00012)
Premature respiratory mortality-long-term (*incidence only*)	Jerrett et al. (2009)	>29 years	β = 0.003971 (0.00133)
	Hospital Admissions		
Respiratory	Pooled estimate: Katsouyanni et al. (2009)	> 65 years	β = 0.00061 (0.00041) natural splines β = 0.00064 (0.00040) penalized splines
Asthma-related emergency department visits	Pooled estimate:	0-99 years	
	Glad et al. (2012)		β = 0.00306 (0.00117)
	Ito et al. (2007)		β = 0.00521 (0.00091)
	Mar and Koenig (2010)		β = 0.01044 (0.00436) (0-17 yr olds) β = 0.00770 (0.00284) (18-99 yr olds)
	Peel et al. (2005)		β = 0.00087 (0.00053)
	Sarnat et al. (2013)		β = 0.00111 (0.00028)
	Wilson et al. (2005)		RR = 1.022 (0.996 – 1.049) per 25
	Other Health Endpoints		
Asthma exacerbations	Pooled estimate: [b]	6–18 years[c]	
	Mortimer et al. (2002)		β = 0.00929 (0.00387)
	O'Connor et al. (2008)		β = 0.00097 (0.00299)
	Schildcrout et al. (2006)		β = 0.00222 (0.00282)
School loss days	Pooled estimate:	5-17 years	
	Chen et al. (2000)		β = 0.015763 (0.004985)
	Gilliland et al. (2001)		β = 0.007824 (0.004445)
Acute respiratory symptoms (MRAD)	Ostro and Rothschild (1989)	18–65 years	β = 0.002596 (0.000776)

[a] Studies highlighted in red represent updates incorporated since the 2008 ozone NAAQS RIA (U.S. EPA, 2008a).
[b] As discussed in sections 5.3 and 5.6.3.1, while we believe that available evidence supports inclusion of estimates of long-term exposure-related respiratory mortality incidence in the core analysis, uncertainty in specifying a lag structure for this endpoint (a key factor in valuation) prevents us from including dollar benefits in the core analysis and instead, these are included as part of the sensitivity analysis exploring this endpoint.
[c] The original study populations were 5 to 12yrs for the O'Conner et al., (2008) and Schildcrout et al., (2006) and 5-9yrs for the Mortimer et al., (2002) study. Based on advice from the SAB-HES, we extended the applied population to 6-18yrs for all three studies, reflecting the common biological basis for the effect in children in the broader age group. See: U.S. EPA-SAB (2004a) and NRC (2002)

Table 5-8. Health Endpoints and Epidemiological Studies Used to Quantify $PM_{2.5}$-Related Health Impacts in the Core Analysis [a]

Endpoint	Study	Study Population	Relative Risk or Effect Estimate (β) (with 95th Percentile Confidence Interval or SE, respectively)
Premature Mortality			
Premature mortality—cohort study, all-cause	Krewski et al. (2009)	> 29 years	RR = 1.06 (1.04–1.06) per 10 μg/m³
	Lepeule et al. (2012)	> 24 years	RR = 1.14 (1.07–1.22) per 10 μg/m³
Premature mortality—all-cause	Woodruff et al. (1997)	Infant (< 1 year)	OR = 1.04 (1.02–1.07) per 10 μg/m³
Chronic Illness			
Nonfatal heart attacks	Peters et al. (2001)	Adults (> 18 years)	OR = 1.62 (1.13–2.34) per 20 μg/m³
	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)
	Kloog et al. (2012)—ICD 460-519 (All Respiratory		β=0.0007 (0.000961)
	Moolgavkar (2000)—ICD 490–496 (Chronic lung disease)	18–64 years	1.02 (1.01–1.03) per 36 μg/m³
	Babin et al. (2007)—ICD 493 (asthma)	< 19 years	β=0.002 (0.004337)
	Sheppard (2003)—ICD 493 (asthma)	< 18	RR = 1.04 (1.01–1.06) per 11.8 μg/m³
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. (2008a)—ICD 426-427; 428; 430-438; 410-414; 429; 440-449 (Cardio-, cerebro- and peripheral vascular disease)		β=0.0008 (0.000107)
	Moolgavkar (2000)—ICD 390–429 (all cardiovascular)	20–64 years	RR=1.04 (t statistic: 4.1) per 10 μg/m³
Asthma-related emergency department visits	Pooled estimate:	All ages	
	Mar et al. (2010)		RR = 1.04 (1.01–1.07) per 7 μg/m³
	Slaughter et al. (2005)		RR = 1.03 (0.98–1.09) per 10 μg/m³
	Glad et al. (2012)		β=0.00392 (0.002843)
Other Health Endpoints			

Acute bronchitis	Dockery et al. (1996)	8–12 years	OR = 1.50 (0.91–2.47) per 14.9 µg/m^3
Asthma exacerbations	Pooled estimate: Ostro et al. (2001) (cough, wheeze, shortness of breath) [b] Mar et al. (2004) (cough, shortness of breath)	6–18 years [b]	OR = 1.03 (0.98–1.07) OR = 1.06 (1.01–1.11) OR = 1.08 (1.00–1.17) per 30 µg/m^3 RR = 1.21 (1–1.47) per RR = 1.13 (0.86–1.48) per 10 µg/m^3
Work loss days	Ostro (1987)	18–65 years	β=0.0046 (0.00036)
Acute respiratory symptoms (MRAD)	Ostro and Rothschild (1989) (Minor restricted activity days)	18–65 years	β=0.00220 (0.000658)
Upper respiratory symptoms	Pope et al. (1991)	Asthmatics, 9–11 years	1.003 (1–1.006) per 10 µg/m^3
Lower respiratory symptoms	Schwartz and Neas (2000)	7–14 years	OR = 1.33 (1.11–1.58) per 15 µg/m^3

[a] Studies highlighted in red represent updates incorporated since the ozone NAAQS RIA (U.S. EPA, 2008a). These updates were introduced in the PM NAAQS RIA (U.S. EPA, 2012b).
[b] The original study populations were 8 to 13 for the Ostro et al. (2001) study and 7 to 12 for the Mar et al. (2004) study. Based on advice from the SAB-HES, we extended the applied population to 6-18, reflecting the common biological basis for the effect in children in the broader age group. See: U.S. EPA-SAB (2004a,b) and NRC (2002).

Table 5-9. Health Endpoints and Epidemiological Studies Used to Quantify Ozone-Related Health Impacts in the Sensitivity Analysis [a]

Endpoint	Study	Study Population	Relative Risk or Effect Estimate (β) (with 95th Percentile Confidence Interval or SE, respectively)
Premature Mortality			
Premature respiratory mortality—long-term	Jerrett et al. (2009)-based models:		
	- non-threshold ozone only (86 cities)	> 29 years	β=0.002664 (0.000969)
	- non-threshold ozone only (96 cities)		β=0.00286 (0.000942)
	- threshold 40 ppb[b]		β=0.00312 (0.00096)
	- threshold 45 ppb		β=0.00336 (0.001)
	- threshold 50 ppb		β=0.00356 0.00106)
	- threshold 55 ppb		β=0.00417 (0.00118)
	- threshold 56 ppb		β=0.00432 (0.00121)
	- threshold 60 ppb		β=0.00402 (0.00137)
Premature mortality—short-term	Smith et al. (2009) (copollutant model with PM_{10})	All ages	β=0.00026 (0.00017)
	Bell et al. (2005)		β=0.00080 (0.00021)
	Levy et al. (2005)		β=0.00112 (0.00018)
	Bell et al. (2004)		β=0.00026 (0.00009)
	Ito et al. (2005)		β=0.00117 (0.00024)
	Schwartz et al. (2005)		β=0.00043 (0.00015)
	Huang et al. (2005) (cardiopulmonary)		β=0.00026 (0.00009)

[a] Studies highlighted in red represent updates incorporated since the 2008 ozone NAAQS RIA (U.S. EPA, 2008b).
[b] All threshold models are ozone-only and based on the full 96 city dataset.

5.6.3.1 Ozone Premature Mortality Effect Coefficients

Core Mortality Effect Coefficients for Short-term ozone Exposure. The overall body of evidence indicates that there is likely to be a causal relationship between short-term ozone exposure and premature death. The 2013 ozone ISA states that:

> "The evaluation of new multi-city studies that examined the association between short-term ozone exposure and mortality found evidence which supports the conclusions of the 2006 ozone AQCD. These new studies reported consistent positive associations between short-term ozone exposure and all-cause (non-accidental) mortality, with associations persisting or increasing in magnitude during the warm season, and provide additional support for associations between ozone exposure and cardiovascular and respiratory mortality" (ozone ISA section 6.6.3, USEPA 2013).

The ISA concludes by stating that, "Although some uncertainties still remain, the collective body of evidence is sufficient to conclude there is likely to be a causal relationship between short-term ozone exposure and total mortality." (ozone ISA section 6.6.3, USEPA 2013a). Regarding potential confounding of the ozone mortality effect by PM, the ISA states, "Overall, across studies, the potential impact of PM indices on ozone-mortality risk estimates tended to be much smaller than the variation in ozone-mortality risk estimates across cities suggesting that ozone effects are independent of the relationship between PM and mortality." (ozone ISA section 6.3.3, USEPA 2013a). However, the ISA does note that the interpretation of the potential confounding effects of PM on ozone-mortality risk estimates requires caution. This caution reflects in part, the every-3rd- and every-6th-day PM sampling schedule (in most cities) which limits the overall sample size available for evaluating potential confounding of the ozone effect by PM. (ozone ISA section 6.3.3, USEPA 2013a).

In their review of the HREA, the SAB's CASAC Ozone Review Panel expressed support for epidemiological studies and corresponding concentration-response functions used in the HREA, which included effect estimates obtained from Smith et al., (2009) and Zanobetti and Schwartz (2008b) (Frey and Samet, 2012, p. 17-18 and Frey, 2014, p, 9). Furthermore, the CASAC specifically noted support for the use of multi-city studies, where available in modeling the health endpoints included in the analysis (Frey and Samet, 2012, p. 15). In addition, they expressed support for modeling total risk, and de-emphasized the importance of estimating risk associated with ozone concentrations above the lowest measure level (LML) from contributing epidemiological studies (Frey and Samet, 2012, p. 10 and 18).

In 2006, the EPA requested an NAS study to answer four key questions regarding ozone-related mortality: (1) how did the epidemiological literature to that point improve our understanding of the size of the ozone-related mortality effect? (2) How best can EPA quantify the level of ozone-related mortality impacts from short-term exposure? (3) How might EPA estimate the change in life expectancy? (4) What methods should EPA use to estimate the monetary value of changes in ozone-related mortality risk and life expectancy?

In 2008, the NAS (NRC, 2008) issued a series of recommendations to the EPA regarding the quantification and valuation of ozone-related short-term mortality. Chief among these was that "…short-term exposure to ambient ozone is likely to contribute to premature deaths" and the committee recommended that "ozone-related mortality be included in future estimates of the health benefits of reducing ozone exposures…" The NAS also recommended that "…the greatest emphasis be placed on the multi-city and NMMAPS studies without exclusion of the meta-analyses" (NRC, 2008). In addition, NAS recommended that EPA "should give little or no weight to the assumption that there is no causal association between estimated reductions in premature mortality and reduced ozone exposure" (NRC, 2008). In 2010, the Health Effects Subcommittee of the Advisory Council on Clean Air Compliance Analysis, while reviewing EPA's *The Benefits and Costs of the Clean Air Act 1990 to 2020* (U.S. EPA, 2011a), also confirmed the NAS recommendation to include ozone mortality benefits (U.S. EPA-SAB, 2010a).

In view of the findings of the ozone ISA, the NAS panel, the HES panel, and the CASAC panel, we include ozone-related premature mortality for short-term exposure in the core health effects analysis using effect coefficients from the Smith et al. (2009) NMMAPS analysis and the Zanobetti and Schwartz (2008) multi-city study. As discussed below, we also include several additional studies as sensitivity analyses. This approach with an emphasis on newer multi-city studies is consistent with recommendations provided by the NAS in their ozone mortality report

(NRC, 2008), "The committee recommends that the greatest emphasis be placed on estimates from new systematic multi-city analyses that use national databases of air pollution and mortality, such as in the NMMAPS, without excluding consideration of meta-analyses of previously published studies." In selecting the Smith et al. (2009) and Zanobetti and Schwartz (2008b) studies, we point both to CASAC support for the use of these two studies in the context of the HREA completed for this NAAQS review (Samet and Frey, 2012, p. 15 and Frey, 2014, p. 9) and the fact that both of these studies are multi-city studies published more recently (as compared with other multi-city studies or meta-analyses included in the sensitivity analyses – see discussion below).

The Smith et al., (2009) study is a reanalysis of the NMMAPS data set focused on evaluating the relationship between short-term ozone exposure and mortality. While this reproduces the core national-scale estimates presented in Bell et al., (2004), it also explores the sensitivity of the mortality effect to different model specifications including (a) regional versus national Bayes-based adjustment,[47] (b) co-pollutants models considering PM_{10}, (c) all-year versus ozone-season based estimates, and (d) consideration of a range of ozone metrics, including the daily 8hr max, which is of particular interest in the context of the RIA given that is the metric that is used in the form of the ozone standard. In addition, the Smith et al. (2009) study does not use the trimmed mean approach employed in the Bell et al. (2004) study in preparing ozone monitor data, which is another advantage.[48] In selecting effect estimates from Smith et al. (2009) for use in the core analysis, we focused on an ozone-only estimate for non-accidental mortality based on the 8hr max metric for the warmer ozone season. In addition, for the sensitivity analysis, we included a copollutants model (ozone and PM_{10}) from Smith et al. (2009) for all-cause mortality which also used the 8hr max ozone metric for the ozone season. As

[47] In Bayesian modeling, effect estimates are "updated" from an assumed prior value using observational data. In the Smith et al (2009) approach, the prior values are either a regional or national mean of the individual effect estimates obtained for each individual city. The Bayesian adjusted city-specific effect estimates are then calculated by updating the selected prior value based on the relative precision of each city-specific estimate and the variation observed across all city-specific individual effect estimates. City-specific estimates are pulled towards the prior value if they have low precision and/or there is low overall variation across estimates. City-specific estimates are given less adjustment if they are precisely estimated and/or there is greater overall variation across estimates.

[48] There are a number of concerns regarding the trimmed mean approach including (1) the potential loss of temporal variation in the data when the approach is used (this could impact the size of the effect estimate) and (2) a lack of complete documentation for the approach which prevents a full reviewing or replication of the technique.

noted in Table 5-6, the decision to use a single pollutant model for the core analysis and reserve the copollutants model for the sensitivity analysis reflects our concern that the reduced sampling frequency for days with copollutants measurements (1/3 and 1/6) can impact characterization of the ozone effect which is the focus of this assessment. In addition, as noted earlier in this section, the latest ozone ISA states that ozone effects are likely to be independent of the relationship between PM and mortality, which further supports the approach of favoring single pollutant models in the core analysis.

The Zanobetti and Smith (2008) study evaluated the relationship between ozone exposure (using an 8hr mean metric for the warm season June-August) and all-cause mortality in 48 U.S. cities using data collected between 1989 and 2000. The study presented single pollutant concentration-response functions based on shorter (0-3 day) and longer (0-20) day lag structures, with the comparison of effects based on these different lag structures being a central focus of the study. For the core analysis, we used the 0-3 day lag based concentration-response function since this had the strongest effect and tighter confidence interval. Note that, because the RIA utilizes the 8hr max ozone metric, we had to covert the effect estimate from Zanobetti and Smith (2008) which is based on an 8hr mean metric to an equivalent effect estimate based on an 8hr max. To do this, we used the ozone metric approach wherein the original effect estimate (and standard error) is multiplied by the appropriate ozone metric adjustment ratio.[49]

Core Mortality Effect Coefficient for Long-term ozone Exposure. We also estimated long-term ozone exposure-related respiratory mortality <u>incidence</u> in the core analysis. The available evidence did not allow us to characterize how long-term exposure to ozone related to the year of onset of mortality (i.e., a cessation lag), which is a necessary input to quantifying the discounted dollar benefits. For this reason, we report the <u>dollar benefits</u> of avoided long-term exposure-related respiratory deaths as a sensitivity analyses (see section 5.7.3.1). Support for

[49] These adjustment ratios are created by (a) obtaining summary air quality (composite monitor values) for each urban study area/ozone season combination reflected in the original epidemiology study, (b) calculating the ratio of the 8hr max to the study-specific air metric (for each of the urban study areas), and (c) taking the average of these urban-study area ratios. Ratio adjustment of the effect estimate does introduce uncertainty into the benefits estimates generated using these adjusted effect estimates, however, adjustments of relatively similar metrics (e.g., 8hr max and 8hr mean), as is the case with the Zanobetti and Schwartz (2008) study, are likely to introduce less uncertainty than adjustments for more disparate ratios (e.g., 24hr or 1hr max ratios to 8hr max equivalents).

modeling long-term exposure-related mortality incidence comes from the final ozone ISA as well as recommendations provided by CASAC in their review of the HREA completed for the current ozone NAAQS review (Frey, 2014, p. 3 and 9).

The final ozone ISA references long-term respiratory mortality in section 7.2.1 (USEPA 2013a) where they state, "The positive results from various designs and locations support a relationship between long-term exposure to ambient ozone concentrations and respiratory health effects and mortality." Later in that chapter, the ISA states that: "The strongest evidence for an association between long-term exposure to ambient ozone concentrations and mortality is derived from associations reported in the Jerrett et al. (2009) study for respiratory mortality that remained robust after adjusting for $PM_{2.5}$ concentrations." (Section 7.7.1, USEPA, 2013a). In that same section, the authors also state that:

> "Coherence and biological plausibility for this observation [the association between long-term exposure and respiratory mortality] is provided by evidence from epidemiologic, controlled human exposure, and animal toxicological studies for the effects of short- and long-term exposure to ozone on respiratory effects (see Sections 6.2 and 7.2). Respiratory mortality is a relatively small portion of total mortality [about 7.6% of all deaths in 2010 were due to respiratory causes (Murphy et al., 2012)], thus it is not surprising that the respiratory mortality signal may be difficult to detect in studies of cardiopulmonary or total mortality."

While the ozone ISA concludes that evidence is *suggestive of a causal association* between total mortality and long-term ozone exposure (section 7.7.1), specifically with regard to respiratory health effects (including mortality), the ISA concludes that there is *likely to be a causal association* (section 7.2.8).

In their review of the HREA completed for the ozone NAAQS review and specifically modeling of the long-term exposure-related respiratory mortality endpoint, the CASAC states that, "The basis for estimating long-term mortality (respiratory) risks relies on a single study, Jerrett et al. (2009), and the HREA should acknowledge the uncertainty and confidence in modeling results from the use of a single study, albeit a good one." And, while they comment on the size of the chronic obstructive pulmonary disease (COPD) mortality effect attributable to ozone (page 7-68) they go on to state that, “the CASAC concurs that Jerrett et al. (2009) is an appropriate study to use at this time as the basis for the long-term mortality risk estimates given its adequacy and the lack of alternative data.” (Frey, C., 2014). This advice supersedes previous advice provided by the HES to include long-term ozone exposure-related mortality only as a sensitivity analysis.

The Jerrett et al. (2009) study was the first to explore the relationship between long-term ozone exposure and respiratory mortality (rather than focusing on cardiopulmonary mortality). Jerrett et al. (2009) exhibits a number of strengths including (a) the study is based on the 1.2 million participant American Cancer Society cohort drawn from all 50 states, DC, and Puerto Rico (included ozone data from 1977 [5 years before enrollment in the cohort began] to 2000); (b) it includes copollutants models that controlled for $PM_{2.5}$; and (c) it explored the potential for a threshold concentration associated with the long-term mortality endpoint. However there are attributes to this study that affect how we interpret the long-term exposure-related respiratory mortality estimates. First, while CASAC notes that Jerrett et al. (2009) is well designed, it is a single study—and so provides the only quantitative basis for estimating this endpoint. By comparison, we estimate short-term exposure-related mortality risk using several studies.

There is also the potential existence and location of a threshold in the C-R function relating mortality and long-term ozone concentrations. That uncertainty could greatly influence our quantitative risk estimates (we address the potential for a threshold in a sensitivity analysis below). The CASAC did address the use of the zero threshold versus threshold based models in the context of the HREA, stating that, “The EPA examined the threshold analysis contained in Jerrett et al. (2009) and found that the mortality model including a threshold at 56 ppb had the lowest log likelihood value of all models examined. However, it is not clear whether the 56 ppb threshold model is a better predictor of respiratory mortality than when using a linear model for the Jerrett et al. data. Different, but valid statistical tests produced different conclusions about the threshold versus linear models. The less stringent test judged the 56 ppb threshold model to be superior to the linear model, but the confidence interval indicates the threshold could occur anywhere from 0 to 60 ppb. Using the more stringent statistical test, none of the threshold models produce better predictions than the linear model. Given these results, the CASAC concurs with the EPA’s planned approach [as stated in the 2nd draft HREA] to conduct a sensitivity analysis evaluating potential thresholds in the C-R functions that relate long-term ozone exposures with respiratory mortality and to not make the threshold models the core analytical procedure in the PA." (Frey, 2014, p. 13-14) Here, the CASAC clearly states their support for inclusion of the zero threshold (linear model) as the core approach, while treating the threshold models as sensitivities.

Reflecting this advice provided by CASAC, we have generated the core benefit estimate for long-term exposure-related respiratory mortality using a non-threshold co-pollutant model (with $PM_{2.5}$) obtained from Jerrett et al. (2009) (see Tables 5-7 and 5-9). Using a co-pollutant model is consistent with the fact that this study applied seasonal average metrics that are insensitive to co-pollutant monitoring for $PM_{2.5}$; we explore the influence of co-pollutant models and thresholds in a sensitivity analysis below.[50] The effect estimates used to model long-term ozone-attributable mortality are calculated using a seasonal average of peak (1-hr maximum) measurements. These long-term exposure metrics can be viewed as long-term exposures to daily peak ozone over the warmer months, as compared with annual average levels such as are used in long-term PM exposure calculations. This increases the need for care in attempting to combine estimates of long-term ozone-attributable mortality and short-term ozone-attributable mortality estimates, in order to avoid double counting. It is also important to keep in mind that our estimates of short-term ozone- attributable mortality are for all-causes, while estimates of long-term ozone-attributable mortality are focused on respiratory-related mortality. This further limits the ability to compare estimates of long-term and short-term exposure related mortality.

[50] See Table 5-6 for additional discussion of the issue involving reduced sampling frequency and modeling of short-term ozone exposure-related mortality.

Sensitivity Analysis: Alternate Mortality Effect Coefficients for Short-term Ozone Exposure. In addition to the ozone-related studies we use for the core estimates, we also evaluate several alternative studies to characterize uncertainty in the core estimates. These alternative studies include a mix of meta-analyses and multi-city studies which have been included in RIAs completed for previous ozone NAAQS reviews, including the review completed in 2008 as well as the Reconsideration completed in 2010 (USEPA, 2008a and USEPA 2010d, respectively). The decision to include a mixture of meta-analyses and multi-city studies in the sensitivity analysis reflects the recommendation from the NRC reference earlier, that in modeling short-term exposure-related mortality for ozone, emphasis be placed on more recent multi-city studies without excluding consideration for meta-analyses.

In selecting effect estimates from each study, we followed the criteria presented in Table 5-6. Consequently, we favored effect estimates reflecting the warmer ozone monitoring period, if available. We also favored effect estimates based on the 8hr max air metric if available. And finally, while we considered multi-pollutant models (providing some coverage for potential confounding by PM), we placed primary emphasis on single-pollutant models since these would typically have a significantly larger dataset with which to specify the ozone mortality effect. The decision to emphasize single-pollutant models and deemphasize copollutants models (specifically in modeling short-term endpoints) reflected observations in the current ozone ISA. In relation to short-term mortality, the authors note limitations of co-pollutant models due to the reduced sampling frequency association with PM in most cities. However, they also note that, "Together, these co-pollutant-adjusted findings across respiratory endpoints provide support for the independent effects of short-term exposures to ambient ozone." (ozone ISA section 6.2.9, USEPA, 2013a). The set of multi-city and meta-analysis studies selected for inclusion in the sensitivity analysis (including effect estimates and standard errors) is presented in Table 5-9. Studies and effect estimates selected for the core analysis are also included in the table for completeness. Detailed discussions of each of these studies can be found in the current ozone ISA as well as the RIA for the ozone NAAQS completed in 2008.

Threshold-Based Effect Coefficients for Long-term Ozone Exposure. As discussed in the HREA completed as part of this NAAQS review (U.S. EPA, 2014b), the exploration of potential thresholds for long-term exposure-related respiratory mortality (as discussed in the Jerrett et al., 2009 study) deserves additional discussion.[51] In their memo clarifying the results of their study (see Sasser, 2014), the authors note that in terms of goodness of fit, long-term health risk models including ozone clearly performed better than models without ozone, indicating the improved predictions of respiratory mortality when ozone is included. In exploring different functional forms, they report that the model including a threshold at 56 ppb had the lowest log-likelihood value of all models evaluated (i.e., linear models and models including thresholds ranging from 40-60 ppb), and thus provided the best overall statistical fit to the data. However, they also note that it is not clear whether the 56 ppb threshold model is a better predictor of respiratory mortality than when using a linear (no-threshold) model for this dataset. Using one statistical test, the model with a threshold at 56 ppb was determined to be statistically superior to the linear model. Using another, more stringent test, none of the threshold models considered were statistically superior to the linear model. Under the less stringent test, although the threshold model produces a statistically superior prediction than the linear model, there is uncertainty about the specific location of the threshold, if one exists. This is because the confidence intervals on the model predictions indicate that a threshold could exist anywhere from 0 to 60 ppb. The authors conclude that considerable caution should be exercised in using any specific threshold, particularly when the more stringent statistical test indicates there is no significantly improved prediction.

Based on this additional information from the authors (Sasser, 2014), we have chosen to reflect the uncertainty about the existence and location of a potential threshold by estimating mortality attributable to long-term ozone exposures using a range of threshold-based effect estimates as sensitivity analyses (see section 5.7.3.1 and Appendix 5B, section 5B.1). Specifically, we generate additional long-term risk results using unique risk models that include

[51] The approach we developed to explore the potential for thresholds related to long-term exposure-related mortality was presented in a memorandum to CASAC which was also released to the public (Sasser, 2014). That memorandum describes additional data obtained from the authors of Jerrett et al. (2009) to support modeling of potential thresholds and also lays out our proposed approach for exploring the impact of potential thresholds on estimates of long-term exposure-related mortality (including presentation of the threshold-based results as a sensitivity analysis and inclusion of the non-threshold model-based results as the core analysis). This plan, including these details related to presentation of the threshold and non-threshold based estimates were supported by CASAC (Frey, 2014, p. 13-14).

a range of thresholds from 40 ppb to 60 ppb in 5 ppb increments, while also including a model with a threshold equal to 56 ppb, which had the lowest log likelihood value for all models examined.[52] In addition, to exploring the impact of potential thresholds, as part of the sensitivity analysis we also explore the impact of using ozone-only (non-threshold) models in estimating long-term exposure-related respiratory mortality.[53]

Ozone Exposure Metric. Both the NMMAPS analysis and the individual time series studies upon which the meta-analyses were based use the 24-hour average or 1-hour maximum ozone concentrations as exposure metrics. The 24-hour average is not the most relevant ozone exposure metric to characterize population-level exposure. Given that the majority of the people tend to be outdoors during the daylight hours and concentrations are highest during the daylight hours, the 24-hour average metric is not appropriate. Moreover, the 1-hour maximum metric uses an exposure window different than that used for the current ozone NAAQS. Together, this means that the most biologically relevant metric, and the one used in the ozone NAAQS since 1997 is the maximum daily 8-hour average ozone. Thus, we have converted ozone mortality health impact functions that use a 24-hour average or 1-hour maximum ozone metric to maximum 8-hour average ozone concentration using standard conversion functions.

This practice is consistent with the form of the current ozone standard. This conversion also does not affect the relative magnitude of the health impact function from a mathematical standpoint. An equivalent change in the 24-hour average, 1-hour maximum and 8-hour maximum will provide the same overall change in incidence of a health effect.[54] The conversion ratios are based on observed relationships between the 24-hour average and 8-hour maximum

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[52] There is a separate effect estimate (and associated standard error) for each of the fitted threshold models estimated in Jerrett et al. (2009). As a result, the sensitivity of estimated mortality attributable to long-term ozone concentrations is affected by both the assumed threshold level (below which there is no effect of ozone) and the effect estimate applied to ozone concentrations above the threshold.

[53] The set of ozone-only non-threshold effect estimates include (a) a value based on the 86 cities for which there are copollutants monitoring data for both ozone and $PM_{2.5}$ (this best compared with the core estimate based on the copollutants non-threshold model) and (b) a value based on the 96 cities for which there is $PM_{2.5}$ data (these 96 cities were used in developing the threshold-based effect estimates used in the analysis).

[54] However, it is important to note that different ozone metrics may not be well-correlated (from either a spatial or temporal standpoint) within a given geographic area which means that application of ratio-converted effect estimates for the same endpoint can result in different incidence estimates for the same location under certain conditions. This introduces uncertainty into the use of these ratio-adjusted effect estimates (see Appendix 5A).

ozone values. For example, in the Bell et al., 2004 analysis of ozone-related premature mortality, the authors found that the relationship between the 24-hour average, the 8-hour maximum, and the 1-hour maximum was 2:1.5:1, so that the derived health impact effect estimate based on the 1-hour maximum should be half that of the effect estimate based on the 24-hour values (and the 8-hour maximum three-quarters of the 24-hour effect estimate).

In the sensitivity analyses for this benefits analysis, we apply national effect estimates based on the pooled multi-city results reported in Bell et al (2004) and the three meta-analysis studies. Bell et al (2004), Bell et al (2005), Levy et al (2005), and Ito et al (2005) all provide national conversion ratios between daily average and 8-hour and 1-hour maxima, based on national data.

5.6.3.2 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 (ED) 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.

With regard to short-term hospital admissions and ED visits, the current ozone ISA states that, "[c]ompared with studies reviewed in the 2006 ozone AQCD, a larger number of recent studies examined hospital admissions and ED visits for specific respiratory outcomes. Although limited in number, both single- and multi-city studies consistently found positive associations between short-term ozone exposures and asthma and COPD hospital admissions and ED visits, with more limited evidence for pneumonia. Consistent with the conclusions of the 2006 ozone AQCD, in studies that conducted seasonal analyses, risk estimates were elevated in the warm season compared to cold season or all-season analyses, specifically for asthma and COPD." (ozone ISA section 6.2.9, USEPA, 2013a). In this same section, the ISA also addresses potential thresholds in effect: "Although the C-R relationship has not been extensively examined, preliminary examinations found no evidence of a threshold between short term ozone exposure and asthma hospital admissions and pediatric asthma ED visits…" Regarding the potential for confounding by other pollutants including PM, the ISA observes that, "Several epidemiologic studies of respiratory morbidity and mortality evaluated the potential confounding effects of copollutants, in particular, PM_{10}, $PM_{2.5}$, or NO_2. In most cases, effect estimates remained robust to the inclusion of copollutants." (ozone ISA section 6.2.9, USEPA 2013a).

Based on consideration for these observation from the ISA, and a thorough review of available epidemiological studies, for the core analysis, we model respiratory hospital admissions (for 65-99yr olds) using effect estimates obtained from Katsouyanni et al., 2009 and asthma-related emergency room visits (for all ages) using several single-city studies. The Katsouyanni et al., 2009 study is for all respiratory hospital admissions, and thus to avoid double counting, we do not provide separate estimates for specific subcategories of respiratory admissions such as asthma. The Katsouyanni et al., 2009 study provides effect estimates specific to the summer season, which is an advantage, however it also utilizes the 1hr max metric, which required adjustment using air metric ratios to generate equivalent 8hr max effect estimates for use in the RIA.[55] The study provides summer season single pollutant effect estimates based both on natural and penalized splines. We used both of these effect estimates and pooled the results using equal-weight averaging. It is also important to note that, while the Katsouyanni et al., 2009 study did include a set of effect estimates based on copollutants modeling (with PM_{10}), these were based on the full year rather than the summer season. Given our focus on warmer ozone season-based models, we only considered the single pollutant models in the RIA.

A number of studies are available to model respiratory ED visits. However, at this time we do not have a valuation function for this endpoint. Since we do have a valuation function available for the narrower category of asthma-related ED visits, for the core estimate, we have

[55] Given that the Katsouyanni et al., 2009 study included a larger number of cities (14), rather than constructing an air metric adjustment ratio based on this set of urban study areas, we used a national ratio to adjust effect estimates to represent the 8hr metric used in the RIA.

focused on asthma-related ED visits (for which we can estimate the economic value), using a set of single city studies together with random-effects pooling to generate a single pooled estimate. The set of single city studies used in this calculation include: Peel et al., (2005) and Sarnat et al., (2013) both for Atlanta, Wilson et al., (2005) and Mar and Koenig (2009) for Seattle, Wilson et al., (2005) for Portland ME, Ito et al., (2007) for New York City, and Glad et al., (2012) for Pittsburgh. We note that of these single city studies, only the Ito et al., (2007) study included a co-pollutant model (for $PM_{2.5}$).[56] In addition, two of the studies required adjustments of their betas to reflect the 8hr max air metric. Specifically, Glad et al., (2012) utilizes the 1hr max air metric, while Sarnat et al., 2013 utilized the 24hr average metric. Each required the use of air metric ratios to adjust their betas. In generating a single pooled benefit estimate for this endpoint, we used random/fixed effects pooling to combine estimates across these single city studies.

5.6.3.3 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 ozone and $PM_{2.5}$. The sources for the effect estimates used to quantify these effects are described below.

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

[56] While we have included copollutants models as sensitivity analyses for mortality, given the reduced role of morbidity endpoints in driving overall dollar benefits, we have not included separate copollutants models for any of the morbidity endpoints as sensitivity analyses. In the case of the Ito et al., (2007) study, we do note, that while the copollutants model does result in a somewhat smaller effect estimate for asthma ED visits as compared with the single pollutant (ozone-only) model, the SE for the copollutants model is also larger (as would be expected).

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.

Based on advice from the SAB-HES (EPA-SAB 2004a), regardless of the age ranges included in the source epidemiology studies, we extend the applied population to ages 6 to 18, reflecting the common biological basis for the effect in children in the broader age group. This age range expansion is also supported by NRC (2002, pp. 8, 116).

To characterize asthma exacerbations in children from exposure to ozone, we selected three multi-city studies (Mortimer et al., 2002, O'Connor et al., 2008, and Schildcrout et al., 2006). Of these three, one of the studies (O'Connor et al., 2008) only included a multi-pollutant model (for $PM_{2.5}$ and NO_2) and consequently, that effect estimate was used in the core analysis. All three of these studies required the application of air metric ratios to adjust effect estimates to represent the 8hr metric used in the RIA.[57] To combine these three estimates into a single pooled estimate, we used equal weights.

Acute Respiratory Symptoms. We estimate one type of acute respiratory symptom related to ozone exposure - MRAD. Minor restricted activity days 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 work and phone work because of difficulty breathing or chest pain.

For ozone, we modeled MRADs using Ostro and Rothschild (1989). This study provides a copollutants model (with $PM_{2.5}$) based on a national sample of 18-64yr olds. The original study used a 24hr average metric and included control for $PM_{2.5}$, which necessitated the use of an air metric ratio to convert the effect estimate to an 8hr max equivalent.

[57] Mortimer et al., (2002) had effect estimates based on an 8hr mean metric, O'Connor et al., (2008) utilized a 24hr metric and Schildcrout et al., (2006) was based on a 1hr max metric. Consequently, all three studies required the application of air metric ratios to produce effect estimates reflecting an 8hr max metric.

School loss days (absences). Children may be absent from school due to respiratory or other acute diseases caused, or aggravated by, exposure to air pollution. Several studies have found a significant association between ozone levels and school absence rates. We use two studies (Gilliland et al., 2001; Chen et al., 2000) to estimate changes in school absences resulting from changes in ozone levels. The Gilliland et al. study estimated the incidence of new periods of absence, while the Chen et al. study examined daily absence rates. We converted the Gilliland et al. estimate to days of absence by multiplying the absence periods by the average duration of an absence. We estimated 1.6 days as the average duration of a school absence, the result of dividing the average daily school absence rate from Chen et al. (2000) and Ransom and Pope (1992) by the episodic absence duration from Gilliland et al. (2001). Thus, each Gilliland et al. period of absence is converted into 1.6 absence days.

Following advice from the National Research Council (NRC, 2002), we calculated reductions in school absences for the full population of school age children, ages five to 17. This is consistent with recent peer-reviewed literature on estimating the impact of ozone exposure on school absences (Hall et al., 2003). We estimated the change in school absences using both Chen et al. (2000) and Gilliland et al. (2001) and then pooled the results using the random effects pooling procedure.

5.6.3.4 Unquantified Human Health Effects

The illustrative emission reduction strategies to reach the proposed and alternative standards described in Chapter 4 would reduce emissions of NOx and VOCs. Although we have quantified many of the health benefits associated with reducing exposure to ozone and $PM_{2.5}$, as shown in Table 5-3, we are unable to quantify the health benefits associated with reducing the potential for NO_2 or VOC exposures due to the absence of air quality modeling data for these pollutants in this analysis. In addition, we are unable to quantify the effects of VOC reductions on ambient $PM_{2.5}$ and associated health effects. Although the method we applied simulated the impact of attaining the proposed and alternative standards on ambient levels of ozone, this method does not simulate how the illustrative emission reductions would affect ambient levels of NO_2 or VOC. Below we provide a qualitative description of these health benefits. In general, previous analyses have shown that the monetized value of these additional health benefits is much smaller than ozone and $PM_{2.5}$-related benefits (U.S. EPA, 2010a, 2010c, 2010d).

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* (NO2 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 percent 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.

The EPA last quantified the value of ozone-related worker productivity in the final Regulatory Impact Analysis supporting the Transport Rule (USEPA, 2011d). That analysis applied information reported in Crocker and Horst (1981) to relate changes in ground-level ozone to changes in the productivity of outdoor citrus workers. That study found that a 10 percent reduction in ozone translated to a 1.4 increase in income **among outdoor citrus workers**. Concerned that this study might not adequately characterize the relationship between ground-level ozone and the productivity of agricultural workers because of the vintage of the underlying data, the Agency subsequently omitted this endpoint.

In 2012, Graff Zivin and Neidell published "The impact of pollution on worker productivity" in the American Economic Review. That study combined data on individual-level daily harvest rates for Outdoor Agricultural Workers (OWAs) with ground-level ozone pollution to characterize changes in worker productivity. The authors used data on harvest rates from a 500-acre farm in the Central Valley of California. That farm produced three crops (blueberries and two types of grapes) and the harvesting laborers were paid through piece rate contracts. The analyses in the paper were based on 2009 and 2010 California growing seasons. The analyses were not affected by: (i) endogenous ozone exposure (because there were limited local sources of ozone precursors); (ii) avoidance behavior (because the work has to be performed outdoors); and (iii) shirking (due to the nature of the piece rate contract).

Table 3 in Graff Zivin and Neidell (2012) reports the main result: A 10 ppb increase in work-day ozone concentration (represented by hourly measurements averaged between 6am and 3pm) will result in a decline of 0.143 (with a standard error of 0.068) in standardized hourly pieces collected on a given work day. The standardized hourly pieces were "the average hourly productivity minus the minimum number of pieces per hour required to reach the piece rate regime, divided by the standard deviation of productivity for each crop" (Graff Zivin and Neidell, 2012; p. 3665). The range of ozone concentrations in the sample was between 10.50 ppb and 86.0 ppb (Table 1 in Graff Zivin and Neidell, 2012). This result is significant and robust under different model specifications designed to test modeling assumptions. Based on the effect estimate and individual-level information in their dataset, the authors estimated the effect of an increase in ozone concentration on the worker productivity, as measured by the average number of pieces collected per hour during a given work day (rather than by *standardized* hourly piece rate that was used in regression modeling). They found a decline of 5.5% in worker productivity due to a 10 ppb increase in average work-day ozone concentration.

While Graff Zivin and Neidell (2012) report the information needed to quantify ozone-related worker productivity, we are still evaluating whether and how to most appropriately apply the limited evidence from this study in a national benefits assessment. An important issue is the generalizability of the results to the appropriate population. We are considering the appropriateness of applying the results of this study to estimate the benefits of increased worker productivity as part of the final ozone RIA, and seek public comment on this approach as an important input to our consideration.

5.6.4 *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 willingness-to-pay (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. 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.

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. Cost-of-illness (COI) estimates generally (although not necessarily in all cases) understate the true value of reducing the risk of a health effect, because they reflect the direct expenditures related to treatment, but not the value of avoided pain and suffering (Harrington and Portney, 1987; Berger, 1987).

We provide unit values for health endpoints (along with information on the distribution of the unit value) in Table 5-10.[58] All values are in constant year 2011$, adjusted for growth in real income for WTP estimates out to 2024 using projections provided by Standard and Poor's,

[58] We note that a number of the endpoints included in Table 5-10 and discussed in this section were only modeled for $PM_{2.5}$ and consequently could be moved to Appendix 5C (as was done with the discussion of effect estimates specific to $PM_{2.5}$). However, given that many of the valuation functions discussed are shared between the two pollutants and given the relatively shorter length of discussions associated with these valuation functions, we have included coverage of all valuation functions (for ozone and $PM_{2.5}$-related endpoints) in this section.

which is discussed in further detail below.[59] Economic theory argues that WTP for most goods (such as environmental protection) will increase if real income increases. Several of the valuation studies used in this analysis were conducted in the late 1980s and early 1990s, and we are in the process of reviewing the literature to update these unit values. The discussion below provides additional details on valuing specific $PM_{2.5}$-related related endpoints.

5.6.4.1 Mortality Valuation

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

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[59] Income growth projections are only currently available in BenMAP through 2024, so both the 2025 and 2038 estimates use income growth only through 2024 and are therefore likely underestimates. Currently, BenMAP does not have an inflation adjustment to 2011$. We ran BenMAP for a currency year of 2010$ and calculated the benefit-per-ton estimates in 2010$. We then adjusted the resulting benefit-per-ton estimates to 2011$ using the Consumer Price Index (CPI-U, all items). This approach slightly underestimates the inflation for medical index and wage index between 2010 and 2011, which affects COI estimates and wage-based estimates.

Table 5-10. Unit Values for Economic Valuation of Health Endpoints (2011$) [a]

Health Endpoint	Central Estimate of Value Per Statistical Incidence: 1990 Income Level	Central Estimate of Value Per Statistical Incidence: 2024 Income Level	Derivation of Distributions of Estimates
Premature Mortality (Value of a Statistical Life)	$8,300,000	$10,000,000	The EPA currently recommends a central VSL of $4.8 million (1990$, 1990 income) based on a Weibull distribution fitted to 26 published VSL estimates (5 contingent valuation and 21 labor market studies). The underlying studies, the distribution parameters, and other useful information are available in Appendix B of the EPA's Guidelines for Preparing Economic Analyses (U.S. EPA, 2010e).
Nonfatal Myocardial Infarction (heart attack)			No distributional information available. Age-specific cost-of-illness values reflect lost earnings and direct medical costs over a 5-year period following a nonfatal MI. Lost earnings estimates are based on Cropper and Krupnick (1990). Direct medical costs are based on simple average of estimates from Russell et al. (1998) and Wittels et al. (1990).
3% discount rate			
Age 0–24	$100,000	$100,000	
Age 25–44	$110,000	$110,000	
Age 45–54	$120,000	$120,000	
Age 55–64	$210,000	$210,000	Lost earnings:
Age 65 and over	$100,000	$100,000	Cropper and Krupnick (1990). Present discounted value of 5 years of lost earnings in 2000$:
7% discount rate			age of onset: at 3% at 7%
Age 0–24	$100,000	$100,000	25–44 $9,000 $8,000
Age 25–44	$110,000	$110,000	45–54 $13,000 $12,000
Age 45–54	$120,000	$120,000	55–65 $77,000 $69,000
Age 55–64	$190,000	$190,000	Direct medical expenses (2000$): An average of:
Age 65 and over	$100,000	$100,000	1. Wittels et al. (1990) ($100,000—no discounting) 2. Russell et al. (1998), 5-year period ($22,000 at 3% discount rate; $21,000 at 7% discount rate)

(continued)

Table 5-10. Unit Values for Economic Valuation of Health Endpoints (2011$) [a] (continued)

Health Endpoint	Central Estimate of Value Per Statistical Incidence		Derivation of Distributions of Estimates
	2000 Income Level	2024 Income Level	
Hospital Admissions			
Chronic Lung Disease (18–64)	$22,000	$22,000	No distributional information available. The COI estimates (lost earnings plus direct medical costs) are based on ICD-9 code-level information (e.g., average hospital care costs, average length of hospital stay, and weighted share of total chronic lung illnesses) reported in Agency for Healthcare Research and Quality (2007) (www.ahrq.gov).
Asthma Admissions (0–64)	$16,000	$16,000	No distributional information available. The COI estimates (lost earnings plus direct medical costs) are based on ICD-9 code-level information (e.g., average hospital care costs, average length of hospital stay, and weighted share of total asthma category illnesses) reported in Agency for Healthcare Research and Quality (2007) (www.ahrq.gov).
All Cardiovascular Age 18–64 Age 65–99	 $44,000 $42,000	 $44,000 $42,000	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 cardiovascular category illnesses) reported in Agency for Healthcare Research and Quality (2007) (www.ahrq.gov).
All respiratory (ages 65+)	$37,000	$37,000	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	$440	$440	No distributional information available. Simple average of two unit COI values (2000$): (1) $310, from Smith et al. (1997) and (2) $260, from Stanford et al. (1999).

(continued)

Table 5-10. Unit Values for Economic Valuation of Health Endpoints (2011$) [a] (continued)

Health Endpoint	Central Estimate of Value Per Statistical Incidence 2000 Income Level	Central Estimate of Value Per Statistical Incidence 2024 Income Level	Derivation of Distributions of Estimates
Respiratory Ailments Not Requiring Hospitalization			
Upper Respiratory Symptoms (URS)	$35	$32	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)	$22	$21	Combinations of the four symptoms for which WTP estimates are available that closely match those listed by Schwartz et al. result in 11 different "symptom clusters," each describing a "type" of LRS. A dollar value was derived for each type of LRS, using mid-range estimates of WTP (IEc, 1994) to avoid each symptom in the cluster and assuming additivity of WTPs. The dollar value for LRS is the average of the dollar values for the 11 different types of LRS. In the absence of information surrounding the frequency with which each of the 11 types of LRS occurs within the LRS symptom complex, we assumed a uniform distribution between $6.9 and $25 (2000$).
Asthma Exacerbations	$56	$60	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 to have a uniform distribution between $16 and $71 (2000$).

(continued)

Table 5-10. Unit Values for Economic Valuation of Health Endpoints (2011$) [a] (continued)

Health Endpoint	Central Estimate of Value Per Statistical Incidence: 2000 Income Level	Central Estimate of Value Per Statistical Incidence: 2024 Income Level	Derivation of Distributions of Estimates
Respiratory Ailments Not Requiring Hospitalization (continued)			
Acute Bronchitis	$460	$500	Assumes a 6-day episode, with the distribution of the daily value specified as uniform with the low and high values based on those recommended for related respiratory symptoms in Neumann et al. (1994). The low daily estimate of $10 is the sum of the mid-range values recommended by IEc (1994) for two symptoms believed to be associated with acute bronchitis: coughing and chest tightness. The high daily estimate was taken to be twice the value of a minor respiratory restricted-activity day, or $110 (2000$).
Work Loss Days (WLDs)	Variable (U.S. median = $150)	Variable (U.S. median = $150)	No distribution available. Point estimate is based on county-specific median annual wages divided by 52 and then by 5—to get median daily wage. U.S. Year 2000 Census, compiled by Geolytics, Inc. (Geolytics, 2002)
School Loss Days	$98	$98	No distribution available. Based on (1) the probability that, if a school child stays home from school, a parent will have to stay home from work to care for the child, and (2) the value of the parent's lost productivity.
Minor Restricted Activity Days (MRADs)	$64	$68	Median WTP estimate to avoid one MRAD from Tolley et al. (1986). Distribution is assumed to be triangular with a minimum of $22 and a maximum of $83, with a most likely value of $52 (2000$). Range is based on assumption that value should exceed WTP for a single mild symptom (the highest estimate for a single symptom—for eye irritation—is $16) and be less than that for a WLD. The triangular distribution acknowledges that the actual value is likely to be closer to the point estimate than either extreme.

[a] All estimates are rounded to two significant digits. Unrounded estimates in 2000$ are available in the Appendix J of the BenMAP user manual (Abt Associates, 2012). Income growth projections are only currently available in BenMAP through 2024, so both the 2025 and 2038 estimates use income growth only through 2024 and are therefore likely underestimates. Currently, BenMAP does not have an inflation adjustment to 2011$. We ran BenMAP for a currency year of 2010$ and calculated the benefit-per-ton estimates in 2010$. We then adjusted the resulting benefit-per-ton estimates to 2011$ using the Consumer Price Index (CPI-U, all items). This approach slightly underestimates the inflation for medical index and wage index between 2010 and 2011, which affects COI estimates and wage-based estimates.

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

Until updated guidance is available, the Agency determined that a single, peer-reviewed estimate applied consistently best reflects the SAB-EEAC advice it has received. Therefore, the Agency has decided to apply the VSL that was vetted and endorsed by the SAB in the Guidelines for Preparing Economic Analyses (U.S. EPA, 2000)[60] while the Agency continues its efforts to update its guidance on this issue. This approach calculates a mean value across VSL estimates derived from 26 labor market and contingent valuation studies published between 1974 and 1991. The mean VSL across these studies is \$4.8 million (1990\$) or \$6.3 million (2000\$).[61] 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 underwent review by the SAB-EEAC. A meeting with the SAB on this paper was held on March 14, 2011 and formal recommendations were transmitted on July 29, 2011 (U.S. EPA-SAB, 2011). EPA is taking SAB's recommendations under advisement.

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

[60] 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.

[61] In this analysis, we adjust the VSL to account for a different currency year (2011\$) and to account for income growth to 2024. After applying these adjustments to the \$6.3 million value, the VSL is \$10 million. Income growth projections are only currently available in BenMAP through 2024, so both the 2025 and 2038 estimates use income growth only through 2024 and are therefore likely underestimates.

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

Although there are several differences between the labor market studies the EPA uses to derive a VSL estimate and the ozone and $PM_{2.5}$ air pollution context addressed here, those differences in the affected populations and the nature of the risks imply both upward and downward adjustments. Table 5-11 lists some of these differences and the expected effect on the VSL estimate for air pollution-related mortality. In the absence of a comprehensive and balanced set of adjustment factors, the EPA believes it is reasonable to continue to use the $4.8 million (1990$) value adjusted for inflation and income growth over time while acknowledging the significant limitations and uncertainties in the available literature.

Table 5-11. Influence of Applied VSL Attributes on the Size of the Economic Benefits of Reductions in the Risk of Premature Death (U.S. EPA, 2006a)

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

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

For $PM_{2.5}$-related premature mortality, we assume that there is a "cessation" lag between exposures and the total realization of changes in health effects. For $PM_{2.5}$, we assumed that some of the incidences of premature mortality related to $PM_{2.5}$ exposures occur in a distributed fashion over the 20 years following exposure and discounted over the period between exposure and premature mortality. Although the structure of the lag is uncertain, the EPA follows the advice of the SAB-HES to assume a segmented lag structure characterized by 30 percent of mortality reductions in the first year, 50 percent over years 2 to 5, and 20 percent over the years 6 to 20 after the reduction in $PM_{2.5}$ (U.S. EPA-SAB, 2004c). 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 percent and 7 percent.[62] 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 $PM_{2.5}$ co-benefits using a 7 percent discount rate are only approximately 10 percent less than the monetized benefits using a 3 percent discount rate. Further discussion of this topic appears in the EPA's *Guidelines for Preparing Economic Analyses* (U.S. EPA, 2010e).

For ozone, we acknowledge substantial uncertainty associated with specifying the lag for long-term respiratory mortality. As stated earlier, it is this uncertainty related to specifying a lag structure which prevents us from including this endpoint as a monetary benefit estimate within the core analysis.[63] In presenting dollar benefit estimates as part of the sensitivity analysis, we include both an assumption of zero lag and a lag structure matching that used for the core $PM_{2.5}$ estimate (the SAB 20 year segmented lag). Inclusion of the zero lag reflects consideration for the possibility that the long-term respiratory mortality estimate captures primarily, an accumulation of short-term mortality effects across the ozone season.[64] The use of the 20 year segmented lag

[62] The choice of a discount rate, and its associated conceptual basis, is a topic of ongoing discussion within the federal government. To comply with OMB 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.

[63] Recall however, that we consider the estimate of reduced incidence of respiratory mortality associated with long-term ozone exposure to have sufficient support in the literature to be included as part of the core estimate.

[64] In presenting risk estimates associated with modeling long-term ozone-related respiratory mortality in the HREA, we noted that: "The effect estimates used in modeling long-term O_3-attributable mortality, utilize a seasonal average of peak (1-hr maximum) measurements. These long-term exposure metrics can be viewed as long-term exposures to daily peak O_3 over the warmer months, as compared with annual average levels such as are used in long-term PM exposure calculations. This increases the need for care in interpreting these long-term O_3-attributable mortality estimates together with the short-term O_3-attributable mortality estimates, in order to avoid double counting." (USEPA, 2014b). This statement was included in the 2nd draft of the HREA that CASAC commented extensively on (p. 7-21) (Frey 2014). While the CASAC made recommendations on specific aspects of our approach for modeling this endpoint (specifically the need to include consideration for potential thresholds as a sensitivity analysis), they did not criticize this observation regarding the potential that this metric could actually represent an

reflects consideration for advice provided by the HES (USEPA-SAB, 2010a), where they state that, "[i]f Alternative Estimates are derived using cohort mortality evidence, there is no evidence in the literature to support a different cessation lag between ozone and particulate matter. The HES therefore recommends using the same cessation lag structure and assumptions as for particulate matter when utilizing cohort mortality evidence for ozone." Dollar benefit estimates generated using both lag assumptions are presented as sensitivity analyses (see section 5.7.3.1).

Uncertainties Specific to Premature Mortality Valuation. The economic benefits associated with reductions in the risk of premature mortality are the largest category of monetized benefits in this RIA. In addition, in prior analyses, the EPA identified valuation of mortality-related benefits as the largest contributor to the range of uncertainty in monetized benefits (Mansfield et al., 2009).[65] 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.

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accumulation of short-term daily peak exposures and that care needed to be taken to avoid double-counting of mortality incidence. Our inclusion of a zero lag reflects the potential that the estimate of long-term exposure-related respiratory mortality could (to a significant extent) capture an accumulation of short-term effects. Note, that we include the zero threshold model together with a 20 year segmented lag model in order to capture a potential range of lag effect and both are given equal coverage in generating the dollar benefit estimates included as a sensitivity analysis for this endpoint.

[65] 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.

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.

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

- **Voluntariness of risks evaluated:** Although job-related mortality risks may differ in several ways from air pollution-related mortality risks, the most important difference may be that job-related risks are incurred voluntarily, or generally assumed to be, whereas air pollution-related risks are incurred involuntarily. Some evidence suggests that people will pay more to reduce involuntarily incurred risks than risks incurred voluntarily (e.g., Lichtenstein and Slovic, 2006). If this is the case, WTP estimates based on wage-risk studies may understate WTP to reduce involuntarily incurred air pollution-related mortality risks.

- **Sudden versus protracted death:** A final important difference related to the nature of the risk may be that some workplace mortality risks tend to involve sudden, catastrophic events, whereas air pollution-related risks tend to involve longer periods of disease and suffering prior to death. Some evidence suggests that WTP to avoid a risk of a protracted death involving prolonged suffering and loss of dignity and personal control is greater than the WTP to avoid a risk (of identical magnitude) of sudden death (e.g., Tsuge et al., 2005; Alberini and Scasny, 2011). To the extent that the mortality risks addressed in this assessment are associated with longer periods of illness or greater pain and suffering than are the risks addressed in the valuation literature, the WTP measurements employed in the present analysis would reflect a downward bias.

- **Self-selection and skill in avoiding risk**: Recent research (Shogren and Stamland, 2002) suggests that VSL estimates based on hedonic wage studies may overstate the average value of a risk reduction. This is based on the fact that the risk-wage trade-off revealed in hedonic studies reflects the preferences of the marginal worker (i.e., that worker who demands the highest compensation for his risk reduction for a given job). This worker must have either a higher workplace risk than the average worker in a given occupation, a lower risk tolerance than the average worker in that occupation, or both. Conversely, the marginal worker should have a higher risk tolerance than workers employed in less-risky sectors. However, the risk estimate used in hedonic studies is generally based on average risk, so the VSL may be biased, in an ambiguous direction, because the wage differential and risk measures do not match.

- **Baseline risk and age:** Recent research (Smith, Pattanayak, and Van Houtven, 2006) finds that because individuals reevaluate their baseline risk of death as they age, the marginal value of risk reductions does not decline with age as predicted by some lifetime consumption models. This research supports findings in recent stated preference studies that suggest only small reductions in the value of mortality risk reductions with increasing age (e.g., Alberini et al., 2004).

5.6.4.2 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 (ICD) (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 the correct currency year using the CPI-U "all items" (Abt Associates, 2012). The resulting national average lost daily wage is $150 (2011$). The total cost-of-illness estimate for an ICD code-specific hospital stay lasting *n* days, then, was the mean hospital charge plus daily lost wage multiplied by *n*. In general, the mean length of stay has decreased since the 2000 database used in the previous version of BenMAP, while the mean hospital charge has increased. We provide the rounded unit values in 2011$ for the COI functions used in this analysis in Table 5-12.

Table 5-12. Unit Values for Hospital Admissions [a]

End Point	ICD Codes	Age Range		Mean Hospital Charge (2011$)	Mean Length of Stay (days)	Total Cost of Illness (unit value in 2011$)
		min.	max.			
HA, Chronic Lung Disease	490–496	18	64	$20,000	3.9	$22,000
HA, Asthma	493	0	64	$15,000	3.0	$16,000
HA, All Cardiovascular	390–429	18	64	$41,000	4.1	$44,000
HA, All Cardiovascular	390–429	65	99	$38,000	4.9	$42,000
HA, All Respiratory	460–519	65	99	$32,000	6.1	$37,000

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

To value asthma emergency department visits, we used a simple average of two estimates from the health economics literature. The first estimate comes from Smith et al. (1997), who reported approximately 1.2 million asthma-related emergency department visits in 1987, at a total cost of $186 million (1987$). The average cost per visit that year was $155; in 2011$, that cost was $480 (using the CPI-U for medical care to adjust to 2011$). The second estimate comes from Stanford et al. (1999), who reported the cost of an average asthma-related emergency department visit based on 1996–1997 data at $400 (using the CPI-U for medical care to adjust to 2011$). A simple average of the two estimates yields a unit value of $440 (2011$).

5.6.4.3 Nonfatal Myocardial Infarctions Valuation

We were not able to identify a suitable WTP value for reductions in the risk of nonfatal heart attacks.[66] 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 5-13):

- 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

[66] We note that this endpoint was only modeled as part of the cobenefits analysis for $PM_{2.5}$ and is not included in the ozone-related benefits analysis. As such, we could have moved this discussion to Appendix 5D (as was done with the discussion of $PM_{2.5}$-related effect estimates). However, since this was the only broader discussion related to valuation which is exclusively related to $PM_{2.5}$ we left it in this section.

of stay for ICD code 410 (myocardial infarction) in the year-2000 Agency for Healthcare Research and Quality (AHRQ) HCUP database is 5.5 days (AHRQ, 2000). However, this may include patients who died in the hospital (not included among our nonfatal myocardial infarction cases), and whose length of stay was therefore substantially shorter than it would be if they had not died.

- Eisenstein et al. (2001) estimated 10-year costs of $45,000 in rounded 1997$ (using a 3% discount rate) for myocardial infarction patients, using statistical prediction (regression) models to estimate inpatient costs. Only inpatient costs (physician fees and hospital costs) were included.

Table 5-13. Alternative Direct Medical Cost of Illness Estimates for Nonfatal Heart Attacks [a]

Study	Direct Medical Costs (2011$)	Over an x-Year Period, for x =
Wittels et al. (1990)	$170,000 [b]	5
Russell et al. (1998)	$34,000 [c]	5
Average (5-year) costs	$100,000	5
Eisenstein et al. (2001)	$76,000 [c]	10

[a] All estimates rounded to two significant digits. Unrounded estimates in 2000$ are available in appendix J of the BenMAP user manual (Abt Associates, 2012).
[b] Wittels et al. (1990) did not appear to discount costs incurred in future years.
[c] Using a 3% discount rate. Discounted values as reported in the study.

As noted above, the estimates from these three studies are substantially different, and we have not adequately resolved the sources of differences in the estimates. Because the wage-related opportunity cost estimates from Cropper and Krupnick (1990) cover a 5-year period, we used estimates for medical costs that similarly cover a 5-year period (i.e., estimates from Wittels et al. (1990) and Russell et al. (1998). We used a simple average of the two 5-year estimates, or rounded to $85,000, and added it to the 5-year opportunity cost estimate. The resulting estimates are given in Table 5-14.

Table 5-14. Estimated Costs Over a 5-Year Period of a Nonfatal Myocardial Infarction (in 2011$) [a]

Age Group	Opportunity Cost	Medical Cost [b]	Total Cost
0–24	$0	$100,000	$100,000
25–44	$12,000 [c]	$100,000	$110,000
45–54	$18,000 [c]	$100,000	$120,000
55–65	$100,000 [c]	$100,000	$210,000
> 65	$0	$100,000	$100,000

[a] All estimates rounded to two significant digits, so estimates may not sum across columns. Unrounded estimates in 2000$ are available in appendix J of the BenMAP user manual (Abt Associates, 2012).
[b] An average of the 5-year costs estimated by Wittels et al. (1990) and Russell et al. (1998).
[c] From Cropper and Krupnick (1990), using a 3% discount rate for illustration.

5.6.4.4 Valuation of Acute Health Events

Asthma exacerbation. Several respiratory symptoms in asthmatics or characterizations of an asthma episode have been associated with exposure to air pollutants. All of these can generally be taken as indications of an asthma exacerbation when they occur in an asthmatic. Therefore, we apply the same set of unit values for all of the variations of “asthma exacerbation”. Specifically, we use a unit value based on the mean WTP estimates for 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.

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 $71 (2011$). 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.

School Loss Days Valuation. To value a school absence, we: (1) estimated the probability that if a school child stays home from school, a parent will have to stay home from work to care for the child; and (2) valued the lost productivity at the parent’s wage. To do this, we estimated the number of families with school-age children in which both parents work, and we valued a school-loss day as the probability that such a day also would result in a work-loss day. We calculated this value by multiplying the proportion of households with school-age children by a measure of lost wages.

We used this method in the absence of a preferable WTP method. However, this approach suffers from several uncertainties. First, it omits willingness to pay to avoid the symptoms/illness that resulted in the school absence; second, it effectively gives zero value to school absences that do not result in work-loss days; and third, it uses conservative assumptions about the wages of the parent staying home with the child. Finally, this method assumes that parents are unable to work from home. If this is not a valid assumption, then there would be no lost wages.

For this valuation approach, we assumed that in a household with two working parents, the female parent will stay home with a sick child. From the Statistical Abstract of the United States (U.S. Census Bureau, 2001), we obtained: (1) the numbers of single, married and "other" (widowed, divorced or separated) working women with children; and (2) the rates of participation in the workforce of single, married and "other" women with children. From these two sets of statistics, we calculated a weighted average participation rate of 72.85 percent. Our estimate of daily lost wage (wages lost if a mother must stay at home with a sick child) is based on the year 2000 median weekly wage among women ages 25 and older (U.S. Census Bureau, 2001). This median weekly wage is $551 (2000$). Dividing by five gives an estimated median daily wage of $103 (2000$). To estimate the expected lost wages on a day when a mother has to stay home with a school-age child, we first estimated the probability that the mother is in the workforce then multiplied that estimate by the daily wage she would lose by missing a workday: 72.85 percent times $103, for a total loss of $75 (2000$). This valuation approach is similar to that used by Hall et al. (2003).

Work Loss Days Valuation. Work loss days are valued at a day's wage. BenMAP-CE calculates county-specific median daily wages from county-specific annual wages (by dividing the annual wage by 52 weeks multiplied by 5 work days per week), on the theory that a worker's vacation days are valued at the same daily rate as work days.

Upper and Lower respiratory symptoms. Lower and upper respiratory symptoms are each considered a complex of symptoms. A dollar value was derived for clusters of these symptoms that most closely match the studies used to calculate incidence (Schwartz and Neas, 2000; Pope et al, 1991) based on mid-range estimates from each cluster (IEc, 1994).

5.6.4.5 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[67] of WTP for health risk reductions is positive, although there is uncertainty about its exact value. Thus, as real income increases, the WTP for environmental improvements also increases. Although many analyses assume that the income elasticity of WTP is unit elastic (i.e., a 10% higher real income level implies a 10% higher WTP to reduce risk changes), empirical evidence suggests that income elasticity is substantially less than one and thus relatively inelastic. As real income rises, the WTP value also rises but at a slower rate than real income.

The effects of real income changes on WTP estimates can influence benefits estimates in two different ways: through real (national average) income growth between the year a WTP study was conducted and the year for which benefits are estimated, and through differences in income between study populations and the affected populations at a particular time. The SAB-EEAC advised the EPA to adjust WTP for increases in real income over time but not to adjust WTP to account for cross-sectional income differences "because of the sensitivity of making such distinctions, and because of insufficient evidence available at present" (U.S. EPA-SAB, 2000). An advisory by another committee associated with the SAB, the Advisory Council on Clean Air Compliance Analysis (SAB-Council), has provided conflicting advice. While agreeing with "the general principle that the willingness to pay to reduce mortality risks is likely to increase with growth in real income" and that "[t]he same increase should be assumed for the 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 can be reconciled, the EPA will continue to adjust valuation estimates to reflect income growth using the methods described below, while providing sensitivity analyses for alternative income growth adjustment factors.

Based on a review of the available income elasticity literature, we adjusted the valuation of human health benefits upward to account for projected growth in real U.S. income. Faced with a dearth of estimates of income elasticities derived from time-series studies, we applied estimates derived from cross-sectional studies in our analysis. Details of the procedure can be found in

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

Kleckner and Neumann (1999). We note that the literature has evolved since the publication of this memo and that an array of newer studies identifying potentially suitable income elasticity estimates are available (IEc, 2012). The EPA anticipates seeking an SAB review of these studies, and its approach to adjusting WTP estimates to account for changes in personal income, in the near future. As such, these newer studies have not yet been incorporated into the benefits analysis. An abbreviated description of the procedure we used to account for WTP for real income growth between 1990 and 2024 is presented below. Income growth projections are only currently available in BenMAP through 2024, so both the 2025 and 2038 estimates use income growth only through 2024 and are therefore likely underestimates.

Reported income elasticities suggest that the severity of a health effect is a primary determinant of the strength of the relationship between changes in real income and WTP. As such, we use different elasticity estimates to adjust the WTP for minor health effects, severe and chronic health effects, and premature mortality. Note that because of the variety of empirical sources used in deriving the income elasticities, there may appear to be inconsistencies in the magnitudes of the income elasticities relative to the severity of the effects (a priori one might expect that more severe outcomes would show less income elasticity of WTP). We have not imposed any additional restrictions on the empirical estimates of income elasticity. One explanation for the seeming inconsistency is the difference in timing of conditions. WTP for minor illnesses is often expressed as a short-term payment to avoid a single episode. WTP for major illnesses and mortality risk reductions are based on longer-term measures of payment (such as wages or annual income). Economic theory suggests that relationships become more elastic as the length of time grows, reflecting the ability to adjust spending over a longer time period (U.S. EPA, 2010e, p. A-9). Based on this theory, it would be expected that WTP for reducing long-term risks would be more elastic than WTP for reducing short-term risks. The relative magnitude of the income elasticity of WTP for visibility compared with those for health effects suggests that visibility is not as much of a necessity as health, thus, WTP is more elastic with respect to income. The elasticity values used to adjust estimates of benefits in 2024 are presented in Table 5-15.[68]

[68] We expect that the WTP for improved visibility in Class 1 areas would also increase with growth in real income.

Table 5-15. 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 2024 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 2024, 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.[69] We used projections of real GDP (in chained 1996 dollars) provided by Standard and Poor's (2000) for the years 2010 to 2024.[70]

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 5-16. Benefits for each of the categories (minor health effects, severe and chronic health effects, premature mortality, and visibility) are adjusted by multiplying the unadjusted benefits by the appropriate adjustment factor. For premature mortality, we applied the income adjustment factor specific to the analysis year, but we do not adjust for income growth over the 20-year cessation lag. Our approach could underestimate the benefits for the later years of the lag.

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[69] 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.

[70] 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.

There is some uncertainty regarding the total costs of illness in the future. Specifically, the nature of medical treatment is changing, including a shift towards more outpatient treatment. Although we adjust the COI estimates for inflation, we do not have data to project COI estimates for the cost of treatment in the future or income growth over time, which leads to an inherent though unavoidable inconsistency between COI- and WTP-based estimates. This approach may under predict benefits in future years because it is likely that increases in real U.S. income would also result in increased COI (due, for example, to increases in wages paid to medical workers) and increased cost of work loss days and lost worker productivity (reflecting that if worker incomes are higher, the losses resulting from reduced worker production would also be higher). In addition, cost-of-illness estimates do not include sequelae costs or pain and suffering, the value of which would likely increase in the future. To the extent that costs would be expected to increase over time, this increase may be partially offset by advancement in medical technology that improves the effectiveness of treatment at lower costs. For these reasons, we believe that the cost-of-illness estimates in this RIA may underestimate (on net) the total economic value of avoided health impacts.

Table 5-16. Adjustment Factors Used to Account for Projected Real Income Growth [a]

Benefit Category	2024
Minor Health Effect	1.07
Severe and Chronic Health Effects	1.22
Premature Mortality	1.20

[a] Based on elasticity values reported in Table 5-15, U.S. Census population projections, and projections of real GDP per capita.

5.7 Benefits Results

As stated in section 5.1 and described in detail in section 5.4.3, we have estimated nationwide benefits for 2025 associated with attainment of alternative ozone standards across the U.S. with the exception of California. We have also estimated the nationwide benefits of attaining in California for 2038. Because of the temporal disconnect between these two scenarios, benefit estimates for each are not totaled and instead, are presented separately (section 5.7.1 for 2025 and section 5.7.2 for post-2025).

In addition to these core incidence and benefits estimates, we also present a number of additional analyses which are intended to inform interpretation of these core benefit estimates (see section 5.7.3). In completing these additional analyses, in many cases, we did not have to generate separate assessments for both scenarios since observations from one scenario could be readily applied to the other (in these cases, we tended to model the 2025 scenario and then discuss application of those observations to the post-2025 scenario).

5.7.1 Benefits of the Proposed and Alternative Annual Primary Ozone Standards for the 2025 Scenario

This section presents incidence reductions and associated dollar benefit estimates associated with the 2025 scenario (i.e., every state apart from California – see section 5.4.3). Applying the impact and valuation functions described previously in this chapter to the estimated changes in ozone yields estimates of the changes in physical damages (e.g., premature mortalities, cases of hospital admissions) and the associated monetary values for those changes. Similarly applying the incidence per ton and dollar per ton values to the estimates of NOx reductions produces estimates of changes in PM-related health effect incidence and associated dollar benefits. Not all known ozone and 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 ozone and PM-related health benefits plus B, the sum of the non-monetized health benefits and welfare co-benefits; this B represents both uncertainty and a bias in this analysis, as it reflects those benefits categories that we are unable to monetize in this analysis.

We follow our standard rounding conventions in presenting these benefits results. After reviewing the presentation of EPA's benefits results, NRC (2002) concluded, "EPA should strive to present the results of the analyses in ways that avoid conveying an unwarranted degree of certainty. Such ways include rounding to few significant digits, increasing the use of graphs, and placing less emphasis on single numbers and greater emphasis on ranges" (p. 161). Following this advice, we round all benefits estimates to two significant digits, and all rounding occurs after final summing of unrounded estimates. As such, totals may not sum across columns or rows. In addition, all incidence estimates are rounded to whole numbers with a maximum of two significant digits.

Table 5-17 shows the population-weighted air quality change for the alternative standards averaged across the continental U.S. Table 5-18 summarizes the tons of VOC and NOx emissions required to simulate attainment of each alternative standard (further differentiated by geographic region including east, west and California). Tables 5-19 through 5-23 present the benefits results for the proposed and alternative ozone standards. Table 5-24 summarizes total benefits by geographic region (including east, west minus California, and California). Note that in presenting estimates related to reductions in ozone (Tables 5-19 and 5-20), we include the full set of core estimates together with a subset of sensitivity analysis results (specifically, alternative estimates for both short-term and long-term mortality). The benefit estimates presented are relative to a 2025 analytical baseline reflecting attainment nationwide (excluding California) of the current primary ozone standards (i.e., 75 ppb) that includes promulgated national regulations and illustrative emissions controls to simulate attainment with 75 ppb.

Table 5-17. Population-Weighted Air Quality Change for the Proposed and Alternative Annual Primary Ozone Standards Relative to Analytical Baseline for 2025[a]

Standard	Population-Weighted Summer Season Ozone Concentration Change (8hr max)[b]
70 ppb	0.5285
65 ppb	1.6317
60 ppb	3.0222

[a] Because we used benefit-per-ton estimates for the $PM_{2.5}$ co-benefits, population-weighted $PM_{2.5}$ changes are not available.
[b] Population weighting based on all ages (demographic used in modeling short-term exposure-related mortality for ozone) for 2025.

Table 5-18. Emission Reductions in Illustrative Emission Reduction Strategies for the Proposed and Alternative Annual Primary Ozone Standards, by Pollutant and Region Relative to Analytical Baseline – Full Attainment (tons)[a]

	70 ppb	65 ppb	60 ppb
NOx			
East	600	1,700	2,800
West	0	110	500
CA	53	110	140
VOC			
East	55	99	150
West	0	7	7
CA	0	0	0

[a] See Chapter 4 for more information on the illustrative emission reduction strategies. The emissions in this table reflect both known and unknown controls. Several recent rules such as Tier 3 will have substantially reduced ozone concentrations by 2025 in the East, thus few additional controls would be needed to reach 70 ppb.

Table 5-19. Estimated Number of Avoided Ozone-Only Health Impacts for the Proposed and Alternative Annual Ozone Standards (Incremental to the Analytical Baseline) for the 2025 Scenario (nationwide benefits of attaining each alternative standard everywhere in the U.S. except California) [a, b]

	Health Effectb	Proposed and Alterative Standards (95th percentile confidence intervals) 70 ppb	65 ppb	60 ppb
Avoided Short-Term Mortality - Core Analysis				
multi-city studies	Smith et al. (2009) (all ages)	200 (97 to 300)	630 (310 to 940)	1,100 (560 to 1,700)
	Zanobetti and Schwartz (2008) (all ages)	340 (180 to 490)	1,000 (560 to 1,500)	1,900 (1,000 to 2,800)
Avoided Long-term Respiratory Mortality - Core Analysis				
multi-city study	Jerrett et al. (2009) (30-99yrs) copollutants model ($PM_{2.5}$)	680 (230 to 1,100)	2,100 (710 to 3,500)	3,900 (1,300 to 6,400)
Avoided Short-Term Mortality - Sensitivity Analysis				
multi-city studies	Smith et al. (2009) (all ages) copollutants model (PM_{10})	160 (-44 to 360)	500 (-140 to 1,100)	920 (-250 to 2,100)
	Schwartz (2005) (all ages)	250 (77 to 420)	780 (240 to 1,300)	1,400 (440 to 2,400)
	Huang et al. (2005) (cardiopulmonary)	240 (88 to 380)	740 (280 to 1,200)	1,400 (510 to 2,200)
	Bell et al. (2004) (all ages)	160 (54 to 270)	510 (170 to 850)	930 (310 to 1,600)
meta-analyses	Bell et al. (2005) (all ages)	520 (250 to 800)	1,600 (780 to 2,500)	3,000 (1,400 to 4,600)
	Ito et al. (2005) (all ages)	730 (440 to 1,000)	2,300 (1,400 to 3,200)	4,200 (2,500 to 5,800)
	Levy et al. (2005) (all ages)	740 (510 to 970)	2,300 (1,600 to 3,000)	4,200 (2,900 to 5,600)
Avoided Long-term Respiratory Mortality - Sensitivity Analysis				
multi-city study	Jerrett et al. (2009) (age 30-99) (86 cities) (ozone-only)	460 (130 to 790)	1,400 (410 to 2,500)	2,600 (760 to 4,500)
	Jerrett et al. (2009) (age 30-99) (96 cites) (ozone-only)	500 (170 to 810)	1,500 (550 to 2,500)	2,800 (1,000 to 4,600)
	Jerrett et al (2009) - copollutant ($PM_{2.5}$) model with: [c]			
	60 ppb threshold	520	1,600	2,900
	56 ppb threshold	410	1,100	1,700
	55 ppb threshold	120	280	510
	50 ppb threhsold	6	58	140
	45 ppb threshold	3	47	105
	40 ppb threshold	<1	10	13
Avoided Morbidity - Core Analysis				
	Hospital admissions - respiratory (age 65+)	360 (-97 to 820)	1,100 (-310 to 2,600)	2,100 (-560 to 4,700)
	Emergency department visits for asthma (all ages)	1,100 (100 to 3,400)	3,500 (330 to 11,000)	6,600 (610 to 20,000)
	Asthma exacerbation (age 6-18)	300,000 (-440,000 to 900,000)	910,000 (-1,300,000 to 2,700,000)	1,700,000 (-2,500,000 to 5,000,000)
	Minor restricted-activity days (age 18-65)	930,000 (380,000 to 1,500,000)	2,900,000 (1,200,000 to 4,500,000)	5,300,000 (2,200,000 to 8,300,000)
	School Loss Days (age 5-17)	330,000 (120,000 to 730,000)	1,000,000 (360,000 to 2,200,000)	1,900,000 (660,000 to 4,500,000)

[a] All incidence estimates are rounded to whole numbers with a maximum of two significant digits.
[b] All incidence estimates are based on ozone-only models unless otherwise noted.
[c] See Appendix 5B, section 5B.1 for additional detail on the threshold-based sensitivity analysis.

Table 5-20. Total Monetized Ozone-Only Benefits for the Proposed and Alternative Annual Ozone Standards (Incremental to the Analytical Baseline) for the 2025 Scenario (nationwide benefits of attaining each alternative standard everywhere in the U.S. except California) (millions of 2011) [a, b]

	Health Effectb	Proposed and Alterative Standards (95th percentile confidence intervals) 70 ppb	65 ppb	60 ppb
Avoided Short-Term Mortality - Core Analysis				
multi-city studies	Smith et al. (2009) (all ages)	$2,000 ($180 to $5,800)	$6,400 ($560 to $18,000)	$12,000 ($1,000 to $33,000)
	Zanobetti and Schwartz (2008) (all ages)	3,400 ($300 to $9,600)	11,000 ($950 to $30,000)	20,000 ($1,700 to $55,000)
Avoided Short-Term Mortality - Sensitivity Analysis				
multi-city studies	Smith et al. (2009) (all ages) copollutants model (PM_{10})	$1,600 (-$390 to $5,900)	$5,100 (-$1,200 to $19,000)	$9,400 (-$2,200 to $34,000)
	Schwartz (2005) (all ages)	$2,500 ($200 to $7,700)	$7,900 ($630 to $24,000)	$15,000 ($1,200 to $44,000)
	Huang et al. (2005) (cardiopulmonary)	$2,400 ($200 to $7,000)	$7,500 ($620 to $22,000)	$14,000 ($1,100 to $41,000)
meta-analyses	Bell et al. (2004) (all ages)	1,700 ($130 to $4,900)	5,200 ($420 to $15,000)	9,500 ($760 to $28,000)
	Bell et al. (2005) (all ages)	5,300 ($470 to $15,000)	17,000 ($1,500 to $48,000)	31,000 ($2,700 to $88,000)
	Ito et al. (2005) (all ages)	$7,400 ($680 to $21,000)	$23,000 ($2,100 to $64,000)	$42,000 ($3,900 to $120,000)
	Levy et al. (2005) (all ages)	$7,500 ($700 to $20,000)	$24,000 ($2,200 to $64,000)	$43,000 ($4,000 to $120,000)
Avoided Long-term Respiratory Mortality - Sensitivity Analysis				
multi-city study	Jerrett et al. (2009) (age 30-99) copollutants model ($PM_{2.5}$) no lag [c]	$6,900 ($560 to $21,000)	$22,000 ($1,700 to $64,000)	$40,000 ($3,200 to $120,000)
	Jerrett et al. (2009) (age 30-99) copollutants model ($PM_{2.5}$) 20 yr segmented lag [d]	$5,600 to $6300 ($460 to $19,000)	$18,000 to $20,000 ($1,400 to $58,000)	$32,000 to $36,000 ($2,600 to $110,000)

[a] All benefits estimates are rounded to whole numbers with a maximum of two significant digits. The monetized value of the ozone-related morbidity benefits are included in the estimates shown in this table for each mortality study (and when combined account for from 4-6% of the total benefits, depending on the total mortality estimate compared against. Note that asthma exacerbations accounts for <<1% of the total).

[b] The sensitivity analysis for long-term exposure-related mortality included an assessment of potential thresholds however this assessment was implemented using incidence estimates (see Table 5-19). Observations from that analysis (in terms of fractional impacts on incidence can be directly applied to these benefit results) (see Appendix 5B, section 5B.1)

[c] A single central-tendency value is provided in each cell, since the zero-lag model used here did not require application of a 3% and 7% discount rates (see footnote d below) (note, however that as with all other entries in this study, we do include a 95th percentile confidence interval range – these are the values within parentheses).

[d] The range (outside of the parentheses) within each cell results from application of a 7% and 3% discount rates in the context of applying the 20year segmented lag (with the 7% resulting in the lower estimate and the 3% the higher estimate). The range presented within the parentheses reflects consideration for the 95th% confidence interval generated for each of these estimates and ranges from a low value (2.5th% CI for the 7% discount-based dollar benefit) to an upper value (97.5th% of the 3% discount-based dollar benefit).

Table 5-21. Estimated Number of Avoided $PM_{2.5}$-Related Health Impacts for the Proposed and Alternative Annual Ozone Standards (Incremental to the Analytical Baseline) for the 2025 Scenario (nationwide benefits of attaining each alternative standard everywhere in the U.S. except California) [a]

	Proposed and Alterative Standards		
Health Effectb	**70ppb**	**65ppb**	**60ppb**
Avoided PM2.5-related Mortality			
Krewski et al. (2009) (adult mortality age 30+)	510	1,400	2,600
Lepeule et al. (2012) (adult mortality age 25+)	1,100	3,300	6,000
Woodruff et al. (1997) (infant mortality)	1	2	5
Avoided PM2.5-related Morbidity			
Non-fatal heart attacks			
Peters et al. (2001) (age >18)	600	1,700	3,100
Pooled estimate of 4 studies (age >18)	64	180	330
Hospital admissions—respiratory (all ages)	150	430	780
Hospital admissions—cardiovascular (age > 18)	180	530	950
Emergency department visits for asthma (all ages)	280	790	1,400
Acute bronchitis (ages 8–12)	790	2,300	4,100
Lower respiratory symptoms (ages 7–14)	10,000	29,000	53,000
Upper respiratory symptoms (asthmatics ages 9–11)	14,000	41,000	75,000
Asthma exacerbation (asthmatics ages 6–18)	17,000	51,000	100,000
Lost work days (ages 18–65)	65,000	180,000	340,000
Minor restricted-activity days (ages 18–65)	380,000	1,100,000	2,000,000

[a] All incidence estimates are rounded to whole numbers with a maximum of two significant digits. Because these estimates were generated using benefit-per-ton estimates, confidence intervals are not available. In general, the 95[th] percentile confidence interval for the health impact function alone ranges from approximately ±30 percent for mortality incidence based on Krewski et al. (2009) and ±46 percent based on Lepeule et al. (2012).

Table 5-22. Monetized $PM_{2.5}$-Related Health Co-Benefits for the Proposed and Alternative Annual Ozone Standards (Incremental to Analytical Baseline) for the 2025

Scenario (nationwide benefits of attaining each alternative standard everywhere in the U.S. except California) (Millions of 2011)[a,b,c]

Monetized Benefits	Proposed and Alterative Standards		
	70 ppb	65 ppb	60 ppb
3% Discount Rate			
Krewski et al. (2009) (adult mortality age 30+)	$4,800	$14,000	$25,000
Lepeule et al. (2012) (adult mortality age 25+)	$11,000	$31,000	$56,000
7% Discount Rate			
Krewski et al. (2009) (adult mortality age 30+)	$4,300	$12,000	$22,000
Lepeule et al. (2012) (adult mortality age 25+)	$9,700	$28,000	$50,000

[a] All estimates are rounded to two significant digits. Because these estimates were generated using benefit-per-ton estimates, confidence intervals are not available. In general, the 95th percentile confidence interval for monetized $PM_{2.5}$ benefits ranges from approximately -90 percent to +180 percent of the central estimates based on Krewski et al. (2009) and Lepeule et al. (2012). Estimates do not include unquantified health benefits noted in Table 5-2 or Section 5.6.3.6 or welfare co-benefits noted in Chapter 6.

[b] The reduction in premature fatalities each year accounts for over 98% of total monetized benefits in this analysis. Mortality risk valuation assumes discounting over the SAB-recommended 20-year segmented lag structure.

Table 5-23. Estimate of Monetized Ozone and $PM_{2.5}$ Benefits for Proposed and Alternative Annual Ozone Standards Incremental to the Analytical Baseline for the 2025 Scenario (nationwide benefits of attaining each alternative standard everywhere in the U.S. except California) – Full Attainment (billions of 2011$) [a]

	Discount Rate	70 ppb	65 ppb	60 ppb
Ozone-only Benefits (range reflects Smith et al., 2009 and Zanobetti and Schwartz, 2008)	[b]	$2.0 to $3.4 +B	$6.4 to $11 +B	$12 to $20 +B
$PM_{2.5}$ Co-benefits (range reflects Krewski et al., 2009 and Lepeule et al., 2012)	3%	$4.8 to $11	$14 to $31	$25 to $56
	7%	$4.3 to $9.7	$12 to $28	$22 to $50
Total Benefits	3%	$6.9 to $14 +B	$20 to $41 +B	$37 to $75 +B
	7%	$6.4 to $13 +B	$19 to $38 +B	$34 to $70 +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 for $PM_{2.5}$ assumes discounting over the SAB-recommended 20-year segmented lag structure. Not all possible benefits are quantified and monetized in this analysis. B is the sum of all unquantified health and welfare co-benefits. Data limitations prevented us from quantifying these endpoints, and as such, these benefits are inherently more uncertain than those benefits that we were able to quantify. These estimates reflect the economic value of avoided morbidities and premature deaths using risk coefficients from the studies noted.

[b] Ozone-only benefits reflect short-term exposure impacts and as such are assumed to occur in the same year as ambient ozone reductions. Consequently, social discounting is not applied to the benefits for this category.

Table 5-24. Regional Breakdown of Monetized Ozone-Specific Benefits Results for the 2025 Scenario (nationwide benefits of attaining each alternative standard everywhere in the U.S. except California) – Full Attainment [a]

Region	Proposed and Alterative Standards		
	70 ppb	**65 ppb**	**60 ppb**
East [b]	99%	96%	92%
California	0%	0%	0%
Rest of West	1%	4%	7%

[a] Because we use benefit-per-ton estimates to calculate the $PM_{2.5}$ co-benefits, a regional breakdown for the co-benefits is not available. Therefore, this table only reflects the ozone benefits.
[b] Includes Texas and those states to the north and east. Several recent rules such as Tier 3 will have substantially reduced ozone concentrations by 2025 in the East, thus few additional controls would be needed to reach 70 ppb.

5.7.2 Benefits of the Proposed and Alternative Annual Primary Ozone Standards for the post-2025 Scenario

This section presents incidence reductions and associated dollar benefit estimates associated with the post-2025 scenario (i.e., nationwide benefits estimates reflecting attainment of alternative standards in California – see section 5.4.3). The same rounding conventions described in section 5.7.1 (for the 2025 estimates) were applied in generating these estimates. As with estimates generated for the 2025 scenario, total monetized health benefits are equal to the subset of monetized ozone and PM-related health benefits plus B, the sum of the non-monetized health benefits and welfare co-benefits. Organization and general content of tables presented here for the post-2025 scenario mirrors that of tables presented in section 5.7.1 for the 2025 scenario and the reader is referred there for further clarification.

Table 5-25. Population-Weighted Air Quality Change for the Proposed and Alternative Annual Primary Ozone Standards Relative to Analytical Baseline for post-2025 [a]

Standard	Population-Weighted Ozone Season Ozone Concentration Change (8hr max)[b]
70 ppb	0.1526
65 ppb	0.3311
60 ppb	0.5007

[a] Because we used benefit-per-ton estimates for the $PM_{2.5}$ co-benefits, population-weighted $PM_{2.5}$ changes are not available.
[b] Population weighting based on all ages (demographic used in modeling short-term exposure-related mortality for ozone) for 2025.

Table 5-26. Estimated Number of Avoided Ozone-Only Health Impacts for the Proposed and Alternative Annual Ozone Standards (Incremental to the Analytical Baseline) for the Post-2025 Scenario (nationwide benefits of attaining each alternative standard just in California) [a, b]

	Health Effectb	Proposed and Alterative Standards (95th percentile confidence intervals) 70 ppb	65 ppb	60 ppb
Avoided Short-Term Mortality - Core Analysis				
multi-city studies	Smith et al. (2009) (all ages)	65 (31 to 97)	140 (68 to 210)	210 (100 to 320)
	Zanobetti and Schwartz (2008) (all ages)	110 (57 to 160)	230 (120 to 340)	350 (190 to 510)
Avoided Long-term Respiratory Mortality - Core Analysis				
multi-city study	Jerrett et al. (2009) (30-99yrs) copollutants model ($PM_{2.5}$)	260 (88 to 430)	560 (190 to 930)	840 (290 to 1,400)
Avoided Short-Term Mortality - Sensitivity Analysis				
multi-city studies	Smith et al. (2009) (all ages) copollutants model (PM_{10})	52 (-14 to 120)	110 (-31 to 250)	170 (-46 to 380)
	Schwartz (2005) (all ages)	80 (25 to 140)	170 (54 to 290)	260 (81 to 440)
	Huang et al. (2005) (cardiopulmonary)	88 (33 to 140)	190 (71 to 310)	290 (110 to 470)
meta-analyses	Bell et al. (2004) (all ages)	52 (17 to 87)	110 (38 to 190)	170 (57 to 280)
	Bell et al. (2005) (all ages)	170 (80 to 250)	360 (170 to 550)	550 (260 to 830)
	Ito et al. (2005) (all ages)	230 (140 to 330)	510 (300 to 710)	770 (460 to 1,100)
	Levy et al. (2005) (all ages)	240 (160 to 310)	510 (350 to 670)	770 (530 to 1,000)
Avoided Long-term Respiratory Mortality - Sensitivity Analysis [c]				
multi-city study	Jerrett et al. (2009) (age 30-99) (86 cities) (ozone-only)	180 (51 to 300)	380 (110 to 660)	570 (160 to 980)
	Jerrett et al. (2009) (age 30-99) (96 cites) (ozone-only)	190 (68 to 310)	410 (150 to 680)	620 (220 to 1,000)
Avoided Morbidity - Core Analysis				
	Hospital admissions - respiratory (age 65+)	120 (-32 to 270)	260 (-69 to 580)	390 (-100 to 880)
	Emergency department visits for asthma (all ages)	320 (29 to 980)	690 (64 to 2,100)	1,000 (97 to 3,200)
	Asthma exacerbation (age 6-18)	97,000 (-140,000 to 290,000)	210,000 (-310,000 to 620,000)	310,000 (-460,000 to 930,000)
	Minor restricted-activity days (age 18-65)	290,000 (120,000 to 460,000)	630,000 (260,000 to 990,000)	950,000 (390,000 to 1,500,000)
	School Loss Days (age 5-17)	110,000 (38,000 to 240,000)	230,000 (81,000 to 500,000)	350,000 (120,000 to 830,000)

[a] All incidence estimates are rounded to whole numbers with a maximum of two significant digits.
[b] All incidence estimates are based on ozone-only models unless otherwise noted.
[c] The sensitivity analysis for long-term exposure-related mortality included an assessment of potential thresholds, which was completed for the 2025 scenario (see Table 5-19). Care should be taken in applying the results of that sensitivity analysis to the post-2025 scenario, although general patterns of impact across the thresholds may apply.

Table 5-27. Total Monetized Ozone-Only Benefits for the Proposed and Alternative Annual Ozone Standards (Incremental to the Analytical Baseline) for the post-2025 Scenario (nationwide benefits of attaining each alternative standard just in California) (millions of 2011) [a, b]

	Health Effectb	Proposed and Alterative Standards (95th percentile confidence intervals) 70 ppb	65 ppb	60 ppb
Avoided Short-Term Mortality - Core Analysis				
multi-city studies	Smith et al. (2009) (all ages)	$660 ($58 to $1,900)	$1,400 ($130 to $4,100)	$2,100 ($190 to $6,100)
	Zanobetti and Schwartz (2008) (all ages)	1,100 ($96 to $3,100)	2,400 ($210 to $6,700)	3,600 ($320 to $10,000)
Avoided Short-Term Mortality - Sensitivity Analysis				
multi-city studies	Smith et al. (2009) (all ages) copollutants model (PM_{10})	$530 (-$120 to $1,900)	$1,100 (-$270 to $4,200)	$1,700 (-$410 to $6,300)
	Schwartz (2005) (all ages)	$820 ($65 to $2,500)	$1,800 ($140 to $5,400)	$2,700 ($210 to $8,100)
	Huang et al. (2005) (cardiopulmonary)	$900 ($74 to $2,600)	$1,900 ($160 to $5,700)	$2,900 ($240 to $8,600)
meta-analyses	Bell et al. (2004) (all ages)	530 ($43 to $1,600)	1,200 ($93 to $3,500)	1,700 ($140 to $5,200)
	Bell et al. (2005) (all ages)	1,700 ($150 to $4,900)	3,700 ($320 to $11,000)	5,600 ($490 to $16,000)
	Ito et al. (2005) (all ages)	$2,400 ($220 to $6,600)	$5,200 ($470 to $14,000)	$7,800 ($710 to $22,000)
	Levy et al. (2005) (all ages)	$2,400 ($220 to $6,500)	$5,200 ($480 to $14,000)	$7,800 ($730 to $21,000)
Avoided Long-term Respiratory Mortality - Sensitivity Analysis				
multi-city study	Jerrett et al. (2009) (age 30-99) copollutants model ($PM_{2.5}$) no lag [c]	$2,700 ($220 to $7,900)	$5,700 ($470 to $17,000)	$8,600 ($700 to $25,000)
	Jerrett et al. (2009) (age 30-99) copollutants model ($PM_{2.5}$) 20 yr segmented lag [d]	$2,200 to $2,400 ($180 to $7,200)	$4,700 to $5,200 ($380 to $15,000)	$7,000 to $7,800 ($570 to $23,000)

[a] All benefits estimates are rounded to whole numbers with a maximum of two significant digits. The monetized value of the ozone-related morbidity benefits are included in the estimates shown in this table for each mortality study (and when combined account for from 4-6% of the total benefits, depending on the total mortality estimate compared against. Note that asthma exacerbations accounts for <<1% of the total).

[b] The sensitivity analysis for long-term exposure-related mortality included an assessment of potential thresholds however this assessment was implemented using incidence estimates (see Table 5-19). Observations from that analysis (in terms of fractional impacts on incidence can be directly applied to these benefit results) (see Appendix 5B, section 5B.1)

[c] A single central-tendency value is provided in each cell, since the zero-lag model used here did not require application of a 3% and 7% discount rates (see footnote d below) (note, however that as with all other entries in this study, we do include a 95th percentile confidence interval range – these are the values within parentheses).

[d] The range (outside of the parentheses) within each cell results from application of a 7% and 3% discount rates in the context of applying the 20year segmented lag (with the 7% resulting in the lower estimate and the 3% the higher estimate). The range presented within the parentheses reflects consideration for the 95th% confidence interval generated for each of these estimates and ranges from a low value (2.5th% CI for the 7% discount-based dollar benefit) to an upper value (97.5th% of the 3% discount-based dollar benefit).

Table 5-28. Estimated Number of Avoided $PM_{2.5}$-Related Health Impacts for the Proposed and Alternative Annual Ozone Standards (Incremental to the Analytical Baseline) for the post-2025 Scenario (nationwide benefits of attaining each alternative standard just in California) [a]

	Proposed and Alterative Standards		
Health Effectb	70 ppb	65 ppb	60 ppb
Avoided PM2.5-related Mortality			
Krewski et al. (2009) (adult mortality age 30+)	45	89	120
Lepeule et al. (2012) (adult mortality age 25+)	100	200	280
Woodruff et al. (1997) (infant mortality)	<1	<1	<1
Avoided PM2.5-related Morbidity			
Non-fatal heart attacks			
Peters et al. (2001) (age >18)	54	110	140
Pooled estimate of 4 studies (age >18)	6	11	16
Hospital admissions—respiratory (all ages)	14	27	37
Hospital admissions—cardiovascular (age > 18)	16	32	45
Emergency department visits for asthma (all ages)	24	46	64
Acute bronchitis (ages 8–12)	67	130	180
Lower respiratory symptoms (ages 7–14)	860	1,700	2,300
Upper respiratory symptoms (asthmatics ages 9–11)	1,200	2,400	3,300
Asthma exacerbation (asthmatics ages 6–18)	1,900	3,800	5,200
Lost work days (ages 18–65)	5,500	11,000	15,000
Minor restricted-activity days (ages 18–65)	32,000	64,000	88,000

[a] All incidence estimates are rounded to whole numbers with a maximum of two significant digits. Because these estimates were generated using benefit-per-ton estimates, confidence intervals are not available. In general, the 95th percentile confidence interval for the health impact function alone ranges from approximately ±30 percent for mortality incidence based on Krewski et al. (2009) and ±46 percent based on Lepeule et al. (2012).

Table 5-29. Monetized $PM_{2.5}$-Related Health Co-Benefits for the Proposed and Alternative Annual Ozone Standards (Incremental to Analytical Baseline) for the post-2025 Scenario (nationwide benefits of attaining each alternative standard just in California) (Millions of 2011) [a,b]

Monetized Benefits	Proposed and Alterative Standards		
	70 ppb	65 ppb	60 ppb
3% Discount Rate			
Krewski et al. (2009) (adult mortality age 30+)	$420	$830	$1,100
Lepeule et al. (2012) (adult mortality age 25+)	$950	$1,900	$2,600
7% Discount Rate			
Krewski et al. (2009) (adult mortality age 30+)	$380	$750	$1,000
Lepeule et al. (2012) (adult mortality age 25+)	$860	$1,700	$2,300

[a] All estimates are rounded to two significant digits. Because these estimates were generated using benefit-per-ton estimates, confidence intervals are not available. In general, the 95th percentile confidence interval for monetized $PM_{2.5}$ benefits ranges from approximately -90 percent to +180 percent of the central estimates based on Krewski et al. (2009) and Lepeule et al. (2012). Estimates do not include unquantified health benefits noted in Table 5-3 or Section 5.6.3.6 or welfare co-benefits noted in Chapter 6.

[b] The reduction in premature fatalities each year accounts for over 98% of total monetized benefits in this analysis. Mortality risk valuation assumes discounting over the SAB-recommended 20-year segmented lag structure.

Table 5-30. Estimate of Monetized Ozone and $PM_{2.5}$ Benefits for Proposed and Alternative Annual Ozone Standards Incremental to the Analytical Baseline for the post-2025 Scenario (nationwide benefits of attaining each alternative standard just in California) – Full Attainment (billions of 2011$) [a]

	Discount Rate	70 ppb	65 ppb	60 ppb
Ozone-only Benefits (range reflects Smith et al., 2009 and Zanobetti and Schwartz, 2008)	[b]	$0.66 to $1.1	$1.4 to $2.4	$2.1 to $3.6
$PM_{2.5}$ Co-benefits (range reflects Krewski et al., 2009 and Lepeule et al., 2012)	3%	$0.42 to $0.95	$0.83 to $1.9	$1.1 to $2.6
	7%	$0.38 to $0.86	$0.75 to $1.7	$1.0 to $2.3
Total Benefits	3%	$1.1 to $2.0 +B	$2.3 to $4.2 +B	$3.4 to $6.2 +B
	7%	$1.1 to $2.0 +B	$2.2 to $4.1 +B	$3.2 to $5.9 +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 for $PM_{2.5}$ assumes discounting over the SAB-recommended 20-year segmented lag structure. Not all possible benefits are quantified and monetized in this analysis. B is the sum of all unquantified health and welfare co-benefits. Data limitations prevented us from quantifying these endpoints, and as such, these benefits are inherently more uncertain than those benefits that we were able to quantify. These estimates reflect the economic value of avoided morbidities and premature deaths using risk coefficients from the studies noted.

[b] Ozone-only benefits reflect short-term exposure impacts and as such are assumed to occur in the same year as ambient ozone reductions. Consequently, social discounting is not applied to the benefits for this category.

Table 5-31. Regional Breakdown of Monetized Ozone-Specific Benefits Results for the post-2025 Scenario (nationwide benefits of attaining each alternative standard just in California) – Full Attainment [a]

Region	Proposed and Alterative Standards		
	70 ppb	65 ppb	60 ppb
East	0%	0%	0%
California	93%	94%	94%
Rest of West	6%	6%	6%

[a] Because we use benefit-per-ton estimates to calculate the $PM_{2.5}$ co-benefits, a regional breakdown for the co-benefits is not available. Therefore, this table only reflects the ozone benefits.

5.7.3 Uncertainty in Benefits Results (including Discussion of Sensitivity Analyses and Supplemental Analyses)

The dollar value of avoided ozone and $PM_{2.5}$ related premature deaths account for 94% to 98% of the total monetized benefits. This is true in part because we are unable to quantify many categories of benefits. The next largest benefit is for reducing the incidence of nonfatal heart attacks. The remaining categories each account for a small percentage of total benefit; however, they represent a large number of avoided incidences affecting many individuals. Comparing an incidence table to the monetary benefits table reveals that the number of incidences avoided and the dollar value for that endpoint do not always closely correspond. For example, for ozone we estimate almost 1,000 times more asthma exacerbations would be avoided than premature mortalities, yet asthma exacerbations account for only a very small fraction (<<1%) of total monetized benefits (see Tables 5-20). 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.

Sources of uncertainty associated with both the modeling of ozone-related benefits and $PM_{2.5}$-related cobenefits are discussed qualitatively in Appendix A. Key assumptions and uncertainties related to the modeling of ozone are presented below.

- We assume that short-term exposure to ozone is associated with mortality and that this relationship holds across the full range of exposure. Furthermore, we assume that long-term exposure to ozone is associated with respiratory mortality and while we favor a no-threshold linear C-R function, we consider the potential impact of thresholds ranging from 40 ppb to 60 ppb.

- In modeling long-term exposure-related respiratory mortality, we acknowledge uncertainty in specifying the nature of the cessation lag and have consequently included two alternative lag structures including a 20 year segmented lag and a zero lag (the zero lag reflects the potential that estimates of long-term exposure-related mortality could actually be capturing the accumulation of short-term exposure-related mortality events). In addition, we acknowledge the value in exploring the impact of potential thresholds in effect on the core incidence and benefits estimates.

$PM_{2.5}$ mortality co-benefits represent a substantial proportion of total monetized benefits (over 98% of the co-benefits), and these estimates have the following key assumptions and uncertainties.

- We assume that all fine particles, regardless of their chemical composition, are equally potent in causing premature mortality. This is an important assumption, because $PM_{2.5}$ produced varies considerably in composition across sources, but the scientific evidence is not yet sufficient to allow differential effects estimates by particle type. The PM ISA, which was twice reviewed by SAB-CASAC, concluded that "many constituents of $PM_{2.5}$ can be linked with multiple health effects, and the evidence is not yet sufficient to allow differentiation of those constituents or sources that are more closely related to specific outcomes" (U.S. EPA, 2009b).

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

- We assume that there is a "cessation" lag between the change in PM exposures and the total realization of changes in mortality effects. Specifically, we assume that some of the incidences of premature mortality related to $PM_{2.5}$ exposures occur in a distributed fashion over the 20 years following exposure based on the advice of the SAB-HES (U.S. EPA-SAB, 2004c), which affects the valuation of mortality benefits at different discount rates.

- We recognize uncertainty associated with application of the benefit-per-ton approach used in modeling $PM_{2.5}$ cobenefits. The benefit-per-ton estimates used here reflect specific geographic patterns of emissions reductions and specific air quality and benefits modeling assumptions associated with the derivation of those estimates (see the TSD describing the calculation of the national benefit-per-ton estimates (U.S. EPA, 2013b) and Fann et al. (2012c)). Consequently, these estimates may not reflect local variability in population density, meteorology, exposure, baseline health incidence rates, or other local factors associated with the current ozone NAAQS review. Therefore, use of these benefit-per-ton values to estimate co-benefits may lead to higher or lower benefit estimates than if co-benefits were calculated based on direct air quality modeling.

In order to evaluate the quantitative impact of specific sources of uncertainty on the core risk estimates, we completed a number of sensitivity analyses, some of which have already been described in presenting the core risk estimates and some of which are presented in Appendix 5B. A brief overview of these sensitivity analyses, including key observations resulting from those observations are presented below in section 5.7.3.1. In addition to these sensitivity analyses, we have also included several supplemental analyses intended to provide additional perspectives on the core incidence and benefits analyses. These supplemental analyses are presented in detail in Appendix 5C and are also briefly summarized below in section 5.7.3.2.

5.7.3.1 Sensitivity Analyses

A number of sensitivity analyses have been completed as part of this RIA. Given the importance of mortality in driving benefits estimates (both for ozone and $PM_{2.5}$ cobenefits), the sensitivity analyses completed have been focused largely on the mortality endpoint. Each sensitivity analysis is briefly described below (including an overview of the approach used in conducting the analysis and key observations). As identified below, several of the sensitivity analyses have been presented earlier parallel to presentation of the core estimates, while others are described in Appendix B.[71]

- **Short-term ozone-exposure related mortality (alternative epidemiological studies and C-R functions):** As described in section 5.6.3.1, in addition to the two core effect estimates we estimated benefits using seven additional effect estimates including four multi-city studies and three meta-analysis studies. This sensitivity analysis showed that the two core incidence and benefits estimates fall within (and towards the lower end of) the broader range resulting from application of the seven alternative effect estimates (see Tables 5-19 and 5-20). This increased our overall confidence in the two core studies, particularly with regard to epidemiological study design and the characterization of the relationship between short-term exposure and mortality.

- **Long-term ozone-exposure related respiratory mortality dollar benefits:** As discussed in section 5.1, we could not specify a cessation lag structure specific to long-term ozone exposure-related respiratory mortality. While we included incidence estimates as a part of the core analysis, we present the associated dollar benefits as a sensitivity analysis (see Tables 5-20 and 5-27). Given uncertainty related to the lag structure, we also included two different lags in modeling these benefits as a sensitivity analysis – a 20

[71] Generally, we gave greater emphasis to those sensitivity analyses examining sources of potential uncertainty directly associated with estimates of ozone-related mortality and have included summaries of those sensitivity analyses results parallel to presentation of core incidence and benefits estimates in sections 5.7.1 and 5.7.2.

year segment lag (as used for $PM_{2.5}$) and a zero lag (see section 5.6.4.1 for additional detail on the lag structures used). The sensitivity analysis suggests that if included in the core benefit estimate, long-term ozone exposure-related mortality could add substantially to the overall benefits (see Table 5-20 and 5-27). Additionally, use of a 20 year segment lag can reduce benefits by 10-20% (relative to a zero lag) depending on the discount rate applied.

- **Long-term ozone-exposure related respiratory mortality and potential thresholds:** As discussed in section 5.6.3.1, while we favor the no-threshold model in estimating benefits related to long-term ozone exposure-related respiratory mortality, as a sensitivity analysis, we evaluated the impact of thresholds ranging from 40-60ppb.[72] These results are included alongside core incidence estimates in Table 5-19 (see Appendix 5B, section 5B1 for additional details). This sensitivity analysis suggested that a threshold of 50ppb or greater could have a substantial impact on estimated benefits, while thresholds below this range have a relatively minor impact.

- **Long-term $PM_{2.5}$ exposure-related mortality and alternative C-R functions (based on expert elicitation):** As discussed in Appendix 5D (section 5D.1) in 2006 we conducted an expert elicitation to help better characterize uncertainty associated with long-term $PM_{2.5}$ exposure-related mortality and specifically the C-R functions used in modeling that endpoint, including the shape of the functions and potential for thresholds in effect. As part of the sensitivity analysis for the current ozone RIA, we applied the set of expert elicitation-based functions to generate an alternative set of $PM_{2.5}$ incidence and benefit estimates (see Appendix 5B, section 5B.2). That sensitivity analysis showed that the two core incidence estimate fall within the range of alternative C-R function based estimates obtained through expert elicitation (see Table 5B-2). This increases overall confidence in the core estimates with regard to the form of the functions and magnitude of the effect estimates.

- **Income elasticity and the willingness-to-pay (WTP) values used for mortality and certain morbidity endpoints:** As described in Appendix 5B (section 5B.3) we examined the impact of alternative assumptions regarding income elasticity (i.e., the degree to which WTP changes as income changes) and the degree of impact on WTP functions used for mortality and for morbidity endpoints. That sensitivity analysis, suggests that alternative assumptions regarding income elasticity could result in a moderate impact on mortality benefits (values ranging from ~90% to ~130% of the core estimate depending on the assumption regarding elasticity). Income elasticity was found to have a far more modest impact on morbidity endpoints modeled using WTP functions.

[72] As noted in section 5.6.3.1, our decision to include benefit estimates based on the no-threshold model in the core analysis reflects recommendations on the HREA provided by CASAC (Frey, 2014).

5.7.3.2 Supplemental Analyses

In addition to the sensitivity analyses described in section 5.7.3.1, we also included a number of supplemental analyses intended to examine other attributes of the core risk estimates (these are briefly described below with additional detail found in Appendix 5C).

- **Consideration for age group-differentiated aspects of short-term ozone exposure-related mortality (including total avoided incidence, life years gained and percent reduction in baseline mortality):** These analyses expand on the basic mortality incidence estimates presented in section 5.7 by considering (a) estimates of the reduction in mortality incidence differentiated by age range (i.e., how the avoided deaths map to different age ranges) and (b) estimates of life years gained by age range and (c) percent reduction in baseline mortality by age range. 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 ozone-attributable deaths occur in populations age 75–99 (see Appendix 5C, section 5C.1 Table 5C-1), but half of the life years would occur in populations younger than 65 (see Appendix 5C, section 5C.1, Table 5C-2). This is because the younger populations have the potential to lose more life years per death than older populations based on changes in ozone exposure for the 2025 scenario. Results presented in Table 5C-3 highlight that when reductions in ozone-attributable mortality (in going from baseline to an alternative standard level) are considered as a percentage of total all-cause baseline mortality, the estimates are relatively small and are fairly constant across age ranges. However, it is important to point out that estimates of total ozone-attributable mortality represent a substantially larger fraction of all-cause baseline mortality than the increment attributable to the simulated reduction in ozone associated with an alternative standard level.

- **Evaluation of mortality impacts relative to the baseline pollutant concentrations (used in generating those mortality estimates) for both short-term ozone exposure-related mortality and long-term $PM_{2.5}$ exposure-related mortality:** In this supplemental analysis, we begin by comparing the distribution of short-term ozone exposure-related mortality against the ozone season-averaged 8hr max values used in deriving those estimates (for the three alternative standards modeled for the 2025 scenario) (see section 5C.2). In making this comparison, we point out that, rather than using a full daily time series of 8hr max values for each grid cell, we used an ozone season-average 8hr max value for incidence modeling. While this simplification did not impact the overall core incidence estimate for this mortality endpoint, it does impact efforts to compare the distribution of mortality estimates against associated daily values (by reducing temporal variation associated with each grid cell calculation). Key observations resulting from this supplemental analysis are that (a) the vast majority of reductions in short-term exposure-related mortality for ozone occur in grid cells with mean 8hr max baseline levels (across the ozone season) between 35 and 55ppb and (b) virtually all of the mortality reductions are associated with ozone levels above the <20ppb range identified within the Ozone ISA as being associated with less confidence in specifying the nature of the C-R function for ozone mortality (ozone ISA, section

2.5.4.4). The other portion of this supplemental analysis (which compares the distribution of $PM_{2.5}$ mortality estimates relative to associated $PM_{2.5}$ levels used in modeling) relies on the concept of the lowest measured level (LML) associated with the epidemiological studies providing the effect estimates used in morality modeling. Specifically, we observe that there is reduced confidence in specifying the nature of the C-R function at exposure levels below the range used in the epidemiological study (i.e., below the LML). In the supplemental analysis, we found that, depending on the mortality study, between 67% and 93% of the mortality estimate is based on modeling involving $PM_{2.5}$ levels above the LML (i.e., levels at which we have increased confidence in specifying the $PM_{2.5}$-mortality relationship) (see Appendix 5C.2).

- **Core Incidence and Dollar Benefits Estimates Reflecting Application of <u>Known Controls for the 2025 Scenario</u>:** In Appendix 5C, section 5C.3, we present a subset of core incidence and benefits for the 2025 scenario reflecting only application of known controls in modeling reductions in ozone to attain each alternative standard level. These known-control based estimates include both ozone-related and $PM_{2.5}$ co-benefit estimates as well as total benefits. As expected, the percent of benefits reflecting application of known controls decreases as you consider more stringent alternative standards. Estimates presented in Appendix 5C.3 suggest that 77%, 59% 34% of total benefits are associated with application of known controls (for the 70, 65 and 60 ppb alternative standards, respectively).

5.8 Discussion

The analysis in this Chapter demonstrates the potential for significant health benefits of the illustrative emissions controls applied to simulate attainment with the alternative primary ozone standards. We estimate that by 2025, the emissions reductions to reach the alternative standards everywhere except California, would have reduced the number of ozone- and $PM_{2.5}$-related premature mortalities and produce substantial non-mortality benefits. Furthermore, emissions reductions required to meet alternative standards in California post-2025 are also likely to produce substantial reductions in these same endpoints. This proposed rule also promises to yield significant welfare impacts as well (see Chapter 6). Even considering the quantified and unquantified uncertainties identified in this chapter, we believe that implementing the alternative standards would have substantial public health benefits that are likely to outweigh the costs for the three alternative standards analyzed (see Chapter 7).

Inherent in any complex RIA such as this one are multiple sources of uncertainty. Some of these we characterized through our quantification of statistical error in the concentration-response relationships and our use of alternate mortality functions. Others, including the projection of atmospheric conditions and source-level emissions, the projection of baseline morbidity rates, incomes and technological development are unquantified. When evaluated within the context of these uncertainties, the health impact and monetized benefits estimates in this RIA can provide useful information regarding the public health benefits associated with the proposed and alternative primary standards.

There are important differences worth noting in the design and analytical objectives of NAAQS RIAs compared to RIAs for implementation rules, such as the Tier 3 (U.S. EPA, 2014c). The NAAQS RIAs illustrate the potential costs and benefits of a revised air quality standard nationwide based on an array of emission reduction strategies for different sources, incremental to implementation of existing regulations and controls needed to attain the current standards. In short, NAAQS RIAs hypothesize, but do not predict, the emission reduction strategies that States may choose to enact when implementing a revised NAAQS. The setting of a NAAQS does not directly result in costs or benefits, and as such, NAAQS RIAs are merely illustrative and are not intended to be added to the costs and benefits of other regulations that result in specific costs of control and emission reductions. By contrast, the emission reductions from implementation rules are generally for specific, well-characterized sources, such as the recent MATS rule (U.S. EPA, 2011e). In general, the EPA is more confident in the magnitude and location of the emission reductions for implementation rules. As such, emission reductions achieved under promulgated implementation rules such as Tier 3 have been reflected in the baseline of this NAAQS analysis. Subsequent implementation rules will be reflected in the baseline for the next ozone NAAQS review. For this reason, the benefits estimated provided in this RIA and all other NAAQS RIAs should not be added to the benefits estimated for implementation rules.

In setting the NAAQS, the EPA considers that ozone concentrations vary over space and time. While the standard is designed to limit concentrations at the highest monitor in an area, it is understood that emission controls put in place to meet the standard of the highest monitor will simultaneously result in lower ozone concentrations throughout the entire area. In fact, the *Health Risk and Exposure Assessment for Ozone* (HREA) (U.S. EPA, 2014b) shows how different standard levels would affect the entire distribution of ozone concentrations, and thus people's exposures and risk, across a selected set of 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 ozone 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, 2014b). While benefits occurring below the standard may be somewhat more uncertain than those occurring above the standard, the EPA considers these to be legitimate components of the total benefits estimate. Though there are greater uncertainties at lower ozone and $PM_{2.5}$ concentrations, there is no evidence of a threshold in short-term ozone or $PM_{2.5}$-related health effects in the epidemiology literature.[73] Given that the epidemiological literature in most cases has not provided estimates based on threshold models, there would be additional uncertainties imposed by assuming thresholds or other non-linear concentration-response functions for the purposes of benefits analysis.

The estimated benefits for the proposed and alternative standards are in addition to the substantial benefits estimated for several recent implementation rules (U.S. EPA, 2009a, 2011d, 2014c). Rules such as Tier 3 and other emission reductions will have substantially reduced ambient ozone concentrations by 2025 in the East, such that few additional controls would be needed to reach 70 ppb in the East beyond the analytical baseline. These rules that have already been promulgated have tremendous combined benefits that explain why the number of avoided premature deaths associated with this NAAQS revision are smaller than were estimated in the previous ozone NAAQS RIA (U.S. EPA, 2006) for the year 2020 and even smaller than the mortality risks estimated for the current year in the ozone HREA (U.S. EPA, 2014b).

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[73] As discussed 5.7.3.1, our modeling of long-term ozone exposure-related respiratory mortality did include consideration of potential thresholds as a sensitivity analysis. However, key points need to be emphasized in the context of interpreting this endpoint: (a) based on recommendations from CASAC the core incidence reduction estimates was based on a non-threshold model (Frey, 2014) and (b) while the incidence estimates for this endpoint were included as core estimates, associated dollar benefit estimates were only included as sensitivity analyses and consequently did not factor in to the core dollar benefit estimates generated.

5.9 References

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APPENDIX 5A: COMPREHENSIVE CHARACTERIZATION OF UNCERTAINTY IN OZONE BENEFITS ANALYSIS

Overview

As noted in Chapter 5, the benefits analysis relies on an array of data inputs—including air quality modeling, health impact functions and valuation estimates among others—which are themselves subject to uncertainty and may also in turn contribute to the overall uncertainty in this analysis. The RIA employs a variety of analytic approaches designed to reduce the extent of the uncertainty and/or characterize the impact that uncertainty has on the final estimate. We strive to incorporate as many quantitative assessments of uncertainty as possible (e.g., Monte Carlo assessments, sensitivity analyses); however, there are some aspects we are only able to characterize qualitatively.

To more comprehensively and systematically address these uncertainties, including those we cannot quantify, we adapt the World Health Organization (WHO) uncertainty framework (WHO, 2008), which provides a means for systematically linking the characterization of uncertainty to the sophistication of the underlying health impact assessment. EPA has applied similar approaches in peer-reviewed analyses of $PM_{2.5}$-related impacts (U.S. EPA, 2010b, 2011, 2012) and ozone-related impacts (U.S. EPA, 2014). EPA's Science Advisory Board (SAB) has supported using a tabular format to qualitatively assess the uncertainties inherent in the quantification and monetization of health impacts, including identifying potential bias, potential magnitude, confidence in our approach, and the level of quantitative assessment of each uncertainty (U.S. EPA-SAB, 1999, 2001, 2004a, 2004b, 2011a, 2011b). The assessments presented here are largely consistent with those previous peer-reviewed assessments.

This appendix focuses on uncertainties inherent in the ozone benefits estimates. For more information regarding the uncertainties inherent in the PM benefits estimates, please see the 2012 PM NAAQS RIA (U.S. EPA, 2012).

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

Table 5A-1 catalogs the most significant sources of uncertainty in the ozone benefits analysis and then characterizes four dimensions of that uncertainty (listed below). The first two dimensions focus on the nature of the uncertainty. The third and fourth dimensions focus on the extent to which the analytic approach chosen in the benefits analysis either minimizes the impact of the uncertainty or quantitatively characterizes its impact.

1) The direction of the bias that a given uncertainty may introduce into the benefits assessment if not taken into account in the analysis approach;

2) The magnitude of the impact that uncertainty is likely to have on the benefits estimate if not taken into account in the analysis approach;

3) The extent to which the analytic approach chosen is likely to minimize the impact of that uncertainty on the benefits estimate; and

4) The extent to which EPA has been able to quantify the residual uncertainty after the preferred analytic approach has been incorporated into the benefits model.

5A.1.1 Direction of Bias

The "direction of bias" column in Table 5A-1 is an assessment of whether, if left unaddressed, an uncertainty would likely lead to an underestimate or overestimate the total monetized benefits. In some cases we indicate that there are reasons why the bias might go either direction, depending upon the true nature of the underlying relationship. Where available, we base the classification of the "direction of bias" on the analysis in the *Integrated Science Assessment for Ozone and Related Photochemical Oxidants* (hereafter, "O_3 ISA") (U.S. EPA, 2013a). Additional sources of information include advice from SAB and the National Academies of Science (NAS), as well as studies from the peer-reviewed literature. In some cases we indicate that there is not sufficient information to estimate whether the uncertainty would likely lead to under or overestimation of benefits; these cases are identified as "unable to determine."

5A.1.2 Magnitude of Impact

The "magnitude of impact" column in Table 5A-1 is an assessment of how much plausible alternative assumptions about the underlying relationship about which we are uncertain could influence the overall monetary benefits. EPA has applied similar classifications in previous risk and benefit analyses (U.S. EPA, 2010b, 2011, 2014), but we have slightly revised the category names and the cut-offs here.[74] The definitions used here are provided below.

[74] In *The Benefits and Costs of the Clean Air Act from 1990 to 2020* (U.S. EPA, 2011), EPA applied a classification of "potentially major" if a plausible alternative assumption or approach could influence the overall monetary benefit estimate by five 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.

- High—if the uncertainty associated with an assumption could influence the total monetized benefits by more than 25%.
- Medium—if the uncertainty associated with an assumption could influence the total monetized benefits by 5% to 25%.
- Low—if the uncertainty associated with an assumption could influence the total monetized benefits by less than 5%.

For each uncertainty, we provide as much quantitative information as is available in the table to support the classification.

Although many of the sources of uncertainty could affect both morbidity and mortality endpoints, because mortality benefits comprise over 94% of the monetized benefits that we are able to quantify in this analysis, uncertainties that affect the mortality estimate have the potential to have larger impacts on the total monetized benefits than uncertainties affecting only morbidity endpoints. One morbidity-related uncertainty that could have a significant impact on the benefits estimate is the extent to which omitted morbidity endpoints are included in the benefits analysis. Including additional morbidity endpoints that are currently not monetized would reduce the fraction of total benefits from mortality. Ultimately, the magnitude classification is determined by professional judgment of EPA staff based on the results of available information, including other U.S. EPA assessments of uncertainty (U.S. EPA, 2010b, 2011)

Based on this assessment, the uncertainties that we classified as high or medium-high impact are: the causal relationship between long-term and short-term ozone exposure and mortality, the shape of the concentration-response function for both categories of ozone-related mortality, and the mortality valuation, specifically for long-term exposure-related mortality. The classification of these uncertainties as "high magnitude" is generally consistent with the results of EPA's Influence Analysis (Mansfield et al., 2009), the Quantitative Health Risk Assessment for Particulate Matter (U.S. EPA, 2010b), and the Benefits and Costs of the Clean Air Act 1990 to 2020 (U.S. EPA, 2011).

5A.1.3 Confidence in Analytic Approach

The "confidence in analytic approach" column of Table 5A-1 is an assessment of the scientific support for the analytic approach chosen (or the inherent assumption made) to account for the relationship about which we are uncertain. In other words, based on the available evidence, how

EPA, 2010b), EPA applied classifications of "low" if the impact would not be expected to impact the interpretation of risk estimates in the context of the PM NAAQS review, "medium" if the impact had the potential to change the interpretation; and "high" if it was likely to influence the interpretation of risk in the context of the PM NAAQS review.

certain are we that EPA's selected approach is the most plausible of the potential alternatives. Similar classifications have been included in previous risk and benefits analyses (U.S. EPA, 2010b, 2011).[75] The three categories used to characterize the degree of confidence are:

- 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; and
- Low—limited data exists to support the selected approach.

Ultimately, the degree of confidence in the analytic approach is EPA staff's professional judgment based on the volume and consistency of supporting evidence, much of which has been evaluated in the O_3 ISA (U.S. EPA, 2013a) and by EPA's independent SAB. The O_3 ISA evaluated the entire body of scientific literature on ozone science and was twice peer-reviewed by EPA's Clean Air Scientific Advisory Committee (CASAC). In general, we regard a conclusion in the O_3 ISA or specific advice from SAB as supporting a high degree of confidence in the selected approach.

Based on this assessment, we have low or low-medium confidence in the evidence available to assess exposure error in epidemiology studies, morbidity valuation, baseline incidence projections for morbidity, and omitted morbidity endpoints. However, because these uncertainties have been classified as having a low or low-medium impact on the magnitude of the benefits, further investment in improving the available evidence would not have a substantial impact on the total monetized benefits.

5A.1.4 Uncertainty Quantification

The column of Table 5A-1 labeled "uncertainty quantification" is an assessment of the extent to which we were able to use quantitative methods to characterize the residual uncertainty in the benefits analysis, after addressing it to the extent feasible in the analytic approach for this RIA. We categorize the level of quantification using the four tiers used in the WHO uncertainty framework (WHO, 2008). The WHO uncertainty framework is a well-established approach to assess uncertainty in risk estimates that systematically links the characterization of uncertainty to the sophistication of the health impact assessment. The advantage of using this framework is that it clearly highlights the level of uncertainty quantification applied in this assessment and the potential sources of uncertainty that require methods

[75] 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.

development in order to assess quantitatively. Specifically, EPA applied this framework in multiple risk and exposure assessments (U.S. EPA, 2010b, 2014), and it has been recommended in EPA guidance documents assessing air toxics-related risk and Superfund site risks (U.S. EPA, 2004 and 2001, respectively). Ultimately, the tier decision is the professional judgment of EPA staff based on the availability of information for this assessment. The tiers used in this assessment are defined below.

- Tier 0—screening level, generic qualitative characterization.
- Tier 1—Scenario-specific qualitative characterization.
- Tier 2—Scenario-specific sensitivity analysis.
- Tier 3—Scenario-specific probabilistic assessment of individual and combined uncertainty.

Within the limits of the data, we strive to use more sophisticated approaches (e.g., Tier 2 or 3) for characterizing uncertainties that have the largest magnitudes and could not be completely addressed through the analytic approach. The uncertainties for which we have conducted probabilistic (Tier 3) assessments in this analysis are mortality causality, the shape of the concentration-response function, and mortality and morbidity valuation. For lower magnitude uncertainties, we include qualitative discussions of the potential impact of uncertainty on risk results (WHO Tier 0/1) and/or completed sensitivity analyses assessing the potential impact of sources of uncertainty on risk results (WHO Tier 2).

5A.2 Organization of the Qualitative Uncertainty Table

Table 5A-1 is organized as follows. The uncertainties are grouped by category (i.e., concentration-response function, valuation, population and baseline incidence, omitted benefits categories, and exposure changes). Within each category, the uncertainties are sorted by magnitude of impact (i.e., high to low) then by confidence in our approach (i.e., low to high). In the table, red (bold) text is used to indicate the uncertainties that likely have a high magnitude of impact on the total benefits estimate. This organization highlights the uncertainty with the largest potential impact and the lowest confidence at the top of each category.

Table 5A-1. Summary of Qualitative Uncertainty for Key Modeling Elements in Ozone Benefits

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Analytical Approach	Uncertainty Quantification
Uncertainties Associated with Concentration-Response Functions				
Causal relationship between short-term ozone exposure and premature mortality	Overestimate, if short-term ozone exposure does not have a causal relationship with premature mortality.	High Mortality generally dominates monetized benefits, so small uncertainties could have large impacts on the total monetized benefits.	High Our approach is consistent with the O_3 Integrated Science Assessment (ISA), which determined that premature mortality has a likely causal relationship with short-term ozone exposure based on the collective body of evidence (p. 6-264). In addition, the NAS recommended that EPA "should give little or no weight to the assumption that there is no causal association between estimated reductions in premature mortality and reduced ozone exposure" (NRC, 2008). In 2010, the Health Effects Subcommittee of the Advisory Council on Clean Air Compliance Analysis, while reviewing EPA's The Benefits and Costs of the Clean Air Act 1990 to 2020 (U.S. EPA, 2011), also confirmed the NAS recommendation to include ozone mortality benefits (U.S. EPA-SAB, 2010).	Tier 1 (qualitative)
	Either	Medium-High	Medium	Tier 2 (sensitivity analysis)

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Analytical Approach	Uncertainty Quantification
Shape of the C-R functions, particularly at low concentrations for short-term ozone exposure-related mortality	The direction of bias that assuming linear-no threshold model or alternative model introduces depends upon the "true" functional from of the relationship and the specific assumptions and data in a particular analysis. For example, if the true function identifies a threshold below which health effects do not occur, benefits may be overestimated if a substantial portion of those benefits were estimated to occur below that threshold. Alternately, if a substantial portion of the benefits occurred above that threshold, the benefits may be underestimated because an assumed linear no-threshold function may not reflect the steeper slope above that threshold to account for all health effects occurring above that threshold.	The magnitude of this impact depends on the fraction of benefits occurring in areas with lower concentrations. Mortality generally dominates monetized benefits, so small uncertainties could have large impacts on total monetized benefits.	The O_3 ISA did not find any evidence that supports a threshold in the relationship between short-term exposure to ozone and mortality within the range of ozone concentrations observed in the United States, and recent evidence suggests that the shape of the ozone-mortality C-R curve remains linear across the full range of ozone concentrations (p. 6-257). Consistent with the O_3 ISA, we assume a log-linear no-threshold model for the concentration-response functions for short-term ozone mortality. However, the ISA notes that there is less certainty in the shape of the C-R function below 20 ppb due to the low density of data in this range (p. 6-254-255).	The comparison of short-term mortality against the associated distribution of (ozone season-averaged) 8hr max ozone levels (see Appendix 5C, section 5C.2) suggests that the vast majority of predicted reductions in mortality are associated with days having 8hr max values that fall within the range of increased confidence in specifying the nature of the mortality response (as identified in the O_3 ISA).
Causal relationship between long-term ozone exposure and	Overestimate, if short-term ozone exposure does not have a causal relationship with premature mortality.	High	High	Tier 1 (qualitative)

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Analytical Approach	Uncertainty Quantification
premature respiratory mortality		The total dollar benefits of reducing ozone levels are comprised mostly of the value placed on reducing the risk of premature death, so small uncertainties could have large impacts on the total monetized benefits.	While the O_3 ISA concludes that evidence is *suggestive of a causal association* between total mortality and long-term ozone exposure (section 7.7.1), specifically with regard to respiratory health effects (including mortality), the ISA concludes that there is *likely to be a causal association* (section 7.2.8). Furthermore, in their review of the HREA completed for the Ozone NAAQS review, the CASAC expressed their support for EPA's plan to include a non-threshold-based modeling of long-term exposure-related respiratory mortality in the core estimate. The CASAC also supported EPA's plan to consider the potential for thresholds in the response (in the range of 40-60 ppb) as a sensitivity analysis in the HREA (see section 5.6.3.1). This same approach (with regard to the core approach and sensitivity analysis for long-term exposure-related morality) has been adopted for the RIA (see below).	
Shape of the C-R functions, particularly at low concentrations for long-term ozone exposure-related respiratory mortality	Either	Medium-High	Medium	Tier 2 (sensitivity analysis)

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Analytical Approach	Uncertainty Quantification
	The direction of bias that assuming linear-no threshold model or alternative model introduces depends upon the "true" functional form of the relationship and the specific assumptions and data in a particular analysis. For example, if the true function identifies a threshold below which health effects do not occur, benefits may be overestimated if a substantial portion of those benefits were estimated to occur below that threshold. Alternately, if a substantial portion of the benefits occurred above that threshold, the benefits may be underestimated because an assumed linear no-threshold function may not reflect the steeper slope above that threshold to account for all health effects occurring above that threshold.	The magnitude of this impact depends on the fraction of benefits occurring in areas with lower concentrations. Note, that due to limitations in our ability to predict the lag structure associated with reductions in long-term ozone exposure-related mortality, we are not including dollar benefits associated with this endpoint in the core benefits analysis (which significantly reduces the role of this source of uncertainty in impact core benefit estimates).	In their memo (see Sasser 2014) clarifying the results of their study (Jerrett et al., 2009) regarding long-term ozone exposure-related respiratory mortality, the study authors note that in terms of goodness of fit, long-term health risk models including ozone clearly performed better than models without ozone, indicating the improved predictions of respiratory mortality when ozone is included. In the article proper, the authors state that, "There was limited evidence that a threshold model specification improved model fit as compared with a non-threshold linear mode…". Furthermore, in the memo referenced above, the authors conclude that considerable caution should be exercised in using any specific threshold, particularly when the more stringent statistical test indicates there is no significantly improved prediction. The CASAC was supportive of the approach EPA used in the HREA of using a non-threshold C-R function based on this study to generate core estimates and consider the impact of potential thresholds (ranging from 40-60 ppb) as a sensitivity analysis.	As part of the sensitivity analyses completed for the RIA, we did examine the potential impact of thresholds (from 40 to 60 ppb) in the response function for long-term exposure-related morality (see Appendix 5B, section 5B.1). That analysis suggested that thresholds between 55 and 60 ppb would have a substantial impact on overall modeled benefits, while thresholds below 50 ppb would have a minor impact on predicted benefits (see Appendix 5B, Table 5B-1).
Exposure error in epidemiology studies	Underestimate (generally) The O_3 ISA states that exposure measurement error can also be an important contributor to uncertainty in effect estimates associated with both short-term and long-term studies (p. 1xii). Together with other factors (e.g., low data density), exposure error can smooth the C-R functions and obscure potential thresholds (p. lxix). In addition, the O_3 ISA states that exposure error can bias effect estimates toward or away from the null and widen confidence intervals (p. lxii).	Medium 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.	Low-Medium Although this underestimation is well documented, including in the O_3 ISA, the SAB has not suggested an approach to adjust for this bias.	Tier 1 (qualitative) (No quantitative method available)
	Unknown	Medium	Medium	Tier 1 (qualitative)

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Analytical Approach	Uncertainty Quantification
Adjustment of risk coefficients to 8-hour maximum	Several of the mortality epidemiological studies were reported for a 24-hour average or 1-hour maximum ozone level. These metrics are not the most relevant to characterizing population-level exposure. Thus, we have converted ozone mortality health impact functions that use these metric to maximum 8-hour average ozone concentration using standard conversion functions. The conversion ratios are based on observed relationships between the 24-hour average and 8-hour maximum in the underlying studies.	This conversion also does not affect the relative magnitude of the health impact function. However, we do note that the pattern of 8hr max concentrations for a particular location over an ozone season could differ from the pattern of 1hr max or 24 hour average metrics for that same location. Consequently, estimates of incidence reductions (and associated dollar benefits) could differ for a particular location depending on the metric used for risk calculations.[76]	This practice is consistent both with the available exposure modeling and with the form of the current ozone standard. However, in some cases, these conversions were not specific to the ozone "warm" season which was the period used in the benefits analysis, which introduces additional uncertainty due to the use of effect estimates based on a mixture of warm season and all year data in the epidemiological studies.	(No quantitative method available)
Confounding by individual risk factors, other than socioeconomic status—e.g., smoking, or ecologic factors, which represent the neighborhood, such as unemployment	Either, depending on the factor and study Individual, social, economic, and demographic covariates can bias the relationship between particulate air pollution and mortality, particularly in cohort studies that rely on regional air pollution levels.	Medium Because mortality dominates monetized benefits, even a small amount of confounding could have medium impacts on total monetized benefits.	Medium To minimize confounding effects, we use risk coefficients that control for individual risk factors to the extent practical.	Tier 2 (sensitivity analysis) (Quantitative methods available but not assessed in this analysis.)
	Either, depending upon the pollutant.	Medium	Medium	Tier 1 (qualitative)

[76] We note that if metric adjustment ratios were derived for individual point locations (reflecting the relationship between metrics for each location) and those metrics were used to generate adjusted C-R functions for each location, then some of this potential uncertainty could be reduced, however available data does not support the application of highly location-specific adjustment ratios (with cross-city or national adjustment ratios being typically used instead).

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Analytical Approach	Uncertainty Quantification
Confounding and effect modification by co-pollutants	Disentangling the health responses of combustion-related pollutants (i.e., PM, SOx, NOx, ozone, and CO) is a challenge. The O_3 ISA defines a confounder as the true cause of the association observed between the exposure and the outcome; in contrast, an effect modifier changes the magnitude of the association between the exposure and the outcome (p. lxii).	Because this uncertainty could affect mortality and because mortality generally dominates monetized benefits, even small uncertainties could have medium impacts on total monetized benefits.	The O_3 ISA states that there is high confidence that unmeasured confounders are not producing the findings when multiple studies are conducted in various settings using different subjects or exposures, such as multi-city studies (p. lxi). When modeling effects of pollutants jointly (e.g., PM and O_3), we apply multi-pollutant effect estimates when those estimates are available to avoid double-counting and satisfy other selection criteria. In addition, we apply multi-city effect estimates when available.	(No quantitative method available)
Application of C-R relationships only to the original study population	Underestimate Estimating health effects for only the original study population may underestimate the whole population benefits of reductions in pollutant exposures.	Low Mortality generally dominates monetized benefits, so further age range expansions for morbidity endpoints would have a small impact on total monetized benefits.	High Following advice from the SAB (U.S. EPA-SAB, 2004a, pg. 7) and NAS (NRC, 2002, pg. 114), we expanded the age range for childhood asthma exacerbations beyond the original study population to ages 6-18.	Tier 2 (sensitivity analysis) (Quantitative methods available but not assessed in this analysis.)
Uncertainties Associated with Economic Valuation				
Mortality Risk Valuation/Value-of-a-Statistical-Life (VSL)	Unknown Some studies suggest that EPA's mortality valuation is too high, while other studies suggest that it is too low. Differences in age, income, risk aversion, altruism, nature of risk (e.g., cancer), and study design could lead to higher or lower estimates of mortality valuation.	High Mortality generally dominates monetized benefits, so moderate uncertainties could have a large effect on total monetized benefits.	Medium The VSL used by EPA is based on 26 labor market and stated preference studies published between 1974 and 1991. EPA is in the process of reviewing this estimate and will issue revised guidance based on the most up-to-date literature and recommendations from the SAB-EEAC in the near future (U.S. EPA, 2010a, U.S. EPA-SAB, 2011c).	Tier 3 (probabilistic) Assessed uncertainty in mortality valuation using a Weibull distribution.
	Underestimate	Medium-High	Low	Tier 2 (sensitivity analysis)

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Analytical Approach	Uncertainty Quantification
Cessation lag structure for long-term ozone mortality	Jerrett et al. (2009) notes that, "Allowing for a 10-year period of exposure to ozone (5 years of follow-up and 5 years before the follow-up period) did not appreciably alter the risk estimates…" We acknowledge substantial uncertainty associated with specifying the lag for long-term respiratory mortality.	Although the cessation lag does not affect the number of premature deaths attributable to long-term ozone exposure, it affects the timing of those deaths and thus the discounted monetized benefits. Mortality generally dominates monetized benefits, so moderate uncertainties could have a large effect on total monetized benefits.	As discussed in section 5.6.4.1, in presenting dollar benefit estimates as part of the sensitivity analysis (presenting dollar benefits for long-term ozone-related morality), we include both an assumption of zero lag and a lag structure matching that used for the core $PM_{2.5}$ estimate (the SAB 20 year segmented lag). Inclusion of the zero lag reflects consideration for the possibility that the long-term respiratory mortality estimate captures primarily an accumulation of short-term mortality effects across the ozone season. The use of the 20 year segmented lag reflects consideration for advice provided by the HES (USEPA-SAB, 2010).	As shown in sensitivity analysis results presented in Appendix 5B, the use of the 20 year segmented lag results in a 10-20% reduction in the total dollar benefit (using a 3% and 7% discount rate, respectively) relative to the alternative approach of applying no lag (i.e., assuming all of the mortality reductions occur in the same year).
	Either	Medium	Medium	Tier 2 (sensitivity analysis)
Income growth adjustments	Income growth increases willingness-to-pay (WTP) valuation estimates, including mortality, over time. From 1997 to 2010, personal income and GDP growth have begun to diverge. If this trend continues, the assumption that per capita GDP growth is a reasonable proxy for income growth may lead to an overstatement of benefits. (IEc, 2012).	Income growth from 1990 to 2020 increases mortality valuation by 20%. Alternate estimates for this adjustment vary by 20% (IEc, 2012). Because we do not adjust for income growth over the 20-year cessation lag, this approach could also underestimate the benefits for the later years of the lag.	Consistent with SAB recommendations (U.S. EPA,-SAB, 2000, pg. 16), we adjust WTP for income growth. Difficult to forecast future income growth. However, in the absence of readily available income data projections, per capita GDP is the best available option.	As shown in Appendix 5B, the use of alternate income growth adjustments would change the monetized benefits by +33% to −14%.
	Underestimate	Low	Low	Tier 3 (probabilistic), where available
Morbidity valuation	Morbidity benefits such as hospital admissions are calculated using cost-of-illness (COI) estimates, which are generally half the WTP to avoid the illness (Alberini and Krupnick, 2000). In addition, the morbidity costs do not reflect physiological responses or sequelae events, such as increased susceptibility for future morbidity.	Even if we doubled the monetized valuation of morbidity endpoints using COI valuation that are currently included in the RIA, the change would still be less than 5% of the monetized benefits. It is unknown how much including sequelae events could increase morbidity valuation.	Although the COI estimates for hospitalizations reflect recent data, we have not yet updated other COI estimates such as for school loss days. The SAB concluded that COI estimates could be used as placeholders where WTP estimates are unavailable, but it is reasonable to presume that this strategy typically understates WTP values (U.S. EPA-SAB, 2004b, pg. 3).	Assessed uncertainty in morbidity valuation using distributions specified in the underlying literature, where available (see Table 5-10).
Uncertainties Associated with Baseline Incidence and Population Projections				
	Either	Low–Medium	Medium	Tier 1 (qualitative)

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Analytical Approach	Uncertainty Quantification
Population estimates and projections	The monetized benefits would change in the same direction as the over- or underestimate in population projections in areas where exposure changes.	Monetized benefits are substantially affected by population density. Comparisons using historical census data show that population projections are ±5% nationally, but projection accuracy can vary by locality. Historical error for Woods & Poole's population projections has been ±8.1% for county-level projections and ±4.1% for states (Woods and Poole, 2012). The magnitude of impact on total monetized benefits depends on the specific location where PM is reduced.	We use population projections for 5-year increments for 304 race/ethnicity/gender/age groups (Woods and Poole, 2012) at Census blocks. Population forecasting is well-established but projections of future migration due to possible catastrophic events are not considered. In addition, projections at the small spatial scales used in this analysis are inherently more uncertain than projections at the county- or state-level.	(No quantitative method available)
Uncertainty in projecting baseline incidence rates for mortality	Unknown Because the mortality rate projections for future years reflect changes in mortality patterns as well as population growth, the projections are unlikely to be biased.	Low-Medium Because mortality generally dominates monetized benefits, small uncertainties could have medium impacts on total monetized benefits.	Medium The county-level baseline mortality rates reflect recent databases (i.e., 2004–2006 data) and are projected for 5-year increments for multiple age groups. This database is generally considered to have relatively low uncertainty (CDC Wonder, 2008). The projections account for both spatial and temporal changes in the population.	Tier 1 (qualitative) (No quantitative method available)
Uncertainty in projecting baseline incidence rates and prevalence rates for morbidity	Either, depending on the health endpoint Morbidity baseline incidence is available for current year only (i.e., no projections available). Assuming current year levels can bias the benefits for a specific endpoint if the data has clear trends over time. Specifically, asthma prevalence rates have increased substantially over the past few years while hospital admissions have decreased substantially.	Low The magnitude varies with the health endpoint, but the overall impact on the total benefits estimate from these morbidity endpoints is likely to be low.	Low-Medium We do not have a method to project future baseline morbidity rates, thus we assume current year levels will continue. While we try to update the baseline incidence and prevalence rates as frequently as practicable, this does not continue trends into the future. Some endpoints such as hospitalizations and ER visits have more recent data (i.e., 2007) stratified by age and geographic location. Other endpoints, such as respiratory symptoms reflect a national average. Asthma prevalence rates reflect recent increases in baseline asthma rates (i.e., 2008).	Tier 1 (qualitative) (No quantitative method available)
Uncertainties Associated with Omitted Benefits Categories				
	Underestimate	Medium-High	Low	Tier 2 (sensitivity analysis)

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Analytical Approach	Uncertainty Quantification
Unquantified ozone health benefit categories, such as worker productivity and long-term mortality	EPA has not included monetized estimates of these benefits categories in the core benefits estimate.	Although the potential magnitude is unknown, including all of the additional endpoints associated with ozone exposure that are currently not monetized could increase the total benefits by a large amount.	Current data and methods are insufficient to value national quantitative estimates of these health effects. The O_3 ISA determined that respiratory effects (including mortality) are causally associated with long-term ozone exposure (p. 2–17). The O_3 ISA also determined that outdoor workers have an increased risk of ozone-related health effects (p. 1-15), and that studies on outdoor workers show consistent evidence that short-term increases in ambient ozone exposure can decrease lung function in healthy adults (p.6-38). Additional studies link short-term ozone exposure to reduced productivity in outdoor workers (Graf Zivin and Neidell, 2013; Crocker and Horst, 1981).	We include sensitivity analyses reflecting long-term mortality which shows that this endpoint could add substantially to the total core benefits range (see Tables 5-20 and 5-27). We are still considering options for updating the worker productivity analysis and including it as a sensitivity analyses.
Uncertainties Associated with Estimated Exposure Changes				
Spatial matching of air quality estimates from epidemiology studies to air quality estimates from air quality modeling	Unknown Epidemiology studies often assume one air quality concentration is representative of an entire urban area when calculating hazard ratios, while benefits are calculated using air quality modeling conducted at 12 km spatial resolution. This spatial mismatch could introduce uncertainty.	Unknown	Low We have not controlled for this potential bias, and the SAB has not suggested an approach to adjust for this bias.	Tier 1 (qualitative) (No quantitative method available)
Uncertainties Associated with the Dollar-per-ton Approach Used in Modeling $PM_{2.5}$ Co-benefits				
	Unknown	Unknown	Medium	Tier 1 (qualitative)

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Analytical Approach	Uncertainty Quantification
Derivation of dollar-per-ton estimates for $PM_{2.5}$	In the analysis used to generate the dollar-per-ton values we assume that all fine particles, regardless of their chemical composition, are equally potent in causing premature mortality. However, the scientific evidence is not yet sufficient to allow differentiation of effect estimates by particle type. We also assume that the health impact function for fine particles is 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 regions that are in attainment with the fine particle standard.		While there are concerns regarding the assumption of a uniform $PM_{2.5}$ toxicity across sectors and a linear C-R function all the way down to zero, these sources of uncertainty impact benefits modeling for $PM_{2.5}$ in general and have been discussed as part of earlier RIAs (see section 5.7.2, of the final PM RIA, U.S. EPA. 2012). We do recognize that, as discussed below (and on pp. 24-25 of US. EPA, 2013), there is increased uncertainty when dollar-per-ton benefits are applied outside of the specific scenario used in their derivation.	(No quantitative method available)
Application of dollar-per-ton estimates in the current Ozone NAAQS review	Unknown	Unknown	Medium	
	As discussed in section 5.4.4, we used a method to calculate the regional benefit-per-ton estimates that is a slightly modified version of the national benefit-per-ton estimates described in the TSD: Estimating the Benefit per Ton of Reducing $PM_{2.5}$ Precursors from 17 Sectors (U.S. EPA, 2013b). The national estimates were derived using the approach published in Fann et al. (2012c), but they have since been updated to reflect the epidemiology studies and Census population data first applied in the final PM NAAQS RIA (U.S. EPA, 2012). These dollar-per-ton estimates were applied to sector-specific NOx emissions reductions modeled as part of the current ozone NAAQS review.	While we acknowledge uncertainty associated with applying dollar-per-ton estimates in the context of this ozone NAAQS review (and outside of the scenario in which they were derived), we are not in a position to characterize the magnitude or direction of any bias that might result from that application.	All benefit-per-ton estimates have inherent limitations, including that the estimates reflect the geographic distribution of the modeled sector emissions, which may not match the emissions reductions anticipated by the proposed standards, and they may not reflect local variability in population density, meteorology, exposure, baseline health incidence rates, or other local factors for any specific locations reflected in benefits modeling. However, the fact that we are modeling regional/national benefits rather than attempting a more spatially refined application of the dollar-per-ton values does reduce uncertainty in the benefits estimates that are generated.	(No quantitative method available)

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APPENDIX 5B: ADDITIONAL SENSITIVITY ANALYSES RELATED TO THE OZONE HEALTH BENEFITS ANALYSIS

Overview

The benefits analysis presented in Chapter 5 of this RIA is based on our current interpretation of the scientific and economic literature. That interpretation requires judgments regarding the best available data, models, and analytical methodologies and the assumptions that are most appropriate to adopt in the face of important uncertainties. The majority of the analytical assumptions used to develop the main estimates of benefits have been reviewed and supported by EPA's independent Science Advisory Board (SAB). Both EPA and the SAB recognize that data and modeling limitations as well as simplifying assumptions can introduce significant uncertainty into the estimates of benefits and that alternative choices exist for some inputs to the analysis, such as the concentration-response functions for mortality.

This appendix assesses the sensitivity of the core benefits to: (a) the potential impact of thresholds in long-term ozone exposure-related mortality in incidence and benefits estimates (section 5B.2), (b) alternative response functions developed through expert elicitation (EE) for long-term $PM_{2.5}$ exposure-related mortality (section 5B.3), and (c) alternative assumptions regarding income elasticity on benefits derived using willingness-to-pay (WTP) functions (section 5B.3).

For the core analysis, we estimated incidence and dollar benefits for two scenarios: 2025 and post-2025 (see Chapter 5, Sections 5.2 and 5.4.3). However, in conducting these sensitivity analyses, we used the 2025 scenario as the basis for making our calculations, since sensitivity analysis findings for this scenario would generally hold for the post-2025 scenario.

In addition to the three sensitivity analyses covered in this appendix, we also included two sensitivity analyses that are covered in detail in Chapter 5. The first of these are estimates of dollar benefits associated with mortality resulting from long-term exposure to ozone. As discussed in Section 5.2, while we felt that we had sufficient confidence to include incidence estimates associated with long-term ozone exposure (based on effect estimates obtained from Jerrett et al., 2009) in the core analysis, limitations in our ability to specify an appropriate lag for reductions in this endpoint meant that we could not include dollar benefit estimates in the core analysis. Instead, we have included those values as sensitivity analyses, including consideration of alternative lag structures (they are presented in Tables 5-20 and 5-27, respectively for the 2025 and post-2025 scenarios).[77] The second of the sensitivity analyses already covered in the chapter addresses alternative models for short-term ozone exposure-related mortality. As discussed in Section 5.6.3.1, in addition to the two epidemiological studies (Smith et al., 2009 and Zanobetti and Schwartz 2008) that provide effect estimates for the core benefits estimates, we have also considered seven additional epidemiological studies as sensitivity analyses. These additional studies include a mix of multi-city and meta-analysis designs. Results of this sensitivity analysis are presented in Tables 5-19 and 5-20 (incidence and benefits, respectively for the 2025 scenario) and Tables 5-26 and 5-27 (incidence and benefits, respectively for the post-2025 scenario).

5B.1 Threshold Sensitivity Analysis for Premature Mortality Incidence and Benefits from Long-term Exposure to Ozone

In estimating long-term ozone mortality, we employed a continuous non-threshold concentration-response (C-R) function relating ozone exposure to premature death. However, as discussed in Section 5.6.3.1, there is uncertainty regarding the potential existence and location of a threshold in the C-R function relating mortality and long-term ozone concentrations. Thus, we have included a sensitivity analysis exploring the impact of potential thresholds in the C-R relationship on estimates of long-term exposure-related mortality that were evaluated in Jerrett et al. (2009), consistent with advice from CASAC (Frey, 2014).

In their memo clarifying the results of their study (Sasser, 2014), the authors note that in terms of goodness of fit, long-term health risk models including ozone clearly performed better than models without ozone, indicating the improved predictions of respiratory mortality when ozone is included. In exploring different functional forms, the authors report that the model including a threshold at 56 ppb had the lowest log-likelihood value of all models evaluated (i.e.,

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[77] The sensitivity analysis-related benefits estimates presented in these tables include (a) benefits estimates reflecting application of a zero lag model and (b) estimates reflecting application of the same 20-year segmented lag used in modeling benefits for $PM_{2.5}$, together with application of a 3% and 7% discount rate. See Sections 5.6.3 and 5.6.4 for additional discussion of lags in relation to long-term ozone-related mortality.

linear models and models including thresholds ranging from 40-60 ppb), and thus provided the best overall statistical fit to the data. However, they also note that it is not clear whether the 56 ppb threshold model is a better predictor of respiratory mortality than when using a linear (no-threshold) model for this dataset. Using one statistical test, the model with a threshold at 56 ppb was determined to be statistically superior to the linear model. Using another, more stringent test, none of the threshold models considered were statistically superior to the linear model. Under the less stringent test, although the threshold model produces a statistically superior prediction than the linear model, there is uncertainty about the specific location of the threshold, if one exists. This is because the confidence intervals on the model predictions indicate that a threshold could exist anywhere from 0 to 60 ppb. The authors conclude that considerable caution should be exercised in using any specific threshold, particularly when the more stringent statistical test indicates there is no significantly improved prediction. Based on this additional information from the authors, we have chosen to reflect the uncertainty about the existence and location of a potential threshold by estimating mortality attributable to long-term ozone exposures using a range of threshold-based effect estimates as sensitivity analyses. Specifically, we estimate long-term ozone mortality benefits using unique risk coefficients that include a range of thresholds from 40 ppb to 60 ppb in 5 ppb increments, while also including a model with a threshold equal to 56 ppb, which had the lowest log-likelihood value for all models examined.[78] Table 5B-1 provides the results of these sensitivity analyses (based on modeling incidence) for 60 ppb, 65 ppb and 70 ppb. We note that the same pattern in terms of relative reductions across thresholds (relative to the core estimate) would hold for the dollar benefit estimates generated for this endpoint.

[78] There is a separate effect estimate (and associated standard error) for each of the fitted threshold models estimated in Jerrett et al. (2009). As a result, the sensitivity of estimated mortality attributable to long-term ozone concentrations is affected by both the assumed threshold level (below which there is no effect of ozone) and the effect estimate applied to ozone concentrations above the threshold.

Table 5B-1. Long-term Ozone Mortality Incidence at Various Assumed Thresholds [a]

Threshold Concentration	70 ppb	65 ppb	60 ppb
No threshold (core model)	680	2,100	3,900
40 ppb	520	1,600	2,900
45 ppb	410	1,100	1,700
50 ppb	120	280	510
55 ppb	5.6	58	140
56 ppb	3.3	47	105
60 ppb	<1	10	13

[a] All estimates rounded to two significant digits.

The results of the sensitivity analysis based on the suite of threshold-based risk coefficients suggests that threshold models can result in substantially lower estimates of ozone-attributable long-term mortality. For example, estimated incidence and dollar benefits for long-term mortality using a model that includes a 55 ppb threshold are approximately 70% less than long-term mortality benefits estimated using the core co-pollutant non-threshold model. Generally, estimated long-term mortality benefits are progressively reduced when using models with increasing thresholds, with the highest threshold considered (60 ppb) removing virtually all of the estimated incidence reduction and associated benefits.

5B.2 Alternative Concentration-Response Functions for $PM_{2.5}$–Related Mortality

In modeling $PM_{2.5}$ cobenefits, we estimate that total dollar benefits are driven largely by reductions in mortality (see Table 5-22 and 5-29). Therefore, it is particularly important to attempt to characterize the uncertainties associated with reductions in premature mortality as modeled in the $PM_{2.5}$ cobenefits analysis. To better understand the concentration-response relationship between $PM_{2.5}$ exposure and premature mortality, the EPA conducted an expert elicitation in 2006 (Roman et al., 2008; IEc, 2006).[79] 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. In this analysis, we present the results derived from the expert elicitation as

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

indicative of the uncertainty associated with a major component of the health impact functions, and we provide the independent estimates derived from each of the twelve experts to better characterize the degree of variability in the expert responses.

In previous RIAs, the EPA presented benefits estimates using concentration-response functions derived from the $PM_{2.5}$ Expert Elicitation (Roman et al., 2008) as a range from the lowest expert value (Expert K) to the highest expert value (Expert E). However, this approach did not indicate the Agency's judgment on what the best estimate of $PM_{2.5}$ benefits may be, and the EPA's independent SAB recommended refinements to the way EPA presented the results of the elicitation (U.S. EPA-SAB, 2008). Therefore, we began to present the cohort-based studies (Krewski et al., 2009; Laden et al., 2006)[80] 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 of other recent multi-state cohort studies conducted in North America, but we have not estimated benefits using the effect coefficients from these studies (see Appendix 5D). Please note that the benefits estimates results presented are not the direct results from the studies or expert elicitation; rather, the estimates are based in part on the effect coefficients provided in those studies or by experts. In addition, because we are using a dollar-per-ton approach in modeling $PM_{2.5}$ cobenefits in this RIA, we cannot generate confidence intervals reflecting the statistical fit characterized in the expert elicitation-based functions.

Even these multiple characterizations based on application of the range of expert elicitation-based effect estimates 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

[80] We have since updated the Harvard Six Cities cohort study from Laden et al. (2006) to use the most recent follow-up publication of this cohort (Lepeule et al, 2012). This study is reflected in the dollar-per-ton values used in this RIA.

parameters and the identification of reasonable upper and lower bounds for input distributions characterizing uncertainty in additional model elements.

The $PM_{2.5}$ expert elicitation and the derivation of effect estimates from the expert elicitation results (used in generating these alternative dollar-per-ton estimates) are described in detail in the 2006 $PM_{2.5}$ NAAQS RIA (U.S. EPA, 2006), the elicitation summary report (IEc, 2006) and Roman et al. (2008), and consequently, we do not present those effect estimates (and associated functional forms) here.

Table 5B-2 presents the results of this sensitivity analysis as completed for the 2025 scenario (overall conclusions generated from this analysis are transferable to the post-2025 scenario). The alternative mortality estimates presented in Table 5B-2 were generated similar to the core cobenefits $PM_{2.5}$ mortality estimates, but applying dollar-per-ton values (for each expert elicitation-based effect estimate) to the sector-level estimates of NOx reductions associated with the 2025 scenario. We have also included the core cobenefits estimates for each alternative standard to facilitate comparison against these alternative sensitivity analysis estimates. Because application of these effect estimates (using a dollar-per-ton approach) represents a linear calculation, results in terms of the relative ranking of the core estimates compared with expert elicitation estimates will remain the same for all alternative standards evaluated. Therefore, we only present estimates for the 70 ppb alternative standard, observing that observations drawn from this sensitivity analysis would hold for other alternative standards considered (and for the post-2025 scenario – for all alternative standards – as well).

Table 5B-2. Application of Alternative (Expert Elicitation-Based Effect Estimates) to the Modeling of $PM_{2.5}$ Co-benefit Estimates for $PM_{2.5}$ (avoided incidence)

C-R Function (and effect estimate)[b]	70 ppb
Krewski et al., (2012) (core model)	**45**
Lepeule et al., (2012) (core model)	**100**
Expert K	10
Expert G	54
Expert L	63
Expert D	65
Expert H	67
Expert J	75
Expert F	88
Expert C	92
Expert I	92
Expert B	95
Expert A	120
Expert E	150

[a] All estimates rounded to two significant digits.
[b] Expert elicitation-based values ordered by magnitude of incidence reduction

The values presented in Table 5B-2 suggest that the two core incidence estimate fall within the range of alternative C-R function based estimates obtained through expert elicitation. This increases overall confidence in the core estimates with regard to the form of the functions and magnitude of the effect estimates. We would expect the relationship between the core estimates and the expert-derived estimates to remain constant for the remaining scenarios as well.

5B.3 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. Income growth projections are only currently available in BenMAP through 2024, so both the 2025 and post-2025 scenario estimates use income growth only through 2024 and are therefore likely underestimates.

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

Table 5B-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[b]	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.
[b] Minor health effects included in this RIA and valued using WTP-based functions include: upper and lower respiratory symptoms, asthma exacerbations, minor restricted activity days, and acute bronchitis.

Table 5B-4. Ranges of Adjustment Factors Used to Account for Projected Real Income Growth to 2024 [a]

Benefit Category	Lower Sensitivity Bound	Upper Sensitivity Bound
Minor Health Effect[b]	1.021	1.170
Premature Mortality	1.043	1.705

[a] Based on elasticity values reported in Table C-4, U.S. Census population projections, and projections of real GDP per capita.
[b] Minor health effects included in this RIA and valued using WTP-based functions include: upper and lower respiratory symptoms, asthma exacerbations, minor restricted activity days, and acute bronchitis.

Table 5B-5. Sensitivity of Monetized Ozone Benefits to Alternative Income Elasticities in 2025 (Millions of 2011$) [a]

Benefit Category	No adjustment		Lower Sensitivity Bound		Upper Sensitivity Bound	
	70 ppb	65 ppb	70 ppb	65 ppb	70 ppb	65 ppb
Minor Health Effect [b]	$66	$200	$67	$210	$77	$240
Premature Mortality [c]	$2,000	$6,400	$2,100	$6,600	$3,500	$11,000

[a] All estimates rounded to two significant digits. Only reflects income growth to 2024.
[b] For purposes of completing this sensitivity analysis, we have included minor restricted activity days (MRADS) based resulting from short-term ozone exposure as the minor health effect evaluated here.
[c] Using short-term mortality effect estimate from Smith et al. (2009) and 3% discount rate. Results using other short-term mortality studies and 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 2025 ranges from 86% to 133% of the main estimate for mortality based on the lower and upper sensitivity bounds on the mortality income adjustment factor. The effect on the value of minor health effects is much less pronounced, ranging from 96% to 108% of the main estimate for minor effects. These observations (in terms of relative impact from alternative elasticities) hold for all three of the alternative standard levels evaluated under both the 2025 and post-205 scenarios.

5B.4 References

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APPENDIX 5C: SUPPLEMENTAL ANALYSES RELATED TO THE OZONE HEALTH BENEFITS ANALYSIS

Overview

A number of additional analyses have been completed to supplement the core estimates generated for the RIA. These analyses give greater insight to the manner in which these impacts are distributed among populations of different ages, the ambient levels of ozone at which the avoided deaths are estimated to occur, and the benefits attributable to the known emissions control measures. These supplemental analyses, which are presented in detail here (and summarized in Section 5.7.3.2) include: (a) age group-differentiated aspects of short-term ozone exposure-related mortality (including total avoided incidence, life years gained and percent reduction in baseline mortality - Section 5C.1), (b) evaluation of mortality impacts relative to the baseline pollutant concentrations (used in generating those mortality estimates) for both short-term ozone exposure-related mortality and long-term $PM_{2.5}$ exposure-related mortality (Section 5C.2)[81] and (c) presentation of core incidence and benefits estimates reflecting application of known controls for the 2025 scenario (Section 5C.3).

5C.1 Age Group-Differentiated Aspects of Short-Term Ozone Exposure-Related Mortality

In their 2008 review of the EPA's approach to estimating ozone-related mortality benefits, NRC indicated, "EPA should consider placing greater emphasis on reporting decreases in age-specific death rates in the relevant population and develop models for consistent calculation of changes in life expectancy and changes in number of deaths at all ages" (NRC, 2008). In addition, NRC noted in an earlier report that "[f]rom a public-health perspective, life-years lost might be more relevant than annual number of mortality cases" (NRC, 2002). This advice is consistent with that of the Health Effects Subcommittee of the Advisory Council on Clean Air Compliance Analysis (SAB-HES), which agreed that "…the interpretation of mortality risk results is enhanced if estimates of lost life-years can be made" (U.S. EPA-SAB, 2004a). To address these recommendations, we use simplifying assumptions to estimate the number of life

[81] The plot of long-term $PM_{2.5}$ exposure-related mortality incidence versus $PM_{2.5}$ levels is taken from previous RIAs and reflects the benefits simulation used to generate the dollar-per-ton estimates used in deriving $PM_{2.5}$-related cobenefits for this analysis (i.e., these plots are not derived using new data specific to this RIA) (see Section 5C.3).

years that might be gained. We also estimate the reduction in the percentage of deaths attributed to ozone resulting from the illustrative emissions reduction strategies to reach the proposed and alternative primary standards. The EPA included similar estimates of life years gained in a previous assessment of ozone and/or $PM_{2.5}$ benefits (U.S. EPA, 2006, 2010c, 2011b), the latter of which was peer reviewed by the SAB-HES (U.S. EPA-SAB, 2010a).

Changes in life years and changes in life expectancy at birth are frequently conflated, thus it is important to distinguish these two very different metrics. Life expectancy varies by age. The Centers for Disease Control and Prevention (CDC) defines life expectancy as the "average number of years of life remaining for persons who have attained a given age" (CDC, 2011). In other words, changes in life expectancy refer to an average change for the entire population, and refer to the future. Over the past 50 years, average life expectancy at birth in the U.S. has increased by 8.4 years (CDC, 2001). For example, life expectancy at birth was estimated in 2007 to be 77.9 years for an average person born in the U.S., but for people surviving to age 60, estimated life expectancy is 82.5 years (i.e., 4.6 years more than life expectancy at birth) (CDC, 2011). Life years, on the other hand, measure the amount of time that an individual loses if they die before the age of their life expectancy. Life years refer to individuals, and refer to the past, e.g., when the individual has already died. If a 60-year old individual dies, we estimate that this individual would lose about 22.5 years of life (i.e., the average population life expectancy for an individual of this age minus this person's age at death).

Due to the use of benefit-per-ton estimates for the $PM_{2.5}$ co-benefits, we are unable to estimate the life years gained by reducing exposure to $PM_{2.5}$ in this analysis. Instead, we refer the reader to the 2012 PM NAAQs RIA for more information about the avoided life years lost from $PM_{2.5}$ exposure (U.S. EPA, 2012b). This analysis found that about half of the avoided PM-related deaths occur in populations age 75 to 99, but half of the avoided life years lost would occur in populations younger than 65 because the younger populations have the potential to lose more life years per death than older populations. In addition, this analysis found that the average individual who would otherwise have died prematurely from PM exposure would gain 16 additional years of life.

Estimated Life Years Gained

For estimating the potential life years gained by reducing exposure to ozone in the U.S. adult population, we use the same general approach as Hubbell (2006) and Fann et al. (2012a). We have not estimated the change in average life expectancy at birth in this RIA. Because life expectancy is an average of the entire population (including both those whose deaths would likely be attributed to air pollution exposure as well as those whose deaths would not), average life expectancy changes associated with air pollution exposure would be expected to always be significantly smaller than the average number of life years lost by an individual who is projected to die prematurely from air pollution exposure.

To estimate the potential distribution of life years gained for population subgroups defined by the age range at which their reduction in air pollution exposure is modeled to occur, we use standard life tables available from the CDC (2014) and the following formula:

$$Total\ Life\ Years = \sum_{i=1}^{n} LE_i \times M_i \qquad (5.2)$$

where LE_i is the average remaining life expectancy for age interval i, M_i is the estimated change in number of deaths in age interval i, and n is the number of age intervals.

To get M_i (the estimated number of avoided premature deaths attributed to changes in ozone exposure for the 2025 scenario), we use a health impact function that incorporates risk coefficients estimated for the adult population in the U.S. and age-specific mortality rates. That is, we use risk coefficients that do not vary by age, but use baseline mortality rates that do. Because mortality rates for younger populations are much lower than mortality rates for older populations, most but not all, of the avoided deaths tend to be in older populations. Table 5C-1 summarizes the number of avoided deaths (by age range) attributable to ozone for each alternative standard for the 2025 scenario. Table 5C-2 summarizes the modeled number of life years gained (for each age range) by reducing ozone for each alternative standard evaluated for the 2025 scenario. We then calculated the average number of life years gained per avoided premature mortality.

Table 5C-1. Potential Reduction in Premature Mortality by Age Range from Attaining Alternate Ozone Standards (2025 scenario) [a, b]

Age Range [b]	Standard Alternative		
	70 ppb	65 ppb	60 ppb
0-4	0.65	1.9	3.6
5-9	0.14	0.44	0.81
10-14	0.15	0.46	0.85
15-19	0.21	0.65	1.2
20-24	0.3	0.9	1.7
25–29	0.61	1.8	3.4
30–34	0.63	1.9	3.5
35–44	3.4	11	19
45–54	8.6	27	49
55–64	22	67	120
65–74	45	140	260
75–84	60	190	340
85–99	59	190	340
Total ozone-attributable mortality	200	630	1,100

[a] Estimates rounded to two significant figures.
[b] Effects calculated using the core Smith et al. (2009) effect estimate for the 2025 scenario

Table 5C-2. Potential Years of Life Gained by Age Range from Attaining Alternate Ozone Standards (2025 Scenario) [a,b]

Age Range [b]	Standard Alternative		
	70 ppb	65 ppb	60 ppb
0-4	51	150	280
5-9	10	30	56
10-14	10	30	55
15-19	13	39	71
20-24	16	49	90
25–29	30	91	170
30–34	28	86	160
35–44	120	380	690
45–54	230	720	1,300
55–64	410	1,300	2,300
65–74	550	1,700	3,100
75–84	390	1,200	2,300
85–99	130	430	790
Total life years gained	2,000	6,200	11,000
Average life years gained per individual	10.0	9.93	9.92

[a] Estimates rounded to two significant figures (except for average life years gained – presented to three significant figures to allow differences across values to be evident.
[b] Effects calculated using the core Smith et al. (2009) effect estimate for the 2025 scenario

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 (see Table 5C-1), but half of the life years would occur in populations younger than

65 (see Table 5C-2). This is because the younger populations have the potential to lose more life years per death than older populations based on changes in ozone exposure for the 2025 scenario. We estimate that the average individual who would otherwise have died prematurely from ozone exposure would gain 10 additional years of life. However, this approach does not account for whether or not people who are older are more likely to be susceptible to the health effects of air pollution or whether that susceptibility was in and of itself caused by air pollution exposure (for a more complete discussion of this issue, see Kunzli et al., 2001).

Percent of Ozone-related Mortality Reduced

To estimate the percentage reduction in all-cause mortality attributed to reduced ozone exposure for the 2025 scenario as a result of the illustrative emissions reduction strategies, we use M_i from the equation above, dividing the number of excess deaths estimated for each alternative standard by the total number of deaths in each county. Table 5C-3 shows the reduction in all-cause mortality attributed to reducing ozone exposure to the proposed primary standards for the 2025 scenario.

Table 5C-3. Estimated Percent Reduction in All-Cause Mortality Attributed to the Proposed Primary Ozone Standards (2025 Scenario) [a]

Age Range [b]	Standard Alternative		
	70 ppb	65 ppb	60 ppb
0-4	0.0183%	0.0547%	0.101%
5-9	0.0176%	0.0538%	0.100%
10-14	0.0178%	0.0542%	0.100%
15-19	0.0180%	0.0546%	0.101%
20-24	0.0177%	0.0536%	0.099%
25–29	0.0177%	0.0535%	0.099%
30–34	0.0176%	0.0535%	0.098%
35–44	0.0174%	0.0535%	0.099%
45–54	0.0174%	0.0536%	0.099%
55–64	0.0179%	0.0554%	0.101%
65–74	0.0179%	0.0557%	0.102%
75–84	0.0176%	0.0549%	0.101%
85–99	0.0164%	0.0522%	0.096%

[a] In order to illustrate the slight variations in percent reductions across age ranges (for a given alternative standard level) we have presented results to three significant figures (rather than two as is typically done for other estimates in this RIA).

Results presented in Table 5C-3 highlight that when reductions in ozone-attributable mortality (in going from baseline to an alternative standard level) are considered as a percentage of total all-cause baseline mortality, the estimates are relatively small and are fairly constant across age ranges. However, it is important to point out that estimates of total ozone-attributable mortality represent a substantially larger fraction of all-cause baseline mortality.

5C.2 Evaluation of Mortality Impacts Relative to the Baseline Pollutant Concentrations (used in generating those mortality estimates) for both Short-Term Ozone Exposure-Related Mortality and Long-Term $PM_{2.5}$ Exposure-Related Mortality

Analysis of baseline ozone levels used in modeling short-term ozone exposure-related mortality

Our review of the current body of scientific literature indicates that a log-linear no-threshold model provides the best estimate of ozone-related short-term mortality (see section 2.5.4.4, in the O_3 ISA, U.S. EPA, 2013a), which was reviewed by the EPA's Clean Air Scientific Advisory Committee. Consistent with this finding, we estimate benefits associated with the full range of ozone exposure. Our confidence in the estimated number of premature deaths avoided (but not in the existence of a causal relationship between ozone and premature mortality) diminishes as we estimate these impacts at successively lower concentrations. However, there are uncertainties inherent in identifying any particular point at which our confidence in reported associations becomes appreciably less, and the scientific evidence provides no clear dividing line. The O_3 ISA noted that the studies indicate reduced certainty in specifying the shape of the C-R function specifically for short-term ozone-attributable respiratory morbidity and mortality, in the range generally below 20 ppb (for these reasons, the ≤ 20 ppb range discussed in the O_3 ISA should be viewed as a more generalized range to be considered qualitatively or semi-quantitatively, along with many other factors, when interpreting the risk estimates rather than as a fixed, bright-line).[82]

[82] While clinical studies have suggested the presence of a threshold for respiratory effects, these should not be used to support specification of population-level thresholds for use in the epidemiological-based risk assessment focusing on short-term exposure-related endpoints. The clinical studies focus on relatively small and clearly defined populations of healthy adults, which are not representative of the broader residential populations typically associated with epidemiological studies, including older individuals and individuals with existing health conditions that place them at greater risk for ozone-related effects. Therefore, the clinical studies are unlikely to have the power to capture population thresholds in a broader and more diverse urban residential population, should those thresholds exist.

Figures 5C-1 and 5C-2 compare the distribution of short-term ozone exposure-related mortality to the underlying distribution of 8hr max baseline ozone levels used in generating those estimates (these two plots present probability and cumulative probability plots, respectively). Both figures are based on the core estimate of short-term mortality generated using effect estimates obtained from Smith et al., 2009.[83] In addition, each figure includes separate plots for the three alternative standard levels being analyzed (with all being based on the 2025 scenario). If we look at Figure 5C-1, we see that approximately 45% of the mortality benefit estimated for the 65 ppb alternative standard is associated with days having baseline ozone level of between 40 and 45 ppb.[84]

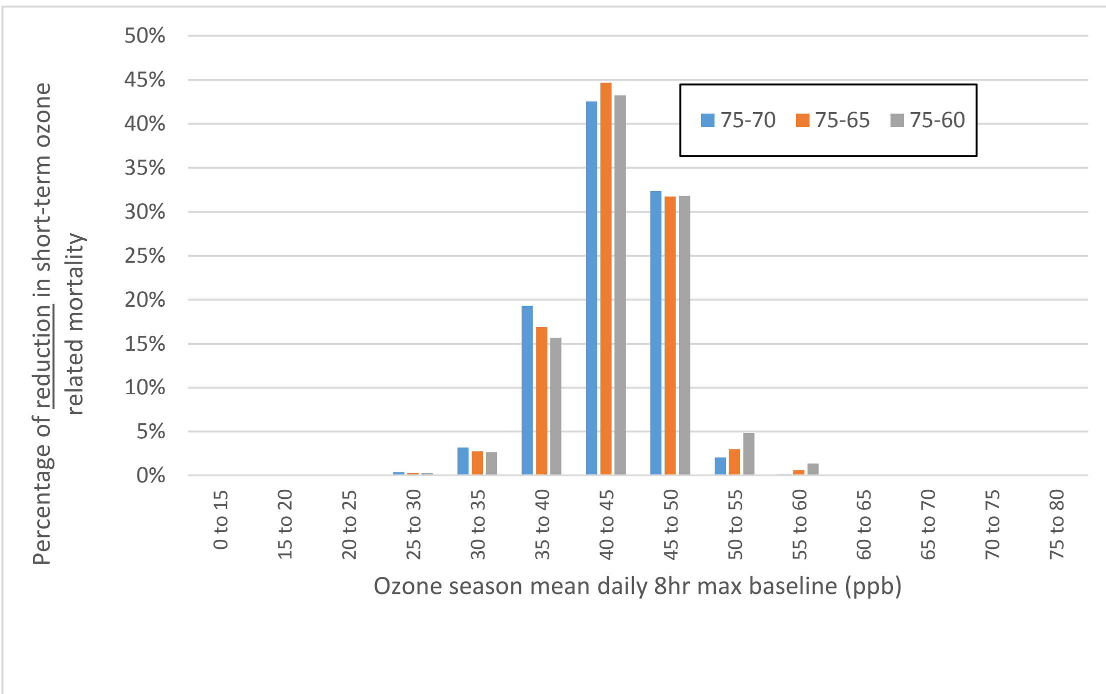


Figure 5C-1. Premature Ozone-related Deaths Avoided for the Alternative Standards (2025 scenario) According to the Baseline Ozone Concentrations

[83] The set of 12km-level mortality estimates (and associated 8hr max baseline values) generated using BenMAP forms the basis for the plots.

[84] As noted later in this section, this baseline range is actually for the mean across the ozone season of 8hr max values within a given grid cell, so the actual distribution of baseline 8hr max values associated with this segment of benefits reductions is likely wider than the 40-45 ppb range.

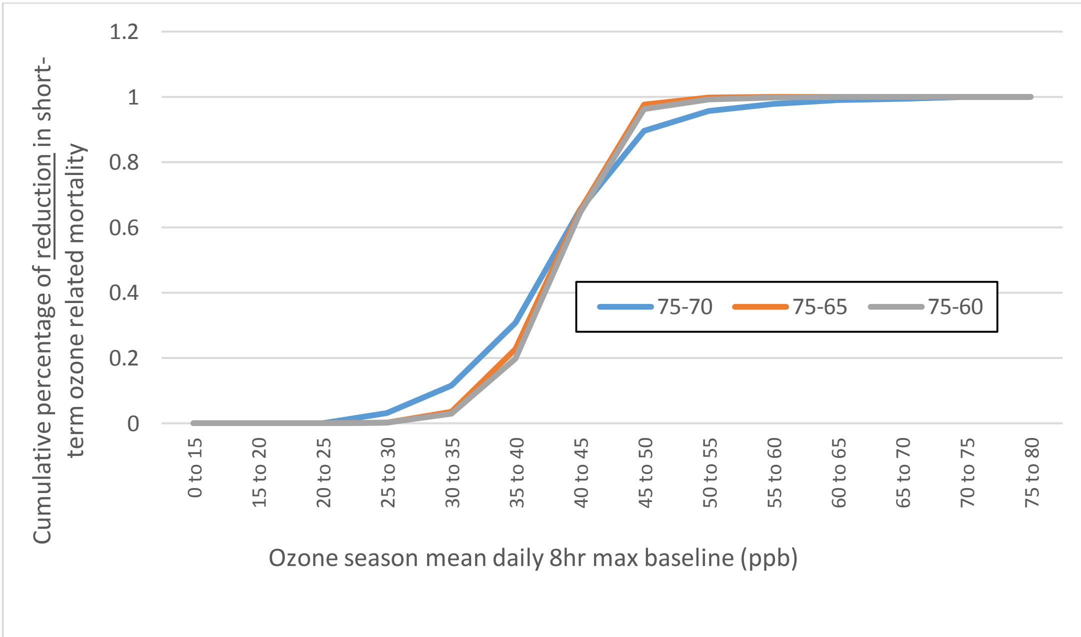


Figure 5C-2. Cumulative Probability Plot of Premature Ozone-related Deaths Avoided for the Alternative Standards (2025 scenario) According to the Baseline Ozone Concentrations

When interpreting these results, it is important to understand that the avoided ozone-related deaths are estimated to occur from ozone reductions in the baseline air quality simulation, which assumes that 75 ppb is already met. When simulating attainment with proposed and alternative standards, we adjust the design value at each monitor exceeding the standard alternative to equal that standard and use an air quality interpolation technique to simulate the change in ozone concentrations surrounding that monitor. This technique tends to simulate the greatest air quality changes nearest the monitor. We estimate benefits using modeled air quality data with 12 km grid cells, which is important because the grid cells are often substantially smaller than counties and ozone concentrations vary spatially within a county. Therefore, there may be a small number of grid cells with concentrations slightly greater than 75 ppb in the gridded baseline even though all monitors could meet an annual standard of 75 ppb. In addition, some grid cells in a county can be below the level of a standard even though the highest monitor value is above that standard. Thus, emissions reductions can lead to benefits in grid cells that are below a standard even within a county with a monitor that exceeds that standard. Furthermore, our approach to simulating attainment can lead to benefits in counties that are below the alternative standard being evaluated. Emissions reduction strategies designed to reduce ozone concentrations at a given monitor will frequently improve air quality in neighboring counties. In order to make a direct comparison between the benefits and costs of these emissions reduction strategies, it is appropriate to include all the benefits occurring as a result of the emissions reduction strategies applied, regardless of where they occur. Therefore, it is not appropriate to estimate the fraction of benefits that occur only in counties that exceed the alternative standards because it would omit benefits attributable to emissions reductions in exceeding counties.

One final caveat in interpreting the information presented in these figures is that in modeling this mortality endpoint, rather than using a true distribution of daily 8hr max ozone levels for each grid cell, due to resource limitations, we used a single mean value for the ozone season within each grid cell. While this will generate the same total ozone benefit estimate for each grid cell compared with application of a full distribution of daily 8hr max values, use of a mean daily value means that an assessment such as this one that considers both the spatial and temporal association between mortality benefit estimates and ozone levels, will be limited somewhat in its treatment of the temporal dimension.

Consideration for the plots presented in Figures 5C-1 and 5C-2 results in a number of observations. The vast majority of reductions in short-term exposure-related mortality for ozone occur in grid cells with mean 8hr max baseline levels (across the ozone season) between 35 and 55ppb. Importantly, virtually all of the mortality reductions are associated with ozone levels above the <20ppb range identified within the O_3 ISA as being associated with less confidence in specifying the nature of the C-R function for ozone mortality (O_3 ISA, section 2.5.4.4).[85] We also note that as we compare patterns across the three alternative standard levels, we see that, as expected, the upper end of the distribution is being shifted downwards as increasingly lower standard levels are analyzed (see Figure 5C-2).

[85] As noted earlier, care needs to be taken in interpreting these mortality vs. ozone air level distributions in the context of the range of reduced confidence (<20ppb) identified in the O_3 ISA. The region of reduced confidence identified by the ISA reflects the composite monitor daily time series values (including 8hr max values) used in short-term mortality studies, while the ozone levels summarized in Figures 5C-1 and 5C-2 are the mean (across the ozone season) of daily 8hr max values within each grid cell. The use of these mean values, while not impacting the total mortality reductions estimated, could significantly reduce variability in the spread of values presented in these two figures.

Concentration Benchmark Analysis for $PM_{2.5}$ Benefit-per-ton Estimates

In general, we are more confident in the magnitude of the risks we estimate from simulated $PM_{2.5}$ concentrations that coincide with the bulk of the observed PM concentrations in the epidemiological studies that are used to estimate the benefits. Likewise, we are less confident in the risk we estimate from simulated $PM_{2.5}$ concentrations that fall below the bulk of the observed data in these studies. Concentration benchmark analyses (e.g., lowest measured level [LML], one standard deviation below the mean of the air quality data in the study, etc.) allow readers to determine the portion of population exposed to annual mean $PM_{2.5}$ levels at or above different concentrations, which provides some insight into the level of uncertainty in the estimated $PM_{2.5}$ mortality benefits. In this analysis, we apply two concentration benchmark approaches (LML and one standard deviation below the mean) that have been incorporated into recent RIAs and EPA's Policy Assessment for Particulate Matter (U.S. EPA, 2011d). There are uncertainties inherent in identifying any particular point at which our confidence in reported associations becomes appreciably less, and the scientific evidence provides no clear dividing line. However, the EPA does not view these concentration benchmarks as a concentration threshold below which we would not quantify health co-benefits of air quality improvements.[86] Rather, the co-benefits estimates reported in this RIA are the best estimates because they reflect the full range of air quality concentrations associated with the emissions reduction strategies. The PM ISA concluded that the scientific evidence collectively is sufficient to conclude that the relationship between long-term $PM_{2.5}$ exposures and mortality is causal and that overall the studies support the use of a no-threshold log-linear model to estimate PM-related long-term mortality (U.S. EPA, 2009b).

For this analysis, policy-specific air quality data is not available, and the compliance strategies are illustrative of what states may choose to do. For this RIA, we are unable to estimate the percentage of premature mortality associated with the emissions reductions at each $PM_{2.5}$ concentration, as we have done for previous rules with air quality modeling (e.g., U.S. EPA, 2011b, 2012a). However, we believe that it is still important to characterize the distribution of exposure to baseline concentrations. As a surrogate measure of mortality impacts, we provide

[86] For a summary of the scientific review statements regarding the lack of a threshold in the $PM_{2.5}$-mortality relationship, see the 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, 2010b).

the percentage of the population exposed at each $PM_{2.5}$ concentration in the baseline of the source apportionment modeling used to calculate the benefit-per-ton estimates for this sector using 12 km grid cells across the contiguous U.S.[87] It is important to note that baseline exposure is only one parameter in the health impact function, along with baseline incidence rates, population and change in air quality. In other words, the percentage of the population exposed to air pollution below the LML is not the same as the percentage of the population experiencing health impacts as a result of a specific emissions reduction policy. The most important aspect, which we are unable to quantify without rule-specific air quality modeling, is the shift in exposure anticipated by implementing the proposed standards. Therefore, caution is warranted when interpreting the LML assessment in this RIA because these results are not consistent with results from RIAs that had air quality modeling.

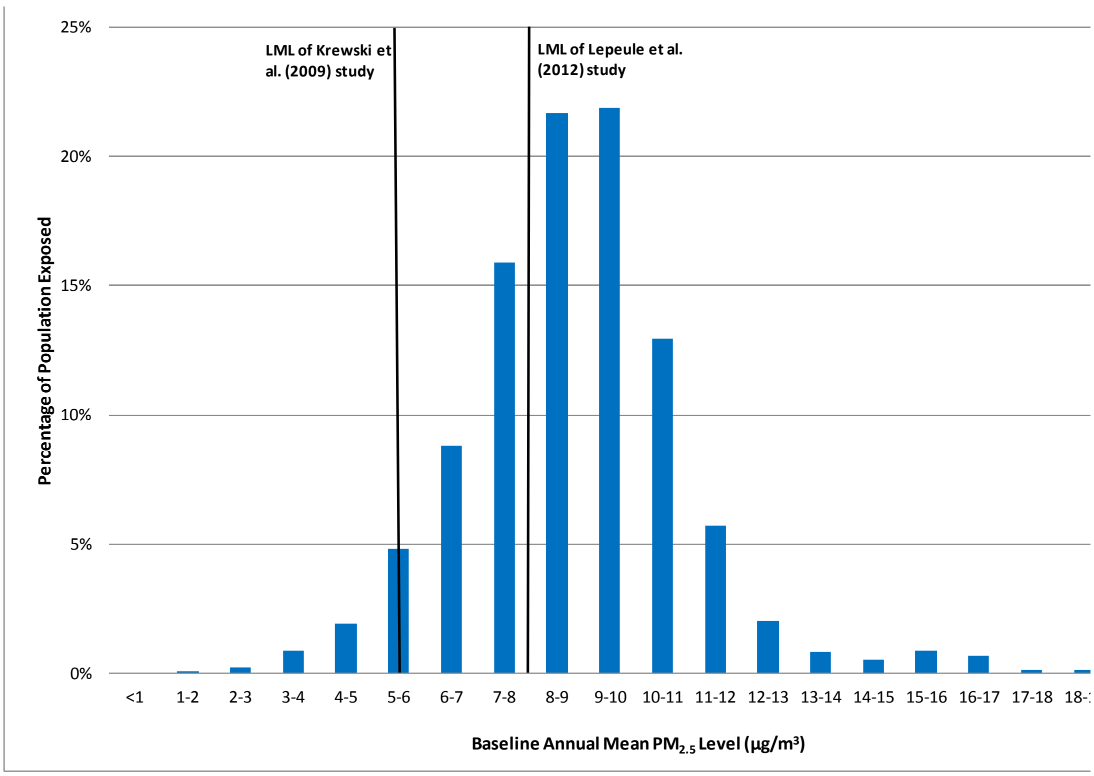
Table 5C-4 provides the percentage of the population exposed above and below two concentration benchmarks (i.e., LML and one standard deviation below the mean) in the modeled baseline for the sector modeling. Figure 5C-3 shows a bar chart of the percentage of the population exposed to various air quality levels in the baseline, and Figure 5C-4 shows a cumulative distribution function of the same data. Both figures identify the LML for each of the major cohort studies.

[87] As noted above, the modeling used to generate the benefit-per-ton estimates does not reflect emissions reductions anticipated from MATS rule. Therefore, the baseline $PM_{2.5}$ concentrations in the LML assessment are higher than would be expected if MATS was reflected.

Table 5C-4. Population Exposure in the Baseline Sector Modeling (used to generate the benefit-per-ton estimates) Above and Below Various Concentrations Benchmarks in the Underlying Epidemiology Studies [a]

Epidemiology Study	Below 1 Standard Deviation. Below AQ Mean	At or Above 1 Standard Deviation Below AQ Mean	Below LML	At or Above LML
Krewski et al. (2009)	89%	11%	7%	93%
Lepeule et al. (2012)	N/A	N/A	23%	67%

[a] One standard deviation below the mean is equivalent to the middle of the range between the 10th and 25th percentile. For Krewski, the LML is 5.8 µg/m³ and one standard deviation below the mean is 11.0 µg/m³. For Lepeule et al., the LML is 8 µg/m³ and we do not have the data for one standard deviation below the mean. It is important to emphasize that although we have lower levels of confidence in levels below the LML for each study, the scientific evidence does not support the existence of a level below which health effects from exposure to $PM_{2.5}$ do not occur.

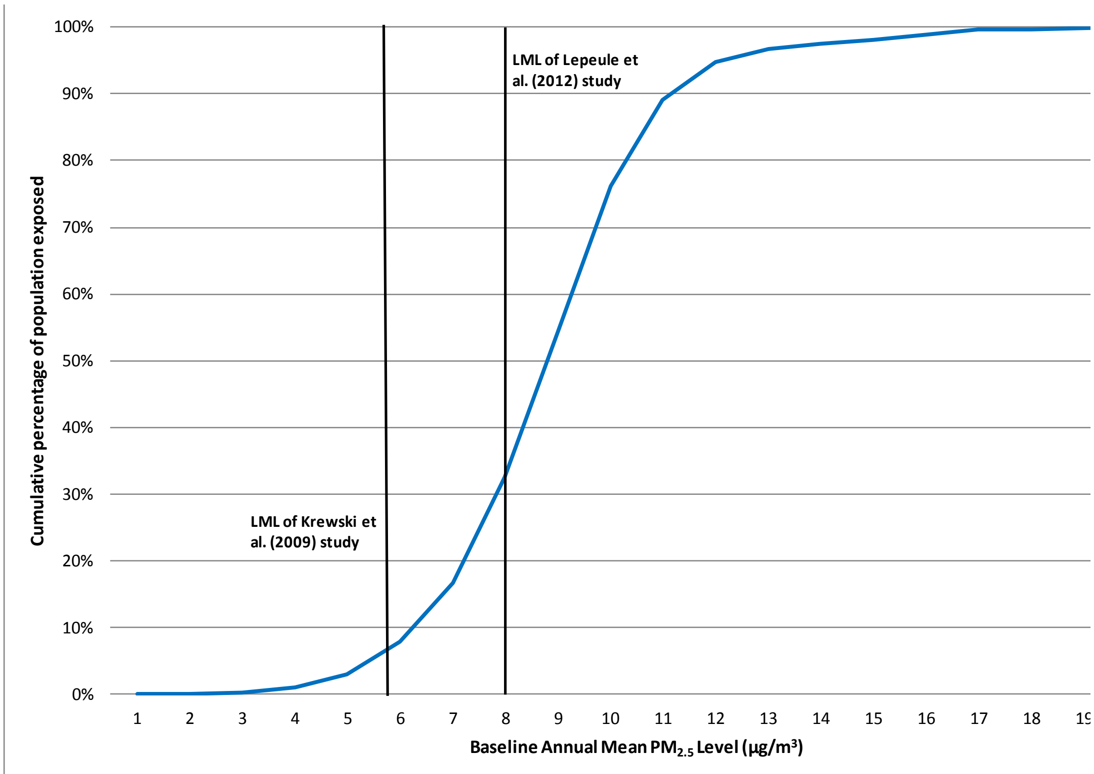


Among the populations exposed to $PM_{2.5}$ in the baseline:

93% are exposed to $PM_{2.5}$ levels at or above the LML of the Krewski et al. (2009) study
67% are exposed to $PM_{2.5}$ levels at or above the LML of the Lepeule et al. (2012) study

Figure 5C-3. Percentage of Adult Population (age 30+) by Annual Mean $PM_{2.5}$ Exposure in the Baseline Sector Modeling (used to generate the benefit-per-ton estimates)*

* This graph shows the population exposure in the modeling baseline used to generate the benefit-per-ton estimates. Similar graphs for analyses with air quality modeling show premature mortality impacts at each $PM_{2.5}$ concentration. Therefore, caution is warranted when interpreting this graph because it is not consistent with similar graphs from RIAs that had air quality modeling (e.g., MATS).



Among the populations exposed to $PM_{2.5}$ in the baseline:

93% are exposed to $PM_{2.5}$ levels at or above the LML of the Krewski et al. (2009) study
67% are exposed to $PM_{2.5}$ levels at or above the LML of the Lepeule et al. (2012) study

Figure 5C-4. Cumulative Distribution of Adult Population (age 30+) by Annual Mean $PM_{2.5}$ Exposure in the Baseline Sector Modeling (used to generate the benefit-per-ton estimates)*

* This graph shows the population exposure in the modeling baseline used to generate the benefit-per-ton estimates. Similar graphs for analyses with air quality modeling show premature mortality impacts at each $PM_{2.5}$ concentration. Therefore, caution is warranted when interpreting this graph because it is not consistent with similar graphs from RIAs that had air quality modeling (e.g., MATS).

5C.3 Core Incidence and Dollar Benefits Estimates Reflecting Application of Known Controls for the 2025 Scenario

This section presents a subset of the core incidence and dollar benefits estimates for the 2025 scenario reflecting only application of known controls in simulating each of the alternative standard levels (i.e., partial-2025 scenario estimates). The presentation of these estimates parallels results summarized for the 2025 and post-2025 scenario in Section 5.7 and the reader is referred to that section for further explanation of the tables and types of estimates included in those tables. However, before presenting detailed benefits summary tables for the partial-2025 scenario, we first (in Table 5C-5) present an overview of the percentage of benefits associated with known controls for each of the alternative standards considered. Then, following that overview table, we present a set of more detailed tables including: core incidence attributable to reductions in ozone (Table 5C-6), core dollar benefits associated with ozone reductions (Table 5C-7), core incidence estimates associated with reductions in $PM_{2.5}$ (Table 5C-8) and core dollar benefits estimates associated with $PM_{2.5}$ reductions (Table 5C-9). A summary of overall core benefits associated with application of known controls is presented in Table 5C-10.

Table 5C-5. Fraction of Total Core Benefits Associated with Partial Attainment (application of known controls) (2025 Scenario)

Category of Benefit	Percentage of Benefits Resulting from Application of Known Controls		
	70 ppb	65 ppb	60 ppb
Ozone Benefits	82%	59%	35%
$PM_{2.5}$ Co-benefits	76%	59%	33%
Total Benefits	77%	59%	34%

Table 5C-6. Estimated Number of Avoided Ozone-Only Health Impacts for the Alternative Annual Primary Ozone Standards (Incremental to the Analytical Baseline) for the Partial Attainment of the 2025 Scenario (known controls) [a,b]

	Health Effectb	Proposed and Alterative Standards (95th percentile confidence intervals) 70 ppb	65 ppb	60 ppb
Avoided Short-Term Mortality - Core Analysis				
multi-city studies	Smith et al. (2009) (all ages)	160 (79 to 250)	370 (180 to 550)	400 (200 to 610)
	Zanobetti and Schwartz (2008) (all ages)	270 (150 to 400)	620 (330 to 900)	680 (360 to 990)
Avoided Long-term Respiratory Mortality - Core Analysis				
multi-city study	Jerrett et al. (2009) (30-99yrs) copollutants model ($PM_{2.5}$)	550 (190 to 910)	1,200 (420 to 2,100)	1,400 (460 to 2,300)
Avoided Short-Term Mortality - Sensitivity Analysis				
multi-city studies	Smith et al. (2009) (all ages) copollutants model (PM10)	130 (-36 to 300)	290 (-80 to 660)	320 (-88 to 730)
	Schwartz (2005) (all ages)	200 (63 to 340)	460 (140 to 770)	500 (160 to 850)
	Huang et al. (2005) (cardiopulmonary)	190 (72 to 310)	430 (160 to 710)	480 (180 to 780)
meta-analyses	Bell et al. (2004) (all ages)	130 (44 to 220)	300 (99 to 490)	330 (110 to 550)
	Bell et al. (2005) (all ages)	430 (200 to 650)	960 (460 to 1,500)	1,100 (510 to 1,600)
	Ito et al. (2005) (all ages)	590 (360 to 830)	1,300 (800 to 1,900)	1,500 (880 to 2,100)
	Levy et al. (2005) (all ages)	600 (410 to 790)	1,400 (930 to 1,800)	1,500 (1,000 to 2,000)
Avoided Long-term Respiratory Mortality - Sensitivity Analysis [c]				
multi-city study	Jerrett et al. (2009) (age 30-99) (86 cities) (ozone-only)	370 (110 to 640)	840 (240 to 1,400)	920 (260 to 1,600)
	Jerrett et al. (2009) (age 30-99) (96 cites) (ozone-only)	400 (140 to 660)	900 (320 to 1,500)	990 (350 to 1,600)
Avoided Morbidity - Core Analysis				
	Hospital admissions - respiratory (age 65+)	300 (-79 to 670)	680 (-180 to 1,500)	740 (-200 to 1,700)
	Emergency department visits for asthma (all ages)	930 (87 to 2,900)	2,100 (200 to 6,600)	2,400 (230 to 7,500)
	Asthma exacerbation (age 6-18)	240,000 (-360,000 to 740,000)	550,000 (-800,000 to 1,600,000)	600,000 (-880,000 to 1,800,000)
	Minor restricted-activity days (age 18-65)	760,000 (310,000 to 1,200,000)	1,700,000 (700,000 to 2,700,000)	1,900,000 (780,000 to 3,000,000)
	School Loss Days (age 5-17)	270,000 (95,000 to 600,000)	600,000 (210,000 to 1,300,000)	660,000 (230,000 to 1,500,000)

[a] All incidence estimates are rounded to whole numbers with a maximum of two significant digits.
[b] All incidence estimates are based on ozone-only models unless otherwise noted.
[c] The sensitivity analysis for long-term exposure-related mortality included an assessment of potential thresholds, which was completed for the 2025 scenario (see Table 5-19). Care should be taken in applying the results of that sensitivity analysis to the partial 2025 scenario, although general patterns of impact across the thresholds may apply.

Table 5C-7. Total Monetized Ozone-Only Benefits for the Alternative Annual Primary Ozone Standards (Incremental to the Analytical Baseline) for the Partial Attainment of the 2025 Scenario (using known controls) [a,b]

	Health Effectb	Proposed and Alterative Standards (95th percentile confidence intervals) 70 ppb	65 ppb	60 ppb
Avoided Short-Term Mortality - Core Analysis				
multi-city studies	Smith et al. (2009) (all ages)	$1,700 ($150 to $4,700)	$3,700 ($330 to $11,000)	$4,100 ($360 to $12,000)
	Zanobetti and Schwartz (2008) (all ages)	2,800 ($250 to $7,900)	6,300 ($560 to $18,000)	6,900 ($610 to $20,000)
Avoided Short-Term Mortality - Sensitivity Analysis				
multi-city studies	Smith et al. (2009) (all ages) copollutants model (PM10)	$1,300 (-$310 to $4,800)	$3,000 (-$710 to $11,000)	$3,300 (-$780 to $12,000)
	Schwartz (2005) (all ages)	$2,100 ($160 to $6,300)	$4,700 ($370 to $14,000)	$5,100 ($410 to $16,000)
	Huang et al. (2005) (cardiopulmonary)	$2,000 ($160 to $5,800)	$4,400 ($370 to $13,000)	$4,900 ($400 to $14,000)
meta-analyses	Bell et al. (2004) (all ages)	1,300 ($110 to $4,000)	3,000 ($240 to $9,100)	3,300 ($270 to $10,000)
	Bell et al. (2005) (all ages)	4,400 ($380 to $12,000)	9,800 ($860 to $28,000)	11,000 ($950 to $31,000)
	Ito et al. (2005) (all ages)	$6,000 ($550 to $17,000)	$14,000 ($1,200 to $38,000)	$15,000 ($1,400 to $42,000)
	Levy et al. (2005) (all ages)	$6,100 ($570 to $17,000)	$14,000 ($1,300 to $38,000)	$15,000 ($1,400 to $41,000)
Avoided Long-term Respiratory Mortality - Sensitivity Analysis				
multi-city study	Jerrett et al. (2009) (age 30-99) copollutants model ($PM_{2.5}$) no lag [c]	$5,600 ($460 to $17,000)	$13,000 ($1,000 to $38,000)	$14,000 ($1,100 to $41,000)
	Jerrett et al. (2009) (age 30-99) copollutants model ($PM_{2.5}$) 20 yr segmented lag [d]	$4,600 to $5,100 ($370 to $15,000)	$10,000 to $11,000 ($840 to $34,000)	$11,000 to $13,000 ($920 to $37,000)

[a] All benefits estimates are rounded to whole numbers with a maximum of two significant digits. The monetized value of the ozone-related morbidity benefits are included in the estimates shown in this table for each mortality study (and when combined account for from 4-6% of the total benefits, depending on the total mortality estimate compared against. Note that asthma exacerbations accounts for <<1% of the total).

[b] The sensitivity analysis for long-term exposure-related mortality included an assessment of potential thresholds however this assessment was implemented using incidence estimates (see Table 5-19). Observations from that analysis (in terms of fractional impacts on incidence can be directly applied to these benefit results) (see Appendix 5B, section 5B.1)

[c] A single central-tendency value is provided in each cell, since the zero-lag model used here did not require application of a 3% and 7% discount rates (see footnote d below) (note, however that as with all other entries in this study, we do include a 95th percentile confidence interval range – these are the values within parentheses).

[d] The range (outside of the parentheses) within each cell results from application of a 7% and 3% discount rates in the context of applying the 20year segmented lag (with the 7% resulting in the lower estimate and the 3% the higher estimate). The range presented within the parentheses reflects consideration for the 95th% confidence interval generated for each of these estimates and ranges from a low value (2.5th% CI for the 7% discount-based dollar benefit) to an upper value (97.5th% of the 3% discount-based dollar benefit).

Table 5C-8. Estimated Number of Avoided $PM_{2.5}$-Related Health Impacts for the Alternative Annual Primary Ozone Standards (Incremental to the Analytical Baseline) for the Partial Attainment of the 2025 Scenario (using known controls) [a,b]

Health Effectb	Proposed and Alterative Standards 60ppb	65ppb	70ppb
Avoided $PM_{2.5}$-related Mortality			
Krewski et al. (2009) (adult mortality age 30+)	390	860	880
Lepeule et al. (2012) (adult mortality age 25+)	870	1,900	2,000
Woodruff et al. (1997) (infant mortality)	1	1	1
Avoided PM2.5-related Morbidity			
Non-fatal heart attacks			
Peters et al. (2001) (age >18)	450	1,000	1,000
Pooled estimate of 4 studies (age >18)	49	110	110
Hospital admissions—respiratory (all ages)	120	260	260
Hospital admissions—cardiovascular (age > 18)	140	310	320
Emergency department visits for asthma (all ages)	210	470	480
Acute bronchitis (ages 8–12)	600	1,300	1,400
Lower respiratory symptoms (ages 7–14)	7,700	17,000	17,000
Upper respiratory symptoms (asthmatics ages 9–11)	11,000	24,000	25,000
Asthma exacerbation (asthmatics ages 6–18)	11,000	25,000	26,000
Lost work days (ages 18–65)	49,000	110,000	110,000
Minor restricted-activity days (ages 18–65)	290,000	640,000	660,000

[a] All incidence estimates are rounded to whole numbers with a maximum of two significant digits. Because these estimates were generated using benefit-per-ton estimates, confidence intervals are not available. In general, the 95[th] percentile confidence interval for the health impact function alone ranges from approximately ±30 percent for mortality incidence based on Krewski et al. (2009) and ±46 percent based on Lepeule et al. (2012).

Table 5C-9. Monetized $PM_{2.5}$-Related Health Co-Benefits for the Alternative Annual Primary Ozone Standards (Incremental to Analytical Baseline) for the Partial Attainment of the 2025 Scenario (using known controls) [a,b,c]

Monetized Benefits	Proposed and Alterative Standards		
	70 ppb	65 ppb	60 ppb
3% Discount Rate			
Krewski et al. (2009) (adult mortality age 30+)	\$3,600	\$8,000	\$8,300
Lepeule et al. (2012) (adult mortality age 25+)	\$8,200	\$18,000	\$19,000
7% Discount Rate			
Krewski et al. (2009) (adult mortality age 30+)	\$3,300	\$7,200	\$7,400
Lepeule et al. (2012) (adult mortality age 25+)	\$7,400	\$16,000	\$17,000

[a] All estimates are rounded to two significant digits. Because these estimates were generated using benefit-per-ton estimates, confidence intervals are not available. In general, the 95th percentile confidence interval for monetized $PM_{2.5}$ benefits ranges from approximately -90 percent to +180 percent of the central estimates based on Krewski et al. (2009) and Lepeule et al. (2012). Estimates do not include unquantified health benefits noted in Table 5-2 or Section 5.6.5 or welfare co-benefits noted in Chapter 6.

[b] The reduction in premature fatalities each year accounts for over 98% of total monetized benefits in this analysis. Mortality risk valuation assumes discounting over the SAB-recommended 20-year segmented lag structure.

Table 5C-10. Combined Estimate of Monetized Ozone and $PM_{2.5}$ Benefits for the Alternative Annual Primary Ozone Standards for the Partial Attainment of the 2025 Scenario (using known controls) (billions of 2011\$) [a,b]

	Discount Rate	70 ppb	65 ppb	60 ppb
Total Benefits	3%	\$5.3 to \$11 +B	\$12 to \$24 +B	\$12 to \$26 +B
	7%	\$5.0 to \$10 +B	\$11 to \$23 +B	\$12 to \$24 +B
Ozone-only Benefits (range reflects Smith et al., 2009 and Zanobetti and Schwartz, 2008)	b	\$1.7 to \$2.8 +B	\$3.7 to \$6.3 +B	\$4.1 to \$6.9 +B
$PM_{2.5}$ Co-benefits (range reflects Krewski et al., 2009 and Lepeule et al., 2012)	3%	\$3.6 to \$8.2 +B	\$8.0 to \$18 +B	\$8.3 to \$19 +B
	7%	\$3.3 to \$7.4 +B	\$7.2 to \$16 +B	\$7.4 to \$17 +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 for $PM_{2.5}$ assumes discounting over the SAB-recommended 20-year segmented lag structure. Not all possible benefits are quantified and monetized in this analysis. B is the sum of all unquantified health and welfare co-benefits. Data limitations prevented us from quantifying these endpoints, and as such, these benefits are inherently more uncertain than those benefits that we were able to quantify. These estimates reflect the economic value of avoided morbidities and premature deaths using risk coefficients from the studies noted.

[b] Ozone-only benefits reflect short-term exposure impacts and as such are assumed to occur in the same year as ambient ozone reductions. Consequently, social discounting is not applied to the benefits for this category.

5C.4 References

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APPENDIX 5D: DISCUSSION OF EFFECT ESTIMATES REFLECTED IN THE DEVELOPMENT OF DOLLAR-PER-TON VALUES USED IN MODELING $PM_{2.5}$ COBENEFITS

Overview

This section describes how we selected effect estimates to estimate the benefits of reducing $PM_{2.5}$ (see Section 5.4.4). This section mirrors that covering effect estimate selection for ozone presented in Section 5.6.3 in terms of organization (in some cases, we have repeated introductory/setup content here to facilitate review by the reader).

The first step in selecting effect coefficients is to identify the health endpoints to be quantified. We base our selection of health endpoints on consistency with the EPA's Integrated Science Assessments (which replace previous "Criteria Documents"), with input and advice from the Health Effects Subcommittee (HES), a scientific review panel specifically established to provide advice on the use of the scientific literature in developing benefits analyses for the *EPA's Report to Congress on The Benefits and Costs of the Clean Air Act 1990 to 2020* (U.S. EPA, 2011a). In addition, we have included more recent epidemiology studies from the PM ISA (U.S. EPA, 2009b) and the PM Provisional Assessment (U.S. EPA, 2012b).[88] 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

[88] The peer-reviewed studies in the *Provisional Assessment* have not yet undergone external review by the Science Advisory Board.

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 5-6 (see section 5.6.3). 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 for an earlier version of the program provide details of the procedures used to combine multiple impact functions (Abt Associates, 2012). In general, we used fixed or random effects models to pool estimates from different single-city studies of the same endpoint. Fixed effect 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 effect 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.[89] Pooled impact functions are used to estimate hospital admissions and asthma exacerbations. When combining evidence across multi-city studies (e.g., cardiovascular hospital admission studies), we use equal weights pooling. The effect estimates drawn from each multi-city study are themselves pooled across a large number of urban areas. For this reason, we elected to give each study an equal weight rather than weighting by the inverse of the variance reported in each study. For more details on methods used to pool incidence estimates, see the BenMAP Manual Appendices (Abt Associates, 2012).

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

The specific studies from which effect estimates were quantified for the core analysis for $PM_{2.5}$ (generated using the dollar-per-ton approach – see Section 5.4.4) are presented in Table 5-8 and repeated below for ease of access in Table 5D-1. We highlight in red those studies that have been added since the benefits analysis conducted for the ozone reconsideration (U.S. EPA, 2010d) or the Ozone NAAQS RIA (U.S. EPA, 2008). 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.

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

Table 5D-1. Health Endpoints and Epidemiological Studies Used to Quantify $PM_{2.5}$-related Health Impacts in the Core Analysis [a]

Endpoint	Study	Study Population	Relative Risk or Effect Estimate (β) (with 95th Percentile Confidence Interval or SE, respectively)
Premature Mortality			
Premature mortality—cohort study, all-cause	Krewski et al. (2009)	> 29 years	RR = 1.06 (1.04–1.06) per 10 μg/m³
	Lepeule et al. (2012)	> 24 years	RR = 1.14 (1.07–1.22) per 10 μg/m³
Premature mortality—all-cause	Woodruff et al. (1997)	Infant (< 1 year)	OR = 1.04 (1.02–1.07) per 10 μg/m³
Chronic Illness			
Nonfatal heart attacks	Peters et al. (2001)	Adults (> 18 years)	OR = 1.62 (1.13–2.34) per 20 μg/m³
	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)
	Kloog et al. (2012)—ICD 460-519 (All Respiratory		β=0.0007 (0.000961)
	Moolgavkar (2000)—ICD 490–496 (Chronic lung disease)	18–64 years	1.02 (1.01–1.03) per 36 μg/m³
	Babin et al. (2007)—ICD 493 (asthma)	< 19 years	β=0.002 (0.004337)
	Sheppard (2003)—ICD 493 (asthma)	< 18	RR = 1.04 (1.01–1.06) per 11.8 μg/m³
Cardiovascular	Pooled estimate:	> 64 years	
	Zanobetti et al. (2009)—ICD 390-459 (all cardiovascular)		β=0.00189 (0.000283)
	Peng et al. (2009)—ICD 426-427; 428; 430-438; 410-414; 429; 440-449 (Cardio-, cerebro- and peripheral vascular disease)		β=0.00068 (0.000214)
	Peng et al. (2008)—ICD 426-427; 428; 430-438; 410-414; 429; 440-449 (Cardio-, cerebro- and peripheral vascular disease)		β=0.00071 (0.00013)
	Bell et al. (2008)—ICD 426-427; 428; 430-438; 410-414; 429; 440-449 (Cardio-, cerebro- and peripheral vascular disease)		β=0.0008 (0.000107)
	Moolgavkar (2000)—ICD 390–429 (all cardiovascular)	20–64 years	RR=1.04 (t statistic: 4.1) per 10 μg/m³
Asthma-related emergency department visits	Pooled estimate:	All ages	
	Mar et al. (2010)		RR = 1.04 (1.01–1.07) per 7 μg/m³
	Slaughter et al. (2005)		RR = 1.03 (0.98–1.09) per 10 μg/m³
	Glad et al. (2012)		β=0.00392 (0.002843)
Other Health Endpoints			
Acute bronchitis	Dockery et al. (1996)	8–12 years	OR = 1.50 (0.91–2.47) per 14.9 μg/m³

Asthma exacerbations	Pooled estimate: Ostro et al. (2001) (cough, wheeze and shortness of breath) [b] Mar et al. (2004) (cough, shortness of breath)	6–18 years [b]	OR = 1.03 (0.98–1.07) OR = 1.06 (1.01–1.11) OR = 1.08 (1.00–1.17) per 30 μg/m^3 RR = 1.21 (1–1.47) per RR = 1.13 (0.86–1.48) per 10 μg/m^3
Work loss days	Ostro (1987)	18–65 years	β=0.0046 (0.00036)
Acute respiratory symptoms (MRAD)	Ostro and Rothschild (1989) (Minor restricted activity days)	18–65 years	β=0.00220 (0.000658)
Upper respiratory symptoms	Pope et al. (1991)	Asthmatics, 9–11 years	1.003 (1–1.006) per 10 μg/m^3
Lower respiratory symptoms	Schwartz and Neas (2000)	7–14 years	OR = 1.33 (1.11–1.58) per 15 μg/m^3

[a] Studies highlighted in red represent updates incorporated since the Ozone NAAQS RIA (U.S. EPA, 2008). These updates were introduced in the PM NAAQS RIA (U.S. EPA, 2012).
[b] The original study populations were 8 to 13 for the Ostro et al. (2001) study and 7 to 12 for the Mar et al. (2004) study. Based on advice from the SAB-HES, we extended the applied population to 6-18, reflecting the common biological basis for the effect in children in the broader age group. See: U.S. EPA-SAB (2004) and NRC (2002).

5D.1 $PM_{2.5}$ Premature Mortality Effect Coefficients

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

Researchers have found statistically significant associations between $PM_{2.5}$ and premature mortality using different types of study designs. Time-series methods have been used to relate short-term (often day-to-day) changes in $PM_{2.5}$ concentrations and changes in daily mortality rates up to several days after a period of elevated $PM_{2.5}$ concentrations. Cohort methods have been used to examine the potential relationship between community-level $PM_{2.5}$ exposures over multiple years (i.e., long-term exposures) and community-level annual mortality rates that have been adjusted for individual level risk factors. When choosing between using short-term studies or cohort studies for estimating mortality benefits, cohort analyses are thought to capture more of the public health impact of exposure to air pollution over time because they account for the effects of long-term exposures as well as some fraction of short-term exposures (Kunzli et al., 2001; NRC, 2002). The NRC stated that "it is essential to use the cohort studies in benefits analysis to capture all important effects from air pollution exposure" (NRC, 2002, p. 108). The NRC further notes that "the overall effect estimates may be a combination of effects from long-term exposure plus some fraction from short-term exposure. The amount of overlap is unknown" (NRC, 2002, p. 108-9). To avoid double counting, we focus on applying the risk coefficients from the long-term cohort studies in estimating the mortality impacts of reductions in $PM_{2.5}$.

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

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

In 2009, the SAB-HES again reviewed the choice of mortality risk coefficients for benefits analysis, concluding that "[t]he Krewski et al. (2009) findings, while informative, have not yet undergone the same degree of peer review as have the aforementioned studies. Thus, the SAB-HES recommends that EPA not use the Krewski et al. (2009) findings for generating the Primary Estimate" (U.S. EPA-SAB, 2010a). Since this time, the Krewski et al. (2009) has undergone additional peer review, which we believe strengthens the support for including this study in this RIA. For example, the PM ISA (U.S. EPA, 2009b) included this study among the key mortality studies. In addition, the risk assessment supporting the PM NAAQS (U.S. EPA, 2010a) 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, 2010a). 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, 2009, 2010b, c).

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

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

Implicit in the calculation of $PM_{2.5}$-related premature mortality impacts are several key assumptions, which are described in further detail later in this Appendix. First, we assume that there is a "cessation" lag in time between the reduction in PM exposure and the full reduction in mortality risk that affects the timing (and thus discounted monetary valuation) of the resulting premature deaths (see Section 5.6.4.1). Second, following conclusions of the PM ISA, we assume that all fine particles are equally potent in causing premature mortality (see Section 5.7.3). Third, following conclusions of the PM ISA, we assume that the health impact function for fine particles is linear within the range of ambient concentrations affected by these standards (see Section 5.7.3).

Alternate Mortality Effect Coefficients for Adults. In addition to the ACS and Six Cities cohorts, several recent cohort studies conducted in North America provide evidence for the relationship between long-term exposure to $PM_{2.5}$ and the risk of premature death. Many of these additional cohort studies are described in the PM ISA (U.S. EPA, 2009b) and the Provisional Assessment (U.S. EPA, 2012b) (and thus not summarized here).[90,91] Table 5D-2 provides the effect estimates from each of these cohort studies for all-cause, cardiovascular, cardiopulmonary, and ischemic heart disease (IHD) mortality, as well as the lowest measured air quality level (LML) and mean concentration in the study.

We also draw upon the results of the 2006 expert elicitation sponsored by the EPA (Roman et al., 2008; IEc, 2006) to demonstrate the sensitivity of the benefits estimates to 12 expert-defined concentration-response functions. The $PM_{2.5}$ expert elicitation and the derivation of effect estimates from the expert elicitation results are described in detail in the 2006 $PM_{2.5}$ NAAQS RIA (U.S. EPA, 2006), the elicitation summary report (IEc, 2006) and Roman et al. (2008), and so we summarize the key attributes of this study relative to the interpretation of the estimates of PM-related mortality reported here. We describe also how the epidemiological literature has evolved since the expert elicitation was conducted in 2005 and 2006.

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

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

Table 5D-2. 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 (µg/m³)	Mean (µg/m³)	Hazard Ratios per 10 µg/m³ Change in $PM_{2.5}$ (95th percentile confidence intervals)			
				All Causes	Cardiovascular	Cardiopulmonary	IHD
Pope et al. (2002)	ACS (age >30)	7.5	18.2	1.06 (1.02–1.11)	1.12 (1.08–1.15)	1.09 (1.03–1.16)	N/A
Laden et al. (2006)	Six Cities (age > 25)	10	16.4	1.16 (1.07–1.26)	1.28 (1.13–1.44)	N/A	N/A
Lipfert et al. (2006)[a]	Veterans (age 39–63)	<14.1	14.3	1.15 (1.05–1.25)	N/A	N/A	N/A
Miller et al. (2007)[b]	WHI (age 50–79)	3.4	13.5	N/A	1.76 (1.25–2.47)	N/A	2.21 (1.17–4.16)
Eftim et al. (2008)	Medicare (age > 65)	6	13.6	1.21 (1.15–1.27)	N/A	N/A	N/A
Zeger et al. (2008)[c]	Medicare (age > 65)	<9.8	13.2	1.068 (1.049–1.087)	N/A	N/A	N/A
Krewski et al. (2009)[d]	ACS (age >30)	5.8	14	1.06 (1.04–1.08)	N/A	1.13 (1.10–1.16)	1.24 (1.19–1.29)
Puett et al. (2009)[b]	NHS (age 30–55)	5.8	13.9	1.26 (1.02–1.54)	N/A	N/A	2.02 (1.07–3.78)
Crouse et al. (2011)[d,e]	Canadian census	1.9	8.7	1.06 (1.01–1.10)	N/A	N/A	N/A
Puett et al. (2011)[f]	Health Professionals (age 40–75)	<14.4	17.8	0.86 (0.70–1.00)	1.02 (0.84–1.23)	N/A	N/A
Lepeule et al. (2012)[d]	Six Cities (age > 25)	8	15.9	1.14 (1.07–1.22)	1.26 (1.14–1.40)	N/A	N/A

[a] Low socio-economic status (SES) men only. Used traffic proximity as a surrogate of exposure.
[b] Women only.
[c] Reflects risks in the Eastern U.S. Risks in the Central U.S. were higher, but the authors found no association in the Western U.S.
[d] Random effects Cox model with individual and ecologic covariates.
[e] Canadian population.
[f] Men with high socioeconomic status only.

The primary goal of the 2006 study was to elicit from a sample of health experts probabilistic distributions describing uncertainty in estimates of the reduction in mortality among the adult U.S. population resulting from reductions in ambient annual average $PM_{2.5}$ levels. These distributions were obtained through a formal interview protocol using methods designed to elicit subjective expert judgments. These experts were selected through a peer-nomination process and included experts in epidemiology, toxicology, and medicine. The elicitation interview consisted of a protocol of carefully structured questions, both qualitative and quantitative, about the nature of the $PM_{2.5}$-mortality relationship designed to build twelve individual distributions for the coefficient (or slope) of the C-R function relating changes in annual average $PM_{2.5}$ exposures to annual, adult all-cause mortality. The elicitation also provided useful information regarding uncertainty characterization in the $PM_{2.5}$-mortality relationship.

Specifically, during their interviews, the experts highlighted several uncertainties inherent within the epidemiology literature, such as causality, concentration thresholds, effect modification, the role of short- and long-term exposures, potential confounding, and exposure misclassification. In Appendix 5C, we evaluate each of these uncertainties in the context of this health impact assessment. For several of these uncertainties, such as causality, we are able to use the expert-derived functions to quantify the impacts of applying different assumptions. The elicitation received favorable peer review in 2006 (Mansfield and Patil, 2006).

Prior to providing a quantitative estimate of the risk of premature death associated with long-term $PM_{2.5}$ exposure, the experts answered a series of "conditioning questions." One such question asked the experts to identify which epidemiological studies they found most informative. The "ideal study attributes"[92] according to the experts included:

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

Although no single epidemiological study completely satisfies each of these criteria, the experts determined that the ACS and Six Cities' cohort studies best satisfy a majority of these ideal attributes. To varying degrees the studies examining these two cohorts are geographically representative; have collected information on individual risk factors; include a large sample size; have collected data on co-pollutants in the case of the ACS study; have accurately characterized the health outcome; include a long (and growing) follow-up period; and, are prospective in

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

nature. The experts also noted a series of limitations in these two cohort studies. In the case of the Six Cities study (Laden et al., 2006), the experts identified the "small sample size, limited number of cities, and concerns about representativeness of the six cities for the U.S. as a whole" as weaknesses. When considering the ACS study (Pope et al., 2002), the experts indicated that the "method of recruitment for the study, which resulted in a group with higher income, more education, and a greater proportion of whites than is representative of the general U.S. population" represented a shortcoming. Several experts also argued that because the ACS study relied upon "…whatever monitors were available to the study…a single monitor represent[ed] exposure for an entire metropolitan area…whereas [the Six Cities study] often had exposures assigned at the county level." Despite these limitations, the experts considered the Pope et al. (2002) extended analysis of the ACS cohort and the Laden et al. (2006) extended analysis of the Six Cities cohort to be particularly influential in their opinions (see Exhibit 3-3 of the elicitation summary report [IEc, 2006]).

Please note that the benefits estimates results presented are not the direct results from the studies or expert elicitation; rather, the estimates are based in part on the effect coefficients provided in those studies or by experts. In addition, the experts provided distributions around their mean $PM_{2.5}$ effect estimates, which provides more information regarding the overall range of uncertainty, and this overall range is larger than the range of the mean effect estimates from each of the experts.

Since the completion of the EPA's expert elicitation in 2006, additional epidemiology literature has become available, including 9 new multi-state cohort studies shown in Table 5-12. This newer literature addresses some of the weaknesses identified in the prior literature. For example, in an attempt to improve its characterization of population exposure the most recent extended analysis of the ACS cohort Krewski et al. (2009) incorporates two case studies that employ more spatially resolved estimates of population exposure.

In light of the availability of this newer literature, we have updated the presentation of results in the RIA. Specifically, we focus the core analysis on results derived from the two most recent studies of the ACS and Six Cities cohorts (Krewski et al., 2009; Lepeule et al., 2012). Because the other multi-state cohorts generally have limited geography and age/gender representativeness, these limitations preclude us from using these studies in our core benefits results, and we instead present the risk coefficients from these other multi-state cohorts in Table 5D-2. In addition, we now include as a sensitivity analysis, mortality estimates based on application of the full set of expert-derived effect estimates (see Appendix 5B, Section 5B.2). We do not combine the expert results in order to preserve the breadth and diversity of opinion on the expert panel (Roman et. al., 2008). This presentation of the expert-derived results is generally consistent with SAB advice (U.S. EPA-SAB, 2008), which recommended that the EPA emphasize that "scientific differences existed only with respect to the magnitude of the effect of $PM_{2.5}$ on mortality, not whether such an effect existed" and that the expert elicitation "supports the conclusion that the benefits of $PM_{2.5}$ control are very likely to be substantial". Although it is possible that the newer literature could revise the experts' quantitative responses if elicited again, we believe that these general conclusions are unlikely to change.

Mortality Effect Coefficients for Infants. In addition to the adult mortality studies described above, several studies show an association between PM exposure and premature mortality in children under 5 years of age.[93] The PM ISA states that less evidence is available regarding the potential impact of $PM_{2.5}$ exposure on infant mortality than on adult mortality and the results of studies in several countries include a range of findings with some finding significant associations. Specifically, the PM ISA concluded that evidence exists for a stronger effect at the post-neonatal period and for respiratory-related mortality, although this trend is not consistent across all studies. In addition, compared to avoided premature deaths estimated for adult mortality, avoided premature deaths for infants are significantly smaller because the number of infants in the population is much smaller than the number of adults and the epidemiology studies on infant mortality provide smaller risk coefficients associated with exposure to $PM_{2.5}$.

In 2004, the SAB-HES noted the release of the WHO Global Burden of Disease Study focusing on ambient air, which cites several recently published time-series studies relating daily PM exposure to mortality in children (U.S. EPA-SAB, 2004). The SAB-HES also cites the study by Belanger et al. (2003) as corroborating findings linking PM exposure to increased respiratory inflammation and infections in children. A study by Chay and Greenstone (2003) found that

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

reductions in TSP caused by the recession of 1981–1982 were statistically associated with reductions in infant mortality at the county level. With regard to the cohort study conducted by Woodruff et al. (1997), the SAB-HES notes several strengths of the study, including the use of a larger cohort drawn from a large number of metropolitan areas and efforts to control for a variety of individual risk factors in infants (e.g., maternal educational level, maternal ethnicity, parental marital status, and maternal smoking status). Based on these findings, the SAB-HES recommended that the EPA incorporate infant mortality into the primary benefits estimate and that infant mortality be evaluated using an impact function developed from the Woodruff et al. (1997) study (U.S. EPA-SAB, 2004).

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

5D.2 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 are estimated in this analysis for $PM_{2.5}$: *respiratory admissions* and *cardiovascular admissions*. There is not sufficient evidence linking $PM_{2.5}$ with other types of hospital admissions. Both asthma- and cardiovascular-related visits have been linked to $PM_{2.5}$ in the United States, though as we note below, we are able to assign an economic value to asthma-related events only. To estimate the effects of $PM_{2.5}$ air pollution reductions on asthma-related ER visits, we use the effect estimate from a study of children 18 and under by Mar et al. (2010), Slaughter et al. (2005), and Glad et al. (2012). The first two studies examined populations 0 to 99 in Washington State, while Glad et al. examined populations 0-99 in Pittsburgh, PA. Mar and colleagues perform their study in Tacoma, while Slaughter and colleagues base their study in Spokane. We apply random/fixed effects pooling to combine evidence across these two studies.

To estimate avoided incidences of cardiovascular hospital admissions associated with $PM_{2.5}$, we used studies by Moolgavkar (2000), Zanobetti et al. (2009), Peng et al. (2008, 2009) and Bell et al., (2008). Only Moolgavkar (2000) provided a separate effect estimate for adults 20 to 64, while the remainder estimate risk among adults over 64.[94] 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

[94] 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.

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 studies by Zanobetti et al. (2009) and Kloog et al. (2012) provide effect coefficients for total respiratory hospital admissions (defined as ICD codes 460–519). We pool these two studies using equal weights. Moolgavkar et al. (2003) examines $PM_{2.5}$ and chronic lung disease hospital admissions (less asthma) in Los Angeles, CA among adults 18 to 64. For children 0 to 18, we pool two studies using random/fixed effects. The first is Babin et al. (2007) which assessed $PM_{2.5}$ and asthma hospital admissions in Washington, DC among children 1 to 18; we adjusted the age range for this study to apply to children 0 to 18. The second is Sheppard et al. (2003) which assessed $PM_{2.5}$ and asthma hospitalizations in Seattle, Washington, among children 0 to 18.

5D.3 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.

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

Based on advice from the SAB-HES (2004), regardless of the age ranges included in the source epidemiology studies, we extend the applied population to ages 6 to 18, reflecting the common biological basis for the effect in children in the broader age group. This age range expansion is also supported by NRC (2002, pp. 8, 116).

To characterize asthma exacerbations in children from exposure to $PM_{2.5}$, 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 semi-arid 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 estimate for PM-related asthma exacerbation incidence. 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 in children.

Acute Respiratory Symptoms. We estimate three types of acute respiratory symptoms related to $PM_{2.5}$ exposure: lower respiratory symptoms, upper respiratory symptoms, and minor restricted activity days (MRAD).

Incidences of lower respiratory symptoms (e.g., wheezing, deep cough) in children aged 7 to 14 were estimated for $PM_{2.5}$ using an effect estimate from Schwartz and Neas (2000). Incidences of upper respiratory symptoms in asthmatic children aged 9 to 11 are estimated for $PM_{2.5}$ using an effect estimate developed from Pope et al. (1991). Because asthmatics have greater sensitivity to stimuli (including air pollution), children with asthma can be more susceptible to a variety of upper respiratory symptoms (e.g., runny or stuffy nose; wet cough; and burning, aching, or red eyes). Research on the effects of air pollution on upper respiratory symptoms has thus focused on effects in asthmatics.

MRADs 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 work 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, 2009b), but proved to be unsuitable for use in this benefits analysis. In particular, the best available study

(Patel et al., 2010) specified a population aged 13–20, which overlaps with the population in which we asses asthma exacerbation. As we describe in detail below, to avoid the chance of double-counting impacts, we do not estimate changes in acute respiratory symptoms and asthma exacerbation among populations of the same age.

Acute Bronchitis. 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,[95] 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).

Work Loss Days. 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.

Work loss days. 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). Ostro (1987) estimated the impact of $PM_{2.5}$ on the incidence of work loss days in a national sample of the adult working population, ages 18 to 65 living in metropolitan areas. Ostro reported that two-week average $PM_{2.5}$ levels were significantly linked to work loss days, but there was some year-to-year variability in the results.

5D.4 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.,

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

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, Washington; Pope et al. (2006), based in Wasatch Range, Utah; Zanobetti and Schwartz (2006), based in Boston, Massachusetts; and, Zanobetti et al. (2009), a multi-city study of 26 U.S. communities. Each of these single and multi-city studies, with the exception of Pope et al. (2006), measure AMIs using hospital discharge rates. Conversely, the Pope et al. (2006) study is based on a large registry with angiographically characterized patients—arguably a more precise indicator of AMI. Because the Pope et al. (2006) study reflected both myocardial infarctions and unstable angina, this produces a more comprehensive estimate of acute ischemic heart disease events than the other studies. However, unlike the Peters study (Peters et al., 2006), Pope and colleagues did not measure the time of symptom onset, and $PM_{2.5}$ data were not measured on an hourly basis.

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

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APPENDIX 5E: INPUTS TO $PM_{2.5}$ COBENEFIT MODELING

Overview

This section presents inputs used in generating $PM_{2.5}$ cobenefits estimates including (a) benefit-per-ton estimates for each sector (dimensioned by mortality study and simulation year)[96] (Table 5E-1), and (b) NOx emissions reductions by sector for both the 2025 and post-2025 scenarios (Table 5E-2).[97] For additional detail on the approach used to generate $PM_{2.5}$ cobenefits estimates and the role played by these two types of inputs, see Chapter 5, Section 5.4.4.

[96] Benefit-per-ton estimates were generated for each of the long-term exposure-related mortality studies used in generating core benefits estimates for this RIA including Krewski et al., 2009 and Lepeule et al., 2012 (see Appendix 5D, section 5D.1). Estimates were available for 2025 and 2030, with those being used to model cobenefits for the 2025 scenario and post-2025 scenario, respectively.

[97] Sector-level NOx reductions (for each alternative standard level) were generated using methods described in Chapter 4, section 4.2 and 4.3. As noted in section 5.4.4, NOx emissions reductions associated with alternative standard levels considered for this NAAQS review involved seven of the 17 sectors for which we had benefit-per-ton values and consequently, the cobenefits $PM_{2.5}$ estimates are based on simulated benefits for those seven sectors.

Table 5E-1. Summary of Effect Estimates from Associated Sectors with Change in Long-Term Exposure to $PM_{2.5}$ in Recent Cohort Studies in North America[a]

Long-term mortality study	Emissions sector																	
	air, locamotive and marine	cement kilns	coke ovens	EGU point	electric arc furnaces	farro alloys	integrated iron and steel	iron and steel	non-EGU point other	non-point other	nonroad	onroad	pulp and paper	refineries	residenti al wood	taconite mining	ocean going vessels	Non-specified source [b]
2025 at 7% social discount																		
Krewski et al., 2009	$7,221	$5,692	$10,438	$5,245	$9,610	$4,358	$13,424	$16,944	$6,341	$7,859	$6,979	$7,620	$3,746	$6,924	$13,598	$5,998	$2,033	$6,433
Lepeule et al., 2012	$16,295	$12,856	$23,578	$11,841	$21,708	$9,842	$30,325	$38,262	$14,316	$17,737	$15,749	$17,177	$8,460	$15,636	$30,706	$13,538	$4,586	$14,514
2025 at 3% social discount																		
Krewski et al., 2009	$8,005	6,311	$11,572	$5,815	$10,654	$4,832	$14,883	$18,783	$7,029	$8,713	$7,736	$8,447	$4,154	$7,676	$15,075	$6,649	$2,254	$7,132
Lepeule et al., 2012	$18,071	14,257	$26,149	$13,132	$24,074	$10,916	$33,630	$42,433	$15,876	$19,671	$17,466	$19,049	$9,383	$17,340	$34,054	$15,014	$5,086	$16,096
2030 at 7% social discount																		
Krewski et al., 2009	$7,829	$6,125	$11,056	$5,591	$10,219	$4,637	$14,264	$18,373	$6,814	$8,469	$7,587	$8,214	$4,017	$7,531	$14,695	$6,403	$2,258	$6,941
Lepeule et al., 2012	$17,662	$13,830	$24,969	$12,621	$23,080	$10,473	$32,217	$41,475	$15,380	$19,109	$17,116	$18,512	$9,070	$17,000	$33,176	$14,451	$5,091	$15,655
2030 at 3% social discount																		
Krewski et al., 2009	$8,680	$6,791	$12,258	$6,199	$11,330	$5,142	$15,815	$20,369	$7,554	$9,389	$8,412	$9,106	$4,454	$8,349	$16,293	$7,099	$2,504	$7,695
Lepeule et al., 2012	$19,587	$15,338	$27,691	$13,996	$25,596	$11,615	$35,729	$45,997	$17,057	$21,192	$18,983	$20,530	$10,059	$18,854	$36,794	$16,026	$5,646	$17,362

[a] Benefit-per-ton estimates reflect application of the 20-year segmented lag used in the core analysis (see section 5.6.4.1) together with either a 3% or 7% social discount rate as noted in the table. In addition, separate sets of benefit-per-ton estimates were generate for 2025 and 2030, reflecting application of appropriate projected demographic and baseline incidence data (see section 5.4.4 for additional detail).

[b] Benefit-per-ton estimates for the non-specified source category were generated as a weighted average of values for the 17 source categories, with weighting based on sector-specific NOx emissions for 2005 obtained from Fann et al., 2012.

Table 5E-2. Sector-Specific NOx Emissions Reductions for Each Alternative Standard Level[a]

Emissions Sector	Alternative Standard Level					
	70ppb		65ppb		60ppb	
	CA NOx	nonCA NOx	CA NOx	nonCA NOx	CA NOx	nonCA NOx
Aircraft, locomotives and marine vessels	-	-	-	-	-	-
Area sources	-	45,708	-	94,340	-	95,332
Cement kilns	-	20,135	-	43,867	-	43,867
Electricity Generating Units	-	32,315	-	217,881	-	244,079
Industrial point sources	-	382,743	-	741,588	-	750,620
Non-road mobile sources	-	4,984	-	12,863	-	12,863
On-road mobile sources	-	-	-	-	-	-
Pulp and paper facilities	-	357	-	617	-	617
Refineries	-	8,243	-	12,384	-	12,411
Residential wood combustion	-	-	-	-	-	-
Unknown sector	53,289	154,343	104,708	752,162	143,916	2,234,709
TOTAL	53,289	648,828	104,708	1,875,702	143,916	3,394,497

[a] All values are tons of NOx reductions (75ppb vs alternative standard). Results are presented both for "CA NOx" (emissions in CA only – used in post-2025 scenario $PM_{2.5}$ cobenefits modeling) and "nonCA NOx" (emissions reductions outside of CA – used in 2025 scenario $PM_{2.5}$ cobenefits modeling)

CHAPTER 6: IMPACTS ON PUBLIC WELFARE OF ATTAINMENT STRATEGIES TO MEET PRIMARY AND SECONDARY OZONE NAAQS

Overview

This chapter provides a discussion of the welfare-related benefits of meeting alternative primary and secondary ozone standards. Welfare benefits of reductions in ambient ozone include increased growth and/or biomass production in sensitive plant species, including forest trees, increased crop yields, reductions in visible foliar injury, increased plant vigor (e.g. decreased susceptibility to harsh weather, disease, insect pest infestation, and competition), and changes in ecosystems and associated ecosystem services. We provide a limited quantitative analysis for effects associated with changes in yields of commercial forests and agriculture, and associated changes in carbon sequestration and storage.

The EPA is proposing to revise the level of the secondary standard to within the range proposed for the primary standard of 65 parts per billion (ppb) to 70 ppb to provide increased protection against vegetation-related effects on public welfare. As an initial matter, the EPA is proposing that ambient ozone concentrations in terms of a three-year average W126 index value within the range from 13 parts per million-hours (ppm-hours) to 17 ppm-hours, would provide the requisite protection against known or anticipated adverse effects to the public welfare, which data analyses indicate would provide air quality in terms of three-year average W126 index values of a range at or below 13 ppm-hours to 17 ppm-hours. Data analyses also indicate that actions taken to attain a standard in the range of 65 ppb to 70 ppb would also improve air quality as measured by the W126 metric. The quantitative analysis in this chapter assesses the welfare benefits of strategies to attain ozone standard levels of 65 to 70 ppb.

In addition to the direct welfare benefits of decreased levels of ambient ozone, the emissions reduction strategies used to demonstrate attainment with alternative ozone standards may result in additional benefits associated with reductions in nitrogen deposition and reductions in ambient concentrations of $PM_{2.5}$ and its components. These additional benefits include reductions in nutrient enrichment and acidification impacts on sensitive aquatic and terrestrial ecosystems and improvements in visibility in state and national parks, wilderness areas, and in the areas where people live and work. We are not able to quantify or monetize these benefits in this RIA.

6.1 Welfare Benefits of Strategies to Attain Primary and Secondary Ozone Standards

The Clean Air Act defines welfare effects to include any non-health effects, including direct economic damages in the form of lost productivity of crops and trees, indirect damages through alteration of ecosystem functions, indirect economic damages through the loss in value of recreational experiences or the existence value of important resources, and direct damages to property, either through impacts on material structures or by soiling of surfaces (Section 302(h) (42 U.S.C. § 7602(h)). For welfare effects associated with changes to ecosystem functions, we use the concept of ecosystem services as a useful framework for analyzing the impact of ecosystem changes on public welfare. Ecosystem services can be generally defined as the benefits that individuals and organizations obtain from ecosystems. The EPA has defined ecological goods and services as the "outputs of ecological functions or processes that directly or indirectly contribute to social welfare or have the potential to do so in the future. Some outputs may be bought and sold, but most are not marketed" (U.S. EPA, 2006). Changes in these services can affect human well-being by affecting security, health, social relationships, and access to basic material goods (MEA, 2005).

This RIA employs reductions in nitrogen oxides (NOx) and volatile organic compound (VOC) emissions to demonstrate attainment with alternative levels of the NAAQS. Reductions in these emissions will result in changes in ambient concentrations of ozone, as well as changes in ambient concentrations of NOx, $PM_{2.5}$ and its components, and deposition of nitrogen. It is appropriate and reasonable to include all the benefits associated with these emissions reductions to provide a comprehensive understanding of the likely public welfare impacts of attaining alternative standards. Table 6-1 shows the welfare effects associated with emissions of NOx and VOC. The following subsections discuss the direct benefits of reducing ambient ozone concentrations and the additional welfare benefits associated with reduced emissions of NOx and VOC.

Table 6-1. Welfare Effects of NOx and VOC Emissions

Pollutant	Atmospheric Effects		Atmospheric and Deposition Effects		Deposition Effects		
	Vegetation Injury (Ozone)	Visibility Impairment	Materials Damage	Climate	Ecosystem Effects—(Organics)	Acidification (freshwater)	Nitrogen Enrichment
NO_x	✓	✓	✓	✓		✓	✓
VOCs	✓	✓	✓		✓		

6.2 Welfare Benefits of Reducing Ozone

Ozone can affect ecological systems, leading to changes in the ecological community and influencing the diversity, health, and vigor of individual species (U.S. EPA, 2013). Ozone causes discernible injury to a wide array of vegetation (U.S. EPA, 2013). In terms of forest productivity and ecosystem diversity, ozone may be the pollutant with the greatest potential for region-scale forest impacts (U.S. EPA, 2013). Studies have demonstrated repeatedly that ozone concentrations observed in polluted areas can have substantial impacts on plant function (De Steiguer et al,. 1990; Pye, 1988).

When ozone is present in ambient air, it can enter the leaves of plants, where it can cause significant cellular damage. Like carbon dioxide 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, 2013; 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, and 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 mycorrhizae, 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, 2013).

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, 2013; Winner, 1994). After injuries have occurred, plants may be capable of repairing the damage to a limited extent (U.S. EPA, 2013). 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, 2013). 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, 2013, 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, 2013). Biomass loss due to ozone exposure affects climate regulation by ecosystems by reducing carbon sequestration. More carbon stays in the atmosphere because carbon uptake by forests is reduced. The studies cited in the Ozone ISA demonstrate a consistent pattern of reduced carbon uptake because of ozone damage, with some of the largest reductions projected over North America (U.S. EPA, 2013).

Ozone also directly contributes to climate change because tropospheric ozone traps heat, leading to increased surface temperatures. Projections of radiative forcing due to changing ozone concentrations over the 21st century show wide variation, due in large part to the uncertainty of future emissions of source gases (U.S. EPA 2014). However, reduction of tropospheric ozone concentrations could provide an important means to slow climate change in addition to the added benefit of improving surface air quality (U.S. EPA, 2014).

While it is clear that increases in tropospheric ozone lead to warming, the precursors of ozone also have competing effects on methane, complicating emissions reduction strategies. A decrease in carbon monoxide or VOC emissions would shorten the lifetime of methane, leading to an overall cooling effect. A decrease in NO_X emissions could lengthen the methane lifetime in certain regions, leading to warming (U.S. EPA, 2014). Additionally, some strategies to reduce ozone precursor emissions could also lead to the reduced formation of aerosols (e.g., nitrates and sulfates) that currently have a cooling effect.

In this RIA, we are able to quantify only a small portion of the welfare impacts associated with reductions in ozone concentrations to meet alternative ozone standards. Using a model of commercial agriculture and forest markets, we are able to analyze the effects on consumers and producers of forest and agricultural products of changes in the W126 index resulting from meeting alternative standards within the proposed range of 70 to 65 ppb, as well as a lower standard level of 60 ppb. We also assess the effects of those changes in commercial agricultural and forest yields on carbon sequestration and storage. This analysis provides limited quantitative information on the welfare benefits of meeting these alternative standards, focused only on one subset of ecosystem services. Commercial and non-commercial forests provide a number of additional services, including medicinal uses, non-commercial food and fiber production, arts and crafts uses, habitat, recreational uses, and cultural uses for Native American tribes. A more complete discussion of these additional ecosystem services is provided in the final Welfare Risk and Exposure Assessment for Ozone (WREA) (U.S. EPA, 2014).

6.3 Additional Welfare Benefits of Strategies to Meet the Ozone NAAQS

Reductions in emissions of NOx and VOC are associated with additional welfare benefits, including reductions in nutrient enrichment and acidification impacts on sensitive aquatic and terrestrial ecosystems and improvements in visibility in state and national parks, wilderness areas, and in the areas where people live and work.

Excess nitrogen deposition can lead to eutrophication of estuarine waters, which is associated with a range of adverse ecological effects. These include 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. HAB are often toxic to fish and shellfish, lead to fish kills and aesthetic impairments of estuaries, and can in some instances 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. 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.

Nutrient enrichment from nitrogen deposition to terrestrial ecosystems is causally linked to alteration of species richness, species composition, and biodiversity (U.S. EPA, 2008b). 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, indicators, and services.

Terrestrial acidification resulting from deposition of nitrogen can result in declines in sensitive tree species, such as red spruce (*Picea rubens*) and sugar maple (*Acer saccharum*), and can also impact other plant communities including shrubs and lichen (U.S. EPA, 2008b). 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, 2008b). Terrestrial acidification affects several important ecosystem services, including declines in habitat for threatened and endangered species, declines in forest aesthetics, declines in forest productivity, and increases in forest soil erosion and reductions in water retention.

Aquatic acidification resulting from deposition of nitrogen can result in effects on health, vigor, and reproductive success for aquatic species; and effects on biodiversity. Deposition of nitrogen results in decreases in the acid neutralizing capacity and increases in inorganic aluminum concentration, which contribute to declines in zooplankton, macro invertebrates, and fish species richness in aquatic ecosystems (U.S. EPA, 2008b).

Reductions in NOx emissions will improve visibility in parks and wilderness areas and in places where people live and work because of their impact on light extinction (U.S. EPA, 2009). 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, 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). While EPA typically estimates the visibility benefits associated with reductions in NOx (U.S. EPA, 2008a), we have not done so here because we do not have estimates of the changes in particulate nitrate needed to calculate changes in light extinction and the resulting changes in economic benefits.

Strategies implemented by state and local governments to reduce emissions of ozone precursors may also impact emissions of CO_2 or other long-lived climate gases. Our ability to quantify the climate effects of the proposed standard levels is limited due to lack of available information on the energy and associated climate gas implications of control technologies assumed in the illustrative control strategy alternatives, remaining uncertainties regarding the impact of ozone precursors on climate change, and lack of available information on the co-controlled greenhouse gas (GHG) emission reductions. As a result, we do not attempt to quantify the impacts of the illustrative attainment scenarios on GHG emissions and impacts.

6.4 Analysis of Commercial Agricultural and Forestry Related Benefits Using the Forest and Agricultural Sector Optimization Model – Greenhouse Gas Version (FASOMGHG)

To estimate the commercial timber effects of ozone induced biomass loss we used the FASOMGHG (Adams et al., 2005) model for the forest and agricultural sectors to calculate the market-based welfare benefits associated with the illustrative attainment strategies for the three alternative ozone standard levels of 70, 65, and 60 ppb incremental to attainment of the current standard level of 75 ppb. The air quality surfaces used are discussed in detail in Chapter 3; the alternative primary standards modeled were recalculated to a W126 index appropriate for use with the exposure-response functions available for trees and crops. This section provides a brief summary of the analytical approach and results of the analysis. More details of the FASOMGHG modeling conducted for this RIA are provided in Appendix 6A, while additional details on the overall FASOMGHG methodology are provided in Appendix 6B of the WREA (U.S. EPA, 2014).

6.4.1 Summary of the Analytical Approach

We used the ozone exposure-response functions evaluated in the ISA (U.S. EPA, 2013) for tree seedlings to calculate relative yield loss (RYL), which is equivalent to relative biomass loss, for trees over their entire life span. The RYL for species were aggregated into average RYL for FASOMGHG forest types, based on mapping tree species to forest types using the *Atlas of United States Trees* (Little, 1971, 1976, 1977, 1978).

We used the NCLAN ozone exposure-response functions evaluated in the ISA to generate RYL for commercial agricultural crops. For those crops that do not have E-R functions, we assign them RYLs for each scenario based on the crop proxy mapping shown in Table 6A-2 of Appendix 6A.[98] The RYL for each crop are aggregated to the regional level by computing a weighted average RYL with weights determined by a county's share of production for the crop. Additional details of the calculations of RYL are provided in Appendix 6A.

Yield gains for both forest species and crops are calculated as the difference in RYL between the 75 ppb standard baseline and the attainment scenarios for 70, 65, and 60 ppb alternative standard levels. The FASOMGHG model requires estimates of yields over a time horizon from 2010 to 2040. As such, we needed to specify yield gains over this range of years. Yield gains are calculated for three periods, 2010 to 2025, 2025-2038, and post-2038. The pre-2025 period has no changes in yields. The 2025-2038 period represents the effects on the W126 index of attainment of the alternative standards across the U.S. with the exception of California. The post-2038 period includes the effects on the W126 index of attaining everywhere across the U.S. including California. There is clearly uncertainty in the path of the yield changes introduced by uncertainties about the specific time pattern of emissions reductions that will be applied in California to attain alterative standards.

Changes in yield are associated with changes in consumer and producer/farmer surplus. Consumer surplus is the difference between what a consumer would be willing to pay for a product and the price they have to pay for the product. Producer surplus refers to the benefit, or profit, a producer receives from providing a good or service at a market price when they would have been willing to sell that good or service at a lower price. In general, increases in yields will cause crop and timber prices to fall. These reductions in prices will have different impacts on consumer and producer surplus. Overall effects on producer and consumer surplus depend on the (1) ability of producers/farmers to substitute other crops that are less ozone sensitive, and (2) responsiveness of demand and supply. The FASOMGHG model estimates changes in consumer and producer surplus and net welfare (sum of consumer and producer surplus). The

[98] For oranges, rice, and tomatoes, which have ozone E-R functions that are not W126-based (they are defined based on alternative measures of ozone concentrations), we directly used the median RYG values under the "13 ppm-hrs" ozone concentration reported in Table G-7 of Lehrer et al. (2007).

FASOMGHG model also provides estimates of the changes in carbon sequestration for the commercial forestry and agricultural sectors.

The model calculates market equilibria under each of the alternative standard scenarios reflecting different forest and agricultural yields resulting from different ozone exposures. By comparing the market equilibria under different scenarios, we can calculate the welfare and carbon sequestration impacts of alternative ozone standards for the U.S. agricultural and forest sector.

6.4.2 Summary of FASOMGHG Results

Tables 6-2 and 6-4 show the estimated changes in consumer and producer surplus associated with attainment of the alternative ozone standards compared to attaining the current standard for the forestry and agricultural sectors, respectively. Tables 6-3 and 6-5 show the percent change in consumer and producer surplus associated with attainment of the alternative ozone standards compared to attaining the current standard for the forestry and agricultural sectors, respectively. Consumer and producer welfare are affected more in the forestry sector than the agricultural sector. In general, consumer welfare increases in both the forestry and agricultural sectors because higher yields lead to lower prices. Because the quantity demanded for most forestry and agricultural commodities is not highly responsive to changes in price, producer surplus often declines when lower prices reduce producer profits more than can be offset by higher yields. In other words, consumers do not increase their demand in response to the falling prices enough to offset the producer's loss of revenue. The increase in consumer welfare is not as large as the loss of producer welfare resulting in net welfare losses in the forestry sector nationally.

Table 6-2. Change in Consumer and Producer Surplus in the Forestry Sector from Attaining Alternative Ozone Standard Levels Compared to Attaining the Current Ozone Standard (Million 2011$)

	Alternative Standard Level	2010	2015	2020	2025	2030	2035	2040
		Change with Respect to Existing Standard						
Consumer Surplus	70 ppb	3	24,592	-8	111	136	225	61
	65 ppb	11	24,596	8	305	337	552	162
	60 ppb	30	24,742	136	719	533	1,094	311
		Change with Respect to Existing Standard						
Producer Surplus	70 ppb	1,203	-22,160	1,768	703	-1,167	523	-52,496
	65 ppb	211	-24,091	-480	1,464	-3,044	-39,691	-10,246
	60 ppb	-233	-23,711	-1,731	1,455	-1,963	-39,291	-8,082

Table 6-3. Percent Change in Consumer and Producer Surplus in the Forestry Sector from Attaining Alternative Ozone Standard Levels Compared to Attaining the Current Ozone Standard

	Alternative Standard Level	2010	2015	2020	2025	2030	2035	2040
		Percent Change with Respect to Existing Standard						
Consumer Surplus	70 ppb	0.00	3.10	0.00	0.01	0.02	0.03	0.01
	65 ppb	0.00	3.10	0.00	0.04	0.04	0.06	0.02
	60 ppb	0.00	3.12	0.02	0.09	0.06	0.12	0.03
		Percent Change with Respect to Existing Standard						
Producer Surplus	70 ppb	0.15%	-2.28	0.18	0.07	-0.12	0.05	-5.04
	65 ppb	0.03%	-2.48	-0.05	0.14	-0.32	-3.88	-0.98
	60 ppb	-0.03%	-2.44	-0.18	0.14	-0.20	-3.84	-0.78

Table 6-4. Change in Consumer and Producer Surplus in the Agricultural Sector from Attaining Alternative Ozone Standard Levels Compared to Attaining the Current Ozone Standard (Million 2011$)

Product	Alternative Standard Level	2010	2015	2020	2025	2030	2035	2040
		Change with Respect to Existing Standard						
Consumer Surplus	70 ppb	0	0	0	113	66	58	-10
	65 ppb	-1	1	4	262	148	289	24
	60 ppb	0	47	9	462	100	408	66
		Change with Respect to Existing Standard						
Producer Surplus	70 ppb	1,202	2,454	1,796	609	-1,144	675	-52,504
	65 ppb	216	530	-407	1,174	-2,933	-39,605	-10,211
	60 ppb	-176	1,015	-1,470	1,320	-1,492	-38,788	-8,119

Table 6-5. Percent Change in Consumer and Producer Surplus in the Agricultural Sector from Attaining Alternative Ozone Standard Levels Compared to Attaining the Current Ozone Standard

Product	Alternative Standard Level	2010	2015	2020	2025	2030	2035	2040
		Percent Change with Respect to Existing Standard						
Consumer Surplus	70 ppb	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	65 ppb	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	60 ppb	0.00	0.00	0.00	0.00	0.00	0.00	0.00
		Percent Change with Respect to Existing Standard						
Producer Surplus	70 ppb	0.17	0.30	0.22	0.07	-0.14	0.08	-5.78
	65 ppb	0.03	0.06	-0.05	0.14	-0.36	-4.53	-1.12
	60 ppb	-0.02	0.12	-0.18	0.15	-0.18	-4.44	-0.89

Since the forestry and agriculture sectors are interlinked and factors affecting one sector can lead to changes in the other, it is important to consider the overall effect of ozone changes in the context of producer and consumer welfare across both sectors. The impacts on consumer surplus are positive for both sectors, with benefits increasing with lower alternative standards. For producer surplus, however, impacts are negative for all alternative standards. Table 6-6 and 6-8 present the annualized surplus (over the period 2010 to 2040) for both sectors using 3 and 7 percent discount rates, while Table 6-7 and 6-9 present the percent change in surplus for both sectors using 3 and 7 percent discount rates.

Table 6-6. Annualized Changes in Consumer and Producer Surplus in Agriculture and Forestry from Attaining Alternative Ozone Standard Levels Compared to Attaining the Current Ozone Standard, 2010-2040, Million 2011$ (3% Discount Rate)

	Alternative Standard Level	Agriculture	Forestry	Total
		Change with Respect to Existing Standard		
Consumer surplus	70 ppb	28	4,552	4,580
	65 ppb	86	4,592	4,678
	60 ppb	132	4,744	4,877
		Change with Respect to Existing Standard		
Producer surplus	70 ppb	-3,601	-4,534	-8,135
	65 ppb	-5,035	-4,524	-9,559
	60 ppb	-4,741	-4,684	-9,425
		Change with Respect to Existing Standard		
Total surplus	70 ppb	-3,573	18	-3,555
	65 ppb	-4,949	68	-4,882
	60 ppb	-4,608	60	-4,548

Table 6-7. Annualized Percent Changes in Consumer and Producer Surplus in Agriculture and Forestry from Attaining Alternative Ozone Standard Levels Compared to Attaining the Current Ozone Standard, 2010-2040, (3% Discount Rate)

	Alternative Standard Level	Agriculture	Forestry	Total
		Percent Change with Respect to Existing Standard		
Consumer surplus	70 ppb	0.00	0.57	0.17
	65 ppb	0.00	0.57	0.17
	60 ppb	0.01	0.59	0.18
		Percent Change with Respect to Existing Standard		
Producer surplus	70 ppb	-0.44	-3.35	-0.85
	65 ppb	-0.62	-3.34	-1.00
	60 ppb	-0.58	-3.46	-0.99
		Percent Change with Respect to Existing Standard		
Total surplus	70 ppb	-0.13	0.00	-1.10
	65 ppb	-0.18	0.01	-0.13
	60 ppb	-0.17	0.01	-0.12

Table 6-8. Annualized Changes in Consumer and Producer Surplus in Agriculture and Forestry from Attaining Alternative Ozone Standard Levels Compared to

Attaining the Current Ozone Standard, 2010-2040, Million 2011$ (7% Discount Rate)

	Alternative Standard Level	Agriculture	Forestry	Total
Consumer surplus		Change with Respect to Existing Standard		
	70 ppb	21	5,570	5,592
	65 ppb	61	5,599	5,660
	60 ppb	100	5,721	5,821
Producer surplus		Change with Respect to Existing Standard		
	70 ppb	-945	-5,561	-6,506
	65 ppb	-2,719	-5,555	-8,273
	60 ppb	-2,635	-5,694	-8,328
Total surplus		Change with Respect to Existing Standard		
	70 ppb	-923	9	-914
	65 ppb	-2,658	45	-2,613
	60 ppb	-2,535	27	-2,508

Table 6-9. Annualized Percent Changes in Consumer and Producer Surplus in Agriculture and Forestry from Attaining Alternative Ozone Standard Levels Compared to Attaining the Current Ozone Standard, 2010-2040, (7% Discount Rate)

	Alternative Standard Level	Agriculture	Forestry	Total
Consumer surplus		Percent Change with Respect to Existing Standard		
	70 ppb	0.00	0.71	0.20
	65 ppb	0.00	0.72	0.21
	60 ppb	0.01	0.73	0.21
Producer surplus		Percent Change with Respect to Existing Standard		
	70 ppb	-0.12	-4.26	-0.70
	65 ppb	-0.34	-4.25	-0.89
	60 ppb	-0.33	-4.36	-0.89
Total surplus		Percent Change with Respect to Existing Standard		
	70 ppb	-0.03	0.00	-0.02
	65 ppb	-0.10	0.00	-0.07
	60 ppb	-0.09	0.00	-0.07

The impacts of the simulations of meeting the existing and alternative ozone standards on carbon sequestration potential in U.S. forest and agricultural sectors are presented in Table 6-10, in millions of metric tons of carbon dioxide equivalence, and the percent change in carbon sequestration potential is presented in Table 6-11. As shown in the table, much greater sequestration changes are projected in the forest sector than in the agricultural sector. The baseline stock of carbon storage decreases over time for agriculture because the agriculture sector GHG emissions sources are released every year and soil carbon sequestration stabilizes over the 30-year period. There are only small increases in net carbon sequestration compared to the existing standard for each of the alternative scenarios modeled.

While we did not quantify the effects in this RIA, increases in growth for trees in urban settings that results from reduced ozone concentrations can have additional benefits from removal of air pollution. The WREA (U.S. EPA, 2014) provides a case study of the potential impacts of ozone reductions on pollution removal in several eastern U.S. urban areas using the i-Tree model. See appendix 6D of the WREA (U.S. EPA, 2014) for details and references for the i-Tree model.

Table 6-10. Increase in Carbon Sequestration from Attaining Alternative Ozone Standard Levels Compared to Attaining the Current Ozone Standard, MMtCO_2e

	Alternative Standard Level	2010	2020	2030	2040	2010-2040*
Agriculture		**Change with Respect to Existing Standard**				
	70 ppb	0	1	3	1	24
	65 ppb	0	2	6	4	62
	60 ppb	1	4	11	11	132
Forestry		**Change with Respect to Existing Standard**				
	70 ppb	-1	-51	100	259	1,537
	65 ppb	-2	-53	323	774	5,207
	60 ppb	-12	-94	465	1,189	7,739

Table 6-11. Percent Change in Carbon Sequestration from Attaining Alternative Ozone Standard Levels Compared to Attaining the Current Ozone Standard

	Alternative Standard Level	2010	2020	2030	2040	2010-2040*
Agriculture		Percent Change with Respect to Existing Standard				
	70 ppb	0.00	0.00	0.03	0.01	0.01
	65 ppb	0.00	0.01	0.05	0.06	0.02
	60 ppb	0.00	0.03	0.10	0.13	0.05
Forestry		Percent Change with Respect to Existing Standard				
	70 ppb	0.00	-0.07	0.13	0.32	0.10
	65 ppb	0.00	-0.07	0.41	0.95	0.34
	60 ppb	-0.02	-0.13	0.60	1.47	0.50

6.5 References

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APPENDIX 6A: METHODS AND DATA USED TO DEVELOP ESTIMATES OF OZONE EFFECTS ON CROP AND FOREST PRODUCTIVITY

Incorporating the impacts of different ambient ozone concentration levels into FASOMGHG requires determining crop yield and forest productivity impacts associated with changes in concentrations. Productivity impacts are required for each crop/region and forest type/region combination included within the model. In this section, we describe our methods for calculating relative yield losses (RYLs) and relative yield gains (RYGs) of crops and tree species under alternative ambient ozone concentration levels.

These data are essential for our market analysis because crop and forest yields play an important role in determining the economic returns to agricultural and forest production activities. Thus, they affect landowner decisions regarding land use, crop mix, forest rotation lengths, production practices, and others. Alterations in ambient ozone concentration levels will therefore change the supply curves of U.S. agricultural and forest commodities, resulting in new market equilibriums. Because both the changes in ozone concentrations and the distribution of ozone-sensitive crops and tree species vary spatially, there may be substantial differences in the net impacts across regions. There may also be distributional impacts as commodity production shifts between regions in response to changes in relative productivity.

6A.1 Methodology

There are several alternative metrics used for assessing ozone concentrations (see Lehrer et al. [2007] for more information). For this assessment, we are using the W126 metric, which is a weighted sum of all ozone concentrations observed from 8 a.m. to 8 p.m available in 2025 and 2038. More specifically, we are using W126 ozone concentration surfaces generated using enhanced Voronoi Neighbor Averaging (eVNA). W126 concentration surfaces based on meeting the current ozone standard[99] were provided by EPA in the previous iteration of this project (2013) and served again as the baseline for this analysis. According to information provided by EPA, the eVNA W126 ozone surface is built from monitor data fused with Community

[99]The current primary and secondary ozone standards are 75 parts per billion (ppb) based on the annual fourth-highest daily maximum 8-hr concentration, averaged over 3 years. For the purposes of calculating impacts on crop yields and forest growth rates, we used the W126 equivalent of the current standard.

Multiscale Air Quality (CMAQ) model-based gradient interpolations. The spatial resolution of the ozone surface in ArcGIS Shapefile format is 12 km.

County-level values were extracted from the eVNA W126 ozone surface using ArcGIS. Only the ozone concentrations for the cropland and forestland portions of the W126 ozone surface are used to derive the county-level average crop and forest W126 ozone levels, respectively. These weighting adjustments were made to better reflect the ozone concentration that would affect the specific portions of each county containing forested land or cropland, rather than basing county-level exposure on the ozone concentration across the whole county. Data from the 2011 USGS National Land Cover Database (NLCD), updated from the previously used 2006 NLCD data are used to extract the cropland and forestland portions from the ozone surface (Jin et al., 2013). The maps below demonstrate the change in forest and cropland area resulting from updating the data from 2006 to 2011.

Table 6A-1. Comparison of Total Cropland and Forestland NLCD Area (sq m)

	2006 NLCD	2011 NLCD	% Increase
Cropland Area	1,730,787,687,000	1,786,333,937,400	3.21
Forestland Area	1,938,825,202,500	1,963,044,107,100	1.25

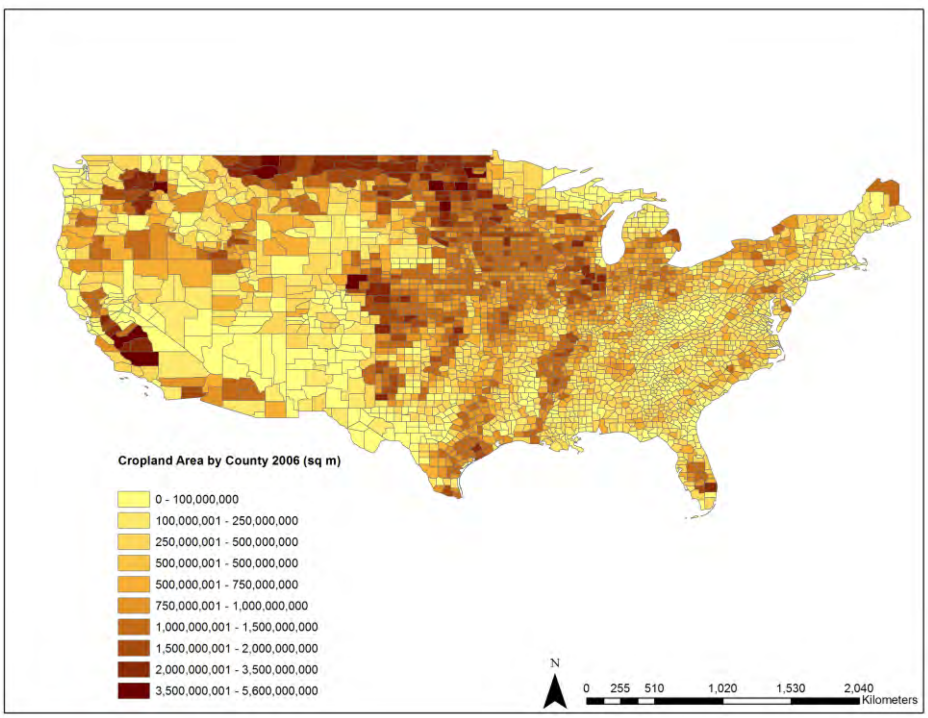

Figure 6A-1. Cropland Area (sq m) by County according to NLCD 2006 Data

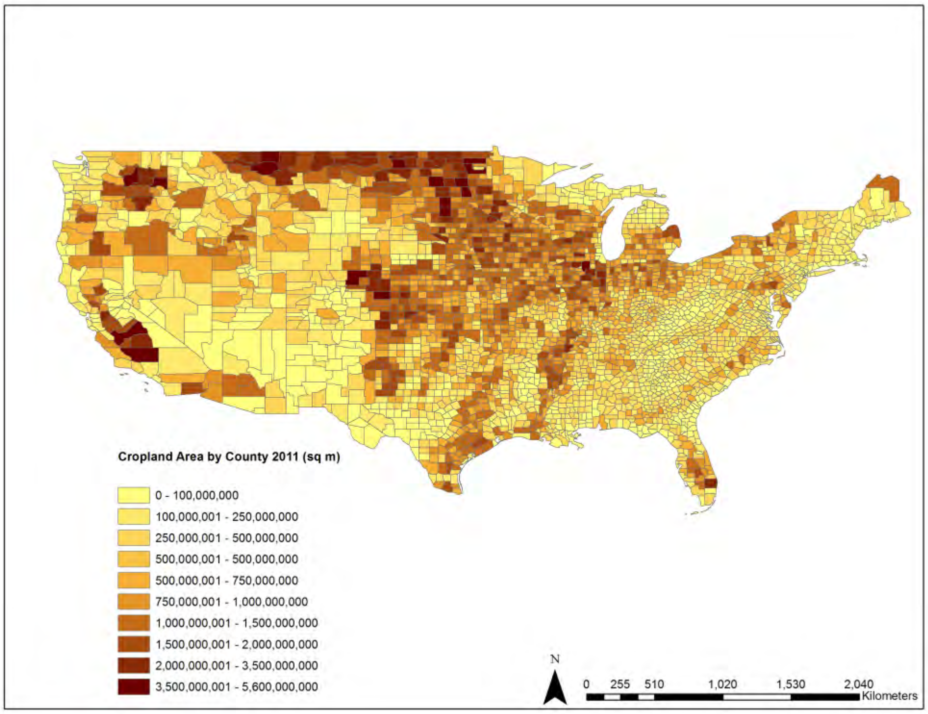

Figure 6A-2. Cropland Area (sq m) by County according to NLCD 2011 Data

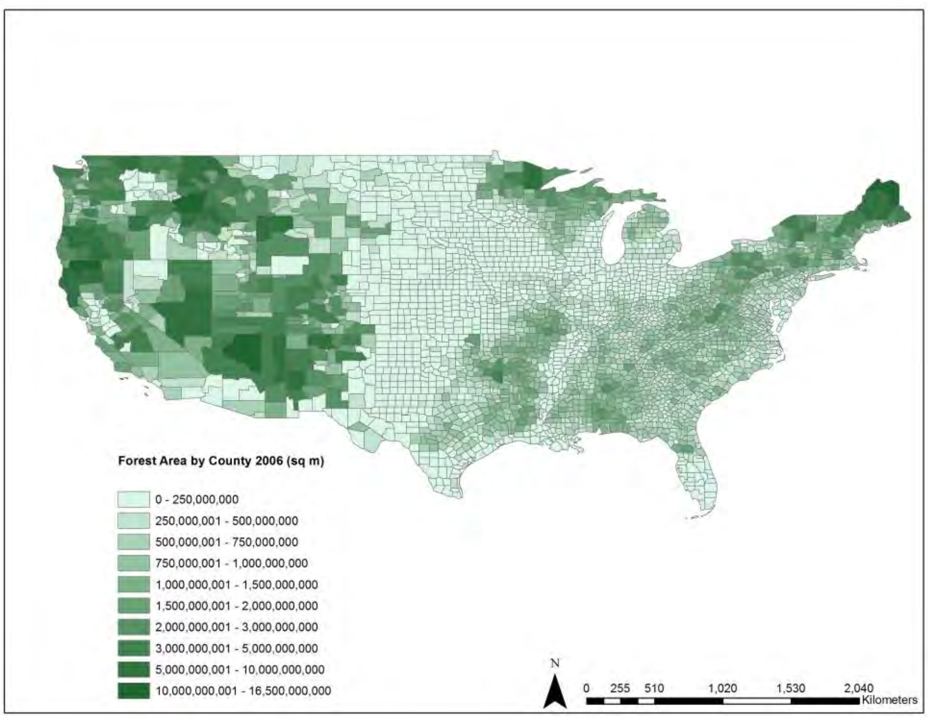

Figure 6A-3. Forest Area (sq m) by County according to NLCD 2006 Data

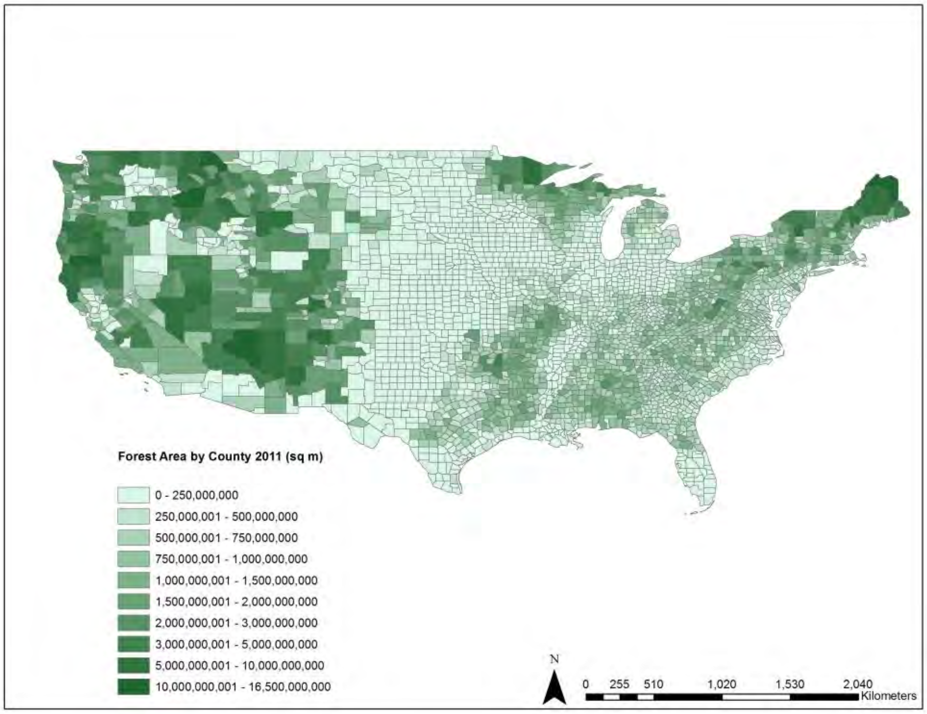

Figure 6A-4. Forest Area (sq m) by County according to NLCD 2011 Data

6A.1.1 Calculation of Relative Yield Loss

The median W126 ozone concentration response (CR) functions for crops and tree seedlings in the 2007 EPA technical report (Lehrer et al., 2007) are used to calculate the RYLs for crops and tree species under each ambient ozone concentration scenario used in this analysis.

Table 6A-2 presents the α and β parameters being used in the W126 ozone CR function for different crops and tree species. The W126 ozone CR function is as follows: $RYL = 1 - e^{-(W126/\alpha)^{\beta}}$.

Table 6A-2. Parameter Values Used for Crops and Tree Species

	α	β
Crops		
Corn	98.3	2.973
Sorghum	205.9	1.963
Soybean	110.0	1.367
Winter wheat	53.7	2.391
Potato	99.5	1.242
Cotton	94.4	1.572
Tree Species		
Ponderosa	159.63	1.1900
Red alder	179.06	1.2377
Black cherry	38.92	0.9921
Tulip poplar	51.38	2.0889
Sugar maple	36.35	5.7785
Eastern white	63.23	1.6582
Red maple	318.12	1.3756
Douglas fir	106.83	5.9631
Quaking aspen	109.81	1.2198
Virginia pine	1,714.64	1.0000

6A.1.1.1 Relative Yield Loss for Crops

Specifically, for crops, we first calculate the FASOMGHG subregion RYLs for crops that have W126 ozone CR functions using the subregion-level, cropland-based ozone concentration values under each scenario. The FASOMGHG subregion-level ozone concentration values are initially calculated for all crops as the simple averages of the county-level ozone concentration values. For crops that do not have W126 ozone CR functions, we assign them W126 ozone CR functions based on the crop proxy mapping shown in Table 6A-3. This crop mapping was based on the authors' judgment and previous experience.[100] In addition, for oranges, rice, and tomatoes, which have ozone CR functions that are not W126-based (they are defined based on alternative measures of ozone levels), we directly used the median RYG values under the 13 ppm-hr ozone level reported in Table G-7 of Lehrer et al. (2007). More details on RYG are presented in further subsections.

[100] Also, note that FASOMGHG defines short-rotation woody trees such as hybrid poplar and willow as crops. Ozone impacts on short-rotation woody trees were based on ozone RYLs for aspen.

Table 6A-3. Mapping of Ozone Impacts on Crops to FASOMGHG Crops

Crops Used for Estimating Ozone Impacts	FASOMGHG Crops
W126 Crops	
Corn	Corn
Cotton	Cotton
Potatoes	Potatoes
Winter wheat	Soft white wheat, hard red winter wheat, soft red winter wheat, durum wheat, hard red spring wheat, oats, barley, rye, sugar beet, grazing wheat, and improved pasture
Sorghum	Sorghum, silage, hay, sugarcane, switchgrass, miscanthus, energy sorghum, and sweet sorghum
Soybeans	Soybeans and canola
Aspen (tree)	Hybrid poplar, willow (FASOMGHG places short-rotation woody biomass production in the crop sector rather than in the forest sector)
Non-W126 Crops	
Oranges	Orange fresh/processed, grapefruit fresh/processed
Rice	Rice
Tomatoes	Tomato fresh/processed

Moreover, for crops that have county-level production data and W126 ozone CR functions (including functions based on proxy crops), we updated the RYLs with production-weighted W126 values. The 2012 USDA Census of Agriculture (Ag Census) county-level production data are used to derive the weighted FASOMGHG subregion RYLs, following Formula (6A.1).

$$wRYL_{ik} = Ozone\ CR\ Function_k\left(\frac{\sum_j Prod_{ijk} * W126_{ij}}{\sum_j Prod_{ijk}}\right), \quad (6A.1)$$

where i denotes FASOMGHG subregion, j indicates county, and k represents crop. Ozone CR Function$_k$ refers to the ozone concentration response function for crop k. Prod$_{ijk}$ represents the county-level production level of crop k, and W126$_{ij}$ represents the cropland-based ozone value for county j in subregion i. Finally, wRYL$_{ik}$ stands for the weighted FASOMGHG subregion RYL for crop k. RYLs are calculated for each ozone concentration level being considered.

6A.1.1.2 Relative Yield Loss for Trees

The ozone CR functions for tree seedlings were used to calculate RYLs for FASOMGHG trees over their whole life span. To derive the FASOMGHG region-level RYLs for trees under each ozone concentration scenario, we used FASOMGHG region ozone values and the mapping in Table 6A-4.

Table 6A-4. Mapping of Ozone Impacts on Forests to FASOMGHG Forest Types

Tree Species Used for Estimating Ozone Impacts	FASOMGHG Forest Type	FASOMGHG Region(s)
Black cherry, tulip poplar	Upland hardwood	SC, SE
Douglas fir	Douglas fir	PNWW
Eastern white pine	Softwood	CB, LS
Ponderosa pine	Softwood	PNWE, PNWW, PSW, RM
Quaking aspen	Hardwood	RM
Quaking aspen, black cherry, red maple, sugar maple, tulip poplar	Hardwood	CB, LS, NE
Red alder	Hardwood	PNWE, PNWW, PSW
Red maple	Bottomland hardwood	SC, SE
Virginia pine	Natural pine, oak-pine, planted pine	SC
Virginia pine, eastern white pine	Natural pine, oak-pine, planted pine	SE
Virginia pine, eastern white pine	Softwood	NE

Note: CB = Corn Belt; LS = Lake States; NE = Northeast; PNWE = Pacific Northwest—East side; PNWW = Pacific Northwest—West side; PSW = Pacific Southwest; RM = Rocky Mountains; SC = South Central; SE = Southeast.

Specifically, the FASOMGHG region-level RYLs are first calculated for each tree species listed in first column of Table 6A-4. Then, a simple average of RYLs for each tree species mapped to a FASOMGHG forest type in a given region is calculated. The mapping of tree species to FASOMGHG forest types is based on Elbert L. Little, Jr.'s *Atlas of United States Trees* (1971, 1976, 1977, 1978). Note that crop RYLs are generated at the FASOMGHG subregion level, whereas forest RYLs are calculated at the FASOMGHG region level, consistent with the greatest level of regional disaggregation available for these sectors within FASOMGHG.

6A.1.1.3 Calculation of Relative Yield Gain

As described by Lehrer et al. (2007), the RYL is the relative yield loss compared with the baseline yield under a "clean air" environment. For implementation within FASOMGHG, we calculate the RYG for crops and trees from moving between ambient ozone concentrations (i.e., RYG is calculated as a change in RYL when moving between scenarios).

Thus, to obtain the RYG for crops and trees under alternative ozone concentrations, we need the RYLs under each scenario. For example, to derive RYG under the current standard 75 ppb scenario relative to current conditions "currcond," we use Formula (6A.2):

$$RYG_{75ppb} = \frac{1-RYL_{75ppb}}{1-RYL_{currcond}} - 1 = \frac{RYL_{currcond}-RYL_{75ppb}}{1-RYL_{currcond}} \tag{6A.2}$$

The FASOMGHG subregion-level crop RYGs and the FASOMGHG region-level tree RYGs for changes associated with moving from one scenario to another were calculated for additional comparisons in the same way.

6A.1.2 Conducting Model Scenarios in FASOMGHG

The current crop/forest budgets included in FASOMGHG are assumed to reflect input/output relationships under current ambient ozone concentrations as these budgets are based on historical data. To model the effects of changing ozone concentrations on the agricultural and forest sectors, the following five scenarios were constructed and run through the model:

1. "Current Conditions" scenario, where no RYGs of crops and trees are considered (assumed to be consistent with current ambient ozone concentration levels);
2. 75 ppb scenario, where crop and forest yields are assumed to increase by the percentages calculated in RYG_{75ppb}, calculated relative to the current scenario;
3. 70 ppb scenario, using RYG_{70ppb}, calculated relative to both current and 75 ppb scenarios
4. 65 ppb scenario, using RYG_{65ppb}, calculated relative to both current and 75 ppb scenarios
5. 60 ppb scenario, using RYG_{60ppb}, calculated relative to both current and 75 ppb scenarios

The time scope of the FASOMGHG model scenarios used for these analyses is 2000–2050, solved in 5-year time steps.[101] The crop and tree RYGs are introduced into the model starting in 2025. During the time-steps of 2025 and 2030, crop and tree RYGs are assumed to be same as the RYGs in 2025. After 2040, RYGs remain constant at the RYGs obtained in 2038. The special case is in 2035, for which the weighted average of RYGs in 2025 and 2038 (60% for RYGs in 2025 and 40% for RYGs in 2038) are assigned to RYGs in 2035. Figure 6A-5 presents

[101] Because of terminal period effects, the model is run out to the 2050 time period, but only results through the 2040 model time period (representative of 2040–2044) are used in the analyses.

the modeling process of simulating ozone scenarios using FASOMGHG. The changes in crop and tree yield growth potentially lead to new market equilibriums for agricultural and forestry commodities, as well as land use changes between agricultural and forestry uses and consequently GHG emissions and sequestration changes.

By comparing the market equilibriums under different scenarios, we can calculate the welfare, land use, and GHG impacts of alternative ozone standards on the U.S. agricultural and forest sector, including changes in consumer and producer welfare, land use allocation, and GHG mitigation potential over time.

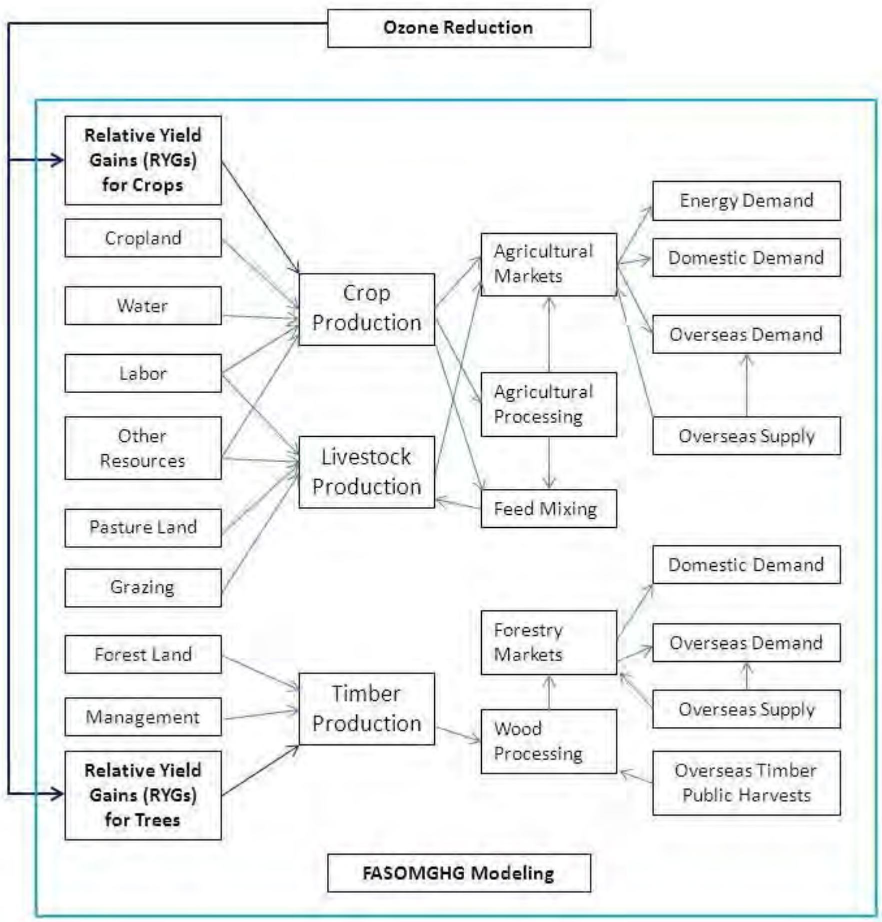

Figure 6A-5. FASOMGHG Modeling Flowchart

6A.1.3 Data Inputs

In this section, we summarize the input data used in the FASOMGHG scenarios specified for this assessment. Following the methods described above, we calculated W126 ozone concentration levels by region and crop. Effects on crop yields and forest productivity were calculated for each FASOMGHG region. We present the values used as model inputs in tabular and map format, with a primary focus here on comparison of the more stringent scenarios to the current standard.

6A.1.3.1 Ambient Ozone Concentration Data

The county-level forested and cropland W126 ozone values were aggregated at regional and sub-regional levels, respectively.

Table 6A-5. Forestland W126 Ozone Values under Alternative Scenarios

		2025				2038			
FASOMGHG Region	C.C.[a]	**75 ppb[b]**	**70 ppb**	**65 ppb**	**60 ppb**	**75 ppb[b]**	**70 ppb**	**65 ppb**	**60 ppb**
CB	11.84	5.24	4.58	3.11	1.99	5.24	4.58	3.10	1.99
GP [c]	8.95	5.15	4.47	3.73	2.87	5.06	4.37	3.63	2.79
LS	6.33	2.83	2.71	2.23	1.83	2.83	2.71	2.23	1.83
NE	8.48	3.65	3.19	2.29	1.64	3.65	3.19	2.29	1.64
PNWE	5.55	2.24	2.24	2.22	2.19	2.16	2.14	2.10	2.06
PNWW	3.79	1.49	1.49	1.49	1.49	1.49	1.48	1.48	1.48
PSW	17.28	6.65	6.65	6.64	6.54	4.52	3.61	2.91	2.45
RM	13.36	7.32	7.29	6.83	5.75	6.69	6.57	6.08	5.07
SC	11.84	4.12	3.46	2.48	1.75	4.11	3.46	2.48	1.75
SE	13.10	3.03	2.81	2.26	1.82	3.03	2.81	2.26	1.82
SW[c]	10.03	6.08	4.58	3.43	2.47	6.04	4.55	3.40	2.44

[a] Current Conditions

[b] Current Standard

[c] GP and SW are modeled as agriculture-only regions in FASOMGHG

Note: CB = Corn Belt; GP = Great Plains; LS = Lake States; NE = Northeast; PNWE = Pacific Northwest—East side; PNWW = Pacific Northwest—West side; PSW = Pacific Southwest; RM = Rocky Mountains; SC = South Central; SE = Southeast; SW = Southwest.

Table 6A-5 displays the agricultural W126 ozone values at the sub-region level. Similar to the forest ozone values, the Pacific Southwest, South Central, Southeast, and Rocky Mountains regions experience the greatest agricultural ozone reductions under the 75 ppb scenario compared with current conditions. The Corn Belt, Southwest, and Northwest regions also see noteworthy agricultural ozone reductions.

Table 6A-6. Cropland W126 Ozone Values under Modeled Scenarios

			2025				2038			
FASOMGHG Region	Sub-region	C.C.[a]	75 ppb[b]	70 ppb	65 ppb	60 ppb	75 ppb[b]	70 ppb	65 ppb	60 ppb
CB	Illinois, Northern	7.84	3.90	3.51	2.47	1.65	3.90	3.51	2.46	1.65
	Illinois, Southern	11.46	6.54	5.79	3.86	2.38	6.53	5.79	3.86	2.38
	Indiana, Northern	10.38	4.65	4.43	3.03	1.99	4.64	4.43	3.02	1.99
	Indiana, Southern	13.13	6.66	6.40	4.22	2.54	6.65	6.40	4.22	2.54
	Iowa, Central	5.71	2.79	2.46	2.06	1.65	2.79	2.46	2.06	1.65
	Iowa, Northeast	6.23	2.69	2.42	2.02	1.67	2.69	2.42	2.02	1.67
	Iowa, Southern	6.65	3.31	2.68	2.07	1.53	3.31	2.68	2.06	1.53
	Iowa, Western	5.74	3.33	2.90	2.51	2.07	3.31	2.90	2.49	2.07
	Missouri	11.50	5.17	4.01	2.76	1.81	5.17	4.01	2.76	1.81
	Ohio, Northeast	12.71	5.34	5.08	3.51	2.35	5.34	5.08	3.51	2.35
	Ohio, Northwest	12.07	5.89	5.65	3.88	2.54	5.88	5.65	3.87	2.54
	Ohio, Southern	13.49	5.93	5.73	3.64	2.12	5.93	5.73	3.64	2.12
GP[c]	Kansas	10.93	8.17	6.88	5.54	4.12	8.02	6.74	5.39	4.03
	Nebraska	8.87	5.81	5.27	4.41	3.24	5.65	5.12	4.26	3.13
	North Dakota	4.46	3.03	3.00	2.86	2.72	3.03	2.96	2.86	2.69
	South Dakota	5.66	3.73	3.50	3.12	2.68	3.67	3.43	3.09	2.62
LS	Michigan	9.50	5.20	4.96	3.57	2.50	5.20	4.96	3.57	2.50
	Minnesota	4.97	2.63	2.49	2.31	2.12	2.63	2.49	2.31	2.11
	Wisconsin	7.02	3.04	2.85	2.25	1.78	3.04	2.85	2.25	1.78

(continued)

			2025				2038			
FASOMGHG Region	**Sub-region**	**C.C.[a]**	**75 ppb**	**70 ppb**	**65 ppb**	**60 ppb**	**75 ppb**	**70 ppb**	**65 ppb**	**60 ppb**
NE	Connecticut	11.90	4.53	3.69	2.71	1.85	4.53	3.69	2.71	1.85
	Delaware	17.45	7.88	6.29	4.15	2.58	7.88	6.29	4.15	2.58
	Maine	3.70	1.62	1.47	1.30	1.20	1.62	1.47	1.30	1.20
	Maryland	17.20	7.37	5.88	3.77	2.29	7.37	5.87	3.77	2.29
	Massachusetts	10.23	2.98	2.39	1.69	1.18	2.98	2.39	1.69	1.18
	New Hampshire	5.84	1.99	1.73	1.32	1.06	1.99	1.73	1.32	1.06
	New Jersey	16.70	6.95	5.42	3.49	2.19	6.93	5.42	3.49	2.19
	New York	8.36	3.93	3.53	2.78	2.18	3.93	3.53	2.78	2.18
	Pennsylvania	12.14	5.37	4.42	2.98	1.97	5.37	4.42	2.98	1.97
	Rhode Island	11.72	4.47	3.71	2.69	1.85	4.47	3.71	2.69	1.85
	Vermont	5.51	2.01	1.84	1.55	1.35	2.01	1.84	1.55	1.34
	West Virginia	10.74	4.63	4.24	2.73	1.68	4.63	4.24	2.73	1.68
PNWE	Oregon	6.66	2.64	2.64	2.62	2.55	2.53	2.48	2.43	2.35
	Washington	4.96	1.79	1.79	1.79	1.78	1.78	1.78	1.77	1.77
PNWW		3.59	1.35	1.35	1.36	1.36	1.34	1.34	1.35	1.35
PSW	California, Northern	21.86	9.61	9.61	9.58	9.49	6.20	4.54	3.21	2.35
	California, Southern	21.46	12.49	12.48	12.43	12.31	6.20	5.52	4.82	4.23

(continued)

FASOMGHG Region	Sub-region	C.C.[a]	2025				2038			
			75 ppb	70 ppb	65 ppb	60 ppb	75 ppb	70 ppb	65 ppb	60 ppb
RM	Arizona	13.53	9.63	9.63	9.17	7.91	8.24	8.11	7.51	6.24
	Colorado	16.33	10.71	10.41	8.98	6.39	10.23	9.88	8.46	6.06
	Idaho	12.75	5.05	5.04	4.79	4.15	4.62	4.57	4.30	3.75
	Montana	6.61	3.25	3.25	3.15	3.00	3.16	3.16	3.09	2.89
	Nevada	15.48	6.57	6.56	6.29	5.58	5.58	5.25	4.75	4.03
	New Mexico	12.58	8.54	8.15	7.40	6.20	8.31	7.89	7.11	5.91
	Utah	18.06	8.62	8.61	7.81	5.85	7.53	7.40	6.58	4.95
	Wyoming	14.28	7.56	7.50	6.79	5.20	7.02	6.91	6.22	4.76
SC	Alabama	13.16	2.89	2.70	2.29	1.91	2.89	2.70	2.29	1.91
	Arkansas	12.30	4.81	3.76	2.57	1.71	4.81	3.76	2.57	1.70
	Kentucky	13.79	5.59	5.35	3.52	2.16	5.59	5.34	3.52	2.16
	Louisiana	9.59	4.62	3.58	2.66	1.95	4.62	3.58	2.66	1.95
	Mississippi	11.21	3.32	2.75	2.11	1.55	3.32	2.75	2.11	1.54
	Tennessee	15.55	4.57	4.22	2.85	1.83	4.57	4.22	2.85	1.83
	Texas, East	9.92	5.67	3.88	2.79	1.99	5.64	3.87	2.77	1.98
SE	Florida	9.16	2.95	2.86	2.76	2.60	2.95	2.86	2.76	2.60
	Georgia	12.79	2.56	2.45	2.21	1.95	2.56	2.45	2.21	1.95
	North Carolina	14.31	3.35	3.08	2.49	2.00	3.35	3.08	2.49	2.00
	South Carolina	12.82	2.18	2.07	1.77	1.52	2.18	2.07	1.77	1.51
	Virginia	12.62	3.95	3.47	2.35	1.59	3.95	3.47	2.35	1.59

(continued)

FASOMGHG Region	Sub-region	C.C.[a]	2025 75 ppb	2025 70 ppb	2025 65 ppb	2025 60 ppb	2038 75 ppb	2038 70 ppb	2038 65 ppb	2038 60 ppb
SW[c]	Oklahoma	11.52	7.44	5.72	4.31	3.07	7.38	5.67	4.25	3.04
	Texas, Central Blacklands	9.17	5.45	4.06	3.05	2.18	5.43	4.05	3.03	2.16
	Texas, Coastal Bend	7.24	5.23	4.13	3.24	2.44	5.23	4.12	3.23	2.42
	Texas, Edwards Plateau	9.02	6.38	5.24	4.28	3.38	6.32	5.17	4.20	3.31
	Texas, High Plains	11.74	8.49	7.50	6.44	5.18	8.36	7.38	6.27	5.03
	Texas, Rolling Plains	10.64	7.13	5.71	4.56	3.45	7.07	5.66	4.48	3.38
	Texas, South	4.33	3.45	2.91	2.48	2.09	3.44	2.90	2.47	2.05
	Texas, Trans Pecos	11.83	7.49	7.09	6.61	5.86	7.38	6.96	6.44	5.68

[a] Current Conditions

[b] Current Standard

[c] GP and SW are modeled as agriculture-only regions in FASOMGHG

Figure 6A-6 presents the incremental ozone reductions under alternative 2038 ozone standards with respect to the current 75 ppb standard. As the standard is tightened from the 70 ppb scenario to the 60 ppb scenario, the greatest ozone reductions are observed in the Pacific Southwest region. Central parts of the Rocky Mountains region extending into northern areas of the Southwest and South central regions also see substantial ozone reductions. These ozone reductions would affect the production of crops and timber that are susceptible to ground-level ozone in these regions.

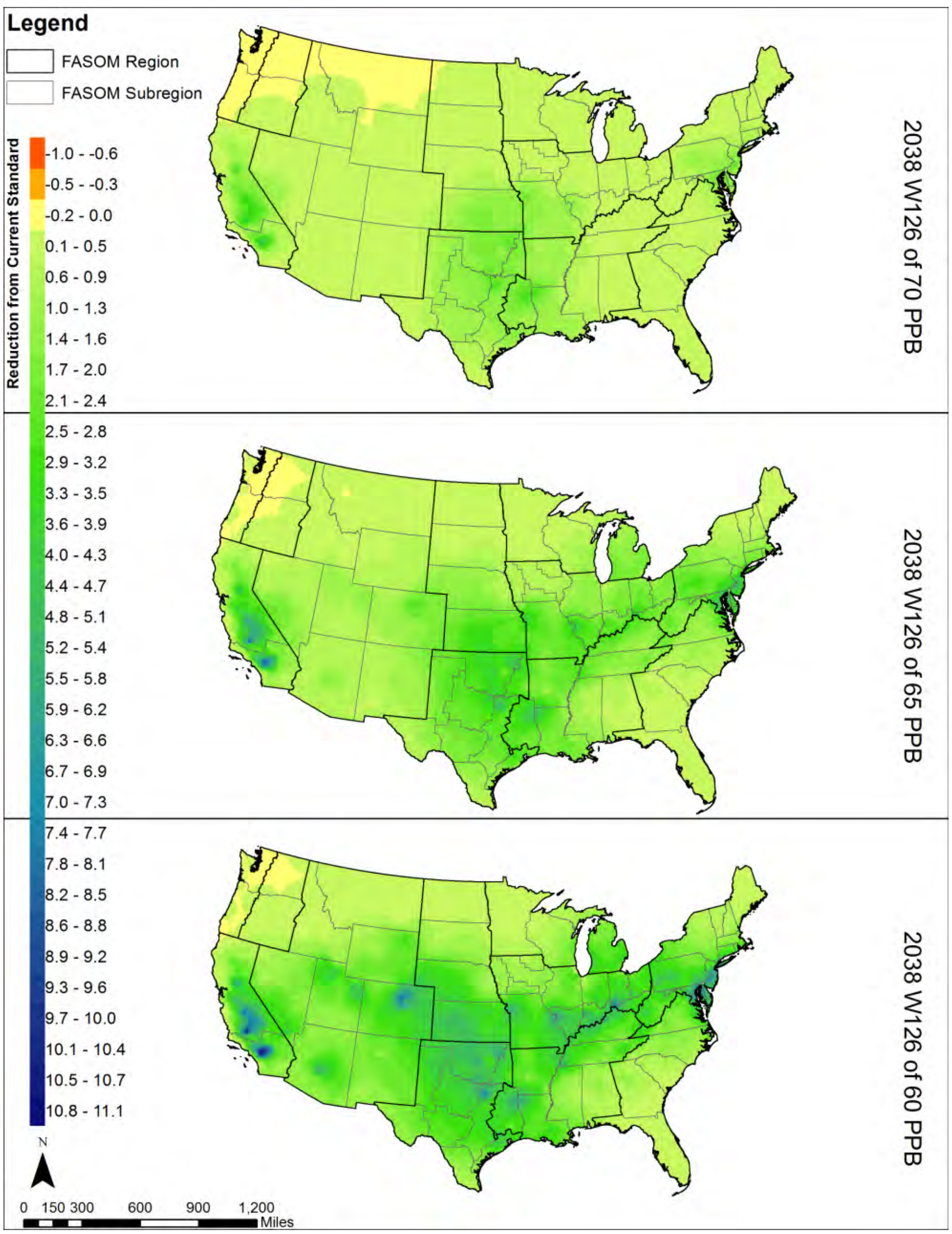

Figure 6A-6. Ozone Reductions with Respect to 75 ppb under Alternative Scenarios

6A.1.3.2 Changes in Crop and Forest Yields with Respect to 75 ppb Scenario

Figures 6A-7 through 6A-12 display major crops' RYGs under alternative ozone standards scenarios at the FASOMGHG subregion level, with respect to the current 75 ppb standard. These are the values that were directly incorporated into FASOMGHG to define the scenarios modeled. Figures 6A-13 and 6A-14 display changes in forest RYGs. As discussed previously, the Rocky Mountains, Corn Belt, and parts of the southern regions of the United States (e.g., within the Pacific Southwest, South Central, and Southeast regions) are shown to experience the most significant further ozone reductions under the alternative policy scenarios. Hence, one would expect to see the most sizable increases in RYGs for crops and tree species grown in those regions. This finding is consistent with our calculations.

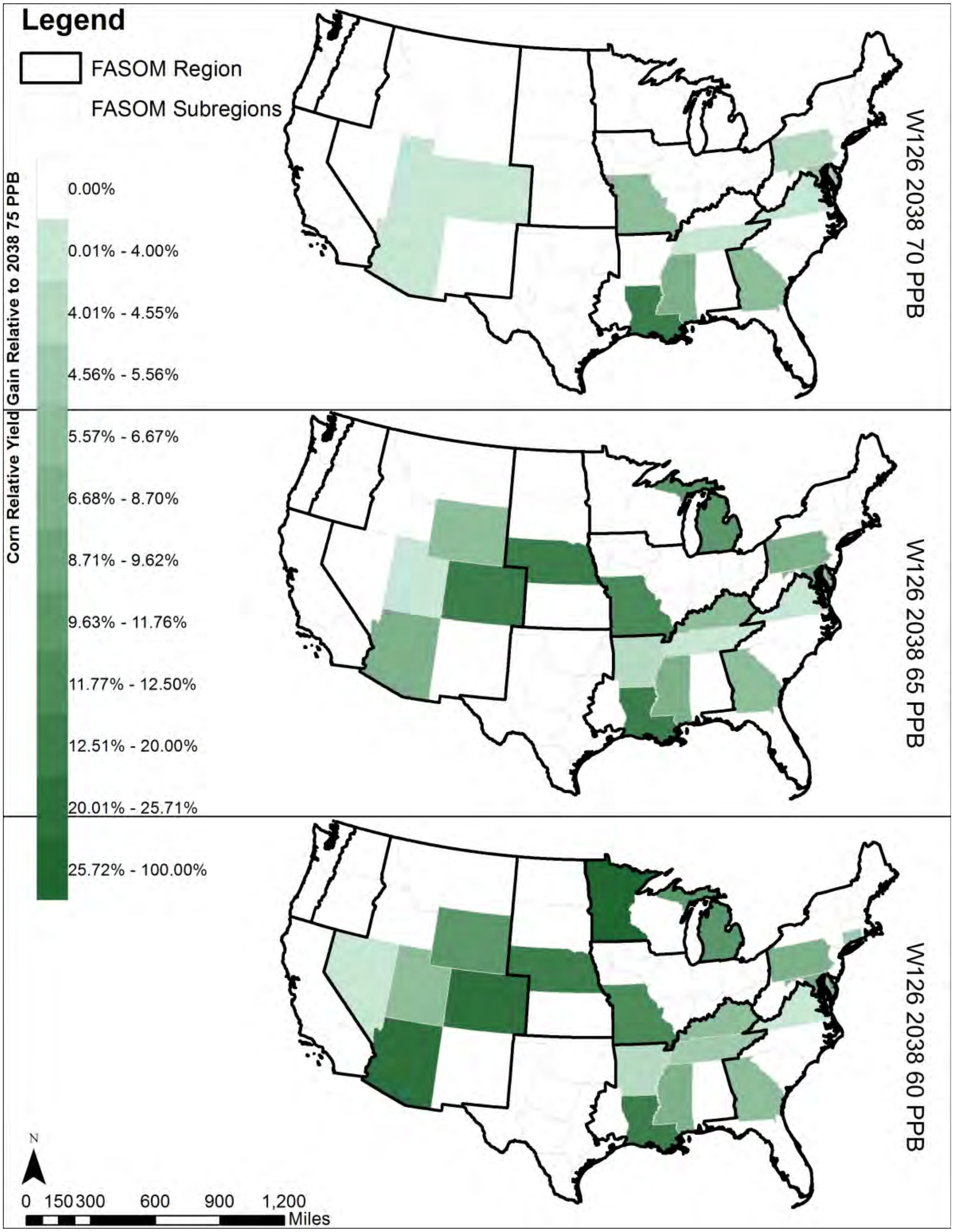

Figure 6A-7. Percentage Changes in Corn RYGs with Respect to the 75 ppb Scenario

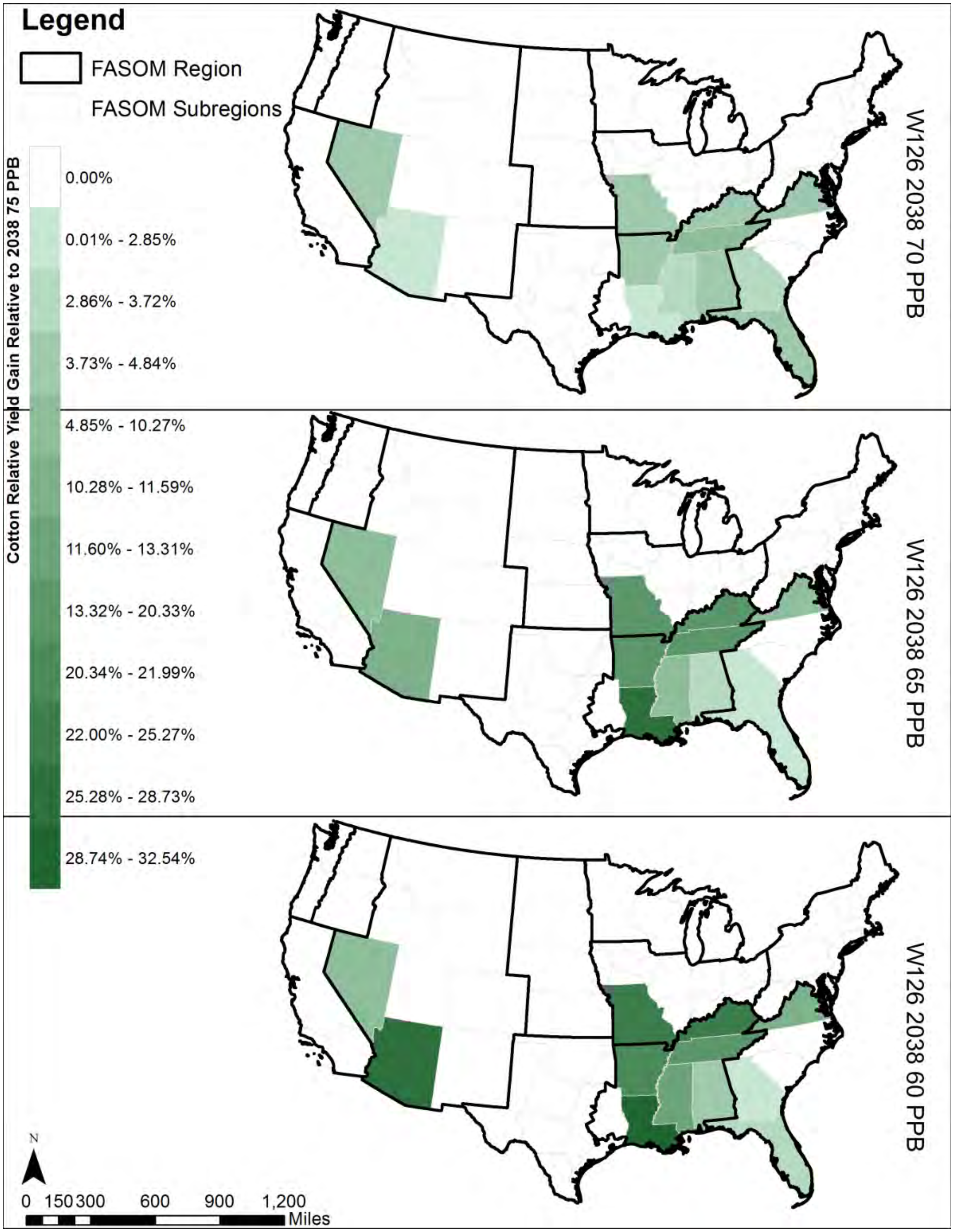

Figure 6A-8. Percentage Changes in Cotton RYGs with Respect to the 75 ppb Scenario

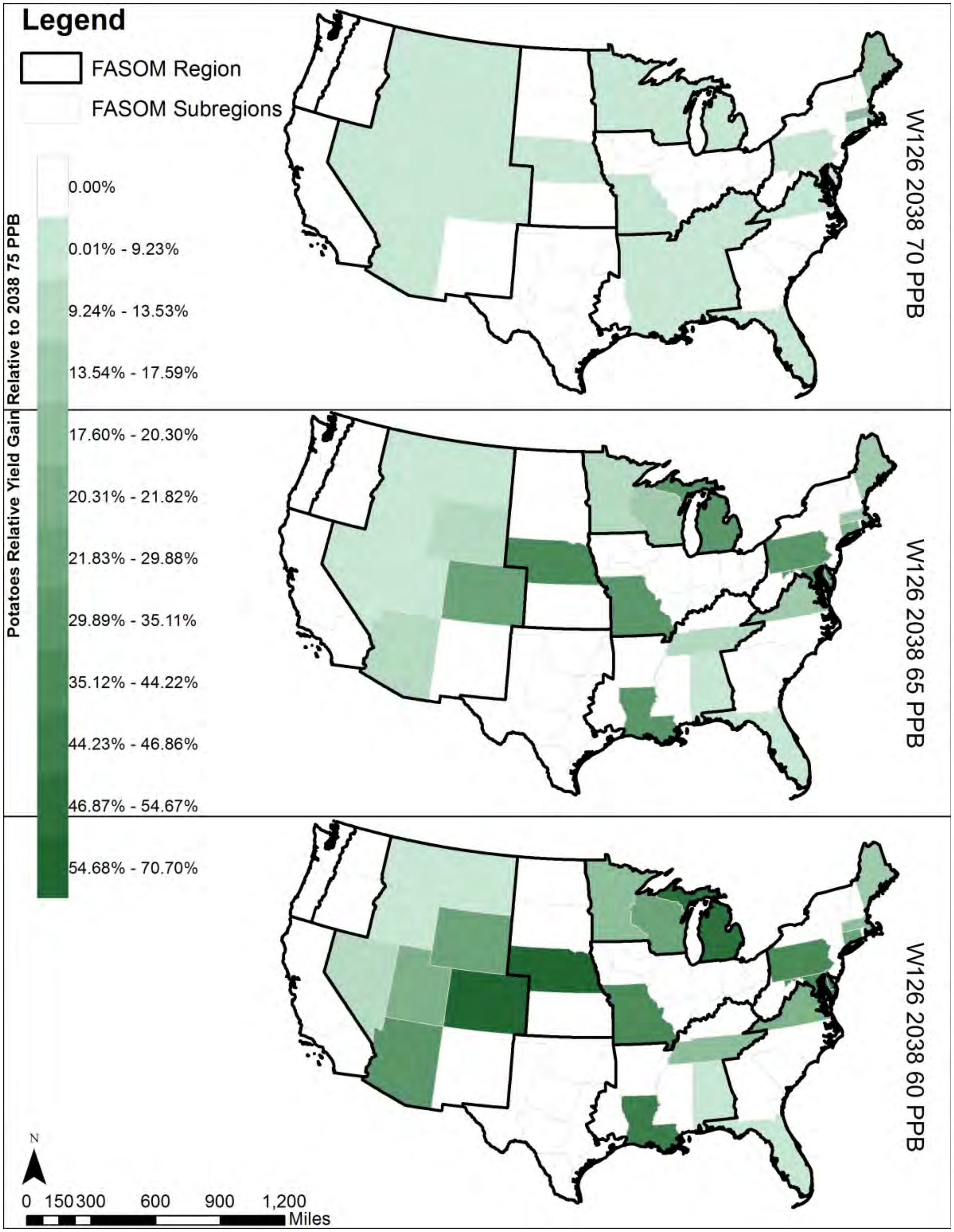

Figure 6A-9. Percentage Changes in Potato RYGs with Respect to the 75 ppb Scenario

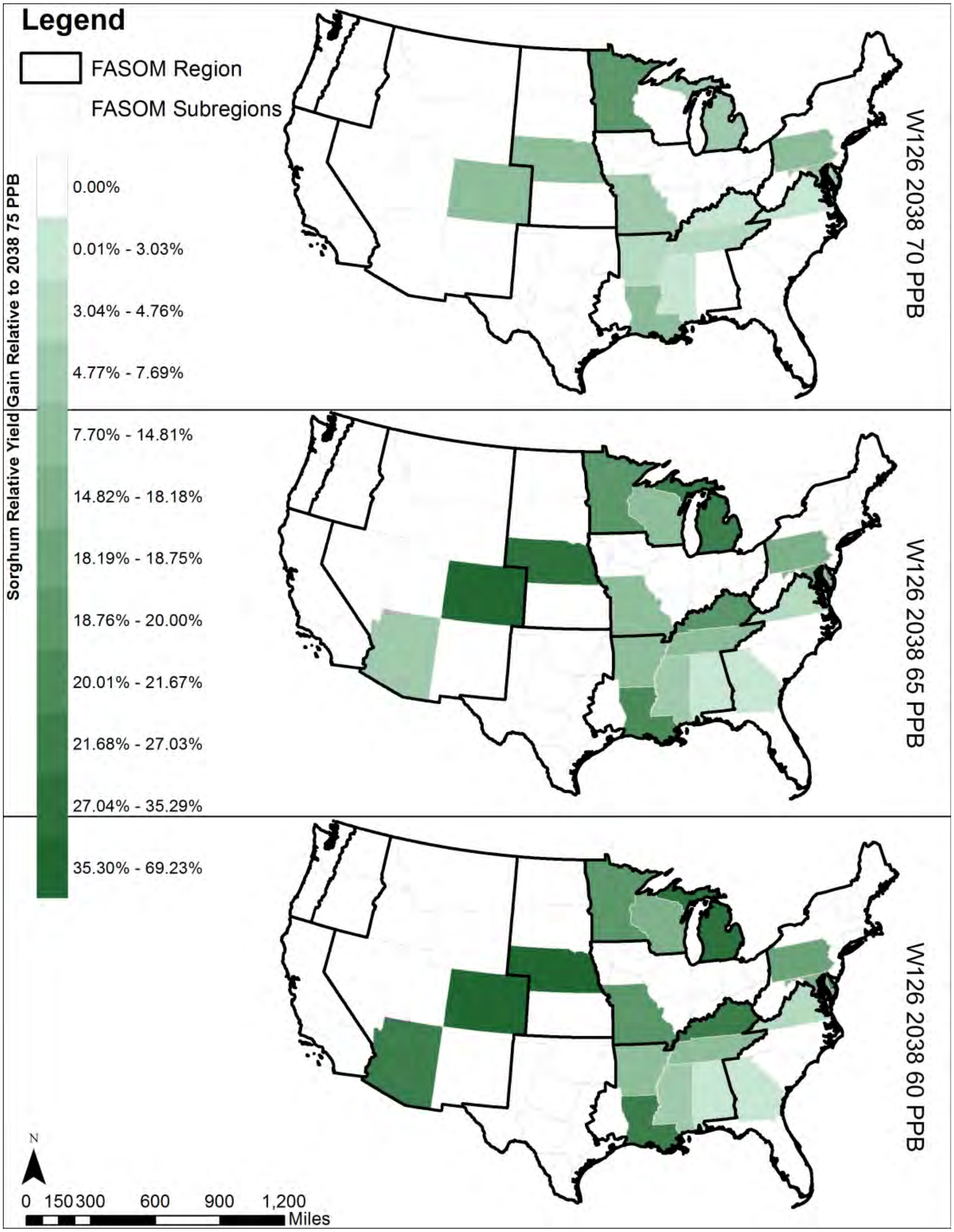

Figure 6A-10. Percentage Changes in Sorghum RYGs with Respect to the 75 ppb Scenario

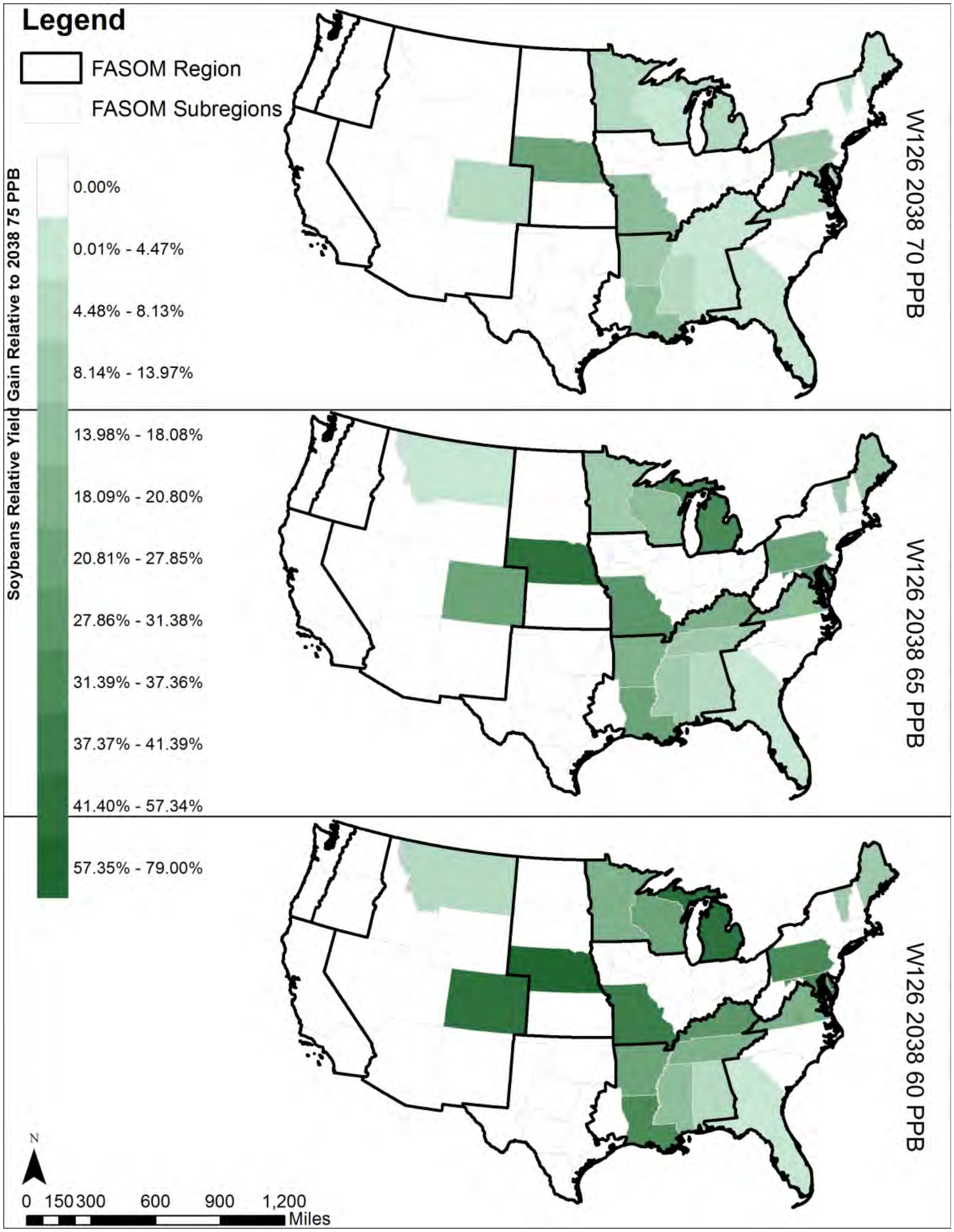

Figure 6A-11. Percentage Changes in Soybean RYGs with Respect to the 75 ppb Scenario

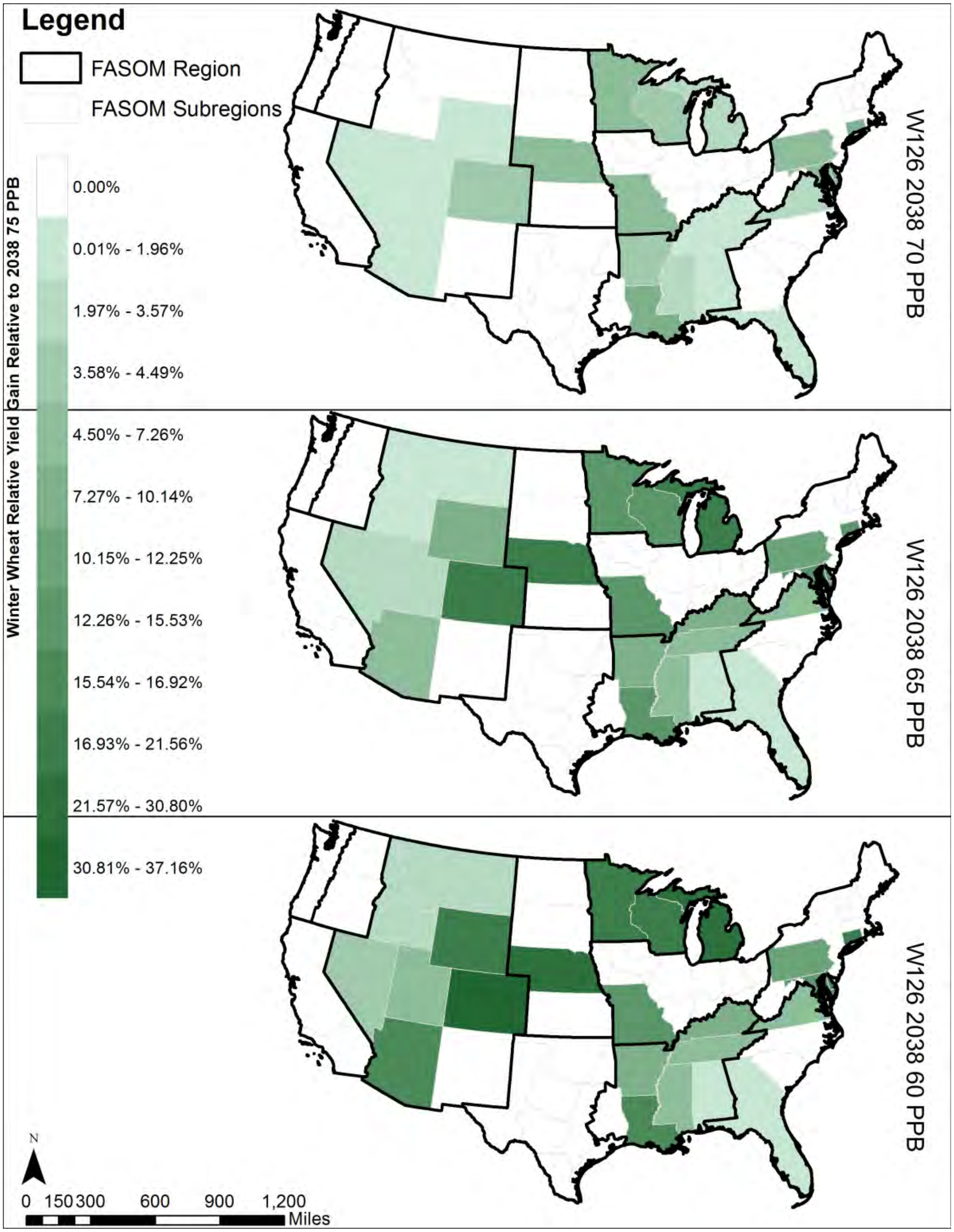

Figure 6A-12. Percentage Changes in Winter Wheat RYGs with Respect to the 75 ppb Scenario

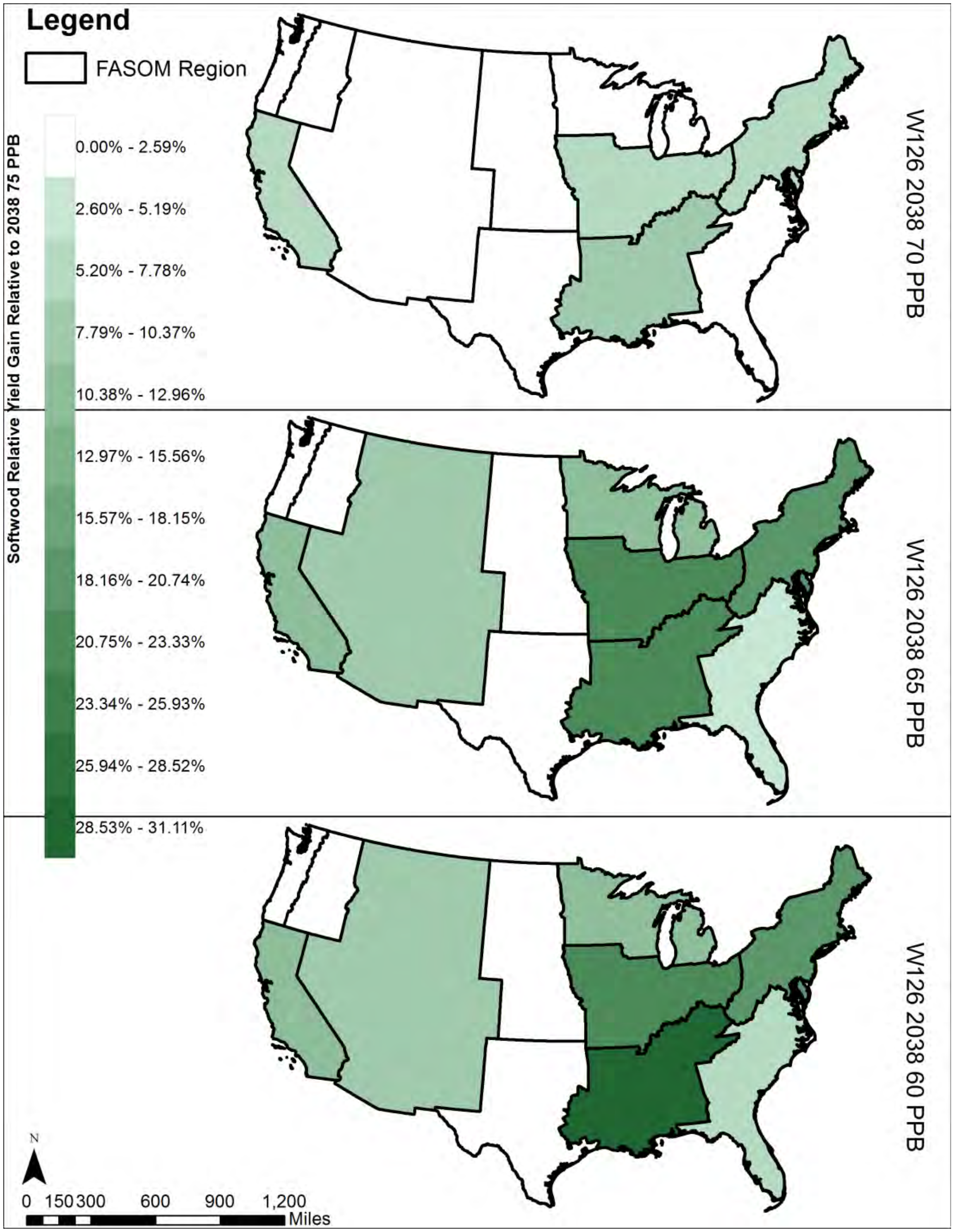

Figure 6A-13. Percentage Changes in Softwood RYGs with Respect to the 75 ppb Scenario

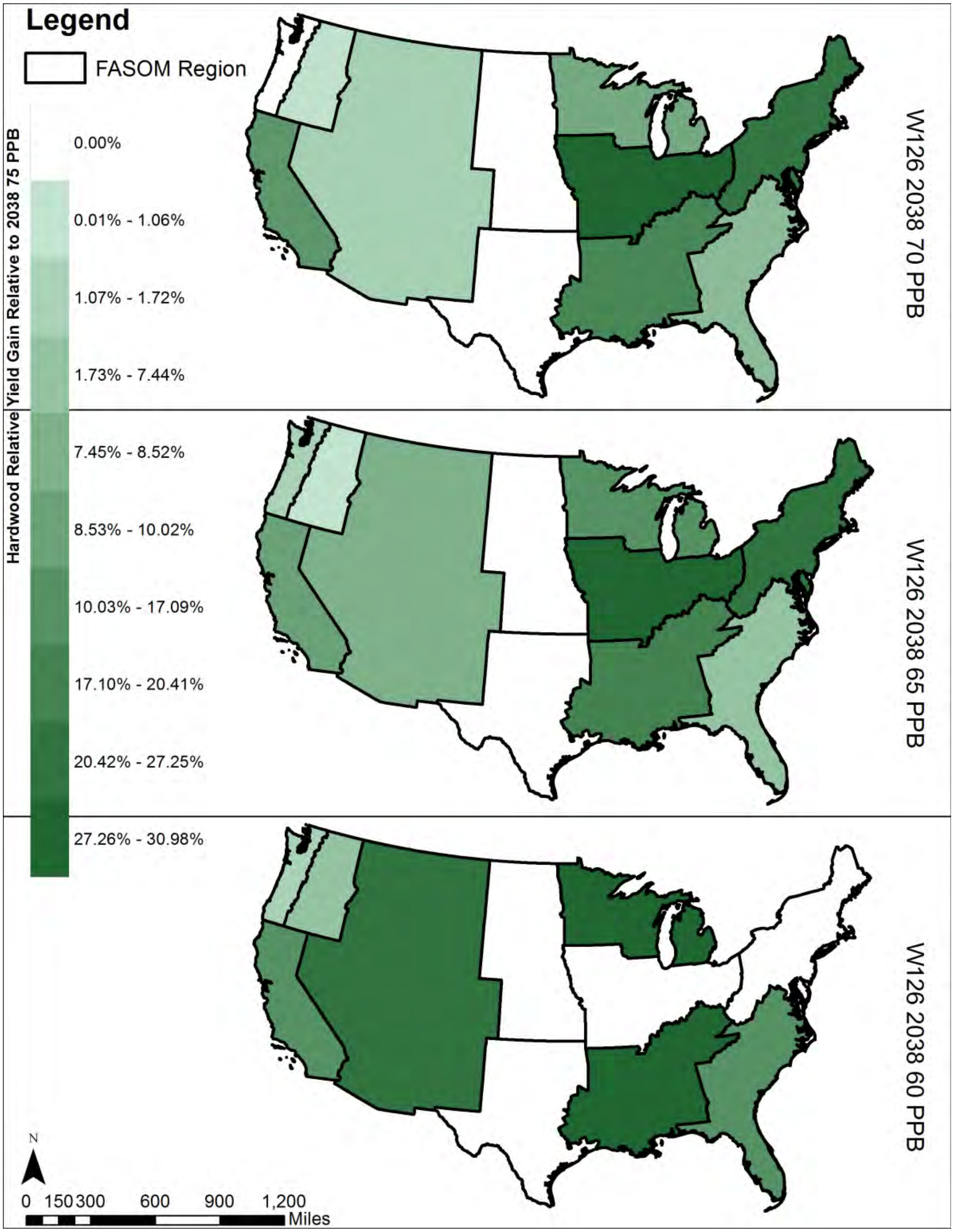

Figure 6A-14. Percentage Changes in Hardwood RYGs with Respect to the 75 ppb Scenario

6A.2 Model Results

FASOMGHG was used to estimate the projected effects of alternative ozone concentration standards on the U.S. agricultural and forestry sectors. As introduced earlier, the comparisons considered for this report focus on the differences between a scenario assuming compliance with the existing 2008 standards (75 ppb) and scenarios in which three more stringent ozone standards are met. Those three scenarios are 70 ppb, 65 ppb, and 60 ppb W126 values. Our analysis included changes to production, prices, forest inventory, land use, welfare, and GHG mitigation potential associated with achieving each of the more stringent standards.

6A.2.1 Agricultural Sector

Ozone negatively affects growth in many plants, leading to lower crop yields. In addition, some crops are more sensitive to ozone than others, so the percentage changes in yield will vary by crop and region. However, reducing ambient ozone concentrations would generally increase agricultural yields and total production, though the reductions in ozone concentrations that would be achieved under a given standard vary across regions. Our analysis began by determining the extent to which current yield losses caused by ozone could be reversed by reducing ozone levels. Increased crop yields lead to a greater available supply of most agricultural crops, which in turn tends to reduce market prices. There is also an overall tendency toward acreage shifting away from ozone-sensitive crops. In general, impacts in the agricultural sector are relatively limited, especially when compared with the forestry sector. By and large, more stringent standards led to increased incremental impacts, but the additional impact in moving to increasingly stringent ozone standards was relatively small.

6A.2.1.1 Production and Prices

Changes in U.S. agricultural production and prices were measured using Fisher indices (sees Tables 6A-7 and 6A-7a).[102] Both primary and secondary commodity production levels are projected to increase by 2040 as a result of heightened productivity. Agricultural production changes were generally relatively small across products, rarely exceeding an increase of 0.50% with respect to the current standard and often changing by 0.02% or less.

[102] The Fisher price index is known as the "ideal" price index. It is calculated as the geometric mean of an index of current prices and an index of past prices.

Table 6A-7. Agricultural Production Fisher Indices (Current conditions =100)

Sector	Policy	2010	2020	2030	2040
Primary Commodities					
Crops	75 ppb	100.0	99.8	100.7	101.3
	70 ppb	100.0	99.8	100.7	101.3
	65 ppb	100.0	99.8	100.8	101.4
	60 ppb	100.0	99.8	100.9	101.4
Livestock	75 ppb	100.0	99.9	100.2	100.3
	70 ppb	100.0	99.9	100.2	100.3
	65 ppb	100.0	99.9	100.2	100.4
	60 ppb	100.0	99.9	100.2	100.5
Farm products[a]	75 ppb	100.0	99.8	100.1	100.3
	70 ppb	100.0	99.8	100.4	100.3
	65 ppb	100.0	99.8	100.5	100.4
	60 ppb	100.0	99.9	100.5	100.5
Secondary Commodities					
Processed	75 ppb	100.0	100.0	100.1	100.1
	70 ppb	100.0	99.9	100.1	100.1
	65 ppb	100.0	99.9	100.1	100.1
	60 ppb	100.0	100.0	100.1	100.1
Meats	75 ppb	100.0	100.0	100.1	100.3
	70 ppb	100.0	100.0	100.1	100.2
	65 ppb	100.0	100.0	100.1	100.3
	60 ppb	100.0	100.0	100.2	100.3
Mixed feeds	75 ppb	100.0	99.9	100.1	100.5
	70 ppb	100.0	99.9	100.1	100.5
	65 ppb	100.0	99.9	100.1	100.5
	60 ppb	100.0	99.9	100.2	100.5

[a] Farm Products is the composite of Crops and Livestock.

Table 6A-7a. Agricultural Price Fisher Indices (Current Conditions = 100)

Sector	Policy	2010	2020	2030	2040
Primary Commodities					
Crops	75 ppb	100.0	100.2	98.5	97.2
	70 ppb	100.0	100.2	98.4	97.0

Sector	Policy	2010	2020	2030	2040
	65 ppb	100.0	100.2	98.3	97.0
	60 ppb	100.0	100.2	98.3	97.0
Livestock	75 ppb	100.1	101.4	99.2	99.9
	70 ppb	100.6	102.2	100.3	97.0
	65 ppb	100.1	102.2	100.6	97.0
	60 ppb	100.1	101.8	100.4	97.0
Farm products[a]	75 ppb	100.0	100.2	100.1	99.9
	70 ppb	100.0	100.2	98.4	97.0
	65 ppb	100.0	100.2	98.3	97.0
	60 ppb	100.0	100.2	98.3	97.0
Secondary Commodities					
Processed	75 ppb	100.1	100.3	98.8	98.3
	70 ppb	100.1	100.3	98.6	98.2
	65 ppb	100.1	100.3	98.2	98.0
	60 ppb	100.1	100.3	98.2	97.9
Meats	75 ppb	100.1	99.8	100.1	99.9
	70 ppb	100.1	99.8	100.1	99.9
	65 ppb	100.1	99.8	100.1	100.0
	60 ppb	100.1	99.8	100.1	100.0
Mixed feeds	75 ppb	100.1	99.7	98.2	99.3
	70 ppb	100.1	99.7	98.1	99.5
	65 ppb	100.1	99.7	97.4	99.3
	60 ppb	100.1	99.7	97.5	99.5

Increased production led to a general decline in market prices because the equilibrium price adjusts to higher levels of supply. This result is consistent with expectations because higher productivity leads to greater supply, which tends to decrease market prices. Changes in price were generally more pronounced than changes in production, with the largest decreases in the Farm Products, Livestock, and Processed categories. Agricultural prices tend to decline by a greater percentage than production increases because the demand for most agricultural commodities is inelastic.[103] However, almost all declines in price were less than 2.0% of prices at the current standards, and most were less than 0.5%.

6A.2.1.2 Crop Acreage

Crop acreage was projected to decline with the introduction of the ozone standards because additional productivity per acre reduces the demand for crop acreage. In aggregate, farmers will be able to meet the demand for agricultural commodities using less land under scenarios with lower ozone concentrations. Consistent with these expectations, the total cropped area is slightly smaller for each model year in the alternative standard cases. However, land allocation also depends on relative returns across various uses and is influenced by forest harvest timing. Changes in land allocation between the agricultural and forestry sectors are discussed later in this section.

Table 6A-8 provides projections of acreage in each of the major U.S. crops, as well as composites of all remaining crops and total cropland. The absolute change relative to the current standard is presented for each alternative standard. Larger changes occurred in sorghum acreage, whereas only minor changes occurred in all other crops, leading to almost no net change in crop acreage across all crops. This shift occurred largely because of differential crop sensitivity to ozone concentrations. Note that the sum of the crop-specific changes will not necessarily equal the total changes shown in Table 6A-8 because some double-cropping is reflected in the model (e.g., soybeans and winter barley).

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[103] Demand elasticities are measures of the responsiveness of the quantity demanded to a change in price. Commodities with inelastic demands are those where consumers change the quantity of a good they purchase by a smaller percentage than the change in market price. Many food products fall into this category because they are relatively low-priced necessities.

Table 6A-8. Major Crop Acreage, Million Acres

Crop	Policy	2010	2020	2030	2040
Corn	75 ppb	91.2	85.8	77.4	70.9
	Change with Respect to Current Standard				
	70 ppb	0.00	0.02	0.00	-0.05
	65 ppb	0.00	0.01	0.06	-0.07
	60 ppb	0.00	0.01	0.04	-0.08
Soybeans	75 ppb	4.9	5.2	5.8	6.9
	Change with Respect to Current Standard				
	70 ppb	0.00	0.00	0.00	0.00
	65 ppb	0.00	0.00	0.00	-0.03
	60 ppb	0.00	0.00	0.00	-0.03
Hay	75 ppb	43.7	41.1	41.7	42.0
	Change with Respect to Current Standard				
	70 ppb	0.00	-0.02	0.03	0.00
	65 ppb	0.00	-0.01	0.05	-0.01
	60 ppb	0.00	0.01	0.06	-0.07
Hard Red Winter Wheat	75 ppb	25.5	23.7	23.2	22.4
	Change with Respect to Current Standard				
	70 ppb	0.00	0.00	0.00	-0.06
	65 ppb	0.00	0.00	-0.06	-0.11
	60 ppb	0.00	0.00	-0.03	-0.10
Cotton	75 ppb	13.6	13.6	14.2	14.0
	Change with Respect to Current Standard				
	70 ppb	0.00	0.00	-0.04	-0.01
	65 ppb	0.00	0.00	-0.05	-0.01
	60 ppb	0.00	0.00	-0.04	0.00
Hard Red Spring Wheat	75 ppb	13.5	12.9	13.4	13.0
	Change with Respect to Current Standard				
	70 ppb	0.00	0.00	0.01	0.01
	65 ppb	0.00	0.01	-0.03	0.00
	60 ppb	0.00	0.01	-0.08	-0.01
Sorghum	75 ppb	73.1	71.9	72.0	72.2
	Change with Respect to Current Standard				
	70 ppb	0.00	0.20	0.02	-4.88
	65 ppb	0.00	0.24	0.02	-4.90
	60 ppb	0.00	0.26	0.09	-4.89
Switch Grass	75 ppb	0.1	0.1	0.1	0.1
	Change with Respect to Current Standard				

Crop	Policy	2010	2020	2030	2040
	70 ppb	0.00	0.00	0.00	0.00
	65 ppb	0.00	0.00	0.00	0.00
	60 ppb	0.00	0.00	0.00	0.00
All Others[a]	75 ppb	352.2	366.7	355.8	340.0
	Change with Respect to Current Standard				
	70 ppb	0.00	0.10	0.01	-0.17
	65 ppb	0.00	0.13	-0.12	-0.32
	60 ppb	0.00	0.19	-0.25	-0.45
Total	75 ppb	5.6	5.4	4.6	4.0
	Change with Respect to Current Standard				
	70 ppb	0.00	0.00	0.01	-0.01
	65 ppb	0.00	-0.01	0.02	0.00
	60 ppb	0.00	-0.01	0.01	0.00

[a] Canola, durum wheat, fresh grapefruit, fresh orange, fresh tomato, grazing wheat, hybrid poplar, oats, potato, processed grapefruit, processed orange, processed tomato, rice, rye, silage, soft red winter wheat, soft white wheat, spring barley, sugar beet, sugarcane, sweet sorghum, winter barley.

6A.2.2 Forestry Sector

As with agricultural crops, ozone diminishes growth in most tree species, and our analysis began by estimating how much of this diminished growth would be reversed under the more stringent ozone standards. Impacts are significantly higher in the forestry sector, especially in hardwood species and species more prevalent in the southern regions. Impacts are more significant for southern regions because of the higher baseline ozone concentrations in the South Central and Southeast regions. Higher initial concentrations resulted in higher reductions to meet the alternative standards and, thus, higher impacts to tree growth. This relationship also contributes to the larger changes in the forestry sector as a whole.

6A.2.2.1 Production and Prices

Reducing ozone concentrations led to increased forest growth, which was reflected in increased production in FASOMGHG. Some of the most substantial ozone standard impacts occurred in saw log and pulp log harvest quantities and prices. Compared with the current standard, alternative standard cases had consistently higher production except for hardwood pulp logs in 2030 and softwood pulp logs in 2040, where production increased only marginally and at times fell below the baseline estimates, especially in 2040. The most significant impacts occurred in hardwood pulp logs, where harvests were projected to be more than 1% higher than under the current standard level by 2040 for the 60 ppb concentration. There are some cases where production of pulp logs and saw logs moved in opposite directions. There are two primary explanations for these trends. The first is that as softwood saw log production expands, the price for softwood saw logs drops, allowing processers to substitute saw logs for cases of production in which pulp logs are traditionally used. The second is that even when the primary log size being harvested is pulp logs, saw logs will generally also be present because of natural variation in tree growth rates (and vice versa for harvest of saw logs). With higher growth rates, there would tend to be more saw logs in stands harvested primarily for pulp logs over time.

The largest changes in production occurred at the 60 ppb level. Changes from the current standard to 70 ppb were fairly small, but increased in 65 ppb, and changes were even larger in 60 ppb. Table 6A-9 presents these changes by major product.

The impact of policy intervention on timber market prices was more substantial than the change in production in terms of percentage changes compared with the current standard. Although increases in production of forest products did not exceed 1.5% compared with the current standard, changes in price were as large as 12.8%. As with agricultural products, many forest products have relatively inelastic demand so prices tend to change by a larger percentage than quantities. Table 6A-10 lists absolute changes with respect to the current standard, whereas Table 6A-11 lists the percentage change in forest product prices for each year, alternative standard, and forest product.

Table 6A-9. Forest Products Production, Million Cubic Feet

Product	Policy	2010	2020	2030	2040
Hardwood Saw logs	75 ppb	3,588	3,394	3,708	4,246
		Change with Respect to Existing Standard			
	70 ppb	1	1	38	3
	65 ppb	1	0	33	-6
	60 ppb	3	0	54	-13
Hardwood Pulp logs	75 ppb	2,448	2,162	2,510	2,220
		Change with Respect to Existing Standard			
	70 ppb	2	1	-29	3
	65 ppb	3	-2	-29	14

	60 ppb	10	-1	-40	25
Softwood saw logs	75 ppb	4,568	5,114	5,449	6,542
		Change with Respect to Existing Standard			
	70 ppb	-1	10	-2	7
	65 ppb	0	22	1	53
	60 ppb	5	35	21	61
Softwood pulp logs	75 ppb	3,437	3,878	4,350	4,298
		Change with Respect to Existing Standard			
	70 ppb	1	-2	3	-17
	65 ppb	0	1	4	-23
	60 ppb	2	1	8	-8

Table 6A-10. Forest Product Prices, U.S. Dollars per Cubic Foot

Product	Policy	2010	2020	2030	2040
Hardwood saw logs	75 ppb	0.80	0.89	0.58	0.33
		Change with Respect to Existing Standard			
	70 ppb	0.00	-0.01	-0.01	-0.02
	65 ppb	0.00	-0.01	-0.02	-0.03
	60 ppb	0.00	-0.01	-0.03	-0.04
Hardwood pulp logs	75 ppb	0.30	0.64	0.45	0.24
		Change with Respect to Current Standard			
	70 ppb	0.00	-0.01	-0.01	-0.01
	65 ppb	0.00	-0.01	-0.03	-0.02
	60 ppb	0.00	-0.02	-0.04	-0.03
Softwood saw logs	75 ppb	2.46	2.08	1.78	1.50
		Change with Respect to Current Standard			
	70 ppb	0.00	0.00	0.00	0.00
	65 ppb	0.00	-0.01	-0.01	-0.02
	60 ppb	-0.01	-0.02	-0.02	-0.04
Softwood pulp logs	75 ppb	1.49	1.33	1.47	1.14
		Change with Respect to Current Standard			
	70 ppb	0.00	0.00	0.00	0.00
	65 ppb	0.00	0.00	-0.01	-0.02
	60 ppb	-0.01	0.00	-0.03	-0.05

Table 6A-11. Forest Product Prices and Percentage Change, U.S. Dollars per Cubic Foot

Product	Policy	2010	2020	2030	2040
Hardwood saw logs	75 ppb	0.80	0.89	0.58	0.33
		% Change with Respect to Existing Standard			
	70 ppb	-0.41%	-0.75%	-2.16%	-5.07%
	65 ppb	-0.24%	-1.00%	-3.89%	-9.11%
	60 ppb	-0.25%	-1.22%	-5.60%	-11.36%
Hardwood pulp logs	75 ppb	0.30	0.64	0.45	0.24
		% Change with Respect to Existing Standard			
	70 ppb	0.07%	-0.96%	-3.19%	-5.15%
	65 ppb	0.12%	-1.92%	-5.57%	-10.53%
	60 ppb	-0.06%	-2.87%	-8.98%	-12.75%
Softwood saw logs	75 ppb	2.46	2.08	1.78	1.50
		% Change with Respect to Existing Standard			
	70 ppb	0.01%	-0.17%	-0.16%	-0.13%
	65 ppb	0.03%	-0.27%	-0.36%	-1.08%
	60 ppb	-0.28%	-0.97%	-1.20%	-2.65%
Softwood pulp logs	75 ppb	1.49	1.33	1.47	1.14
		% Change with Respect to Existing Standard			
	70 ppb	-0.05%	0.28%	-0.28%	-0.13%
	65 ppb	-0.18%	0.10%	-0.76%	-1.90%
	60 ppb	-0.57%	-0.27%	-1.87%	-4.11%

6A.2.2.2 Forest Acres Harvested

Harvested acres are projected to decline in hardwoods as a result of higher productivity in the policy cases. Conversely softwood acres harvested increases over time. The difference between the hardwood harvested acres in the current standard case and in the alternative standards widens from 2010 to 2040, increasing to a difference of more than 4% under the 60 ppb case. The impact to total acres of softwood harvested shows a less uniform pattern, with the largest impacts under the 65 ppb scenario, and an increase of up to 6% of harvested acres by 2040 compared to 2010. Table 6A-12 presents the model results for forest acres harvested.

Table 6A-12. Forest Acres Harvested, Thousand Acres

Product	Policy	2010	2020	2030	2040
Total hardwood	75 ppb	14,923	11,701	12,277	13,138
		Change with Respect to Current Standard			
	70 ppb	4	50	-139	-151
	65 ppb	8	37	-265	-352
	60 ppb	37	50	-377	-513
Total softwood	75 ppb	17,539	16,158	14,911	19,031
		Change with Respect to Current Standard			
	70 ppb	-11	35	47	106
	65 ppb	-13	107	223	361
	60 ppb	-2	142	328	314

6A.2.2.3 Forest Inventory

Under FASOMGHG definitions, existing inventory includes only trees that have been standing since the initial model year of 2000. All trees planted since then, including both reforestation and afforestation, are included in new inventory. The model projected significant increases in existing inventory for hardwood species under the current standard, and consistent with the increase in the acres harvested of softwood, the existing inventory of softwoods declines through 2040 under the current standard. The difference in responses is partially explained by differential sensitivity to ozone between species. Hardwood species show a much higher sensitivity to ozone levels and are thus modeled to respond more dramatically to reductions in ozone concentration.

Some relatively large differences between ozone standards occurred in the forest inventory projections. For example, existing hardwood inventory was projected to be 4.0% small under the 60 ppb case than the 70 ppb case by 2040. New hardwood inventory is similarly sensitive, with the model projecting a 2% decrease for this same comparison. For new and existing inventory of both hardwoods and softwoods, the largest impacts occurred at the 70 ppb standard. This type of nonlinear response can occur because of differences in the relative impacts on alternative forest and agricultural products that lead to land reallocation. Table 6A-13 presents the model results for forest inventory.

Table 6A-13. Existing and New Forest Inventory, Million Cubic Feet

Product	Policy	2010	2020	2030	2040
Existing Hardwood	75 ppb	281,924	273,055	292,685	306,296
	Change with Respect to Existing Standard				
	70 ppb	0	-64	1,983	4,998
	65 ppb	0	-138	5,974	14,937
	60 ppb	0	-278	8,896	22,548
Existing Softwood	75 ppb	184,828	153,771	135,137	133,794
	Change with Respect to Existing Standard				
	70 ppb	0	-19	185	494
	65 ppb	0	-38	683	1,583
	60 ppb	-6	-77	1,059	2,553
New Hardwood	75 ppb	1,932	9,437	18,872	29,732
	Change with Respect to Existing Standard				
	70 ppb	0	-152	-107	77
	65 ppb	0	-150	-63	577
	60 ppb	1	-156	0	931
New Softwood	75 ppb	8,837	64,254	114,454	128,581
	Change with Respect to Existing Standard				
	70 ppb	0	-42	-17	-497
	65 ppb	4	6	-128	-1,164
	60 ppb	7	-18	-312	-1,798

6A.2.3 Cross-Sectoral Policy Impacts

One of the advantages of a model such as FASOMGHG for analysis of impacts on major land-using activities is the ability to account for shifts in land use. Differentiated impacts on productivity across products will lead to changes in market prices and in the relative profitability of alternative land uses. In response, landowners will change their allocation of land across different productive activities, which will contribute to market impacts. In addition, these changes in land use have implications for GHG emissions and other environmental impacts. In this section, we discuss changes in land use, net GHG emissions, and producer and consumer welfare across the agricultural and forest sectors.

6A.2.3.1 Land Use

FASOMGHG projected changes in eight land use categories: existing forest, reforestation, afforestation, cropland, pasture, cropland pasture,[104] and lands enrolled in the Conservation Research Program (CRP).[105] The largest impacts under the current standard were projected in afforestation. The general projected pattern under the alternative standards within these categories was a decline in reforested area, afforested area, and cropland in 2030 and 2040, coupled with increases in the pasture and cropland pastured areas. There was no change in acreage retained in CRP or rangeland.

The incremental impact of more stringent ozone standards appears to be non-linear in some cases, especially in existing and reforested areas. Existing forest exhibits a general decline in the area under the 70 ppb scenario; in the 60 ppb scenario, however, there are small declines through 2020, followed by large increases in 2030 and 2040. This is due to the different trends in the hardwood and softwood species. Table 6A-14 presents the model results by major land use type.

Table 6A-14. Land Use by Major Category, Thousand Acres

Product	Policy	2010	2020	2030	2040
Existing forest	75 ppb	257,565	201,587	161,426	131,210
		Change with Respect to Current Standard			
	70 ppb	-7	82	-113	-269
	65 ppb	0	0	0	0
	60 ppb	-2	-84	159	691
Reforested	75 ppb	72,201	117,974	148,837	172,267
		Change with Respect to Current Standard			
	70 ppb	-10	-7	-56	-189
	65 ppb	-16	52	-212	-878
	60 ppb	28	107	-267	-1,346
Afforested	75 ppb	14,086	10,886	6,404	11,474
		Change with Respect to Current Standard			

[104] Cropland pasture is managed land suitable for crop production (i.e., relatively high productivity) that is being used as pasture.

[105] Rangeland estimates are also included, but rangeland is held fixed in FASOMGHG by assumption because it cannot be allocated to any other use.

Product	Policy	2010	2020	2030	2040
	70 ppb	0	0	-134	-134
	65 ppb	0	0	-309	-309
	60 ppb	0	0	-427	-427
Cropland	75 ppb	311,713	313,325	304,227	292,678
	Change with Respect to Current Standard				
	70 ppb	0	96	-38	-60
	65 ppb	0	124	-102	-157
	60 ppb	0	180	-157	-200
Pasture	75 ppb	84,280	85,049	86,133	82,571
	Change with Respect to Current Standard				
	70 ppb	3	-7	86	64
	65 ppb	10	-11	189	168
	60 ppb	10	24	257	263
Cropland pasture	75 ppb	45,381	44,634	55,061	61,042
	Change with Respect to Current Standard				
	70 ppb	0	1	29	51
	65 ppb	0	1	161	216
	60 ppb	0	6	235	277
Rangeland	75 ppb	302,210	301,104	300,049	299,039
	Change with Respect to Current Standard				
	70 ppb	0	0	0	0
	65 ppb	0	0	0	0
	60 ppb	0	0	0	0
CRP	75 ppb	36,879	36,659	36,659	36,659
	Change with Respect to Current Standard				
	70 ppb	0	0	0	0
	65 ppb	0	0	0	0
	60 ppb	0	0	0	0

6A.2.3.2 Welfare

Welfare impacts resulting from the implementation of alternative standard levels followed the same pattern between the agriculture and forestry sectors, although it was more pronounced in forestry. Consumer surplus typically increased in both cases as higher productivity under reduced ozone conditions tended to increase total production and reduce market prices. Because demand for most forestry and agricultural commodities is inelastic, there are more instances in which producer surplus declines. In some year/ozone concentration combinations, the effect of falling prices on producer profits more than outweighs the effects of higher production levels.

Percentage changes in agricultural sector consumer and producer surplus between the current standard and the alternative standards were relatively small in many cases, with the largest percentage change being a 5.8% decline in producer surplus in the 2040 model period. However, the agricultural sector is a very large market, and even small percentage changes in welfare can result in annualized values of tens or even hundreds of millions of dollars. Table 6A-15 provides consumer and producer surplus for the agricultural sectors under the current standard, along with the change in surplus for each alternative standard. There is considerable variability in the magnitude of consumer and producer impacts from year to year, which is not surprising given the dynamic nature of the model and numerous adjustments taking place over time in response to changes in net returns associated with alternative land uses.

Table 6A-15. Consumer and Producer Surplus in Agriculture, Million 2010 U.S. Dollars

	Policy	**2010**	**2015**	**2020**	**2025**	**2030**	**2035**	**2040**
Consumer Surplus	75 ppb	1,907,219	1,928,147	1,955,266	1,983,375	2,013,518	2,042,566	2,071,338
		Change with Respect to Existing Standard						
	70 ppb	0	0	0	113	66	58	-10
	65 ppb	-1	1	4	262	148	289	24
	60 ppb	0	47	9	462	100	408	66
Producer Surplus	75 ppb	718,105	824,802	815,996	865,123	817,707	874,549	908,352
		Change with Respect to Existing Standard						
	70 ppb	1,202	2,454	1,796	609	-1,144	675	-52,504
	65 ppb	216	530	-407	1,174	-2,933	-39,605	-10,211
	60 ppb	-176	1,015	-1,470	1,320	-1,492	-38,788	-8,119

The impacts of the scenarios with more stringent ozone standards were larger in the forestry sector, with bigger increases in consumer surplus and greater declines in producer surplus. Table 6A-16 presents the model results of the welfare analysis in the forestry sector.

Table 6A-16. Consumer and Producer Surplus in Forestry, Million 2010 U.S. Dollars

	Policy	2010	2015	2020	2025	2030	2035	2040
Consumer surplus	75 ppb	715,634	760,957	801,653	819,407	867,332	885,687	926,837
	Change with Respect to Current Standard							
	70 ppb	3	24,592	-9	-2	70	167	72
	65 ppb	12	24,595	4	43	189	263	138
	60 ppb	30	24,695	127	257	433	686	245
Producer surplus	75 ppb	93,795	147,719	154,137	144,888	147,039	147,797	132,850
	Change with Respect to Current Standard							
	70 ppb	1	-24,614	-28	94	-24	-152	8
	65 ppb	-5	-24,621	-73	291	-111	-86	-35
	60 ppb	-57	-24,727	-261	135	-471	-503	37

Because of the complex dynamics of the agriculture and forestry sectors and variability in welfare impacts over time, it is often helpful to summarize the impacts in terms of annualized values. Table 6A-17 and Table 6A-18 summarize the annualized impacts of alternative ozone standards on consumer and producer surplus in the agricultural and forestry sectors for 2010–2044 at a discount rate of 3% and 7% respectively.[106] The impacts of alternative standards on consumer surplus are positive for each of the tighter standards for both agricultural and forestry sectors, with the benefits increasing with more stringent requirements. For producer surplus, on the other hand, annualized impacts are negative for all of the tighter standards for both agriculture and forestry sector, becoming more negative as stringency is increased. Overall, total surplus declines for the agriculture sector while rises slightly for the forest sector across alternative standard levels, leading to a net decrease for both sectors.

Table 6A-17. Annualized Changes in Consumer and Producer Surplus in Agriculture and Forestry, 2010–2044, Million 2010 U.S. Dollars (3% Discount Rate)

	Policy	Agriculture	Forestry	Total
Consumer surplus	75 ppb	1,969,838	805,563	2,775,401
	Change with Respect to Current Standard			

[106] Each model period in FASOMGHG is representative of the 5-year period starting with that year, so results reported for 2040 are representative of 2040–2044. Thus, we use values through 2044 in the annualization calculations.

	70 ppb	28	4,552	4,580
	65 ppb	86	4,592	4,678
	60 ppb	132	4,744	4,877
Producer surplus	75 ppb	817,744	135,478	953,222
		Change with Respect to Current Standard		
	70 ppb	-3,601	-4,534	-8,135
	65 ppb	-5,035	-4,524	-9,559
	60 ppb	-4,741	-4,684	-9,425
Total surplus	75 ppb	2,787,582	941,041	3,728,623
	70 ppb	-3,573	18	-3,555
	65 ppb	-4,949	68	-4,882
	60 ppb	-4,608	60	-4,548

Table 6A-18. Annualized Changes in Consumer and Producer Surplus in Agriculture and Forestry, 2010–2044, Million 2010 U.S. Dollars (7% Discount Rate)

Product	Policy	Agriculture	Forestry	Total
	75 ppb	1,951,843	782,748	2,734,591
		Change with Respect to Existing Standard		
Consumer surplus	70 ppb	21	5,570	5,592
	65 ppb	61	5,599	5,660
	60 ppb	100	5,721	5,821
	75 ppb	800,022	130,681	930,703
		Change with Respect to Existing Standard		
Producer surplus	70 ppb	-945	-5,561	-6,506
	65 ppb	-2,719	-5,555	-8,273
	60 ppb	-2,635	-5,694	-8,328
	75 ppb	2,751,865	913,429	3,665,294
		Change with Respect to Existing Standard		
Total surplus	70 ppb	-923	9	-914
	65 ppb	-2,658	45	-2,613
	60 ppb	-2,535	27	-2,508

6A.2.3.3 Greenhouse Mitigation Potential

The capacity for both the agricultural and forest sectors to sequester carbon is enhanced in each of the alternative standard cases, with increasing magnitude as policy stringency is increased. Although FASOMGHG projects fewer acres of forestland and total cropland, the accelerated storage of carbon in trees and forestland and cropland soils outweighs any decline from reductions in covered area. Carbon storage in both sectors is consistently higher in the alternative standard cases, with the gap widening over time (see Figure 6A-15 for change in forest carbon stock). By 2040, the agricultural sector sequestered 0.01%~0.05%% more carbon under the alternative standard cases and the forestry sector up to 0.5% more, resulting in gains of more than 1,500 million metric tons CO_2 equivalent ($MMtCO_2e$). Table 6A-19 presents carbon sequestration projections under the current standard and changes under each alternative standard.[107] Note that negative values in the row for the current standard indicate sequestration or carbon storage. Negative values in the change rows indicate that the alternative standard stores more carbon than the current standard (and vice versa for positive changes).

Notice that for the agricultural sector, the overall stock of net GHG would decrease over time in the baseline because cropping activities involve fertilizer and chemical usage, fossil fuels, running machinery, livestock emissions from enteric fermentation and manure management, and so forth—all these GHG emissions are being released each year, while soil carbon sequestration moves toward equilibrium within 25 years of a change in tillage. As soil carbon reaches equilibrium, little additional sequestration is taking place each year but annual emissions from other sources continue. Thus, over time, the annual emissions tend to outweigh the increase in carbon stocked in agricultural soils, and net stock of GHG tends to become less negative and eventually positive relative to the starting point.[108]

[107] These are total stocks of net GHG emissions over time, not annual emissions. If the total stock of GHG is becoming more negative over time, more net sequestration is taking place than emissions. If the total stock of GHG is becoming less negative or positive over time, emissions are greater than the increase in sequestration.

[108] This change is consistent with the fact that U.S. agriculture is a net source of emissions on an annual basis. The value of the total GHG stock associated with agriculture is starting at a negative value because of the FASOMGHG convention of accounting for total carbon sequestration present in agricultural soils in the first year of the model run. A large stock of carbon is sequestered, but it does not increase by much over time.

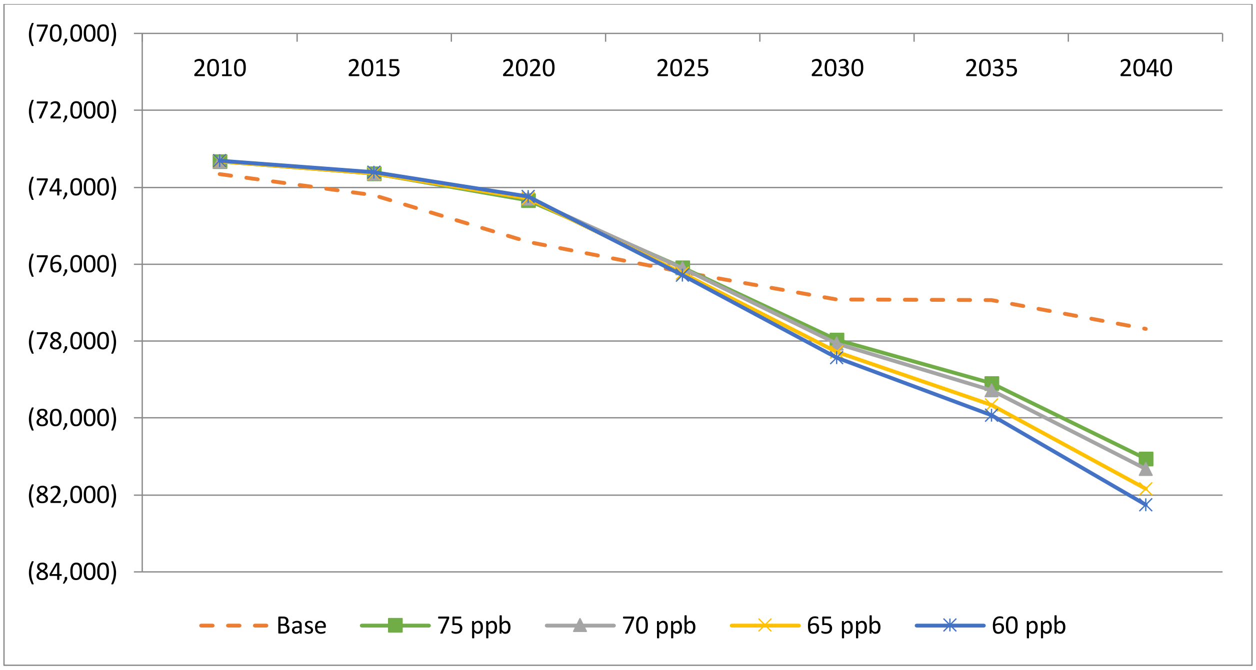

Figure 6A-15. Carbon Storage in Forestry Sector, MMtCO₂e

Table 6A-19. Carbon Storage, $MMtCO_2e$

Product	Policy	2010	2020	2030	2040	2010-2044
Agriculture	75 ppb	-18,621	-15,240	-11,772	-8,009	-268,210
	Change with Respect to Existing Standard					
	70 ppb	0	-1	-3	-1	-24
	65 ppb	0	-2	-6	-4	-62
	60 ppb	-1	-4	-11	-11	-132
Forestry	75 ppb	-73,321	-74,338	-77,963	-81,063	-1,533,424
	Change with Respect to Existing Standard					
	70 ppb	1	51	-100	-259	-1,537
	65 ppb	2	53	-323	-774	-5,207
	60 ppb	12	94	-465	-1,189	-7,739

Changes in forestry sector carbon sequestration are largely driven by changes in forest management, which include the increases in tree yield in the lower ozone environments. The increased sequestration in this category outweighs losses in sequestration in the other major forestry categories: afforestation and forest soil. Table 6A-20 presents the detailed changes in forestry carbon sequestration.

Table 6A-20. Forestry Carbon Sequestration, $MMtCO_2e$

Product	Policy	2010	2020	2030	2040
Afforestation, Trees	75 ppb	-730	-1,594	-963	-1,502
	Change with Respect to Existing Standard				
	70 ppb	0	0	22	29
	65 ppb	0	0	48	64
	60 ppb	0	0	66	88
Afforestion, Soils	75 ppb	-767	-598	-550	-1,018
	Change with Respect to Existing Standard				
	70 ppb	0	0	11	11
	65 ppb	0	0	27	27
	60 ppb	0	0	37	37
Forest Management	75 ppb	-39,827	-37,995	-40,023	-39,556
	Change with Respect to Existing Standard				
	70 ppb	1	43	-125	-291
	65 ppb	2	44	-387	-852
	60 ppb	11	77	-553	-1,298
Forest Soils	75 ppb	-28,320	-27,698	-27,243	-27,474
	Change with Respect to Existing Standard				
	70 ppb	0	10	5	4
	65 ppb	1	13	20	19
	60 ppb	1	21	27	28

6A.3 Summary

Impacts to both sectors generally mirror one another, although they are more prominent in the forestry sector. Not only are tree species more responsive to changes in ozone, but the largest reductions to meet the alternative standards will occur in regions with large forestry sectors: South Central, Southeast, and Rocky Mountains. Reductions in agricultural regions are comparatively moderate. Productivity of both crops and forests is projected to increase at each of the alternative standard levels. This increase in supply resulted in decreased prices for forest products and agricultural commodities, which benefits consumer welfare while reducing producer welfare. Unless there are significant changes to wood products markets in particular, producers will be forced to sell at reduced prices to absorb the increased supply. Nonetheless, gains to consumers become increasingly large with more stringent ozone standards. Gains to agricultural producers in the most stringent case are associated with a decline in forestry returns that results in a net shift in land use toward agriculture.

Increased productivity is also projected to affect land use both within and between the agricultural and forest sectors. Within sectors, acreage is projected to shift from crops and tree species that are more sensitive to ozone to those that are less sensitive because productivity in the former will be more substantially affected by reductions in ozone concentrations. For ozone-sensitive crops and species, producers are projected to require less land to produce at the same or higher levels. Forest acreage in particular is projected to decline sharply, driven by declines in both reforestation and afforestation.

Despite reductions in crop and forest area, carbon sequestration is expected to increase over time, led almost entirely by increased forest sequestration. Although there is less reforestation and afforestation and lower sequestration in new inventory, the change is a result of existing inventories becoming so much larger as trees grow faster. Lower sequestration in new inventory is outweighed by increased inventory in standing forests, represented in the model as a change in forest management.

Increased stringency in the ozone standard generally produces larger impacts on all of the model outputs. However, the additional impact of moving from the current standard to 65 ppb, or to 60 ppb was sometimes marginal compared with changes occurring between the current standard to 70 ppb. In particular, the impacts to the forestry sector, most notably in forest inventories and the forest sector welfare analysis, tended to increase at a decreasing rate after meeting the 70 ppb standard.

The model results are subject to several limitations: First, the ozone concentration response functions applied to crops and trees were using “median” parameters in Lehrer et al. (2007)—the RYLs and RYGs calculated are thus “median” ones; second, the use of crop proxy mapping and the forest-type mapping due to incomplete data specified in Section 6.1 adds to the uncertainty of these model results; third, the potential changes in tree species mixes within forest types due to ground ozone-level changes were not considered; and last, the international trade component in FASOMGHG that assumes USDA-based future projections under current conditions may present another uncertainty for the model results, especially when soybeans and wheat are among the major crop commodities for U.S. exports and have relatively large responses to changed ozone environments.

6A.4 References

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CHAPTER 7: ENGINEERING COST ANALYSIS AND ECONOMIC IMPACTS

Overview

This chapter summarizes the data sources and methodologies used to estimate engineering costs of attaining the alternative, more stringent levels for the ozone primary standards analyzed in this regulatory impact analysis (RIA). The chapter also provides estimates of the engineering costs of control strategies presented in Chapter 4 for the alternative standards of 70, 65, and 60 ppb. The discussion is presented as follows: Section 7.1 presents the costs associated with the application of known controls and is followed by discussions about the challenges of estimating costs for unknown controls (Section 7.2); costs associated with emissions reductions from unknown controls that are needed to demonstrate full attainment of the alternative standards analyzed (Section 7.3); total compliance cost estimates (Section 7.4); updated methodology (Section 7.5); economic impacts (Section 7.6); and the uncertainties and limitations associated with these components of the RIA (Section 7.7).

The engineering costs described in this chapter generally include the costs of purchasing, installing, operating, and maintaining the referenced technologies. The costs associated with monitoring, testing, reporting, and record keeping for affected sources are not included in the annualized cost estimates. For a variety of reasons, actual control costs may vary from the estimates the 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 the EPA anticipates that state and local governments will consider programs that are best suited for local conditions. Also, the EPA recognizes the unknown emissions control portion of the engineering cost estimates (Section 7.3) reflects substantial uncertainty about the sectors and technologies that might become available for cost-effective application in the future.

The engineering cost estimates are limited in their scope. This analysis focuses on the emissions reductions needed for attainment of a range of alternative revised standards. The EPA understands that some states will incur costs both designing State Implementation Plans (SIPs) for and implementing new control strategies to meet final revised standards. However, the EPA does not know what specific actions states will take to design their SIPs to meet final 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.

7.1 Estimating Engineering Compliance Costs

7.1.1 Methods and Data

After designing the hypothetical control strategy using the methodology discussed in Chapter 4, the EPA used the Control Strategy Tool (CoST) (U.S. EPA, 2014a) to estimate engineering control costs for non-electric generating unit (non-EGU point) point, nonpoint and mobile nonroad sources. CoST calculates engineering costs using one of two different methods: (1) an equation that incorporates key operating unit information, such as unit design capacity or stack flow rate, or (2) an average annualized cost-per-ton factor multiplied by the total tons of reduction of a pollutant. Most control cost information within CoST was developed based on the cost-per-ton approach because estimating engineering costs using an equation requires more detailed data, and parameters used in these equations may not be readily available or broadly representative across sources within the emissions inventory. The cost equations used in CoST estimate annual, capital and/or operating and maintenance (O&M) costs and are used primarily for some larger sources such as industrial/commercial/institutional (ICI) boilers and petroleum refinery process heaters. Information on CoST control measures information, including cost-per-ton factors and cost equations can be found at www.epa.gov/ttnecas1/cost.htm. Costs for selective reduction catalysts (SCR) applied as part of the analysis for reducing NO_x at coal-fired electric generating units (EGUs) were estimated using documentation for the Integrated Planning Model (IPM) (Sargent & Lundy, 2013).

Capital costs are converted to annual costs using the capital recovery factor (CRF).[109] Where possible, 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. For more information on this cost methodology, refer to the EPA Air Pollution Control Cost Manual (U.S. EPA, 2003) and EPA's Guidelines for Preparing Economic Analyses, Chapter 6 (US. EPA, 2014b).

Engineering costs will differ depending on the quantity of emissions reduced, emissions unit capacity, or stack flow, which can vary over time. Engineering costs will also differ in nominal terms by the year for which the costs are calculated (e.g., 2011$ versus 2008$).[110] For capital investment, in order to attain standards in 2025 we assume capital investment occurs at the beginning of 2025. We make this simplifying assumption because we do not know what all firms making capital investments for control measures will do and when they will do it. Our estimates of annualized costs include annualized capital and annual O&M costs for those controls included in our known 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 and 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 2025. The EUAC method is discussed in detail in the EPA Air Pollution Control Cost Manual (U.S. EPA, 2003). The controls applied and their respective engineering costs are described in the Chapter 7 Appendix.

[109] 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.

[110] The engineering costs will not be any different in real (inflation-adjusted) terms if calculated in 2011 versus other year dollars, if the other-year dollars are properly adjusted. For this analysis, all costs are reported in real 2011 dollars.

7.1.2 *Compliance Cost Estimates for Known Controls*

In this section, we provide engineering cost estimates for the known controls identified in Chapter 4 that include control technologies for EGUs, non-EGU point, nonpoint and mobile nonroad sources. Onroad mobile source controls were not applied because they are largely addressed in existing rules such as the recent Tier 3 rule. Engineering costs generally refer to the equipment installation expense, the site preparation costs for the application, and annual operating and maintenance costs. Note that in many cases the application of these control strategies does not result in areas reaching attainment for the alternative ozone standards of 70, 65, and 60 ppb and additional emission reductions beyond known controls are needed.

The EPA evaluated the costs of all known NO_x controls contained in the CoST control measures database for this RIA and found that all nonpoint and nonroad sector controls were below $14,000 per ton of NO_x emission reduction, and that the bulk of the non-EGU point source controls were below this cost. Overall, for all NO_x controls prior to application of any cost cap, controls costing less than $14,000 per ton account for 96 percent of known emission reductions. Figure 7-1 represents the marginal cost curve for all the NOx control measures contained in the CoST database. This is an incomplete representation of the marginal abatement cost curve for all NOx abatement, because we do not have information on the control measures and costs for the remaining uncontrolled NOx emissions (see discussion in section 7.2). The controls above $14,000 were investigated and we determined that the higher cost controls were primarily due to errors in the cost equations for a few types of sources (mainly ICI Boilers and Process Heaters) for certain source sizes. We are taking steps to correct these equations for the final ozone RIA. As a result of the error mentioned above, a cost cap of $14,000 per ton of NO_x emissions reduction for the non-EGU point source controls was applied, to remove the incorrectly applied controls from the analysis. A small number of NO_x controls were applied for non-EGU point sources above $14,000 per ton but these had little impact on the overall emission reductions or control cost. A significant portion of the EGU SCR controls were above this level; no cost cap was applied to the EGU SCR controls. Note that control costs for California were lower than for other areas because there were no EGU SCR controls applied in California because there were no coal-fired utility boilers without SCR already in place.

We intended to apply a similar cost cap but inadvertently applied a slightly higher cap of $15,000 per ton of VOC emission reduction. At the beginning of the analysis we were anticipating VOC reductions would play a relatively minor role in the analysis so we did not do a separate marginal cost analysis for VOC. However, for the final ozone RIA we will conduct a separate analysis for VOC and will make costing decisions accordingly.

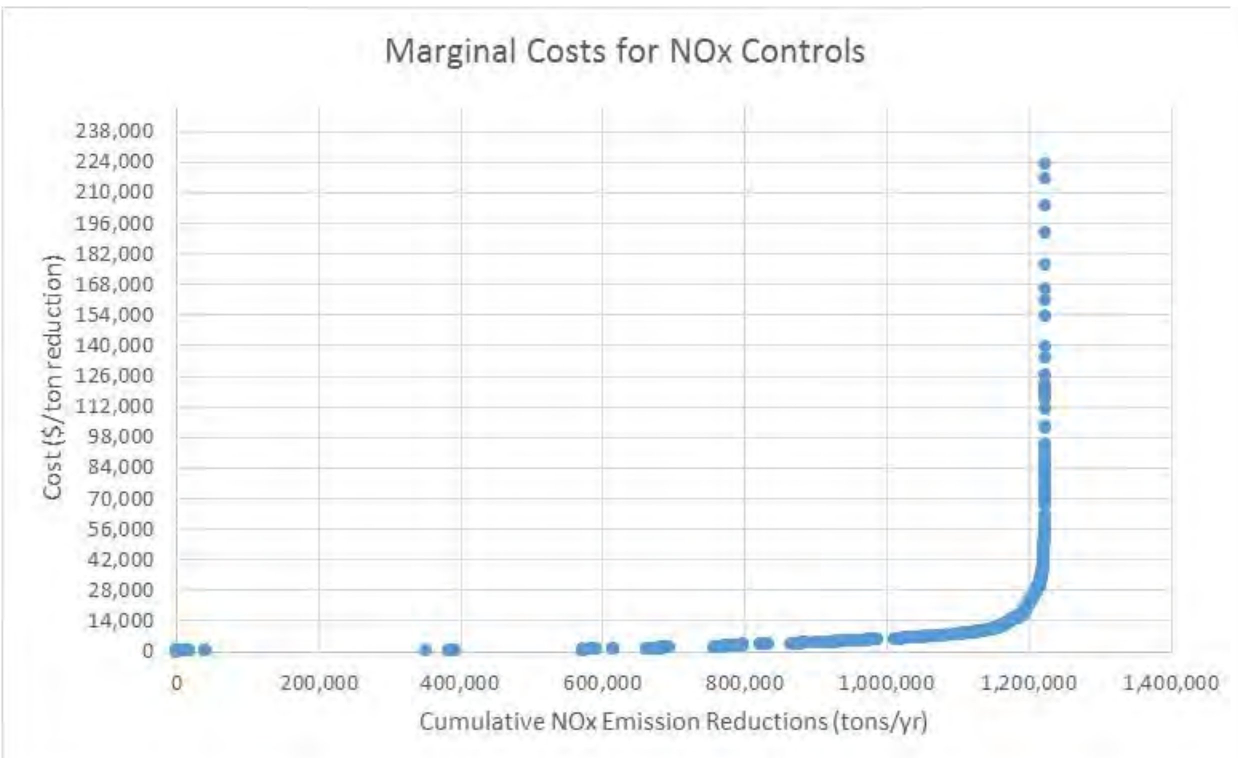

Figure 7-1. Marginal Costs for Known NO_x Controls for All Source Sectors (EGU, non-EGU Point, Nonpoint, and Nonroad)

See Tables 7-1 through 7-3 for summaries of control costs from the application of known controls for alternative standards of 70, 65, and 60 ppb. Costs are listed by sector for both eastern and western U.S., except California and presented at 3 and 7 percent discount rates. Note that any incremental costs for known controls for California (post-2025) for alternative standards of 70, 65, and 60 ppb are zero because all known controls for California were applied in the demonstration of attainment for the baseline standard of 75 ppb.

These numbers reflect the engineering costs annualized at discount rates of 3 percent and 7 percent, which is to the extent possible consistent with the guidance provided in the Office of Management and Budget's (OMB) (2003) Circular A-4. Discount rates refer to the rate at which capital costs are annualized.[111] A higher discount, or interest, rate results in a larger annualized cost of capital estimate. It is important to note that it is not possible to estimate both 3 percent and 7 percent discount rates for a number of the controls included in this analysis. Because we obtain control cost data from many sources, we are not always able to obtain consistent data across original data sources.[112] 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.

In addition, the EGU control costs were not estimated for either 3 or 7 percent. The interest rate used for this analysis reflects an internal rate of return of 11.51 percent for retrofit controls as described in the IPM v5.13 documentation (U.S. EPA, 2013).

For non-EGU point source controls, some disaggregated data is available, and we were able to calculate those costs at both 3 and 7 percent discount rates for those controls. For the alternative standards analyzed in this RIA, approximately 23 percent 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 EGU, nonpoint, or nonroad source controls, total annualized costs for these sectors are assumed to be calculated using a 7 percent discount rate. Because we do not have a full set of costs at the 3 percent discount rate, the 3 percent columns in

[111] In this analysis, the discount rate refers to the interest rate used in the discounted cash flow analysis to determine the present value of future cash flows. A social discount rate is a discount rate used in computing the value of monies spent on social projects or investments, such as environmental protection. The social discount rate is directly analogous to the discount rate we use in the engineering cost analysis, as well as certain rates used in corporate finance (e.g., hurdle rate or a project appropriate discount rate), so the mathematics are identical.

[112] Data sources can include states and technical studies, which do not always include the original data source.

Tables 7-1 to 7-3 reflect the sum of some non-EGU point source controls at a 3 percent discount rate, some non-EGU point source controls at a 7 percent discount rate, and the other sectors at a 7 percent discount rate. With the exception of the 3 percent Total Annualized Cost estimates in Tables 7-1 to 7-3, engineering cost estimates presented throughout this chapter and elsewhere in this document are based on a 7 percent discount rate.

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 $1.6 billion for an alternative annual standard of 70 ppb, $4.2 billion for a 65 ppb alternative standard, and $4.4 billion for a 60 ppb alternative standard. Costs of NO_x controls in terms of dollars per ton of NO_x reduction for the alternative standards analyses were approximately $12,000/ton on average for the EGU sector with a range of $2,000/ton to $38,000/ton; $3,000/ton for the non-EGU point sector on average, with a range of $17 to $90,000/ton; $1,100/ton for the nonpoint sector on average ,with a range of $520 to $2,200/ton; and $4,600/ton for the nonroad sector on average with a range of $3,300/ton to $5,300/ton. The overall average cost range was $2,300 to $3,000/ton for all sectors combined.

The cost trend in terms of dollars per ton of emissions reduction is increasing for some sectors and decreasing for others. The variation is small for most sectors and depends on the sources that happen to be in the geographic areas of control. In general, the same set of controls is being applied at each level of the alternative standards analyzed. The primary difference in the control strategies for the alternative standards is the greater size of the geographic area of control as the stringency of the alternative standards increases. Overall for all sectors as a whole, the average cost per ton is actually increasing slightly with increasing stringency of the alternative standard analyzed. The reason for slight increase in cost per ton relative to the increase in stringency is that newly affected areas are estimated to obtain emissions reductions using slightly more expensive known technologies as the alternative standards become more stringent.

Table 7-1. Summary of Known Annualized Control Costs by Sector for 70 ppb for 2025 - U.S., except California (millions of 2011$)[a]

Geographic Area	Emissions Sector	Known Control Costs	
		7 Percent Discount Rate	**3 Percent Discount Rate**
East	EGU	310[b]	310[b]
	Non-EGU Point	640	620[c]
	Nonpoint	610	610[d]
	Nonroad	22	22[d]
	Total	**1,600**	1,600[e]
West	EGU	-	-
	Non-EGU Point	-	-
	Nonpoint	-	-
	Nonroad	-	-
	Total	-	-
	Total Known Control Costs	**1,600**	**1,600[e]**

[a] All values are rounded to two significant figures.
[b] EGU control cost data is calculated using an 11.51 percent for retrofit controls.
[c] Non-EGU control cost data is calculated at a 3% interest where control cost equations are utilized.
[d] Nonpoint and nonroad control costs are calculated using a 7% interest rate because no 3% data exists.
[e] 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.

Table 7-2. Summary of Known Annualized Control Costs by Sector for 65 ppb for 2025 - U.S., except California (millions of 2011$)[a]

Geographic Area	Emissions Sector	Known Control Costs	
		7 Percent Discount Rate	3 Percent Discount Rate
East	EGU	1,500[b]	1,500[b]
	Non-EGU Point	1,100	1,100[c]
	Nonpoint	1,100	1,100[d]
	Nonroad	53	53[d]
	Total	**3,800**	**3,800[e]**
West	EGU	230[b]	230[b]
	Non-EGU Point	86	86[c]
	Nonpoint	87	87[d]
	Nonroad	5.7	5.7[d]
	Total	**410**	**410[e]**
	Total Known Control Costs	**4,200**	**4,200[e]**

[a] All values are rounded to two significant figures.
[b] EGU control cost data is calculated using an 11.51 percent for retrofit controls.
[c] Non-EGU control cost data is calculated at a 3% interest where control cost equations are utilized.
[d] Nonpoint and nonroad control costs are calculated using a 7% interest rate because no 3% data exists.
[e] 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.

Table 7-3. Summary of Known Annualized Control Costs by Sector for 60 ppb for 2025 - U.S., except California (millions of 2011$)[a]

Geographic Area	Emissions Sector	Known Control Costs	
		7 Percent Discount Rate	3 Percent Discount Rate
East	EGU	1,500[b]	1,500[b]
	Non-EGU Point	1,200	1,100[c]
	Nonpoint	1,100	1,100[d]
	Nonroad	53	53[d]
	Total	**3,800**	**3,700[e]**
West	EGU	400	400[b]
	Non-EGU Point	98	97[c]
	Nonpoint	88	88[d]
	Nonroad	5.7	5.7[d]
	Total	**590**	**590[e]**
	Total Known Control Costs	**4,400**	**4,400[e]**

[a] All values are rounded to two significant figures.
[b] EGU control cost data is calculated using an 11.51 percent for retrofit controls.
[c] Non-EGU control cost data is calculated at a 3% interest where control cost equations are utilized.
[d] Nonpoint and nonroad control costs are calculated using a 7% interest rate because no 3% data exists.
[e] 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.

7.2 The Challenge of Estimating Costs for Unknown Controls

As described in Chapter 4, the known control measures were applied to EGU, non-EGU point, nonpoint (area), and nonroad mobile sources for demonstration of attainment with the current and alternative standards. Table 4-7 lists the specific control technologies applied in the known control analysis. There were several areas where known controls did not achieve enough emissions reductions to attain the alternative standards of 70, 65, and 60 ppb. To complete the analysis, the EPA then estimated the additional emissions reductions beyond known controls needed to reach attainment, also referred to as unknown controls. For information on the methodology used to develop the emissions reductions estimates, see Chapter 3.

The estimation of engineering costs for unspecified emission reductions needed to reach attainment many years in the future is inherently a difficult task. This is because it is likely that the abatement supply function will shift out or change shape over time due to a variety of economic, technical, and regulatory influences. Our experience with Clean Air Act implementation shows that numerous factors, such as technical change and development of innovative strategies, can lead to emissions reductions that may not seem possible today, while potentially reducing costs over time. For example, facility-level data collected through the U.S. Census Bureau's Pollution Abatement Costs and Expenditures (PACE) survey suggests that this may have happened in the manufacturing sector in recent decades. Based on surveys of approximately 20,000 plants classified in manufacturing industries, the PACE data show during the 1994-2005 time period, a period of increasing regulatory stringency, spending on air pollution abatement as a percentage of revenues decreased for the manufacturing sector.[113] Although exogenous factors such as changes in economic conditions may have

[113] The Pollution Abatement Costs and Expenditures (PACE) survey collects facility-level data on pollution abatement capital expenditures and operating costs for compliance with local, state, and federal regulations and voluntary or market-driven pollution abatement activities. In 2005, the most recent year PACE data were collected, the U.S. manufacturing sector spent $3.9 billion dollars on air capital expenditures and incurred $8.6 billion dollars in operating costs for air pollution prevention and treatment. These figures represent less than 3% of total new capital expenditures and less than 0.18% of total revenue for the manufacturing sector, respectively. These percentages have declined since 1994, when air capital expenditures were less than 4% of total new capital expenditures and air pollution abatement operating costs were less than 0.2% of total revenue. Levinson (2009) finds that most of the pollution reductions in the U.S. come from changes in technology as opposed to changes in imports or changes in the types of domestically produced goods. He finds that even though manufacturing output increased by 24% from 1987 to 2001, emissions of four common air pollutants from the sector declined 25% over that time period and the most important factor contributing to the decrease in pollution is technical change or innovation.

contributed to the relative share in costs of pollution abatement, it is also possible that technological change and innovation may have contributed to this relative decline.

In addition to considering the potential for technological innovation, it is also important to understand that EPA's control strategy tools largely focus on a limited set of emissions inventory sectors, whereas abatement opportunities exist in other sectors. EPA's control strategy tools undergo continuous improvement, and as the need for additional abatement opportunities grows, more evaluation of uncontrolled emissions takes place. During these evaluations, additional abatement opportunities from applying identified controls typically are found.

This section discusses various factors that must be considered in developing a methodology for estimating the costs of unknown controls. First, we explain why the abatement supply curve from known controls presented in the previous section provides an incomplete picture of all currently available abatement opportunities. Second, we show how, as time passes and the EPA reviews NAAQS standards, relevant information is revealed in the current RIA development process that was not available to analysts developing RIAs for previous reviews, such as unforeseen regulatory programs or other exogenous factors that account for significant emissions reductions. Third, we discuss a related issue, that technical change may affect the marginal abatement cost curve. Additionally, we present evidence from the literature that regulatory action can act as a forcing function for technical change. Fourth, we discuss how regulatory costs can decrease over time as regulated entities gain experience reducing emissions and reduce per unit costs, commonly referred to as "learning by doing". Fifth, we discuss how NOx offset prices could serve as reasonable proxies for the costs associated with emissions reductions from unknown controls. Finally, we describe how we use this information to help inform the unknown control cost methodology applied in section 7.3.

7.2.1 Incomplete Characterization of NOx Marginal Abatement Cost Curves

Underlying the selection of controls as described in Appendix 4A is the concept of the marginal abatement cost curve (MACC). The marginal abatement cost curve (MACC) is a representation of how the marginal cost of additional emissions abatement changes with increasing levels of abatement. Adding new technologies, or changing either the abatement amount or cost of the technology, will change the shape of the overall MACC.

In developing engineering cost estimates in section 7.1, the focus was largely on end-of-pipe controls and only includes limited process-oriented control measures, such as switching to lower-emitting fuel or energy sources and the installation of energy efficiency measures. These measures can result in significant emissions abatement, but are not reflected in the marginal abatement cost curve based on traditional control measures. As a result, the MACC derived in the previous section from known controls represents an incomplete supply curve that partially captures the “true” abatement supply. An illustrative, hypothetical depiction of an “observed but incomplete” MACC and the “true” underlying MACC is presented in Figure 7-2.

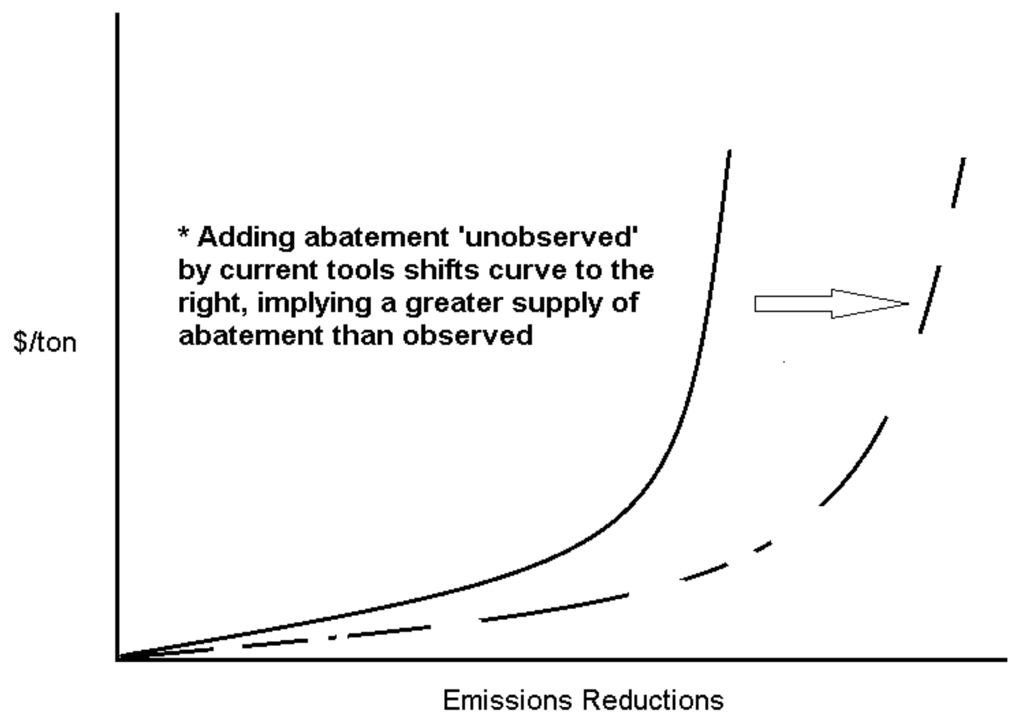

Figure 7-2. Observed but incomplete MACC (solid line) based on known controls identified by current tools and complete MACC (dashed line) where gaps indicate abatement not identified by current tools

In the figure, the solid line traces out a hypothetical observed MACC, while the dashed line characterizes the combination of observed and unobserved abatement possibilities. The inclusion of the unobserved abatement pushes the abatement supply out.

Due to the incomplete characterization of the full range of the MACC, it is important to understand the composition of the cost information that is available to construct the partial

MACC. The nature of available information on the cost of NOx abatement measures is somewhat complex. The highest cost per ton estimates are often associated with controls that achieve very small total reductions in NOx or are special cases. For example, in some cases, controls have been developed primarily to address other pollutant emissions, such as SO_2, but achieve NOx reductions as a co-benefit. These controls are well characterized in the CoST database because they have been applied for SO_2 control, but the degree to which sources would adopt those controls for NOx is uncertain, and it is unlikely that those very high per ton cost measures would be the efficient marginal abatement cost (MAC) level for the targeted level of NOx emissions reductions needed for attainment. In addition, there are some controls that have been identified as applicable for some sources even though they have very high marginal costs per ton of NOx, because those controls are required to meet other provisions of the CAA. For example, LNB+SCR for process heaters is relatively common at refineries, mainly due to a New Source Review enforcement initiative that has been underway since 2001. These plants are being required to install the controls regardless of cost as part of the enforcement action, and thus the costs may not represent the actual marginal cost at the level of abatement needed to reach the NAAQS attainment targets.

Lack of information about the MAC for emissions reductions not characterized in CoST is not an indication that controlling those tons is necessarily more difficult than controlling NOx from other sources that are in the database, or that the MAC for those tons is necessarily higher than all of the costs of controls already in the database. Some sectors are controlled at a higher rate than others, and in those cases, getting additional NOx reductions may indeed require higher cost controls. For example, EGU NOx has been heavily controlled, and the additional SCR units applied in the analysis are relatively more expensive per ton than the typical SCRs that have been applied in past analyses. However, other sectors may not be as well-controlled, and lower cost controls may be available.

7.2.2 Comparison of Baseline Emissions and Controls across Ozone NAAQS RIAs from 1997 to 2014

While each ozone NAAQS analysis since 1997 has required at least some emissions reductions from controls that were unknown at the time of the analysis, evidence suggests that over time new information on exogenous factors affecting baseline emissions and emissions controls becomes available that can shift emissions reductions from the unknown to the known category. Some exogenous factors that might affect baseline emissions include changes in economic conditions that may affect production levels, as well as plant closures and openings. Baseline emissions may also be affected by EPA or state regulations that require specific controls that may not be fully characterized in the set of known controls applied in an earlier RIA, or by EPA or state regulations targeting other pollutants, e.g., air toxics, that may result in reductions in NOx or VOC as a co-benefit. For example, in the 1997 ozone NAAQS RIA, the NOx emissions reductions from the mobile source Tier 2 standards were not included as known controls, even though the RIA acknowledged the potential for these standards to provide substantial cost effective controls. Likewise, the 2008 ozone NAAQS RIA did not include controls on EGUs reflecting the Mercury and Air Toxics Standards or the Clean Power Plan. As a result, emissions reductions from unknown controls were much higher in some regions of the U.S. than they are in this RIA.

Furthermore, many of these emission reductions may be achieved at a cost less than was originally applied to emissions reductions from unknown tons. Several of the large NOx-reducing regulations issued between the 1997 and 2008 RIAs had *ex ante* estimates of costs per ton of NOx reduced well below the $10,000 per ton value applied to emissions reductions from unknown controls in the 1997 Ozone NAAQS RIA. Table 7-4 provides information on the cost per ton for NOx reductions from five major NOx-reducing regulations. To the extent possible, we determined from the RIA for each regulation the projected emissions reductions in 2010 (or the closest year) and the estimated costs in 2010. Costs were adjusted from the year reported in the RIAs to constant 2010 dollars. In some cases, only an annualized cost was reported. In those cases we divided the year specific emissions projection by the annualized cost, recognizing that this may under or overstate the actual year specific cost per ton. Where possible, we report the separate costs for NOx emissions reductions. However, for most programs, total costs which include reductions in multiple pollutants are reported. As such, the cost per ton of NOx alone is likely overstated.

Table 7-4. Emissions and Cost Information for Major NOx Rules Issued Between 1997 and 2008

Regulation	Projected NOx Emissions Reductions in 2010 (thousands)	Total Annual Cost in 2010 (Million 2010$)	Average Cost/Tons NOx Reduced (2010$)

	(A)	(B)	(C = B/A)
NOx SIP Call[a]	1,141	2,515	$2,204
Clean Air Interstate Rule[b]	1,200	$3,034	$2,528
Tier 2 Standards[c]	1,236	$5,256	$4,253
Heavy Duty Diesel Engines[d]	403	$4,570	$11,340
Nonroad Diesel[e]	203 (in 2015)	$660	$3,251

[a] Costs and emissions reductions obtained from Table ES-2 in the Regulatory Impact Analysis for the NOx SIP Call, FIP, and Section 126 Petitions, Volume 2: Health and Welfare Benefits (U.S. EPA, 1998).
[b] Emissions reductions obtained from Table 7-2 and Costs obtained from Table 7-3 in the Regulatory Impact Analysis for the Final Clean Air Interstate Rule (U.S. EPA, 2005).
[c] Costs and emissions reductions obtained from Appendix VI-C in the Regulatory Impact Analysis - Control of Air Pollution from New Motor Vehicles: Tier 2 Motor Vehicle Emissions Standards and Gasoline Sulfur Control Requirements (U.S. EPA, 1999).
[d] Emission reductions obtained from Table II.B-4, and costs obtained from Table V.D-1 in the Regulatory Impact Analysis: Heavy-Duty Engine and Vehicle Standards and Highway Diesel Fuel Sulfur Control Requirements (U.S. EPA, 2000)
[e] .Emissions reductions obtained from Table 8.6-1 (NOx+NMHC), and costs obtained from Table 8.5-2 (NOx+NMHC) in the Final Regulatory Impact Analysis: Control of Emissions from Nonroad Diesel Engines (U.S. EPA, 2004). In this RIA, no information was provided for NOx emissions alone, in all cases, NOx+non-methane hydrocarbons (NMHC) was provided. Calculated cost per ton is thus the average for NOx and NMHC emissions reductions, rather than just for NOx emissions reductions.

The NOx State Implementation Plan (the "NOx SIP call") was partially represented in the 1997 RIA. The other rules in Table 7-4 were not included in 1997 RIA, although the RIA notes that the Tier 2 standards may be a way to get additional reductions to meet the ozone NAAQS. Table 7-4 shows that for these five major regulations, which account for 4.1 million tons of NOx reductions, the expected average cost per ton ranged from about $2,200 to $11,300 per ton of NOx reduced. Only in the single case of the heavy duty diesel engine rule was the cost per ton NOx reduced expected to be greater than the $10,000/ton value used for unknown controls in the 1997 RIA. This suggests that unknown controls may be implemented at lower cost than the highest point of the MACC for known controls anticipated in the RIA.

Comparing the MACC over time and across analyses is complicated because of (1) differences in the regions or areas affected by the proposed changes to the ozone standards, (2) the information available at the time of the analyses, and (3) analytical assumptions made about the universe of sources that can be controlled. Table 7-5 provides information about assumptions used in generating the costs of control measures for the 1997, 2008, and 2014 RIAs.

Table 7-5. Comparison of Key Assumptions Used in Developing Estimates of NOx Emissions Controls and Costs across Past and Current Ozone NAAQS Regulatory Impact Analyses [a]

	1997	2008	2014
Sectors Excluded	No additional utility NOx controls beyond Title IV and OTAG recommendations. Some mobile source control measures excluded due to mismatch between attainment dates and implementation timelines for the control measures.	Utility NOx controls beyond baseline only applied in the East (including East TX)	On-road mobile sources.
Geographic Definition of Control Areas	Exceeding county plus CMSA for county exceeding the standard. If no CMSA, just use the exceeding county (12 areas in East, 7 areas in West)	Non-EGU point and area controls applied to counties exceeding the standard plus surrounding counties out to 200km. Some additional controls were placed on large point sources in counties touching the buffer. For mobile controls, both local (within 200 km buffer) and statewide controls were applied. Counties outside the state in which the exceeding county resides are excluded. Controls were applied to all states in the OTC excepting VT.	NOx controls for 70 ppb were applied to counties exceeding the standard, then surrounding counties within 200 km that were also within state boundaries (Texas, California) and within OTC boundaries including counties closest to exceeding counties first (Northeast). Where more controls were needed they were applied to counties in other states within the region, nearest the county exceeding the standard (Oklahoma, Arkansas, Louisiana and Kansas for Houston). VOC controls for 70 ppb were applied to special areas and counties within 100 km from them (California North and South, Dallas, Houston, Baltimore, and New York City). NO_X controls for 65 and 60 ppb were applied in regions (California, Southwest, Central, Midwest, Northeast). VOC controls were applied in special areas plus counties within 100 km from these special areas (Denver, Dallas, Houston, Chicago, New York City)
Cost Cap	$10,000/ton (1990 dollars) – justification is that states generally have not chosen to require existing sources to apply control measures with incremental costs above the threshold, even in severe areas like the South Coast of CA. Sensitivity on $7,000/ton to $20,000/ton.	$23,000/ton (2006$) for non-EGU point and area sources – 98% of possible reductions are achieved at 82% of total costs. Based on evaluation of marginal cost curves for all counties in control areas (1,300 counties).	For NOx: $14,000/ton for point non-EGU sources, although an error resulted in a small number of point non-EGU controls above $14,000/ton. No nonpoint (area) source NOx controls were above $14,000/ton so cost caps had no effect on nonpoint source controls. For ICI boilers, costs were calculated through equations without cost constraints.

	1997	2008	2014
			For VOC: $15,000/ton for point non-EGU and nonpoint (area) sources
Minimum Emission Reduction	Not clearly defined	For NOx: 5 tons for point non-EGU and non-point sources For VOC: 1 ton for point non-EGU and non-point	For NOx: 5 tons for point non-EGU and non-point For VOC: 1 ton for point non-EGU and non-point
Minimum Emissions	Not clearly defined	50 tons for NOx	25 tons for NOx
Assumed Control Effectiveness (% of intended effect)	95 to 100 percent	Variable	Variable
Rule Penetration[b]	Obtained from published reports from state and local agencies	75% for onroad and nonroad SCR and diesel particulate filters	75% penetration in CA and 25% in TX and Northeast for nonroad

[a] These analyses also included controls for VOC emissions, however, those controls were generally very local in nature. The current analysis is focused primarily on controlling NOx to meet the alternative ozone standards, with relatively modest VOC controls where they are expected to help reach attainment. Also, for the 1997 and 2008 RIAs, attainment of the PM standards in place at the time of the analysis was assumed, and as a result, some additional NOx measures were already in place.

[b] Percent of county-level mobile or area source inventory affected by control measure.

7.2.3 Impact of Technological Innovation and Diffusion

In general, the MACC at any particular point in time for a defined set of emitting sectors will be an increasing function of the level of abatement. That is, marginal costs are increasing as the amount of emissions are reduced. However, in regulatory analyses of NAAQS, we are typically assessing costs of abatement in a future year or years selected to represent implementation of the standards. As such, a MACC constructed based on currently available information on abatement opportunities will not be the best representation of a future MACC. MACC in the future can differ from MACC in the present due to technological innovation and diffusion, such as the introduction of new technologies or improvements in effectiveness or applicability of existing technologies. Additionally, environmental policy can create incentives and constraints that influence the rate and direction of technical change (Jaffe et al. 2002) as well as the rate of diffusion and adoption of the innovations (Sterner and Turnheim 2009) .

In the context of emissions controls examined in this RIA, technological innovation and diffusion can affect the MACC in several ways. The following bullets present some examples of the potential effects of technical change:

Case 1: New control technologies can be developed that cost less than existing technologies.

Case 2: A new control technology is developed to address an uncontrolled emissions source, at a higher marginal cost than existing technologies, but still lower than the cost threshold value.

Case 3: The efficiency of an existing control measure increases. In some cases, the control efficiency of a measure can be improved through technological advances.

Case 4: The cost of an existing control measure decreases.

Case 5: The applicability of an existing control measure to other emissions sources increases.

Overall, these five cases describe ways that technological change can reduce both the amount of unidentified abatement needed, decrease the MAC, decrease average costs, and decrease total costs relative to the case where it is assumed that technological change does not occur in response to increased demand for abatement. It is also possible in cases where there is a strictly binding emissions reduction target that new technologies can be introduced and adopted with much higher marginal costs. However, if there are cost off-ramps, such as those provided by Section 185 of the CAA, those higher cost technologies will not be adopted.

Regulatory policies can also help induce technological change when a standard cannot be met either (1) with existing technology or (2) with existing technology at an acceptable cost, but over time market demand will provide incentives for industry to invest in research and development of appropriate technologies. These incentives are discussed in Gerard and Lave (2005), who demonstrate that the 1970 Clean Air Act induced significant technical change that reduced emissions for 1975 and 1976 automobiles. Those mandated improvements went beyond the capabilities of existing technologies by using regulatory pressure to incentivize the development of catalytic converting technology in 1975. Induced technological change can correspond to Cases 1 through 3 above.

There are many other 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 NOx burners for NOx emissions
- Scrubbers that achieve 95 percent or 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 with lower emissions 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
- Increasing market penetration of gas-electric hybrid vehicles and cleaner fuels

These technologies were not commercially available two decades ago, and some were not even in existence. Yet today, all of these technologies are on the market, and many are widely employed. Several are key components of major pollution regulatory programs.

As Brunnermeier and Cohen (2003) demonstrate, there is a positive correlation, other things held constant, between environmental innovations (measured as the number of relevant environmental patent applications) and specific regulations imposed on an industry (measured in terms of the frequency of government compliance inspections). Lanjouw and Mody (1996) show empirically a positive relationship between responses to environmental regulations (i.e., increases in pollution abatement expenditure) and new technology (i.e., relevant patent applications) in the United States, Japan, and Germany. They show that in each of these countries, even though on different timelines, the share of environmental patents increased considerably in response to stricter environmental regulations. Similarly, Popp (2004) studied the relationship between environmental regulation and new technology focusing on SO_2 and NO_x. The study was performed using patent data from the United States, Japan, and Germany. Popp found that more stringent regulation enhanced domestic patenting by domestic inventors.

While regulation may influence the direction and intensity of emissions-related research and development activities, "crowding out" of investment resources may occur as resources are directed away from other opportunities, potentially leading to opportunity costs that offset savings resulting from research and development successes (Popp and Newell 2012). In a study that links energy-related patent activity and firm financial data, Popp and Newell (2012) find that while increases in alternative energy patents result in fewer patents for other energy technologies, this result is due to firm-level profit-maximizing behavior rather than constraints on the magnitude of research and development resources. Alternatively, Kneller and Manderson (2012) find evidence in the United Kingdom that environment-related research and development resulting from more stringent regulation may crowd out other research and development activities but that environment-related capital does not crowd out non-environmental capital. Another factor to consider is the degree to which a particular sector is likely to be close to fully controlled, e.g., in comparing existing emissions with uncontrolled emissions levels, is the percent of control close to 100 percent? In those cases, achieving additional reductions through technological change is likely to be more difficult and costly, because the benefits of investment in those technologies is smaller, due to smaller remaining potential for abatement.

7.2.4 Learning by Doing

What is known as "learning by doing" or "learning curve impacts" has also made it possible to achieve greater emissions reductions than had been feasible earlier, or reduce the costs of emissions control relative 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. This type of change corresponds to case 4 in the discussion of the ways technological change can affect the MACC that appeared earlier. Such impacts have been identified to occur in a number of studies conducted for various production processes. These impacts would manifest themselves as a lowering of expected costs for operation of technologies in the future below what they may otherwise have been. For example, Rubin et al. 2004 show that capital costs of Flue gas desulphurization (FGD) and selective catalytic reduction (SCR) systems have decreased over time as a result of research and development activities and learning by doing, among other factors, and that failing to account for these technological dynamics can lead to incorrect estimates of future regulatory costs.

Rubin et al. (2012) also note that when technologies succeed, costs tend to fall over time. They offer the example of post-combustion SO_2 and NO_x combustion systems. After an increase in costs during an initial commercialization period, costs decreased by at least 50 percent over the course of two decades. Table 9.5 in the 1997 Ozone NAAQS RIA summarizes historical and projected "progress ratios" for existing technologies. These ratios show declining costs over time, due to learning by doing, economies of scale, reductions in O&M costs, and technological improvements in manufacturing processes. There are other discrete examples, for example prices of the catalyst used in operating SCR have dropped dramatically over time. From 1980 to 2005, catalyst prices dropped by roughly 85 percent (Cichanowicz, 2010). This follows the "learning curve," which finds that production and implementation costs decrease as learning and repetitive use occurs. In addition, Table 3a.7 in Appendix 3a of the 2008 Ozone NAAQS RIA lists controls applied to new source types (Case 3). For example, SCR is now applicable to the cement manufacturing sector, and SNCR is now applicable to a large number of additional boiler source categories. In some cases, more effective controls were determined to be applicable where in past cases, less effective controls were applied. For example, for industrial and manufacturing incinerators, where previously SNCR was the NOx control technology, SCR was applied in 2008, increasing the control efficiency from 45 percent to 90 percent.

The magnitude of learning curve impacts on pollution control costs has been estimated for a variety of sectors as part of the cost analyses done for the Direct Cost Estimates Report for the

Second EPA Section 812 Prospective Analysis of the Clean Air Act Amendments of 1990.[114] In the Report, learning curve adjustments were included for those sectors and technologies for which learning curve data was available. For all technologies and industries, a default learning rate of 10 percent was adopted based on SAB advice. No adjustments were used for on-road and non-road controls. The 10 percent adjustment is a 10 percent cost reduction per doubling of emission reductions. The literature supports a rate of up to 20 percent for many technologies (Dutton and Thomas, 1984). The impact of this on costs in the Report was to reduce costs of local controls in nonattainment areas by 9.9 percent in 2020.

A typical learning curve adjustment is to reduce either capital or operation and maintenance costs by a certain percentage given a doubling of output from that sector or for that technology. In other words, capital or operation and maintenance costs will be reduced by some percentage for every doubling of output for the given sector or technology. In addition, learning by doing may also lead to instances where existing control technologies are found to be applicable to additional sources. This corresponds to Case 5 in the discussion in section 7.2.1. For example, scrubber technologies applied to electric utilities have been adapted to apply to industrial boilers. As a result of this increased applicability due to learning, potential abatement has increased at a cost less than the cost threshold. For this RIA, however, we do not have the necessary data to properly generate control costs that reflect learning curve impacts.

7.2.5 Using Regional NOx Offset Prices to Estimate Costs of Unknown Emissions Controls

In ozone nonattainment areas, new sources interested in locating in that area and existing sources interested in expanding are required to offset any emissions increases. If those emissions increases are NO_x emissions, the source typically purchases NO_x emission reduction credits (ERCs), or offsets, from within that particular nonattainment area. Within nonattainment areas, offset prices fluctuate because of changes in the available supply of offsets and changes in demand for offsets. Offset supply increases when facilities shut down or when they make

[114] Industrial Economics, Incorporated and E.H. Pechan and Associates, Direct Cost Estimates for the Clean Air Act Second Section 812 Prospective Analysis: Final Report, prepared for U.S. EPA, Office of Air and Radiation, February 2011. Available at http://www.epa.gov/cleanairactbenefits/feb11/costfullreport.pdf.

process or other changes that reduce emissions permanently. Offset demand depends on the industrial base in a given area and fluctuates with changes in economic growth. For example, in the San Joaquin Valley, in recent years offset prices have increased because of increased oil and gas industry development.

We identified historical NO_x offset prices in several nonattainment areas, including the San Joaquin Valley and the South Coast in California, Houston, TX, and New York region. For the San Joaquin Valley Air Pollution Control District, we collected information on NO_x offset prices using the California Air Resources Board's *Emission Reduction Offset Transaction Cost Summary Reports* for 2002 through 2013[115]. For the South Coast Air Quality Management District, we collected information on prices for perpetual NO_x RECLAIM Trading Credit (RTC) for 2003 through 2012 from the *Listing of Trade Registrations*.[116] Lastly, we collected information on NOx offset prices in the Houston-Galveston nonattainment area for 2010 through 2013 from the *Trade Report*[117] and the New York-New Jersey-Connecticut nonattainment area from 2000 through 2013 from industry representatives.

Table 7-6 presents the price data we were able to collect for these four regions, adjusted to 2011 dollars using the Gross Domestic Product Implicit Price Deflator. The offset prices in this table are denominated in units of perpetual tons, or tons per year. The prices constitute average of the trades in the regions for the year given.

[115] http://www.arb.ca.gov/nsr/erco/erco.htm

[116] http://www.aqmd.gov/home/programs/business/about-reclaim/reclaim-trading-credits

[117] http://www.tceq.state.tx.us/airquality/banking/mass_ect_prog.html

Table 7-6. Average NO_x Offset Prices for Four Areas (2011$) [a]

	Annualized NOx Offset Prices ($/ton)			
	San Joaquin Valley	**California South Coast**	**Houston TX**	**New York Region**
2000	N/A	N/A	N/A	25,000
2001	N/A	N/A	N/A	12,000
2002	36,000	N/A	N/A	12,000
2003	28,000	N/A	N/A	12,000
2004	25,000	12,000	N/A	12,000
2005	25,000	31,000	N/A	11,000
2006	21,000	163,000	N/A	11,000
2007	21,000	206,000	N/A	N/A
2008	48,000	210,000	N/A	N/A
2009	58,000	128,000	N/A	N/A
2010	62,000	98,000	36,000	N/A
2011	64,000	56,000	N/A	N/A
2012	47,000	47,000	N/A	N/A
2013	42,000	N/A	97,000	4,000
Average	40,000	106,000	66,000	12,000
Maximum	64,000	210,000	97,000	25,000

[a] All values are rounded to two significant figures.

The data series for the California regions are more complete than those for Houston and the New York region. We are working to obtain more complete data series for future analysis.

To more directly compare offset prices to potential annual costs for unidentified emissions controls, we annualized the tons per year prices using the same engineering cost equations as used in the main analysis to estimate annualized control cost. We converted the offset cost to an annual costs by using the capital recovery factor (CRF) discussed in Section 7.1.1. In a capital cost context, the CRF incorporates the interest rate and lifetime of the purchased capital. In this instance, although the offsets are perpetual in nature, we assumed a lifetime of 20 years in order to make the cost basis more comparable to the control cost estimates. Also, we used 7 percent for the interest rate. Table 7-7 presents the average and maximum annualized NO_x offset prices in 2011 dollars.

Table 7-7. Annualized NOx Offset Prices for Four Areas (2011$)[a]

	Annualized NOx Offset Prices ($/ton)			
	San Joaquin Valley	California South Coast	Houston TX	New York Region
Average	$ 4,000	$ 10,000	$ 6,000	$ 1,000
Maximum	$ 6,000	$ 20,000	$ 9,000	$ 2,000

[a] All values are rounded to two significant figures.

From an economic perspective, these offset prices may represent the shadow value of a ton of emissions. It is possible that these offset prices could serve as reasonable proxies for the costs associated with emissions reductions from unknown controls. The cost information informing the known control strategy traces out an incomplete marginal abatement cost curve in that, as discussed in Chapter 4, the controls used in the known control analysis are primarily end-of-pipe technologies. The known control estimates for NO_x do not account for other forms of abatement, switching to lower emitting fuels or increasing energy efficiency, for example. The estimates also do not account for institutional or market arrangements that allow firms to buy or sell emissions offsets in nonattainment regions with emissions constraints. These voluntary exchanges may enable abatement at lower costs than may otherwise be available. The benefit of these market transaction data is that the prices are revealed by the interaction of offset supply and demand in regions with differentiated characteristics and air quality profiles.

7.2.6 Conclusion

The preceding sections have discussed the ways in which various factors might affect the observed marginal abatement costs and the resulting total abatement costs estimated in this RIA. Based on past experience with Clean Air Act implementation, the EPA believes that it is reasonable to anticipate that the marginal cost of emissions reductions will decline over time due to technological improvements and more widespread adoption of previously considered niche control technologies as well as the development of innovative strategies.[118] As the EPA continuously improves its data and tools, we expect to better characterize the currently unobserved pieces of the MACC. As a result of our consideration of these complexities, we are

[118] See Chapter 4, Section 4.5 for additional discussion of uncertainties associated with predicting technological advancements that may occur between now and 2025.

currently unable to quantitatively predict future shifts in the abatement supply curve because many factors are intertwined and data are incomplete or highly uncertain.

7.3 Compliance Cost Estimates for Unknown Emissions Controls

This section presents the methodology and results for the costs of emissions reductions from unidentified controls needed for attainment of the alternative ozone standards. As discussed in Chapter 4, the application of the modeled control strategy was not successful in reaching full nationwide attainment of the alternate ozone standards. Many areas remained in nonattainment under all four alternate standard scenarios. Therefore, the engineering costs detailed in Section 7.1 represent only the costs of partial attainment.

7.3.1 Methods

Prior to presenting the methodology for estimating costs for unspecified emission reductions in this RIA, it is important to provide information from EPA's Science Advisory Board Council Advisory,[119] dated June 8, 2007, on the issue of estimating costs of unidentified control measures:

> *812 Council Advisory, Direct Cost Report, Unidentified Measures (charge question 2.a)*
>
> *"The Project Team has been unable to identify measures that yield sufficient emission reductions to comply with the National Ambient Air Quality Standards (NAAQS) and relies on unidentified pollution control measures to make up the difference. Emission reductions attributed to unidentified measures appear to account for a large share of emission reductions required for a few large metropolitan areas but a relatively small share of emission reductions in other locations and nationwide.*
>
> *"The Council agrees with the Project Team that there is little credibility and hence limited value to assigning costs to these unidentified measures. It suggests taking great care in reporting cost estimates in cases where unidentified measures account for a significant share of emission reductions. At a minimum, the components of the total cost associated with identified and unidentified measures should be clearly distinguished. In some cases, it may be preferable to not quantify the costs of unidentified measures and to*

[119] U.S. Environmental Protection Agency. June 2007. Advisory Council on Clean Air Compliance Analysis (COUNCIL), Council Advisory on OAR's Direct Cost Report and Uncertainty Analysis Plan. Washington, DC.

simply report the quantity and share of emissions reductions attributed to these measures.

"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."

While we have considered alternative methodologies to predict future abatement supply curves, we are currently unable to quantitatively predict future shifts in the supply curve with sufficient confidence to use in this RIA. For most NAAQS RIAs prepared during the past five years, EPA estimated the costs for unidentified controls using a pair of methodologies: what we termed a "fixed cost" approach, following the SAB advice, and a "hybrid" approach that has not yet been reviewed by the SAB. We now refer to the fixed-cost approach as the "average cost" approach because it more accurately characterizes the concepts underlying the approach. The average cost methodology uses an assumed national average cost per ton for unidentified controls needed for attainment, as well as two alternative assumed values employed for sensitivity analysis. The range of estimates reflects different assumptions about the cost of additional emissions reductions beyond those in the modeled control strategy. While we use a constant cost per ton of emissions reduction to estimate the costs of the emissions reductions beyond known controls, this does not imply that marginal costs are not increasing in needed emissions abatement. Rather, the average cost per ton is intended to capture what might be the total costs associated with the abatement of the emissions reductions from unknown controls.

The alternative estimates implicitly reflect different assumptions about the amount of technological progress and innovation in emission reduction strategies. The average cost methodology reflects a view that because no cost data exists for unspecified future strategies, it is unclear whether approaches using hypothetical cost curves will be more accurate or less accurate in forecasting total national costs of unspecified controls than an average-cost approach that uses a range of national cost per ton values.

The hybrid cost methodology assumed increasing marginal costs of control along an upward-sloping marginal cost curve. The hybrid cost methodology assumed the rate of increase in the marginal costs of abatement is proportional to the weighted ratio of the amount of abatement using identified controls to the remaining needed abatement using unidentified controls.[120] Under this approach, the relative costs of unspecified controls in different geographic areas reflected the expectation that average per-ton control costs are likely to be higher in areas needing a higher ratio of emissions reductions from unspecified and known controls. However, the weight, which reflected the anticipated degree of difficulty of achieving needed emissions reductions, and the ratios that informed the slope of the marginal abatement cost curve in previous NAAQS analyses were strong assumptions that have not been empirically tested.

When used to estimate costs for end-of-pipe technologies, the hybrid methodology assumed all emissions reductions come from the highest cost margin of the abatement supply curve which, as explained in the previous section, is unlikely for much of the unobserved abatement capacity in the present and future. For example, EPA's control strategy tools largely focus on a limited set of emissions inventory sectors, whereas abatement opportunities exist in other sectors. When new abatement opportunities are identified in other sectors, they typically are not at the higher end of the cost curve.

For areas needing significant additional emission reductions, much pollution abatement is likely needed from sources within regulated sectors that historically have not been intensively regulated. However, if national standards become more stringent, new regions or firms will be added to the regulated domain. These new entrants, with their relatively untapped abatement supply, will contribute to an outward shift in abatement supply. The newly regulated regions and firms will also face new incentives for technical change and innovation that may lower costs over the long run by developing new, more efficient compliance strategies. Because the point of departure for the hybrid approach cost curve is based on our current database, which includes only existing controls, it will systematically overstate future costs if any cost-reducing technological change occurs.

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[120] See, for example, Section 7.2 and Appendix &.A.2 is the December 2012 RIA for the final PM NAAQS, available at http://www.epa.gov/ttn/ecas/regdata/RIAs/finalria.pdf.

As noted in previous NAAQS analyses, the EPA continues to explore other sources of information to inform the estimates of extrapolated costs. For this RIA we examined the full set of known controls, examined evidence that suggests that over time new information and data emerges that shifts emissions reductions from the unknown to the known category, as well as explored whether NOx offset prices can serve as reasonable proxies for the costs of emissions reductions not identified by current tools.

Based upon deliberations informing this discussion, the EPA Council's advice, and the requirements of E.O. 12866 and OMB circular A-4, which provides guidance on the estimation of benefits and costs of regulations, in this RIA, we follow the Council recommendations by using an average cost per ton as a central estimate and conduct sensitivity analysis using alternative average costs to explore how sensitive total costs are to these assumptions. While the average cost methodology has limitations, we agree with the Council that the approach is both transparent and strikes a balance between the likelihood that some unidentified abatement would arise at lower segments of the identified cost curve while other sources of abatement may come at the higher cost margin.

While the known control analysis limited the application of controls with costs above \$14,000 per ton, we examined the full set of controls available for application in regions needing emissions reductions. The MAC curves from this analysis are presented above in Section 7.1. For NOx controls, a total of 1.22 million tons of reductions are available, and about 1.18 million of these tons are available for less than \$15,000 per ton. The known reductions available for less than \$15,000 per ton represent about 96 percent of the total known reductions in areas needing emissions reductions to meet an alternative standard level. In addition, the average cost per ton across all of these abatement opportunities is about \$3,400 per ton. As a result, we decided to use \$15,000 per ton NOx as the main estimate for the extrapolated cost analysis. This assumed cost is representative of higher cost controls available in the analysis. If, for example, the true costs of the unidentified controls are distributed at the upper end of the identified control costs depicted in Figure 7-1, \$15,000 per ton may under-estimate the average value of the unidentified abatement. Alternatively, the assumed value might overestimate the '"unobserved" abatement discussed in Section 7.2. The results shown in Table 7-4 (per ton cost of NOx reductions for five major NOx-reducing rules) and Table 7-7 (annualized per ton cost of NOs offsets in four regions) may suggest that the $15,000 per ton assumed average cost may be over-estimating the average value of the unidentified abatement.

Because of this uncertainty, we use alternative assumptions of the average cost in the Appendix, a first sensitivity analysis using an assumed cost of $10,000 per ton and a second sensitivity analysis using an assumed $20,000 per ton.[121] This range is inclusive of the annualized NOx offset prices observed in recent years in the areas likely to need unknown controls to achieve the proposed standard, and if anything, suggests the central estimate of $15,000/ton is conservative. EPA requests comments on the methods presented to estimate emission reductions needed beyond known controls including the parameter estimate of $15,000/ton.

Because cost changes due to technological change will be available on a national-level, it makes sense to use national-level average cost per ton in the primary analysis. However, as indicated by the variation in NOx offset prices across regions shown in Table 7-6, regional factors may play a significant role in the estimation of control costs. As a result, the EPA will continue to explore alternative methodologies and sources of regional information that may make the average cost methodology more regionally specific for the RIA for the final rule.

7.3.2 Unknown Compliance Cost Estimates

Table 7-8 presents the extrapolated control cost estimates for the East and West in 2025, except for California for the alternative standards using an assumed average cost of $15,000/ton. Values of $10,000/ton and $20,000/ton are used for the sensitivity analyses found in Appendix 7.2.

[121] As shown in Section 7.1, we also performed a similar analysis for VOC controls, which indicated that about 52 percent of VOC controls available in the analysis for less than $15,000 per ton, with an average of about $12,000 per ton. While a limited amount of extrapolated VOC emissions reductions were needed for the 60 ppb alternative level for the East (41,000 tons), we decided to use the same $15,000 per ton (with $10,000 and $20,000 per ton for the sensitivity analysis) for VOC controls for simplicity.

Table 7-8. Extrapolated Control Costs in 2025 by Alternative Standard for 2025 -- U.S., except California (millions of 2011$)

Alternative Level	Geographic Area	Extrapolated Cost
70 ppb	East	2,300
	West	-
	Total	**2,300**
65 ppb	East	11,000
	West	-
	Total	**11,000**
60 ppb	East	28,000
	West	5,200
	Total	**34,000**

[a] All values are rounded to two significant figures. Extrapolated costs are based on the average-cost methodology using a $15,000/ton assumed average cost.

Table 7-9 presents the extrapolated control cost estimates for post-2025 for California across the alternative standards using an assumed average cost of $15,000/ton.

Table 7-9. Extrapolated Control Costs in 2025 by Alternative Standard for Post-2025 -- California (millions of 2011$)

Alternative Level	Geographic Area	Extrapolated Costs
70 ppb	California	800
65 ppb	California	1,600
60 ppb	California	2,200

[a] All values are rounded to two significant figures. Extrapolated costs are based on the average-cost methodology using a $15,000/ton assumed average cost.

7.4 Total Compliance Cost Estimates

As discussed throughout this RIA, we present the primary costs and benefits estimates for 2025. We assume that potential nonattainment areas everywhere in the U.S., excluding California, will be designated such that they are required to reach attainment by 2025, and we developed our projected baselines for emissions, air quality, and populations for 2025.

In estimating the incremental costs and benefits of potential alternative standards, we recognize that there are several areas that are not required to meet the existing ozone standard by 2025. The Clean Air Act allows areas with more significant air quality problems to take additional time to reach the existing standard. Several areas in California are not required to meet the existing standard by 2025 and may not be required to meet a revised standard until sometime between 2032 and December 31, 2037.[122] We were not able to project emissions and air quality beyond 2025 for California, however, we adjusted baseline air quality to reflect mobile source emissions reductions for California that would occur between 2025 and 2030; these emissions reductions were the result of mobile source regulations expected to be fully implemented by 2030. While there is uncertainty about the precise timing of emissions reductions and related costs for California, we assume costs occur through the end of 2037 and beginning of 2038. In addition, we model benefits for California using projected population demographics for 2038.

Because of the different timing for incurring costs and accruing benefits and for ease of discussion throughout the analyses, we refer to the different time periods for potential attainment as 2025 and post-2025 to reflect that (1) we did not project emissions and air quality for any year other than 2025; (2) for California, emissions controls and associated costs are assumed to occur through the end of 2037 and beginning of 2038; and (3) for California benefits are modeled using population demographics in 2038. It is not straightforward to discount the post-2025 results for California to compare with or add to the 2025 results for the rest of the U.S. While we estimate benefits using 2038 information, we do not have good information on precisely when the costs of controls will be incurred. Because of these differences in timing related to California attaining a revised standard, the separate costs and benefits estimates for post-2025 should not be added to the primary estimates for 2025.

Tables 7-9 and 7-10 present summaries of the total national annual costs (known and extrapolated) of attaining the alternative standards of 70, 65, and 60 ppb. 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, where available, to the known control estimates at a 7 percent discount rate where the costs could not be determined for the 3 percent rate; we added these to the extrapolated costs at a 7 percent discount rate. Table 7-10

[122] The EPA will likely finalize designations for a revised ozone NAAQS in late 2017. Depending on the precise timing of the effective date of those designations, nonattainment areas classified as Severe 15 will likely have to attain sometime between late 2032 and early 2033 and nonattainment areas classified as Extreme will likely have to attain by December 31, 203.

presents the total national annual costs by alternative standard for 2025 for all of the U.S., except California. Table 7-11 presents the total national annual costs by alternative standard for post-2025 for California.

Table 7-10. Summary of Total Control Costs (Known and Extrapolated) by Alternative Level for 2025 - U.S., except California (millions of 2011$, 7% Discount Rate)[a]

Alternative Level	Geographic Area	Total Control Costs (Known and Extrapolated)
70 ppb	East	3,900
	West	-
	Total	**$3,900**
65 ppb	East	15,000
	West	400
	Total	**$15,000**
60 ppb	East	33,000
	West	5,800
	Total	**$39,000**

[a] All values are rounded to two significant figures. Extrapolated costs are based on the fixed-cost methodology.

Table 7-11. Summary of Total Control Costs (Known and Extrapolated) by Alternative Level for post-2025 - California (millions of 2011$, 7% Discount Rate)[a]

Alternative Level	Geographic Area	Total Control Costs (Known and Extrapolated)
70 ppb	California	800
65 ppb	California	1,600
60 ppb	California	2,200

[a] All values are rounded to two significant figures. Extrapolated costs are based on the fixed-cost methodology.

7.5 Updated Methodology Presented in this RIA

The cost analysis presented in this chapter incorporates an array of methodological and technical updates that the EPA has adopted since the previous review of the ozone standards in 2008 and proposed reconsideration in 2010. The updates to models, methods, and data are too numerous to be able to quantitatively estimate the impact of any of the updates individually. Therefore, we present the major updates below qualitatively. Many of these changes reflect updates to inputs to the cost analysis, but are discussed here for completeness. Below we note the aspects of this analysis that differ from the 2008 RIA as well as the 2010 reconsideration RIA

(U.S. EPA, 2010). A few overarching changes that are worth mentioning: the incremental costs and benefits for this analysis are measured from a baseline of the current 75 ppb ozone standard; in the previous analysis the baseline was the 84 ppb ozone standard. Also, the currency year was updated from 2006$ to 2011$.

Emissions and Air Quality Updates

The base year emissions for this analysis are 2011, and the future analysis year is 2025 (previously the base year was 2002 and the future analysis year was 2020). Key changes in the emission estimates include: increased accuracy of stationary source emissions estimates, updates in models and Annual Energy Outlook (AEO) projections for mobile sources and EGUs, inclusion of oil and gas sector emissions and numerous updates to the nonpoint emissions. In addition, from 2002 to 2011 there have been a number of changes in the energy system, cleaner mobile sources, and economic changes that have resulted in reduced emissions since the last ozone analysis. There are additional federal control programs included in the emissions projections to 2025, including Tier 3, MATS, and the Clean Power Plan.

Air quality monitor design values were updated to reflect 2009 through 2013 air quality in contrast the 2008 and 2010 analyses which used 2000 through 2004 air quality. As shown in Chapter 2 Figure 2-2, ozone design values are generally decreasing over time. For example, the figure shows that for the period of years from 2000 through 2004 the 90th percentile concentrations ranged from 85.9 to 102.8 ppb; the 75th percentile values for the same period ranged from 78.8 to 87.3 ppb. In contrast, the design values for the period of 2009 through 2013 the 90th percentile concentrations ranged from 77 to 86.2 ppb and the 75th percentile concentrations ranged from 70.8 to 81 ppb. These design values are the basis for future year air quality projections. A quick comparison across analyses reveals that in the 2008 analysis 89 counties were projected to exceed the higher end of the proposed range of 70ppb in the future year (2020), while only 9 counties are projected to exceed 70 ppb in this analysis for 2025. The same is true for the lower end of the proposed range, in 2008 231 counties were projected to exceed 65 ppb and that number has decreased to 68 counties in this analysis.

As emissions decrease in the base year and air quality design values decrease, the result is a smaller incremental change in air quality needed to meet the revised standards as compared to the 2008 and 2010 analyses. The smaller increment needed to achieve attainment also means that fewer controls are needed across a reduced number of geographic areas

Control Strategy Updates

Many improvements were made in the non-EGU point control measures used in this analysis that make the data more accurate and defensible. These changes include: removal of incorrect links between control measures and SCCs; updates to cost equations where more recent data was available to improve their accuracy; inclusion of information that has recently become available concerning known and emerging technologies for reducing NOx emissions; and revising costs and control efficiencies from control measures in the dataset based on recently obtained information from industry and multi-jurisdictional organizations (e.g., Ozone Transport Commission and Lake Michigan Air Directors Consortium). In addition, a different mix of known control measures across EGU, mobile, nonpoint and non-EGU point were applied due to the recently promulgated rules mentioned above.

Costing methodology updates have occurred since previous ozone analyses, the 'hybrid' approach was not utilized to estimate costs of emission reductions needed beyond known controls. Holding cost methodology constant (average/fixed cost approach), fewer emissions reductions were needed beyond known controls in this analysis due to the increased application of known control measures mentioned above.

The above mentioned technical and methodological changes resulted in the lower cost estimates in this analysis. The most influential factors on the cost analysis were the lower number of exceeding counties and increased data accuracy of the known control measures. The effect of fewer exceeding counties is that fewer emissions reductions are needed, and therefore the costs are lower. The effect of additional and lower-cost known control measures being applied is a lowering of the emissions reductions needed beyond known controls, which are typically the higher cost emissions reductions.

7.6 Economic Impacts

7.6.1 Introduction

This section addresses the potential economic impacts of the illustrative control strategies for the potential alternative ozone standards. The control costs are uncertain for several reasons. The controls that the states ultimately choose to implement will likely differ from the illustrative control strategies for which costs are estimated in earlier sections of this chapter. The flexibility afforded to states by the Clean Air Act also allows them to adopt programs that include design elements that may mitigate or promote particular economic impacts based on their individual priorities. The cost estimates become more uncertain because of the length of time before they will be implemented. By the 2025 and post-2025 time frames, changes in technology, changes in implemented regulations, and changes in relative prices will all add to the uncertainty in the cost analysis. Finally, the portion of costs that is extrapolated is not allocated to particular sectors.

Economic impacts focus on the behavioral response to the costs imposed by a policy being analyzed. The responses typically analyzed are market changes in prices, quantities produced and purchased, changes in international trade, changes in profitability, facility closures, and employment. Often, these behavioral changes are used to estimate social costs if there is indication that the social costs differ from the estimate of control costs because behavioral change results in other ways of meeting the requirements (e.g., facilities choosing to reduce emissions by producing less rather than adding pollution control devices).

The potential alternative ozone standards are anticipated to impact multiple markets in many times and places. Computable General Equilibrium (CGE) models are designed to address such problems. To support the Final Ozone NAAQS of March 2008 (Final Ozone NAAQS Regulatory Impact Analysis), among other rulemakings, the EPA used the Economic Model for Policy Analysis (EMPAX) to estimate the market impacts of the portion of the cost that was associated with the application of known controls (excluding the extrapolated costs). EMPAX is a dynamic computable general equilibrium (CGE) model that forecasts a new equilibrium for the entire economy after a policy intervention. While the external Council on Clean Air Compliance Analysis (Council) peer review of The Benefits and Costs of the Clean Air Act from 1990 to 2020(Hammitt 2010) stated that inclusion of benefits in an economy-wide model, specifically adapted for use in that study, "represent[ed] a significant step forward in benefit-cost analysis," EPA recognizes that serious technical challenges remain when attempting to evaluate the benefits and costs of potential regulatory actions using economy-wide models. Consistent with the Council's advice regarding the importance of including benefit-side effects demonstrated by the Benefits and Costs of the Clean Air Act from 1990 to 2020, and the lack of available multi-year air quality projections needed to include these benefit-side effects, EPA has not conducted CGE modeling for this analysis.

However, the EPA recognizes that serious technical challenges remain when attempting to evaluate the impacts of potential regulatory actions using economy-wide models. The EPA is therefore establishing a new Science Advisory Board (SAB) panel on economy-wide modeling to consider the technical merits and challenges of using this analytical tool to evaluate costs, benefits, and economic impacts in regulatory development. The EPA will use the recommendations and advice of this SAB panel as an input into its process for improving benefit-cost and economic impact analyses that are used to inform decision-making at the Agency. The panel will also be asked to identify potential paths forward for improvements that could address the challenges posed when economy-wide models are used to evaluate the effects of regulations.

The advice from the SAB panel formed specifically to address the subject of economy-wide modeling will not be available in time for this analysis. Given the ongoing SAB panel on economy-wide modeling, and the uncertain nature of costs, this section proceeds with a qualitative discussion of market impacts.

7.6.2 Summary of Market Impacts

Consider an added cost to produce a good associated with the pollution control required to reach the alternative ozone standards. Such a good is either one developed for the consumer (called a consumption good), or one used in the production of other goods for consumption (called an intermediate good). Some goods are both consumption and intermediate goods. First, consider the direct impact on the market facing the increased cost. In this case for the market facing the increased cost, the price will go up and the amount sold will go down. The magnitude of these shifts depends on a number of factors. The greater the unit cost increase relative to the price of the good the greater will be the changes. The more responsive a consumer is to a change in the price of a consumption good or the more responsive a purchase of an intermediate good is the greater will be the changes. For the alternative ozone standards, many goods will have direct changes in costs of production. This makes the assumption of isolated markets too simple. With multiple intermediate goods affected, then the intermediate goods and consumption goods they are used to produce are affected. As fewer intermediate goods and consumption goods are purchased at a higher price, other intermediate goods and consumption goods that serve as substitutes become more attractive and more are sold at a higher price. All of these market changes lead to changes in income, which can lead to changes in purchases of consumption goods. Quantities of intermediate goods used to reduce emissions would also change. Considering all of these changes, it is not possible to qualitatively conclude the direction of price and quantity changes for any single market. Any conclusions about changes in international trade, profits, closures, or social cost is impossible in a qualitative analysis.

7.7 Uncertainties and Limitations

The EPA acknowledges several important limitations of this analysis, which include the following:

Boundary of the cost analysis: In this engineering cost analysis we include only the impacts to the regulated industry, such as the costs for purchase, installation, operation, and maintenance of control equipment over the lifetime of the equipment. As mentioned above, recordkeeping, reporting, testing and monitoring costs are not included. In some cases, costs are estimated for changes to a process such as switching from one fuel to another less polluting fuel. Additional profit or income may be generated by industries supplying the regulated industry, especially for control equipment manufacturers, distributors, or service providers. These types of secondary impacts are not included in this engineering cost analysis.

Cost and effectiveness of control measures: Our application of control measures reflect average retrofit factors and equipment lives that are applied on a national scale. We do not account for regional or local variation in capital and annual cost items such as energy, labor, materials, and others. Our estimates of control measure costs may over- or under-estimate the costs depending on how the difficulty of actual retrofitting and equipment life compares with our control assumptions. In addition, our estimates of control efficiencies for the known controls assume that the control devices are properly installed and maintained. There is also variability in scale of application that is difficult to reflect for small area sources of emissions.

Discount rate: Because we obtain control cost data from many sources, we are not always able to obtain consistent data across original data sources. If disaggregated control cost data are unavailable (i.e., where capital, equipment life value, and operation and maintenance [O&M] costs are not separated out), the EPA typically assumes that the estimated control costs are annualized using a 7 percent discount rate. When disaggregated control cost data are available (i.e., where capital, equipment life value, and O&M costs are explicit), we can recalculate costs using a 3 percent discount rate. In general, we have some disaggregated data available for non-EGU point source controls, and we do not have any disaggregated control cost data for area source controls. In addition, these discount rates are consistent with OMB guidance, but the actual real discount rates may vary regionally or locally.

Known control costs: We estimate that there is an accuracy range of +/- 30 percent for non-EGU point source control costs. This level of accuracy is described in the EPA Air Pollution Control Cost Manual, which is a basis for the estimation of non-EGU control cost estimates included in this RIA. This level of accuracy is consistent with either the budget or bid/tender-level of cost estimation as defined by the AACE International.[123] The accuracy for nonpoint control costs estimates has not been determined, but it is likely no more accurate than those for non-EGU point source control costs.

Differences between ex ante and ex post compliance cost estimates: In comparing regulatory cost estimates before and after regulation, *ex ante* cost estimate predictions often overestimate or underestimate costs. Harrington *et al.* (2000) surveyed the predicted and actual costs of 28 federal and state rules, including 21 issued by the U.S. Environmental Protection Agency and the Occupational Safety and Health Administration (OSHA). In 14 of the 28 rules, predicted total costs were overestimated, while analysts underestimated costs in three of the remaining rules. In

[123] AACE International. Recommended Practice No. 18R-97. Cost Estimate Classification System – As Applied in Engineering, Procurement, and Construction For the Process Industries. Revised on November 29, 2011. Available at http://www.aacei.org/non/rps/18R-97.pdf.

EPA rules where per-unit costs were specifically evaluated, costs of regulations were overestimated in five cases, underestimated in four cases, and accurately estimated in four cases (Harrington et al. 2000). The collection of literature regarding the accuracy of cost estimates seems to reflect these splits. The "Retrospective Study of the Costs of EPA Regulations" found that several of the case studies[124] suggested that cost estimates were over-estimated. However, the EPA stated in the report that the small number of regulatory actions covered and data and analytical challenges associated with the case studies limited the certainty of this conclusion.

Costs of unknown controls (extrapolated costs): In addition to the application of known controls, the EPA assumes the application of unidentified future controls that make possible the additional emissions reductions needed beyond known controls for attainment in the projection year for this analysis.

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[124] The four case studies in the 2014 *Retrospective Study of the Costs of EPA Regulations* examine five EPA regulations: the 2001/2004 National Emission Standards for Hazardous Air Pollutants and Effluent Limitations Guidelines, Pretreatment Standards, and New Source Performance Standards on the Pulp and Paper Industry; Critical Use Exemptions for Use of Methyl Bromide for Growing Open Field Fresh Strawberries in California for the 2004-2008 Seasons; the 2001 National Primary Drinking Water Regulations for Arsenic; and the 1998 Locomotive Emission Standards.

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APPENDIX 7A: ENGINEERING COST ANALYSIS

Overview

Chapter 7 describes the engineering cost analysis approach that EPA used in applying to demonstrate attainment of alternative ozone standard levels of 70, 65, and 60 ppb. This Appendix contains more detailed information about the control costs of the known control strategy analyses by control measure as well as sensitivity analyses for the fixed cost approach used to estimate costs for the unknown emissions controls.

7A.1 Cost of Known Controls in Alternative Standards Analyses

This section presents costs of known controls for the alternative standards analyses. Costs are in terms of 2011 dollars and include values for all portions of the U.S. that were part of the analyses. However, because all available known controls for California were applied as part of the baseline analysis, no known controls were available for the alternative standards analyses in California so these costs do not include any known control costs for California. Costs for the alternative standard analyses are incremental to the baseline attainment demonstration. Tables 7A-1 and 7A-2 present the Costs for known controls by measure for the 70ppb alternative standard analysis for NOx and VOC respectively. Tables 7A-3 and 7A-4 present the Costs for known controls by measure for the 65ppb alternative standard analysis, and Tables 7A-5 and 7A-6 present the costs for known controls by measure for the 60 ppb alternative standard analysis.

Table 7A-1. Costs for Known NOx Controls in the 70 ppb Analysis (millions of 2011$)[a]

NO_x Control Measure	Cost
Adjust Air to Fuel Ratio and Ignition Retard - Gas Fired IC Engines	0.35
Biosolid Injection Technology - Cement Kilns	0.42
Ignition Retard - IC Engines	0.037
Low Emission Combustion - Gas Fired Lean Burn IC Engines	1
Low NOx Burner - Commercial/Institutional Boilers & IC Engines	6.6
Low NOx Burner - Industr/Commercial/Institutional (ICI) Boilers	1.4
Low NOx Burner - Industrial Combustion	0.37
Low NOx Burner - Lime Kilns	0.28
Low NOx Burner - Natural Gas-Fired Turbines	8.4
Low NOx Burner - Residential Furnaces	5.2
Low NOx Burner - Residential Water Heaters & Space Heaters	13
Low NOx Burner and Flue Gas Recirculation - Coke Oven/Blast Furnace	0.13
Low NOx Burner and Flue Gas Recirculation - Iron & Steel Mills - Reheating	0.026
Low NOx Burner and SCR - Coal-Fired ICI Boilers	4.1
Low NOx Burner and SCR - Industr/Commercial/Institutional Boilers	27

NO_x Control Measure	Cost
Natural Gas Reburn - Natural Gas-Fired EGU Boilers	0
Nonroad Diesel Retrofits & Engine Rebuilds - e.g., Construction Equipment	29
Non-Selective Catalytic Reduction (NSCR) - 4 Cycle Rich Burn IC Engines	10
OXY-Firing - Glass Manufacturing	20
Selective Catalytic Reduction (SCR) - Cement Kilns	16
Selective Catalytic Reduction (SCR) - Fluid Catalytic Cracking Units	4
Selective Catalytic Reduction (SCR) - IC Engines, Diesel	5.5
Selective Catalytic Reduction (SCR) - ICI Boilers	8.5
Selective Catalytic Reduction (SCR) - Industrial Incinerators	4
Selective Catalytic Reduction (SCR) - Petroleum Refinery Gas-Fired Process Heaters	3.1
Selective Catalytic Reduction (SCR) - Sludge Incineration	1.4
Selective Non-Catalytic Reduction (SNCR) - Comm./Inst. Incinerators	0.029
Selective Non-Catalytic Reduction (SNCR) - Industrial Incinerators	0.24
Selective Non-Catalytic Reduction (SNCR) - Utility Boilers	0.43

[a] All values are rounded to two significant figures.

Table 7A-2. Costs for Known VOC Controls in the 70 ppb Analysis (millions of 2011$) [a]

VOC Control Measure	Cost
Control of Fugitive Releases - Oil & Natural Gas Production	0.044
Control Technology Guidelines - Wood Furniture Surface Coating	1.9
Flare - Petroleum Flare	0.36
Gas Recovery - Municipal Solid Waste Landfill	0.27
Improved Work Practices, Material Substitution, Add-On Controls - Printing	0.4
Incineration - Other	0.84
Incineration - Surface Coating	200
Low-VOC Coatings and Add-On Controls - Surface Coating	0.04
LPV Relief Valve - Underground Tanks	7.1
Permanent Total Enclosure (PTE) - Surface Coating	3.5
RACT - Graphic Arts	20
Reduced Solvent Utilization - Surface Coating	2.7
Reformulation - Architectural Coatings	140
Reformulation - Industrial Adhesives	6.4
Reformulation-Process Modification - Automobile Refinishing	38
Reformulation-Process Modification - Cold Cleaning	3.3
Reformulation-Process Modification - Cutback Asphalt	0.02
Reformulation-Process Modification - Surface Coating	13
Solvent Recovery System - Printing/Publishing	0.038
Wastewater Treatment Controls- POTWs	0.73

[a] All values are rounded to two significant figures.

Table 7A-3. Costs for Known NO_x Controls in the 65 ppb Analysis (millions of 2011$) [a]

NOx Control Measure	Cost
Adjust Air to Fuel Ratio and Ignition Retard - Gas Fired IC Engines	12
Biosolid Injection Technology - Cement Kilns	2.7
Ignition Retard - IC Engines	0.96
Low Emission Combustion - Gas Fired Lean Burn IC Engines	110
Low NOx Burner - Coal Cleaning	0.77
Low NOx Burner - Commercial/Institutional Boilers & IC Engines	40
Low NOx Burner - Industr/Commercial/Institutional (ICI) Boilers	41
Low NOx Burner - Industrial Combustion	3.3
Low NOx Burner - Lime Kilns	4.8
Low NOx Burner - Miscellaneous Sources	0.058
Low NOx Burner - Natural Gas-Fired Turbines	44
Low NOx Burner - Residential Furnaces	16
Low NOx Burner - Residential Water Heaters & Space Heaters	110
Low NOx Burner - Steel Foundry Furnaces	0.27
Low NOx Burner - Surface Coating Ovens	0.092
Low NOx Burner and Flue Gas Recirculation - Coke Oven/Blast Furnace	2.2
Low NOx Burner and Flue Gas Recirculation - Iron & Steel Mills - Reheating	0.55
Low NOx Burner and Over Fire Air - Utility Boilers	1.6
Low NOx Burner and SCR - Coal-Fired ICI Boilers	87
Low NOx Burner and SCR - Industr/Commercial/Institutional Boilers	210
Low NOx Burner and SNCR - Industr/Commercial/Institutional Boilers	3.6
Natural Gas Reburn - Natural Gas-Fired EGU Boilers	1.2
Nonroad Diesel Retrofits & Engine Rebuilds - e.g., Construction Equipment	58
Non-Selective Catalytic Reduction (NSCR) - 4 Cycle Rich Burn IC Engines	170
Non-Selective Catalytic Reduction (NSCR) - Nitric Acid Mfg	1.4
OXY-Firing - Glass Manufacturing	140
SCR and Flue Gas Recirculation - Fluid Catalytic Cracking Units	0.91
SCR and Flue Gas Recirculation - ICI Boilers	4.7
SCR and Flue Gas Recirculation - Process Heaters	2.9
Selective Catalytic Reduction (SCR) - Ammonia Mfg	15
Selective Catalytic Reduction (SCR) - Cement Kilns	230
Selective Catalytic Reduction (SCR) - Fluid Catalytic Cracking Units	19
Selective Catalytic Reduction (SCR) - IC Engines, Diesel	24
Selective Catalytic Reduction (SCR) - ICI Boilers	76
Selective Catalytic Reduction (SCR) - Industrial Combustion	24
Selective Catalytic Reduction (SCR) - Industrial Incinerators	5.6
Selective Catalytic Reduction (SCR) - Iron Ore Processing	1.3
Selective Catalytic Reduction (SCR) - Petroleum Refinery Gas-Fired Process Heaters	61
Selective Catalytic Reduction (SCR) - Process Heaters	0.26
Selective Catalytic Reduction (SCR) - Sludge Incineration	39
Selective Catalytic Reduction (SCR) - Space Heaters	1.3
Selective Catalytic Reduction (SCR) - Utility Boilers	1,700
Selective Non-Catalytic Reduction (SNCR) - Coke Mfg	6.4
Selective Non-Catalytic Reduction (SNCR) - Comm./Inst. Incinerators	2.9
Selective Non-Catalytic Reduction (SNCR) - ICI Boilers	0.75
Selective Non-Catalytic Reduction (SNCR) - Industrial Combustion	0.065
Selective Non-Catalytic Reduction (SNCR) - Industrial Incinerators	2.8

NOx Control Measure	Cost
Selective Non-Catalytic Reduction (SNCR) - Miscellaneous	0.25
Selective Non-Catalytic Reduction (SNCR) - Municipal Waste Combustors	2.5
Selective Non-Catalytic Reduction (SNCR) - Utility Boilers	0.46
Ultra-Low NOx Burner - Process Heaters	1.8

[a] All values are rounded to two significant figures.

Table 7A-4. Costs for Known VOC Controls in the 65 ppb Analysis (millions of 2011$) [a]

VOC Control Measure	Cost
Control of Fugitive Releases - Oil & Natural Gas Production	0.083
Control Technology Guidelines - Wood Furniture Surface Coating	3
Flare - Petroleum Flare	0.36
Gas Recovery - Municipal Solid Waste Landfill	0.37
Improved Work Practices, Material Substitution, Add-On Controls - Printing	1.4
Incineration - Other	0.91
Incineration - Surface Coating	370
Low VOC Adhesives and Improved Application Methods - Industrial Adhesives	0.06
Low-VOC Coatings and Add-On Controls - Surface Coating	0.57
LPV Relief Valve - Underground Tanks	13
Permanent Total Enclosure (PTE) - Surface Coating	17
Petroleum and Solvent Evaporation - Surface Coating Operations	0.037
RACT - Graphic Arts	38
Reduced Solvent Utilization - Surface Coating	3.1
Reformulation - Architectural Coatings	300
Reformulation - Industrial Adhesives	6.1
Reformulation-Process Modification - Automobile Refinishing	62
Reformulation-Process Modification - Cutback Asphalt	0.075
Reformulation-Process Modification - Surface Coating	24
Solvent Recovery System - Printing/Publishing	1.3
Solvent Substitution and Improved Application Methods - Fiberglass Boat Mfg	0.059
Wastewater Treatment Controls- POTWs	0.82

[a] All values are rounded to two significant figures.

Table 7A-5. Costs for Known NOx Controls in the 60 ppb Analysis (millions of 2011$) [a]

NO_x Control Measure	Cost
Adjust Air to Fuel Ratio and Ignition Retard - Gas Fired IC Engines	12
Biosolid Injection Technology - Cement Kilns	2.7
Ignition Retard - IC Engines	1
Low Emission Combustion - Gas Fired Lean Burn IC Engines	120
Low NOx Burner - Coal Cleaning	0.77
Low NOx Burner - Commercial/Institutional Boilers & IC Engines	40
Low NOx Burner - Industr/Commercial/Institutional (ICI) Boilers	42
Low NOx Burner - Industrial Combustion	3.3
Low NOx Burner - Lime Kilns	4.8
Low NOx Burner - Miscellaneous Sources	0.077
Low NOx Burner - Natural Gas-Fired Turbines	47
Low NOx Burner - Residential Furnaces	17
Low NOx Burner - Residential Water Heaters & Space Heaters	120
Low NOx Burner - Steel Foundry Furnaces	0.27
Low NOx Burner - Surface Coating Ovens	0.092
Low NOx Burner and Flue Gas Recirculation - Coke Oven/Blast Furnace	2.2
Low NOx Burner and Flue Gas Recirculation - Iron & Steel Mills - Reheating	0.55
Low NOx Burner and Over Fire Air - Utility Boilers	1.6
Low NOx Burner and SCR - Coal-Fired ICI Boilers	87
Low NOx Burner and SCR - Industr/Commercial/Institutional Boilers	210
Low NOx Burner and SNCR - Industr/Commercial/Institutional Boilers	3.6
Natural Gas Reburn - Natural Gas-Fired EGU Boilers	1.3
Nonroad Diesel Retrofits & Engine Rebuilds - e.g., Construction Equipment	58
Non-Selective Catalytic Reduction (NSCR) - 4 Cycle Rich Burn IC Engines	170
Non-Selective Catalytic Reduction (NSCR) - Nitric Acid Mfg	1.4
OXY-Firing - Glass Manufacturing	140
SCR and Flue Gas Recirculation - Fluid Catalytic Cracking Units	0.91
SCR and Flue Gas Recirculation - ICI Boilers	4.7
SCR and Flue Gas Recirculation - Process Heaters	2.9
Selective Catalytic Reduction (SCR) - Ammonia Mfg	15
Selective Catalytic Reduction (SCR) - Cement Kilns	230
Selective Catalytic Reduction (SCR) - Fluid Catalytic Cracking Units	19
Selective Catalytic Reduction (SCR) - IC Engines, Diesel	26
Selective Catalytic Reduction (SCR) - ICI Boilers	76
Selective Catalytic Reduction (SCR) - Industrial Combustion	24
Selective Catalytic Reduction (SCR) - Industrial Incinerators	5.6
Selective Catalytic Reduction (SCR) - Iron Ore Processing	1.3
Selective Catalytic Reduction (SCR) - Petroleum Refinery Gas-Fired Process Heaters	61
Selective Catalytic Reduction (SCR) - Process Heaters	0.26
Selective Catalytic Reduction (SCR) - Sludge Incineration	39
Selective Catalytic Reduction (SCR) - Space Heaters	1.5
Selective Catalytic Reduction (SCR) - Utility Boilers	1,900
Selective Non-Catalytic Reduction (SNCR) - Coke Mfg	6.4
Selective Non-Catalytic Reduction (SNCR) - Comm./Inst. Incinerators	2.9
Selective Non-Catalytic Reduction (SNCR) - ICI Boilers	0.75
Selective Non-Catalytic Reduction (SNCR) - Industrial Combustion	0.087
Selective Non-Catalytic Reduction (SNCR) - Industrial Incinerators	2.8

NO_x Control Measure	Cost
Selective Non-Catalytic Reduction (SNCR) - Miscellaneous	0.25
Selective Non-Catalytic Reduction (SNCR) - Municipal Waste Combustors	2.5
Selective Non-Catalytic Reduction (SNCR) - Utility Boilers	0.46
Ultra-Low NOx Burner - Process Heaters	1.8

[a] All values are rounded to two significant figures.

Table 7A-6. Costs for Known VOC Controls in the 60 ppb Analysis (millions of 2011$) [a]

VOC Control Measure	Cost
Control of Fugitive Releases - Oil & Natural Gas Production	0.089
Control Technology Guidelines - Wood Furniture Surface Coating	3.3
Flare - Petroleum Flare	0.36
Gas Recovery - Municipal Solid Waste Landfill	0.41
Improved Work Practices, Material Substitution, Add-On Controls - Printing	1.4
Incineration - Other	0.91
Incineration - Surface Coating	370
Low VOC Adhesives and Improved Application Methods - Industrial Adhesives	0.064
Low-VOC Coatings and Add-On Controls - Surface Coating	0.97
LPV Relief Valve - Underground Tanks	13
Permanent Total Enclosure (PTE) - Surface Coating	28
Petroleum and Solvent Evaporation - Surface Coating Operations	0.071
RACT - Graphic Arts	40
Reduced Solvent Utilization - Surface Coating	3.2
Reformulation - Architectural Coatings	310
Reformulation - Industrial Adhesives	6.1
Reformulation-Process Modification - Automobile Refinishing	66
Reformulation-Process Modification - Cutback Asphalt	0.087
Reformulation-Process Modification - Surface Coating	25
Solvent Recovery System - Printing/Publishing	1.3
Solvent Substitution and Improved Application Methods - Fiberglass Boat Mfg	0.059
Wastewater Treatment Controls- POTWs	0.82

[a] All values are rounded to two significant figures.

7A.2 Alternative Estimates of Costs Associated with Emissions Reductions from Unknown Controls

This section presents alternative estimates of the extrapolated control costs using alternative average cost per ton of emission reductions from unknown controls. The alternative values used are $10,000/ton and $20,000, as well as the $15,000/ton value used as the primary estimate in the RIA. Table 7A-7 presents the alternative estimates for the East and West Regions in 2025, without California, while Table 7A-8 presents the estimates for post-2025 California.

Table 7A-7. Extrapolated Control Costs in 2025 by Alternative Standard for 2025 U.S., except California, using Alternative Average Cost Assumptions (millions of 2011$)

Alternative Level	Geographic Area	Extrapolated Cost		
		$10,000/ton	$15,000/ton	$20,000/ton
70 ppb	East	1,500	2,300	3,100
	West	-	-	-
65 ppb	East	7,500	11,000	15,000
	West		-	
60 ppb	East	19,000	28,000	38,000
	West	3,500	5,200	7,000

[a] All values are rounded to two significant figures.

Table 7A-8. Extrapolated Control Costs in 2025 by Alternative Standard for Post-2025 California, using Alternative Average Cost Assumptions (millions of 2011$)

Alternative Level	Geographic Area	Extrapolated Cost		
		$10,000/ton	$15,000/ton	$20,000/ton
70 ppb	California	530	800	1,100
65 ppb	California	1,000	1,600	2,100
60 ppb	California	1,400	2,200	2,900

[a] All values are rounded to two significant figures.

Note, by definition, the lower per ton estimate is 50% less than the upper end estimate, and the range between the extrapolated cost bounds becomes larger as the alternative levels become more stringent and rely more heavily on unknown controls.

Tables 7A-9 presents the estimates of the total control costs for 2025 East and West, without California, when using the alternative per ton cost assumptions for emissions reductions from unknown controls. Tables 7A-10 presents the estimates of the total control costs for post-2025 California when using the alternative per ton cost assumptions for emissions reductions from unknown controls.

Table 7A-9. Summary of Total Control Costs (Known and Extrapolated) by Alternative Level for 2025 - U.S. using Alternative Cost Assumption for Extrapolated Costs, except California (millions of 2011$)[a]

Alternative Level	Geographic Area	Total Control Costs (Known and Extrapolated)		
		Extrapolated Cost = $10,000/ton	Extrapolated Cost = $15,000/ton	Extrapolated Cost = $20,000/ton
70 ppb	East	3,100	3,900	4,700
	West	-	-	-
65 ppb	East	11,000	15,000	19,000

Alternative Level	Geographic Area	Total Control Costs (Known and Extrapolated)		
		Extrapolated Cost = $10,000/ton	Extrapolated Cost = $15,000/ton	Extrapolated Cost = $20,000/ton
	West	400	400	400
60 ppb	East	23,000	33,000	42,000
	West	4,100	5,800	7,600

[a] All values are rounded to two significant figures.

Table 7A-10. Summary of Total Control Costs (Known and Extrapolated) by Alternative Level for Post-2025 California - U.S. using Alternative Cost Assumption for Extrapolated Costs (millions of 2011$)[a]

Alternative Level	Geographic Area	Total Control Cost		
		Extrapolated Cost = $10,000/ton	Extrapolated Cost = $15,000/ton	Extrapolated Cost = $20,000/ton
70 ppb	California	530	800	1,100
65 ppb	California	1,000	1,600	2,100
60 ppb	California	1,400	2,200	2,900

[a] All values are rounded to two significant figures.

CHAPTER 8: COMPARISON OF COSTS AND BENEFITS

Overview

The EPA has performed an illustrative analysis to estimate the costs and human health benefits of nationally attaining alternative ozone standards. The EPA Administrator is proposing to revise the level of the primary ozone standard to within a range of 65 to 70 ppb and is soliciting comment on alternative standard levels below 65 ppb, as low as 60 ppb. Per Executive Order 12866 and the guidelines of OMB Circular A-4, this Regulatory Impact Analysis (RIA) presents the analyses of the following alternative standard levels -- 60 ppb, 65 ppb, and 70 ppb. This chapter summarizes these results and discusses the implications of the analysis. The cost and benefit estimates below are calculated incremental to a 2025 baseline assuming attainment of the existing ozone standard of 75 ppb and incorporating air quality improvements achieved through the projected implementation of existing regulations.

8.1 Results

In this RIA we present the primary costs and benefits estimates for full attainment in 2025. For analytical purposes, we assume that almost all areas of the country will meet each alternative standard level in 2025 through the adoption of technologies at least as effective as the control strategies used in this illustration. It is expected that some costs and benefits will begin occurring earlier, as states begin implementing control measures to attain earlier or to show progress towards attainment. For California, we provide estimates of the costs and benefits of attaining the standard in a post-2025 time frame.

In estimating the incremental costs and benefits of potential alternative standard levels, we recognize that there are several areas that are not required to meet the existing ozone standard by 2025. The Clean Air Act allows areas with more significant air quality problems to take additional time to reach the existing standard. Several areas in California are not required to meet the existing standard by 2025, and depending on how areas are ultimately designated for a revised standard, many areas may not be required to meet a revised standard until sometime between 2032 and December 31, 2037. We were not able to project emissions and air quality beyond 2025 for California; however, we adjusted baseline air quality to reflect mobile source emissions reductions for California that would occur between 2025 and 2030; these emissions reductions were the result of mobile source regulations expected to be fully implemented by 2030. While there is uncertainty about the precise timing of emissions reductions and related costs for California, we assume costs occur through the end of 2037 and beginning of 2038. In addition, we model benefits for California using projected population demographics for 2038.

Because of the different timing for incurring costs and accruing benefits and for ease of discussion throughout the analyses, we refer to the different time periods for potential attainment as 2025 and post-2025 to reflect that (1) we did not project emissions and air quality for any year other than 2025; (2) for California, emissions controls and associated costs are assumed to occur through the end of 2037 and beginning of 2038; and (3) for California benefits are modeled using population demographics in 2038. It is not straightforward to discount the post-2025 results for California to compare with or add to the 2025 results for the rest of the U.S. While we estimate benefits using 2038 information, we do not have good information on precisely when the costs of controls will be incurred. Because of these differences in timing related to California attaining a revised standard, the separate costs and benefits estimates for post-2025 should not be added to the primary estimates for 2025.

By the 2030s, various mobile source rules, such as the onroad and nonroad diesel rules are expected to be fully implemented. Because California will likely not have all of its areas in attainment with a revised standard until sometime after its attainment date for the existing standard, it is important to reflect the impact these mobile source rules might have on the emissions that affect ozone nonattainment. To reflect the emissions reductions that are expected from these rules, we subtract those from the estimates of the emissions reductions that might be needed for California to fully attain in 2025, making our analysis more consistent with full attainment later than 2025. The EPA did the analysis this way to be consistent with the requirements in the Clean Air Act and because forcing full attainment in California in an earlier year would likely lead to overstating costs due to (1) benefits those areas might enjoy from existing federal or state programs implemented between 2025 and the future potential attainment year, (2) the likelihood that energy efficiency and cleaner technologies will be further implemented, and/or (3) the potential decline in costs of existing technologies due to economies of scale or improvements in the efficiency of installing and operating controls (‘learning by doing’).

Tables 8-1 and 8-2 summarize the costs and benefits of the three potential alternative standard levels analyzed and shows the net benefits for each of the levels across a range of modeling assumptions related to the calculation of costs and benefits. Tables 8-3 and 8-4 provide information on the costs by geographic region for the U.S., except California in 2025 and on the costs for California for post-2025. Tables 8-5 and 8-6 provide a regional breakdown of benefits for 2025 and a regional breakdown of benefits for post-2025.

The estimates for benefits reflect the variability in the functions available for estimating the largest source of benefits – avoided premature mortality associated with simulated reductions in ozone and $PM_{2.5}$ (as a co-benefit). The low end of the range of net benefits is constructed by subtracting the cost from the lowest benefit, while the high end of the range is constructed by subtracting the cost from the highest benefit. Following these tables is a discussion of the implications of these estimates, as well as the uncertainties and limitations that should be considered in interpreting the estimates.

In the RIA we provide estimates of costs of emissions reductions to attain the proposed standards in three regions -- California, the rest of the western U.S., and the eastern U.S. In addition, we provide estimates of the benefits that accrue to each of these three regions resulting from (i) control strategies applied within the region, (ii) reductions in transport of ozone associated with emissions reductions in other regions, and (iii) the control strategies for which the regional cost estimates are generated. These benefits are not directly comparable to the costs of control strategies in a region because the benefits include benefits not associated with those control strategies.

The net benefits of emissions reductions strategies in a specific region would be the benefits of the emissions reductions occurring both within and outside of the region minus the costs of the emissions reductions. Because the air quality modeling is done the national level, we do not estimate separately the nationwide benefits associated with the emissions reductions occurring in any specific region.[125] As a result, we are only able to provide net benefits estimates at the national level. The difference between the costs for a specific region and the

[125] For California, we provide separate estimates of the costs and nationwide estimates of benefits, so it is appropriate to calculate net benefits. As such, we provide net benefits for the post-2025 California analysis.

benefits accruing to that region is not an estimate of net benefits of the emissions reductions in that region.

Table 8-1. Total Costs, Total Monetized Benefits, and Net Benefits in 2025 for U.S., except California (billions of 2011$)[a]

	Total Costs	Monetized Benefits	Net Benefits
	7% Discount Rate	7% Discount Rate	7% Discount Rate
Proposed Alternative Standard Levels			
70	**$3.9**	**$6.4 to $13**	**$2.5 to $9.1**
65	**$15**	**$19 to $38**	**$4 to $23**
Alternative Standard Level			
60	$39	$34 to $70	($5) to $31

[a] EPA believes that providing comparisons of social costs and social benefits at 3 and 7 percent is appropriate. Estimating multiple years of costs and benefits is not possible for this RIA due to data and resource limitations. As a result, we provide a snapshot of costs and benefits in 2025, using the best available information to approximate social costs and social benefits recognizing uncertainties and limitations in those estimates.

Table 8-2. Total Costs, Total Monetized Benefits, and Net Benefits of Control Strategies Applied in California, Post-2025 (billions of 2011$)[a]

	Total Costs	Monetized Benefits	Net Benefits
	7% Discount Rate	7% Discount Rate	7% Discount Rate
Proposed Alternative Standard Levels			
70	**$0.80**	**$1.1 to $2**	**$0.3 to $1.2**
65	**$1.6**	**$2.2 to $4.1**	**$0.6 to $2.5**
Alternative Standard Level			
60	$2.2	$3.2 to $5.9	$1 to $3.7

[a] EPA believes that providing comparisons of social costs and social benefits at 3 and 7 percent is appropriate. Estimating multiple years of costs and benefits is not possible for this RIA due to data and resource limitations. As a result, we provide a snapshot of costs and benefits in 2025, using the best available information to approximate social costs and social benefits recognizing uncertainties and limitations in those estimates.

EPA believes that providing comparisons of social costs and social benefits at 3 and 7 percent is appropriate. Ideally, streams of social costs and social benefits over time would be estimated and the net present values of each would be compared to determine net benefits of the illustrative attainment strategies. The three different uses of discounting in the RIA – (i) construction of annualized engineering costs, (ii) adjusting the value of mortality risk for lags in mortality risk decreases, and (iii) adjusting the cost of illness for non-fatal heart attacks to adjust for lags in follow up costs -- are all appropriate. Our estimates of net benefits are the approximations of the net value (in 2025) of benefits attributable to emissions reductions needed to attain just for the year 2025.

Table 8-3. Summary of Total Control Costs (Known and Extrapolated) by Alternative Level for 2025 - U.S., except California (billions of 2011$, 7% Discount Rate)[a]

Alternative Level	Geographic Area	Total Control Costs (Known and Extrapolated)
70 ppb	East	3.9
	West	-
	Total	**$3.9**
65 ppb	East	15
	West	0.4
	Total	**$15**
60 ppb	East	33
	West	5.8
	Total	**$39**

[a] All values are rounded to two significant figures. Extrapolated costs are based on the average-cost methodology.

Table 8-4. Summary of Total Control Costs (Known and Extrapolated) by Alternative Level for post-2025 - California (billions of 2011$, 7% Discount Rate)[a]

Alternative Level	Geographic Area	Total Control Costs (Known and Extrapolated)
70 ppb	California	$0.8
65 ppb	California	$1.6
60 ppb	California	$2.2

[a] All values are rounded to two significant figures. Extrapolated costs are based on the average-cost methodology.

Table 8-5. Regional Breakdown of Monetized Ozone-Specific Benefits Results for the 2025 Scenario (nationwide benefits of attaining each alternative standard everywhere in the U.S. except California) – Full Attainment [a]

Region	Proposed and Alterative Standards		
	70 ppb	**65 ppb**	**60 ppb**
East [b]	99%	96%	92%
California	0%	0%	0%
Rest of West	1%	4%	7%

[a] Because we use benefit-per-ton estimates to calculate the $PM_{2.5}$ co-benefits, a regional breakdown for the co-benefits is not available. Therefore, this table only reflects the ozone benefits.
[b] Includes Texas and those states to the north and east. Several recent rules such as Tier 3 will have substantially reduced ozone concentrations by 2025 in the East, thus few additional controls would be needed to reach 70 ppb.

Table 8-6. Regional Breakdown of Monetized Ozone-Specific Benefits Results for the post-2025 Scenario (nationwide benefits of attaining each alternative standard just in California) – Full Attainment [a]

Region	Proposed and Alterative Standards		
	70 ppb	65 ppb	60 ppb
East	0%	0%	0%
California	93%	94%	94%
Rest of West	6%	6%	6%

[a] Because we use benefit-per-ton estimates to calculate the $PM_{2.5}$ co-benefits, a regional breakdown for the co-benefits is not available. Therefore, this table only reflects the ozone benefits.

In this RIA, we quantify an array of adverse health impacts attributable to ozone and $PM_{2.5}$. The Integrated Science Assessment for Ozone and Related Photochemical Oxidants ("Ozone ISA") (U.S. EPA, 2013a) identifies the human health effects associated with ozone exposure, which include premature death and a variety of illnesses associated with acute (days-long) and chronic (months to years-long) exposures. Similarly, the Integrated Science Assessment for Particulate Matter ("PM ISA") (U.S. EPA, 2009) identifies the human health effects associated with ambient particles, which include premature death and a variety of illnesses associated with acute and chronic exposures. Air pollution can affect human health in a variety of ways, and in Table 8-7 we summarize the "categories" of effects and describe those that we could quantify in our "core" benefits estimates and those we were unable to quantify because we lacked the data, time or techniques.

Table 8-7. Human Health Effects of Pollutants Potentially Affected by Strategies to Attain the Primary Ozone 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 ozone	Premature mortality based on short-term exposure (all ages)	✓	✓	Section 5.6
	Premature respiratory mortality based on long-term exposure (age 30–99)	✓	[a]	Section 5.6
Reduced incidence of morbidity from exposure to ozone	Hospital admissions—respiratory causes (age > 65)	✓	✓	Section 5.6
	Emergency department visits for asthma (all ages)	✓	✓	Section 5.6
	Asthma exacerbation (age 6-18)	✓	✓	
	Minor restricted-activity days (age 18–65)	✓	✓	Section 5.6
	School absence days (age 5–17)	✓	✓	Section 5.6
	Decreased outdoor worker productivity (age 18–65)	[a]	[a]	Section 5.6

Benefits Category	Specific Effect	Effect Has Been Quantified	Effect Has Been Monetized	More Information
	Other respiratory effects (e.g., mediation use, pulmonary inflammation, decrements in lung functioning)	—	—	ozone ISA [c]
	Cardiovascular (e.g., hospital admissions, emergency department visits)	—	—	ozone ISA [c]
	Reproductive and developmental effects (e.g., reduced birthweight, restricted fetal growth)	—	—	ozone ISA [c]
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 of PM RIA
	Infant mortality (age <1)	✓	✓	Section 5.6 of PM RIA
Reduced incidence of morbidity from exposure to $PM_{2.5}$	Non-fatal heart attacks (age > 18)	✓	✓	Section 5.6 of PM RIA
	Hospital admissions—respiratory (all ages)	✓	✓	Section 5.6 of PM RIA
	Hospital admissions—cardiovascular (age >20)	✓	✓	Section 5.6 of PM RIA
	Emergency department visits for asthma (all ages)	✓	✓	Section 5.6 of PM RIA
	Acute bronchitis (age 8–12)	✓	✓	Section 5.6 of PM RIA
	Lower respiratory symptoms (age 7–14)	✓	✓	Section 5.6 of PM RIA
	Upper respiratory symptoms (asthmatics age 9–11)	✓	✓	Section 5.6 of PM RIA
	Asthma exacerbation (asthmatics age 6–18)	✓	✓	Section 5.6 of PM RIA
	Lost work days (age 18–65)	✓	✓	Section 5.6 of PM RIA
	Minor restricted-activity days (age 18–65)	✓	✓	Section 5.6 of PM RIA
	Chronic Bronchitis (age >26)	—	—	Section 5.6 of PM RIA
	Emergency department visits for cardiovascular effects (all ages)	—	—	Section 5.6 of PM RIA
	Strokes and cerebrovascular disease (age 50–79)	—	—	Section 5.6 of PM RIA
	Other cardiovascular effects (e.g., other ages)	—	—	PM ISA [b]
	Other respiratory effects (e.g., pulmonary function, non-asthma ER visits, non-bronchitis chronic diseases, other ages and populations)	—	—	PM ISA [b]
	Reproductive and developmental effects (e.g., low birth weight, pre-term births, etc.)	—	—	PM ISA [b,c]
	Cancer, mutagenicity, and genotoxicity effects	—	—	PM ISA [b,c]
	Asthma hospital admissions (all ages)	—	—	NO_2 ISA [d]

Benefits Category	Specific Effect	Effect Has Been Quantified	Effect Has Been Monetized	More Information
Reduced incidence of morbidity from exposure to NO_2	Chronic lung disease hospital admissions (age > 65)	—	—	NO_2 ISA [d]
	Respiratory emergency department visits (all ages)	—	—	NO_2 ISA [d]
	Asthma exacerbation (asthmatics age 4–18)	—	—	NO_2 ISA [d]
	Acute respiratory symptoms (age 7–14)	—	—	NO_2 ISA [d]
	Premature mortality	—	—	NO_2 ISA [b,c]
	Other respiratory effects (e.g., airway hyperresponsiveness and inflammation, lung function, other ages and populations)	—	—	NO_2 ISA [b,c]

[a] We are in the process of considering an update to the worker productivity analysis for ozone based on more recent literature.
[b] We assess these benefits qualitatively because we do not have sufficient confidence in available data or methods.
[c] We assess these benefits qualitatively because current evidence is only suggestive of causality or there are other significant concerns over the strength of the association.
[d] We assess these benefits qualitatively due to time and resource limitations for this analysis.

8.2 Discussion of Results

The costs and benefits presented in this RIA incorporate an array of methodological and technical changes that the EPA has adopted since the previous review of the ozone standards in 2008 (U.S. EPA, 2008) and proposed reconsideration in 2010 (U.S. EPA 2010).

Several factors contributed to lower cost estimates, including shifting the baseline year from 2020 to 2025, allowing for more time to attain and for Federal measures to work. The baseline starting point has changed from 84 ppb to 75 ppb, substantially reducing the amount of emissions reductions needed and the associated costs of attainment. Also, we have identified additional known controls, which are less expensive per ton than unknown controls. Lastly, there are fewer counties exceeding the alternative standards analyzed, therefore fewer emissions reductions are needed for attainment, resulting in lower cost estimates.

While the costs presented in this analysis decreased compared to the prior analyses, the benefits estimates remained about the same despite the baseline differences. The main factors that affected the benefits estimates included the updated analysis year of 2025, which affects population projections, baseline mortality rates, and income growth adjustment. The Value of a Statistical Life was revised, and we removed thresholds and the assumption of no causality for ozone mortality (which as assumed in 2008). The differences resulted in a tighter benefits range, but the total benefits are about the same as the previous ozone RIA analyses.

8.2.1 Relative Contribution of PM Benefits to Total Benefits

Because of the relatively strong relationship between $PM_{2.5}$ concentrations and premature mortality, PM co-benefits resulting from reductions in NOx emissions can make up a large fraction of total monetized benefits, depending on the specific PM mortality impact function used, and on the relative magnitude of ozone benefits, which is dependent on the specific ozone mortality function assumed. PM co-benefits based on daily average concentrations are calculated over the entire year, while ozone related benefits are calculated only during the summer ozone season. Because the control strategies evaluated in this RIA are assumed to operate year round rather than only during the ozone season, this means that PM benefits will accumulate during both the ozone season and the rest of the year.

For primary benefits estimates in 2025, $PM_{2.5}$ co-benefits account for between 70 and 75 percent of co-benefits, depending on the standard analyzed and on the choice of ozone and PM mortality functions used.[126] The estimate with the lowest fraction from PM co-benefits occurs when we model benefits for the lowest alternative standard level analyzed (60ppb) and add the lower bound core estimates for both ozone-related and $PM_{2.5}$ (co-benefit) related mortality. The estimate with the highest fraction from PM co-benefits results from modeling the highest alternative standard level analyzed (70ppb) based on combining the high-bound core ozone and $PM_{2.5}$ (co-benefit) related mortality estimates.

8.2.2 Developing Future Control Strategies with Limited Data

Because of relatively higher ozone levels in several large urban areas (Southern California, Houston, and the Northeastern urban corridor, including New York and Philadelphia) and because of limitations associated with the data on currently known emissions control technologies, the EPA recognized that known and reasonably anticipated emissions controls would likely not be sufficient to bring some areas into attainment with either the existing or alternative, more stringent ozone standard levels. Therefore, we designed this analysis in two stages: the first stage focused on analyzing the air quality improvements that could be achieved through application of documented, well-characterized emissions controls, and the costs and

[126] For the separate results for post-2025, $PM_{2.5}$ co-benefits account for between 30 and 45 percent of total benefits, again depending on the standard level analyzed and the choice of ozone and PM mortality functions.

benefits associated with those controls. The second stage took the emissions reductions beyond known controls and used an extrapolation method to estimate the costs and benefits of these additional emissions reductions needed to bring all areas into full attainment with the alternative standard levels analyzed.

The structure of the RIA reflects this two-stage analytical approach. Separate chapters are provided for the emissions, air quality, and cost impacts of modeled controls. We used the information currently available to develop reasonable approximations of the costs and benefits of the extrapolated portion of the emissions reductions necessary to reach attainment. However, because of the uncertainty associated with the extrapolation of costs, we judged it appropriate to provide separate estimates of the costs and benefits for partial attainment (based on known controls) and full attainment (based on known controls and extrapolation), as well as an overall estimate for reaching full attainment. There is a single chapter on benefits, because the methodology for estimating benefits does not change between stages. However, in that chapter, we again provide separate estimates of the benefits associated with the partial attainment and full attainment portions of the analysis.

In both stages of the analysis, it should be recognized that all estimates of future costs and benefits are not intended to be forecasts of the actual costs and benefits of implementing potentially revised standards. Ultimately, states and local areas will be responsible for developing and implementing emissions control programs to reach attainment with the ozone NAAQS, with the timing of attainment being determined by future decisions by states and the EPA. Our estimates are intended to provide information on the general magnitude of the costs and benefits of alternative standard levels rather than on precise predictions of control measures, costs, or benefits. With these caveats, we expect that this analysis can provide a reasonable picture of the types of emissions controls that are currently available, the direct costs of those controls, the levels of emissions reductions that may be achieved with these controls, the air quality impact that can be expected to result from reducing emissions, and the public health benefits of reductions in ambient ozone levels. This analysis identifies those areas of the U.S. where our existing knowledge of control strategies is not sufficient to allow us to model attainment, and where additional data or research may be needed to develop strategies for attainment.

In many ways, RIAs for proposed actions are learning processes that can yield valuable information about the technical and policy issues that are associated with a particular regulatory action. This is especially true for RIAs for proposed NAAQS, where we are required to stretch our understanding of both science and technology to develop scenarios that illustrate how certain we are about how economically feasible the attainment of these standards might be regionally. The proposed ozone NAAQS RIA provided great challenges when compared to previous RIAs primarily because as we tighten standards across multiple pollutants with overlapping precursors (e.g., the recent tightening of the $PM_{2.5}$ standards), we move further down the list of cost-effective known and available controls in our database. With the more stringent NAAQS, more areas will need to find new ways of reducing emissions. While we can speculate on what some of these technologies might look like based on new developments in energy efficiency and clean technology, the specific technological path in different nonattainment areas is not clear.

Because of the uncertainty regarding the development of future emissions reduction strategies, a significant portion of the analysis is based on extrapolating from available data on known control technologies to generate the emissions reductions necessary to reach full attainment of an alternative ozone NAAQS and the resulting costs and benefits. Studies indicate that it is not uncommon for pre-regulatory cost estimates to be higher than later estimates, in part because of difficulty in predicting technological changes. Over longer time horizons, such as the time allowed for areas with high levels of ozone pollution to meet the ozone NAAQS, the opportunity for technical change is greater (See Chapter 7, Section 7.2 for additional discussion). Also, because of the nature of the extrapolation method for benefits (which focuses on reductions in ozone only at monitors that exceed the NAAQS), we generally understate the total benefits that would result from implementing additional emissions controls to fully attain the ozone NAAQS (i.e., assuming that the application of control strategies would result in ozone reductions both at nonattainment and attainment monitors). On the other hand, the possibility also exists that benefits are overestimated, because it is possible that new technical changes might not meet the specifications, development time lines, or cost estimates provided in this analysis.

8.3 Framing Uncertainty

This section includes a qualitative presentation of key factors that (1) could impact how air quality changes over time; (2) could impact the timing for meeting an alternative standard; (3)

are difficult to predict and quantify; and (4) introduce additional uncertainty into this analysis.[127] These factors, summarized in Table 8.8 below, include energy development, distribution, and use trends; land use development patterns; economic factors; energy and research and development policies; climate signal changes; and the influence of technological change. Additional factors that could have an impact on how air quality changes over time include environmental indicators other than climate change and societal preferences and attitudes toward the environment and conservation; the potential direction and magnitude of these additional factors is less clear.

These key factors can impact the estimated baseline air quality used in the analysis, and as a result the types of control measures and associated costs needed to meet an alternative standard. In addition, some combinations of the key factors could have significant effects beyond the effects of any individual factor. We cannot estimate the probability that any one factor or combination of factors will occur, but we do believe that they introduce additional, broader uncertainties about future trends that provide important context for the costs and benefits presented in this analysis.

Table 8-8. Relevant Factors and Their Potential Implications for Attainment

Individual Factors	Potential Implications for NAAQS Attainment	Information on Trends
Energy -- Extraction, conversion, distribution and storage, efficiency, international energy trends	Geopolitics, reserves, international and domestic demand, and technological breakthroughs in energy technologies can drive fuel prices up or down. If more renewable sources of energy are employed and use of natural gas increases, then emissions may be lower, potentially lowering attainment costs.	Recently there has been an increase in domestic production of oil and a relative decrease in imported oil, in addition to policies and investments geared toward the development of alternative fuels and energy efficiency.[128] This is likely a result of all of these activities, which have led to a reduction of U.S. dependence on imports of foreign oil. Several trends have emerged within the last ten years and are expected to continue, including increased natural gas production and consumption, renewable energy installations,

[127] OMB Circular A-4 indicates that qualitative discussions should be included in analyses whenever there is insufficient data to quantify uncertainty.

[128] http://energy.gov/articles/us-domestic-oil-production-exceeds-imports-first-time-18-years

Individual Factors	Potential Implications for NAAQS Attainment	Information on Trends
		and energy efficiency technology installations.[129]
Land Use Development Patterns – Design of urban areas, vehicle-miles travelled	A move toward denser urban settlements, slowing of growth in vehicle miles travelled (VMT) and increased use of public transit could decrease emissions, potentially lowering attainment costs.[130]	Recent trends in VMT illustrate some of the uncertainty around future emissions from mobile sources[131]. In 2006, projections of VMT showed a sustained increase,[132] yet VMT growth slowed in recent years and actually declined in 2008 and 2009.[133] Between 2000 and 2010 average growth in VMT was 0.8%, as compared to 2.9% from the previous decade.
The Economy	An increase in economic growth, investment in technologies that have high energy use, and a return of U.S. manufacturing could lead to higher emissions making attainment potentially more costly. A slowing of the economy, investments in energy efficient technologies, and a continuation of a service-based economy could lead to lower emissions making attainment potentially less costly.[134]	Affluence leads to increased consumption and energy use. However, this increase may not be proportional. Energy and materials is not directly proportional to economic growth, decreasing or stabilizing over time in spite of continued economic growth.[135]
Policies[a] – Energy efficiency, energy security, direction of research and development, renewable energy	A move toward energy security and independence would mean an increased use of domestic energy sources. If this results in a fuel mix where emissions decrease, then attainment could likely be less costly.	State and local policies related to energy efficiency, cleaner energy, energy security[136], as well as the direction of research and development of technology can have a direct or indirect effect on emissions. Policies that result in energy efficiency, renewable energy, the use

[129] http://www.eia.gov/forecasts/aeo/er/pdf/0383er%282014%29.pdf; http://energy.gov/sites/prod/files/2014/08/f18/2013%20Wind%20Technologies%20Market%20Report%20Presentation.pdf; http://www.eia.gov/electricity/monthly/update/archive/april2014/; http://www.ercot.com/content/news/presentations/2014/GCPA%20%2002%20Oct%202013%20FINAL.pdf; http://energy.gov/eere/sunshot/photovoltaics.

[130] For example, see Cervero (1998), the Center for Clean Air Policy's Transportation Emissions Guidebook (http://www.trb.org/Main/Blurbs/156164.aspx). For ongoing research see http://apps.trb.org/cmsfeed/TRBNetProjectDisplay.asp?ProjectID=3092.

[131] For example, see the Transportation Research Board's National Cooperative Highway Research Program (NCHRP) 2014.

[132] https://www.fhwa.dot.gov/policy/2006cpr/chap9.htm#body

[133] https://www.fhwa.dot.gov/policyinformation/travel_monitoring/13jantvt/page2.cfm

[134] For example, Bo (2011), and http://www.epa.gov/region1/airquality/nox.html for manufactures contributions to NO_X emissions.

[135] UNEP 2011, http://www.unep.org/resourcepanel/decoupling/files/pdf/decoupling_report_english.pdf

[136] For example, http://www2.epa.gov/laws-regulations/summary-energy-independence-and-security-act.

Individual Factors	Potential Implications for NAAQS Attainment	Information on Trends
	If not, attainment could likely be more costly. A move toward investments in fuel efficiency and low emissions fuels could decrease emissions and likely lower attainment costs.	of cleaner fuels and conservation measures would likely result in decreased emissions and likely decrease attainment costs.[137] Growth in energy demand has stayed well below growth in gross domestic product, likely as a result of technological advances, federal, state and local energy efficiency standards and policies, and other macroeconomic factors.[138] U.S. productivity per energy expended relative to other countries suggests that additional efficiency gains are possible.[139]
Intensity, Location and Outcome of the Climate Change Signal	Strong climate signals that bring high temperatures could increase ozone, likely making attainment more costly.	Uncertainty exists regarding how the climate signal will interact with air quality, as well as with other factors. However, research demonstrates that in areas where there are both high levels of emissions and high temperatures, attaining an ozone standard will likely be much harder. The magnitudes of these impacts will depend on atmospheric chemical and physical processes, as well as anthropogenic activities that increase or decrease NOx and/or VOC emissions.[140]
Technological Change -- Including emissions reductions technologies and other technological developments	Innovation in production and emissions control technologies, learning that lower costs, and breakthroughs in battery/energy storage technologies for use with renewable energy could improve air quality, reducing emissions and likely lowering attainment costs.	Examples of emerging technologies include carbon capture and sequestration (CCS), battery technologies, emerging advanced biofuels, which could all have breakthroughs that could impact fuel use. Similarly, shifts in industrial production processes, such as a move from using primary metals to more recycling could impact energy use[141] .

[a] Policies refer to any policies or regulations that are not environmental regulations set by U.S. EPA, states, tribes, or local authorities.

8.4 Key Observations from the Analysis

The following are key observations about the RIA results.

- **Tightening the ozone standards can incur significant, but uncertain, costs.** Our estimates of costs for a set of modeled NOx and VOC controls comprise only a small part of the estimated costs of full attainment. These estimated costs for the modeled set of

[137] For example, see http://www.dsireusa.org/solar/solarpolicyguide/.

[138] http://bipartisanpolicy.org/sites/default/files/BPC%20SEPI%20Energy%20Report%202013_0.pdf, p. 5.

[139] http://bipartisanpolicy.org/sites/default/files/BPC%20SEPI%20Energy%20Report%202013_0.pdf, p. 69.

[140] See Jacobs (2009).

[141] http://www.eia.gov/todayinenergy/detail.cfm?id=16211

controls are still uncertain, but they are based on the best available information on control technologies, and have their basis in real, tested technologies. Estimating costs of full attainment was based on a generalized relationship between emissions and ozone levels. This introduces significant uncertainty into the calculation of the emissions reductions that might be needed to reach full attainment.

- **Tightening the ozone standards can also result in significant benefits.** Estimates of benefits are driven largely by projected reductions in ozone-related short-term mortality and co-benefits associated with reductions in $PM_{2.5}$-related long-term mortality. Although using a benefit-per-ton approach in modeling $PM_{2.5}$-related cobenefits (rather than direct modeling) has increased uncertainty, this approach is peer-reviewed and robust. We also modeled reductions in ozone-related long-term respiratory mortality, however due to concerns over potential double counting of benefits and limitations in our ability to project the lag-structure of reductions in this mortality endpoint, we did not include these estimates as part of the core benefit estimate. In addition to these mortality endpoints, we did quantify a wide-range of morbidity endpoints for both ozone and $PM_{2.5,}$ although these contribute only minimally to total monetized benefits.

- **Air quality modeling approach can introduce uncertainty.** Based on air quality modeling sensitivity analyses, there is significant spatial variability in the relationship between local and regional NOx emission reductions and ozone levels across urban areas. We performed a national scale air quality modeling analysis to estimate ozone concentrations for the future base case year of 2025. To accomplish this, we modeled multiple emissions cases for 2025, including the 2025 base case and twelve (12) 2025 emissions sensitivity simulations. The 12 emissions sensitivity simulations were used to develop ozone sensitivity factors (ppb/ton) from the modeled response of ozone to changes in NOx and VOC emissions from various sources and locations. These ozone sensitivity factors were then used to determine the amount of emissions reductions needed to reach the 2025 baseline and evaluate potential alternative standard levels of 70, 65, and 60 ppb incremental to the baseline. We used the estimated emissions reductions needed to reach each of these standard levels to analyze the costs and benefits of alternative standard levels.

- **Available technologies that might achieve NOx and VOC reductions to attain alternative ozone NAAQS are not sufficient.** In some areas of the U.S., the information we have about existing controls does not result in sufficient emissions reductions needed to meet the existing standard. After applying existing rules and the illustrative known controls across the nation (excluding California), in order to reach 70 ppb we were able to identify controls that reduce overall NOx emissions by 490,000 tons and VOC emissions by 55,000 tons. In order to reach 65 ppb we were able to identify controls that reduce overall NOx emissions by 1,100,000 tons and VOC emissions by 110,000 tons. After these reductions, in order to reach 70 ppb over 150,000 tons of NOx emissions remained, and in order to reach 65 ppb over 750,000 tons of NOx emissions remained.

- **California costs and benefits are highly uncertain.** California faces large challenges in meeting any alternative standard, but their largest challenges may be in attaining the existing standard. Because our analysis suggested that all available controls would be exhausted in attempting to meet the current 75 ppb standard, all of the benefits and costs of lower standards in California are based on the application of unknown controls. Both the benefits and the costs associated with the assumed NOx and VOC reductions in California are particularly uncertain.

- **Some EPA existing mobile source programs will help areas reach attainment.** These programs promise to continue to help areas reduce ozone concentrations beyond 2025. In California, continued implementation of mobile source rules, including the onroad and nonroad diesel rules and the locomotive and marine engines rule, are projected to reduce NOx emissions by an additional 14,000 tons and VOC emissions by an additional 6,300 tons between 2025 and 2030. These additional reductions will likely reduce the overall emissions reductions needed for attainment relative to what California might have needed to reduce from other sectors if attainment were to be required in 2025.
- **The economic impacts (i.e., social costs) of the cost of these modeled controls were not included in this analysis.** Incorporating the economic impact of the extrapolated portion of the costs was too uncertain to be included as part of these estimates, and it was determined best to keep the modeled and extrapolated costs on the same basis.
- **Costs and benefits will depend on implementation timeframes.** States will ultimately select the specific timelines for implementation as part of their State Implementation Plans. To the extent that states seek classification as extreme nonattainment areas, the timeline for implementation may be extended beyond 2025, meaning that the amount of emissions reductions that will be required in 2025 will be less, and costs and benefits in 2025 will be lower.

8.5 References

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CHAPTER 9: STATUTORY AND EXECUTIVE ORDER IMPACT ANALYSIS

Overview

This section explains the statutory and executive orders applicable to EPA rules, and discusses EPA's actions taken pursuant to these orders.

9.1 National Technology Transfer and Advancement Act

Section 12(d) of the National Technology Transfer and Advancement Act of 1995 (NTTAA), Public Law No. 104-113, §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.

Today's proposed rulemaking does not involve technical standards. Therefore, the EPA is not considering the use of any voluntary consensus standards.

9.2 Paperwork Reduction Act

This action does not impose an information collection burden under the provisions of the Paperwork Reduction Act, 44 U.S.C. 3501 et seq. There are no information collection requirements directly associated with the establishment of a NAAQS under section 109 of the CAA.

Burden means the total time, effort, or financial resources expended by persons to generate, maintain, retain, or disclose or provide information to or for a federal agency. This includes the time needed to review instructions; develop, acquire, install, and utilize technology and systems for the purposes of collecting, validating, and verifying information, processing and maintaining information, and disclosing and providing information; adjust the existing ways to comply with any previously applicable instructions and requirements; train personnel to be able to respond to a collection of information; search data sources; complete and review the collection of information; and transmit or otherwise disclose the information. Burden is defined at 5 CFR 1320.3(b).

An agency may not conduct or sponsor, and a person is not required to respond to a collection of information unless it displays a currently valid OMB control number. The OMB control numbers for the EPA's regulations in 40 CFR are listed in 40 CFR part 9. Per the *Implementation of the 2008 National Ambient Air Quality Standards for Ozone: State Implementation Plan Requirements,* the annual burden for this information collection averaged over the first 3 years is estimated to be a total of 40,000 labor hours per year at an annual labor cost of $2.4 million (present value) over the 3-year period or approximately $91,000 per state for the 26 state respondents, including the District of Columbia. The average annual reporting burden is 690 hours per response, with approximately 2 responses per state for 58 state respondents. There are no capital or operating and maintenance costs associated with the proposed rule requirements.

In addition, per the draft *Supporting Statement for Revisions to Ambient Air Monitoring Regulations for Ozone (Proposed Rule)* (EPA ICR Number 0940), as part of the ozone NAAQS proposed rulemaking EPA is proposing revisions to the length of the required ozone monitoring season (see Section VI of the preamble). The draft Information Collection Request (ICR) is estimated to involve 158 respondents for a total average cost of approximately $24 million (total capital, and labor and operation and maintenance costs) plus a total burden of 339,930 hours for the support of all operational aspects of the entire ozone monitoring network. The labor costs associated with these hours are approximately $19.8 million. Also included in the total costs are other costs of operation and maintenance of $2.2 million and equipment and contract costs of $2.1 million. EPA typically funds 60 percent of the approximately $24 million through grants to the State/local agencies. In addition to the costs at the State, local, and Tribal air quality management agencies, there is a burden to EPA of 41,418 hours and $2.6 million.

9.3 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 today's proposed 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 today's 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 ozone 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). We continue to be interested in the potential impacts of the proposed rule on small entities and welcome comments on issues related to such impacts.

9.4 Unfunded Mandates Reform Act

Title II of the Unfunded Mandates Reform Act of 1995 (UMRA), Public Law 104-4, establishes requirements for federal agencies to assess the effects of their regulatory actions on state, local, and tribal governments and the private sector. Under section 202 of the UMRA, the EPA generally must prepare a written statement, including a cost-benefit analysis, for proposed and final rules with "federal mandates" that may result in expenditures to state, local, and tribal governments, in the aggregate, or to the private sector, of $100 million or more in any 1 year. Before promulgating an EPA rule for which a written statement is needed, section 205 of the UMRA generally requires the EPA to identify and consider a reasonable number of regulatory alternatives and to adopt the least costly, most cost-effective or least burdensome alternative that achieves the objectives of the rule. The provisions of section 205 do not apply when they are inconsistent with applicable law. Moreover, section 205 allows the EPA to adopt an alternative other than the least costly, most cost-effective or least burdensome alternative if the Administrator publishes with the final rule an explanation for why that alternative was not adopted. Before the EPA establishes any regulatory requirements that may significantly or uniquely affect small governments, including tribal governments, it must have developed under section 203 of the UMRA a small government agency plan. The plan must provide for notifying potentially affected small governments, enabling officials of affected small governments to have meaningful and timely input in the development of the EPA regulatory proposals with significant federal intergovernmental mandates, and informing, educating, and advising small governments on compliance with the regulatory requirements.

Today's proposed rule contains no federal mandates (under the regulatory provisions of Title II of the UMRA) for state, local, or tribal governments or the private sector. The proposed rule 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 proposed rule contains no regulatory requirements that might significantly or uniquely affect small governments. Furthermore, as indicated previously, 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 (RIA) pursuant to the Unfunded Mandates Reform Act would not furnish any information that the court could consider in reviewing the NAAQS). Accordingly, the EPA has determined that the provisions of sections 202, 203, and 205 of the UMRA do not apply to this proposed decision. 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, EPA will address, as appropriate, unfunded mandates if and when it proposes any revisions to 40 CFR parts 51 or 58.

9.5 Executive Order 12866: Regulatory Planning and Review

Under section 3(f)(1) of Executive Order (EO) 12866 (58 FR 51735, October 4, 1993), the ozone NAAQS 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 EO 12866 and any changes made in response to OMB recommendations have been documented in the docket for this action. In addition, the EPA prepared this RIA of the potential costs and benefits associated with this action. A copy of the analysis is available in the RIA docket (EPA-HQ-OAR-2013-0169) and the analysis is briefly summarized here. The RIA estimates the costs and monetized human health and welfare benefits of attaining four alternative ozone NAAQS nationwide. Specifically, the RIA examines the alternatives of 75 ppb, 70 ppb, 65 ppb, and 60 ppb. The RIA contains illustrative analyses that consider a limited number of emissions control scenarios that states and Regional Planning Organizations might implement to achieve these alternative ozone NAAQS. However, the Clean Air Act (CAA) and judicial decisions make clear that the economic and technical feasibility of attaining ambient standards are not to be considered in setting or revising NAAQS, although such factors may be considered in the development of state plans to implement the standards. Accordingly, although this RIA has been prepared, the results of the RIA have not been considered in issuing today’s proposed rule.

9.6 Executive Order 12898: Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations

Executive Order 12898 (59 FR 7629 (Feb. 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.

To gain a better understanding of the populations within the areas potentially impacted by a revised ozone NAAQS, the EPA conducted a proximity analysis that examined socio-demographic attributes of populations in these areas. The areas of interest for these analyses were defined as all counties contained within any Core-Based Statistical Area (CBSA) with at least one monitor with a current (2011-2013) design value above the proposed range of standard levels (65 to 70 ppb), as well as counties not in a CBSA with a current (2011-2013) design value above the proposed range of standard levels. The tabulation of results from this analysis are presented below. Table 9-1 shows the percent of minority populations within CBSAs with monitors above the levels of the proposed ozone standard levels, as well as the national percentages for comparison. Table 9-2 shows the percent of populations of different ages, education, and income level for those same areas. Additional details can be found in Appendix 9A.

Table 9-1. Summary of Population Totals and Demographic Categories for Areas of Interest and National Perspective

Demographic Summary	Population	White	African American	Native American	Other or Multiracial[1]	Minority/Non-White Hispanic[a]
Area of Interest Total						
65 ppb	221,431,286	153,706,027	30,429,108	1,726,110	35,570,041	67,725,259
70 ppb	193,316,836	132,112,738	27,193,155	1,488,364	32,522,579	61,204,098
% of Area of Interest Total						
65 ppb		69%	14%	1%	16%	31%
70 ppb		68%	14%	1%	17%	32%
National Total	312,861,256	226,405,205	39,475,216	2,952,087	44,028,748	86,456,051
% of National Total		72%	13%	1%	14%	28%

[a] The race *Minority/Non-White Hispanic* field is computed by subtracting the white population from the total population.

Table 9-2. Summary of Population Totals and Demographic Categories for Areas of Interest and National Perspective

Demographic Summary	Linguistically Isolated	Age 0 – 4	Age 0 - 17	Age 65+	Without a HS Diploma	Low Income[2]
Area of Interest Total						
65 ppb	13,072,109	14,695,948	54,008,810	27,163,990	20,914,891	67,027,700
70 ppb	12,179,896	12,849,637	47,296,147	23,518,071	18,495,474	58,296,224
% of Area of Interest Total						
65 ppb	6%	7%	24%	12%	14%	30%

[1] Appendix 9A clarifies that "other or multiracial" is derived from individual reporting on Census forms and includes citations to the specific 2010 Census data used in this analysis.

[2] Appendix 9A clarifies that "low income" in this analysis is defined as income two times the poverty line or less.

70 ppb	6%	7%	24%	12%	15%	30%
National Total	19,196,507	20,465,065	75,217,176	40,830,262	30,952,789	101,429,436
% of National Total	6%	7%	24%	13%	15%	32%

The proposed rule will establish uniform national standards for ozone air pollution. This analysis identifies, on a limited basis, the subpopulations that may be exposed to elevated ozone concentrations, who thus are expected to benefit most from this regulation. This analysis does not identify the demographic characteristics of the most highly affected individuals or communities; nor does it quantify the level of risk faced by those individuals or communities. To the extent that any minority, low-income or indigenous subpopulation is disproportionately impacted by ozone levels because it resides in an area of interest, that subpopulation also stands to see increased environmental and health benefits from the emission reductions called for by this proposed rule. Available data suggest that the counties most likely to experience risk reductions from the proposed rule are approximately 14% African American, 1% Native American, 16-17% Other and multi-racial, and 31-32% Minority/Hispanic, which are approximately equal to the proportions of these populations represented in the U.S.

9.7 Executive Order 13045: Protection of Children from Environmental Health & Safety Risks

Executive Order 13045, "Protection of Children from Environmental Health Risks and Safety Risks" (62 FR 19885, April 23, 1997) applies to any rule that: (1) is determined to be "economically significant" as defined under Executive Order 12866, and (2) concerns an environmental health or safety risk that the EPA has reason to believe may have a disproportionate effect on children. If the regulatory action meets both criteria, the Agency must evaluate the environmental health or safety effects of the planned rule on children, and explain why the planned regulation is preferable to other potentially effective and reasonably feasible alternatives considered by the Agency.

Today's proposed rule is subject to Executive Order 13045 because it is an economically significant regulatory action as defined by Executive Order 12866, and we believe that the environmental health risk addressed by this action may have a disproportionate effect on children. The proposed rule will establish uniform national ambient air quality standards for ozone; these standards are designed to protect public health with an adequate margin of safety, as required by CAA section 109. However, the protection offered by these standards may be especially important for children because children, especially children with asthma, along with other sensitive population subgroups such as all people with lung disease and people active outdoors, are potentially susceptible to health effects resulting from ozone exposure. Because children are considered a potentially susceptible population, we have carefully evaluated the environmental health effects of exposure to ozone pollution among children. Discussions of the results of the evaluation of the scientific evidence, policy considerations, and the exposure and risk assessments pertaining to children occurs throughout the preamble. Table 9-2 above includes a summary of available data that indicates that the counties most likely to experience risk reductions from the proposed rule are comprised of approximately 24% of the population between the ages of zero and 17, which is approximately equal to the proportions of this segment of the population represented in the U.S.

9.8 Executive Order 13132: Federalism

Executive Order 13132, entitled "Federalism" (64 FR 43255, August 10, 1999), requires EPA to develop an accountable process to ensure "meaningful and timely input by state and local officials in the development of regulatory policies that have federalism implications." "Policies that have federalism implications" are defined in the Executive Order to include regulations that 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."

Today's proposed rule 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 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 E (above) on UMRA, this rule does not impose significant costs on state, local, or tribal governments or the private sector. Thus, Executive Order 13132 does not apply to this rule.

However, as also noted in section D (above) on UMRA, EPA recognizes that states will have a substantial interest in this rule and any corresponding revisions to associated SIP requirements and air quality surveillance requirements, 40 CFR part 51 and 40 CFR part 58, respectively. Therefore, in the spirit of Executive Order 13132, and consistent with the EPA policy to promote communications between the EPA and state and local governments, the EPA has specifically solicited comment on today's proposed rule from state and local officials.

9.9 Executive Order 13175: Consultation and Coordination with Indian Tribal Governments

Executive Order 13175, entitled "Consultation and Coordination with Indian Tribal Governments" (65 FR 67249, November 9, 2000), requires the EPA to develop an accountable process to ensure "meaningful and timely input by tribal officials in the development of regulatory policies that have tribal implications." This rule concerns the establishment of ozone NAAQS. The Tribal Authority Rule gives tribes the opportunity to develop and implement CAA programs such as the ozone 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.

Today's proposed rule does not have tribal implications, as specified in Executive Order 13175. 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. In addition, tribes are not obligated to conduct ambient monitoring for ozone or to adopt the ambient monitoring requirements of 40 CFR part 58. Thus, Executive Order 13175 does not apply to this rule.

The EPA specifically solicits comment on this rule from tribal officials. Prior to finalization of this proposal, the EPA intends to conduct outreach consistent with the EPA Policy on Consultation and Coordination with Indian Tribes. Outreach to tribal environmental professionals will be conducted through participation in Tribal Air call, which is sponsored by the National Tribal Air Association. In addition, the EPA intends to offer formal consultation to the tribes during the public comment period. If consultation is requested, a summary of the result of that consultation will be presented in the notice of final rulemaking and will be available in the docket.

9.10 Executive Order 13211: Actions that Significantly Affect Energy Supply, Distribution, or Use

Today's proposed rule is not a "significant energy action" as defined in Executive Order 13211, "Actions Concerning Regulations That Significantly Affect Energy Supply, Distribution, or Use" (66 FR 28355 (May 22, 2001)) because in the Agency's judgment it is not likely to have a significant adverse effect on the supply, distribution, or use of energy. The purpose of this rule is to establish revised NAAQS for ozone. The rule does not prescribe specific pollution control strategies by which these ambient standards will be met. Such strategies will be 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. Thus, the EPA concludes that today's proposed rule is not likely to have any adverse energy effects and does not constitute a significant energy action as defined in Executive Order 13211.

Application of the modeled illustrative control strategy containing known controls for power plants, shown in Chapter 7, means that 3 percent of the total projected coal-fired EGU capacity nationwide in 2025 could be affected by controls for the alternative standard level of 70 ppb. Similarly, 21 percent of total projected coal-fired EGU capacity in 2025 could be affected for the alternative standard level of 65 ppb, and 22 percent for the alternative standard level of 60 ppb. In addition, some fuel switching might occur that could alter these percentages, though we are unable to estimate the effect on energy impacts from fuel switching. Controls on EGUs powered by fuels other than coal were not part of this illustrative analysis, and thus, would not be affected. In addition, we are unable to estimate energy impacts resulting from application of controls to non-EGUs or mobile sources. It is important to note that the estimates presented above are just one illustrative strategy and states may choose to apply control on sources other than EGUs for the purposes of attaining a more stringent ozone standard.

APPENDIX 9A: SOCIO-DEMOGRAPHIC CHARACTERISTICS OF POPULATIONS IN CORE BASED STATISTICAL AREAS WITH OZONE MONITORS EXCEEDING PROPOSED OZONE STANDARDS

Overview

This appendix describes a limited screening-level analysis of the socio-demographic characteristics of populations living in areas with an ozone monitor with a current (2011-2013) design value exceeding the proposed range of ozone standard levels, 65 to 70 parts per billion (ppb). This analysis does not include a quantitative assessment of exposure and/or risk for specific populations of potential interest from an environmental justice (EJ) perspective, and therefore it cannot be used to draw any conclusions regarding potential disparities in exposure or risk across populations of interest from an EJ perspective. This appendix describes the technical approach used in the analysis, discusses uncertainties and limitations associated with the analysis, and presents results.

Executive Order 12898, *Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations* (59 FR 7629; Feb. 16, 1994), 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. . In addition, Executive Order 13045, *Protection of Children from Environmental Health Risks and Safety Risks* (62 FR 19885, April 23, 1997) applies to any rule that: (1) is determined to be "economically significant" as defined under Executive Order 12866, and (2) concerns an environmental health or safety risk that the EPA has reason to believe may have a disproportionate effect on children. Accordingly, the Environmental Protection Agency's (EPA) Office of Air Quality Planning and Standards (OAQPS) has conducted a limited analysis of population demographics in some areas that may be affected by the proposed revisions to the National Ambient Air Quality Standards (NAAQS) for ozone.

The EPA Administrator is proposing to revise the NAAQS for ozone from the current level of 75 ppb to within the range of 65 to 70 ppb, while also soliciting comment on retaining the current standard and levels down to 60 ppb. This proposed rule will establish uniform national standards for ozone in ambient air. The proposed revisions would improve public health protection for at-risk groups, especially children. The Agency has elected to conduct a limited analysis of key socio-demographic characteristics of populations living in areas of interest, defined for this analysis as any Core Based Statistical Area (CBSA) with at least one county having an ozone monitor with a current (2011-2013) design value exceeding the proposed range of ozone standard levels (65 to 70 ppb), and also including individual counties not included in a CBSA with a design value exceeding the proposed range of standards.

9A.1 Design of Analysis

To gain a better understanding of the populations within the areas of interest, the EPA conducted an analysis at the county level for this ozone NAAQS review. The areas of interest for these analyses were defined as all counties contained within any CBSA with at least one monitor with a current (2011-2013) design value above the proposed range of standard levels (65 to 70 ppb) as well as counties not in a CBSA with a current (2011-2013) design value above the proposed range of standard levels (65 ppb to 70 ppb). The areas of interest were designed to try to capture population and communities most likely to benefit from improved air quality resulting from implementation of the proposed ozone NAAQS revisions.

At the lower end of the range of proposed standard levels of 65 ppb, this definition of the areas of interest resulted in 953 counties being analyzed, 888 in 265 CBSAs and 65 outside CBSAs. At the upper end of the range of proposed standard levels of 70 ppb, this resulted in 707 counties being analyzed, 680 in 182 CBSAs and 27 outside CBSAs. The demographic variables used in this analysis include race, ethnicity, age, economic and education data. Details on these demographic groups are provided in the following section (9A.1.1).

To compare the demographic data in the areas of interest to the national data, the data from the identified counties were aggregated to represent the areas of interest identified by the lower and upper end of the proposed range of standard levels. This analysis identifies, on a limited basis, the subpopulations that are most likely to experience reductions in ozone concentrations as a result of actions taken to meet the proposed range of standard levels and thus are expected to benefit most from this regulation. This analysis does not identify the demographic characteristics of the most highly affected individuals or communities nor does it quantify the level of risk faced by those individuals or communities. To the extent that any minority, low-income or indigenous subpopulation is disproportionately impacted by ozone levels because they reside in an area of interest, that subpopulation also stands to see increased environmental and health benefit from meeting the more protective proposed standard levels.

The aggregated demographic sub-population values across the areas of interest are compared to the national data and are listed in Table 9A-2 in Section 9A-3.

9A.1.1 Demographic Variables Included in Analysis

This analysis includes race, ethnicity, and age data derived from the 2010 Census SF1 dataset[i] and economic and education data from the Census Bureau's 2006-2010 American Community Survey (ACS) 5-Year Estimates.[ii] This data is summarized in Table 9A-1.

Table 9A-1. Census Derived Demographic Data

Race, Ethnicity, and Age Data (Census 2010 block-level SF1 data)*	
Parameter	Definition
Population	Total population
White	Number of whites (may include Hispanics)
African Americans	Number of African Americans (may include Hispanics)
Native Americans	Number of Native Americas (may include Hispanics)
Other and multiracial	Number of other race and multiracial (may include Hispanics)
Minority/Non-white Hispanic	Total Population less White Population
Age 0 to 4	Number of people age 0 to 4
Age 0 to 17	Number of people age 0 to 17
Age 65 and up	Number of people age 65 and up
Economic and Education Date (2006-2010 ACS)*	
Parameter	Definition
Poverty	Number of people living in households with income below the poverty line
2 x Poverty	Number of people living in households with income below twice the poverty line
Linguistic isolation	Number of people linguistically isolated
Education level	Adults without a high school diploma

*Census 2010 does not currently report this data for the Virgin Islands, Guam, American Samoa, and the Northern Marianas; Census 2000 data are used for these areas.

As noted above, the EPA uses population data collected by the 2010 Census. All data is stored at the block level. For those indicators available from the Census at the block group, but not block level, the EPA assigns a block the same percentage as the block group of which it is a part. For example, a block is assigned the same percentage of people living below the national poverty line as the block group in which it is contained. Nationally, a census block contains about 50 people on average; and a block group contains about 26 blocks on average, or about 1,350 people. (For comparison, a census tract is larger than a block group, with each tract containing an average of 3 block groups, or about 4,300 people). For this analysis the data was aggregated to the county level.

Data on race, ethnicity and age for all census blocks in the country except for the Virgin Islands, Guam, American Samoa, and the Northern Marianas were obtained from the 2010 Census SF1 dataset. This dataset gives a breakdown of the population for each census block among different racial and ethnic classifications, including: White, African American or Black, Hispanic or Latino, American Indian or Native Alaskan, Asian, Native Hawaiian or other South Pacific Islander, other race, and two or more races. Data on age distributions in the U.S. and Puerto Rico were obtained at the census block level from the 2010 Census of Population and Housing Summary File 1 (SF1) short form, Table P12. SF1 contains the information compiled from the questions asked of all people about every housing unit. Data on poverty status, education level, and linguistic isolation in the U.S. and Puerto Rico were obtained at the block group level from the Census Bureau's 2006-2010 ACS.[iii] Data for the Virgin Islands and other island territories (Guam, American Samoa, and the Northern Marianas) were retrieved from similar tables, which are available through the Census' American Fact Finder internet portal, www.factfinder.census.gov. "Minority" means a person, as defined by the U.S. Bureau of Census, who is a: (1) Black American (a person having origins in any of the black racial groups of Africa); (2) Hispanic person (a person of Mexican, Puerto Rican, Cuban, Central or South American, or other Spanish culture or origin, regardless of race); (3) Asian American or Pacific Islander (a person having origins in any of the original peoples of the Far East, Southeast Asia, the Indian subcontinent, or the Pacific Islands); or (4) American Indian or Alaskan Native (a person having origins in any of the original people of North America and maintain cultural identification through tribal affiliation or community recognition). The Minority/Non-White Hispanic field is computed by subtracting the white population from the total population. Compared to white populations in the U.S., numerous studies have found that minority populations live in closer proximity to pollution sources, experience worse health, and have less ability to participate in environmental decision making. [iv]

The percentage of people living below the national poverty line is defined by the Census as the percentage of residents whose household income is at or below the poverty guidelines updated periodically in the Federal Register by the U.S. Department of Health and Human Services under the authority of 42 U.S.C. 9902(2).[v] Low income has been linked in many studies to poor health, lack of access to health care, closer proximity to pollution sources, and greater susceptibility to illnesses caused by exposure to air pollution. [vi] Low income in this appendix is defined as 2 times the national poverty line.

The percentage of residents whose age is less than 5 years includes infants and children – all of whom are considered a sensitive subpopulation for many forms of environmental contaminants, including air pollution.[vii] The reasons for children's increased sensitivity to air pollution are manifold and include: still-developing respiratory and other bodily systems; smaller body size in proportion to inhaled contaminants; and varying behavior patterns including longer durations spent outdoors at high breathing rates compared to adults.[viii] These factors lead to increased morbidity and mortality risks for children from exposure to air pollutants.[ix] Furthermore, minority and low-income children are at even greater risk, as the increased susceptibility for each of these demographic groups compounds such a child's overall vulnerability.[x]

The percentage of residents of age 65 years and over is considered a sensitive subpopulation for many forms of environmental contamination, including air pollution.[xi] The increased susceptibility of this subpopulation stems not only from their age, but also from their poorer health and lower fitness levels.[xii] Those age 65 years and up have a higher mortality risk—both long-term and short-term—than other populations from air pollution, as well as increased morbidity risks including cardiovascular illness and respiratory disease.[xiii]

The percentage of residents lacking a high school diploma is considered a relevant because lack of education may indirectly increase the effects of environmental contamination, including air pollution. Low education may decrease access to health care and information about environmental risks and how to respond appropriately to such risks.[xiv] Studies have revealed that low education appears to be a factor in health disparities, multi-morbidity, and mortality in general; and lower education may increase the relative risk of air pollution and premature mortality.[xv] In particular, this association was established after examining the relationship between an increase in particulate matter and mortality among persons with lower education.[xvi]

The percentage of residents experiencing linguistic isolation is considered a relevant because linguistic isolation may render households less able to identify and mitigate environmental harms by limiting both access to health care and access to information about environmental risks and how to respond appropriately to those risks.[xvii] Studies have revealed that counties with higher concentrations of immigrants and linguistically isolated households have more hazardous waste generators and more proposed Superfund sites, compared to other counties.[xviii] For example, a significant cancer risk was found between linguistically isolated households and exposure from Toxics Release Inventory (TRI) releases in the San Francisco Bay area.[xix] Finally, immigrants may encounter discrimination and prejudice which impact vulnerability.

9A.2 Considerations in Evaluating and Interpreting Results

This analysis characterizes the socio-demographic attributes of populations located in areas defined by a county or a CBSA containing a county with a monitor design value greater than 65 ppb and 70 ppb. Therefore, the results of this analysis can only be used to inform whether there are differences in the composition of populations residing within these areas relative to the nation as a whole. As noted earlier, the purpose of the analysis is to determine whether populations of interest from an EJ perspective have a higher representation in areas that exceed the proposed range of ozone standard levels, and thus may be more affected by strategies to attain alternative standards. This analysis does not include a quantitative assessment of exposure and/or risk for specific populations of potential interest from an EJ-perspective, and therefore it cannot be used to draw any conclusions regarding potential disparities in exposure or risk across populations of interest from an EJ perspective. Nor can it be used to draw conclusions about any disparities in the health and environmental benefits that result from strategies to attain alternative ozone standards.

In order to clearly identify disparities in risk between populations of interest, we would need to conduct rigorous site-specific population-level exposure and risk assessments that take into account short-term mobility (daily patterns of travel linked for example to school or work) or long-term mobility (families moving into or out of specific block groups).

9A.3 Presentation of Results

This section presents a summary of the results for the assessment of demographic characteristics of populations in areas with ozone monitors with measured values greater than the levels of the proposed range of ozone standards. The results are also provided in tabular form.

As a whole, the demographic distributions within the areas of interest estimated for the proposed range of standard levels (i.e., 65 ppb to 70 ppb) correspond well to the national averages. Table 9A-2 presents these results and the raw data used in this assessment. The population totals and subtotals by demographic group as well as the percentages of the demographic groups for the nation and for the lower and upper end of the proposed range of standard levels (65 ppb and 70 ppb) are shown. Most of the sub-populations are within a few percentage points of the national average. The largest difference is between the national and study area percentages for the Minority/Non-White Hispanic demographic group and that is a difference of only 4%. Overall, these qualitative results support the determination that the proposed rule will tend to benefit geographic areas that have a higher proportion of minority and low income residents than the national average. Perhaps more importantly, these results provide EPA much needed insight into how and with whom to best target efforts to inform communities about the proposed rule and otherwise ensure their meaningful involvement in the rulemaking process.

Table 9A-2 Summary of Population Totals and Demographic Categories for Areas of Interest and National Perspective

Demographic Summary	Population	White	African American	Native American	Other or Multiracial	Minority/ Non-White Hispanic[a]
Area of Interest Total						
65 ppb	221,431,286	153,706,027	30,429,108	1,726,110	35,570,041	67,725,259
70 ppb	193,316,836	132,112,738	27,193,155	1,488,364	32,522,579	61,204,098
% of Area of Interest Total						
65 ppb		69%	14%	1%	16%	31%
70 ppb		68%	14%	1%	17%	32%
National Total	312,861,256	226,405,205	39,475,216	2,952,087	44,028,748	86,456,051
% of National Total		72%	13%	1%	14%	28%

[a] The race *Minority/Non-White Hispanic* field is computed by subtracting the white population from the total population.

Demographic Summary	Linguistically Isolated	Age 0 - 4	Age 0 - 17	Age 65+	Without a HS Diploma	Low Income
Area of Interest Total						
65 ppb	13,072,109	14,695,948	54,008,810	27,163,990	20,914,891	67,027,700
70 ppb	12,179,896	12,849,637	47,296,147	23,518,071	18,495,474	58,296,224
% of Area of Interest Total						
65 ppb	6%	7%	24%	12%	14%	30%
70 ppb	6%	7%	24%	12%	15%	30%
National Total	19,196,507	20,465,065	75,217,176	40,830,262	30,952,789	101,429,436
% of National Total	6%	7%	24%	13%	15%	32%

[i] 2010 Census Summary File 1 Delivered via FTP, http://www2.census.gov/census_2010/04-Summary_File_1/
[ii] U.S. Census Bureau 2006-2010 American Community Survey 5-Year Estimates, http://www.census.gov/acs/www/data_documentation/2010_release/
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CHAPTER 10: QUALITATIVE DISCUSSION OF EMPLOYMENT IMPACTS OF AIR QUALITY

Overview

Executive Order 13563 directs federal agencies to consider regulatory impacts on job creation and employment: "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". Although benefit-cost analyses do not typically include a separate analysis of regulation-induced employment impacts,[144] during periods of sustained high unemployment, such impacts are of particular concern and questions may arise about their existence and magnitude. This chapter discusses some, but not all, possible types of labor impacts that may result from measures to decrease NOx emissions.

Section 10.1 describes the theoretical framework used to analyze regulation-induced employment impacts, discussing how economic theory alone cannot predict whether such impacts are positive or negative. Section 10.2 presents an overview of the peer-reviewed literature relevant to evaluating the effect of environmental regulation on employment. Section 10.3 discusses employment related to installation of NOx controls on coal and gas-fired electric generating units, industrial boilers, and cement kilns.

10.1 Economic Theory and Employment

Regulatory employment impacts are difficult to disentangle from other economic changes affecting employment decisions over time and across regions and industries. Labor market responses to regulation are complex. They depend on labor demand and supply elasticities and possible labor market imperfections (e.g., wage stickiness, long-term unemployment, etc). The unit of measurement (e.g., number of jobs, types of job hours worked, and earnings) may affect observability of that response. Net employment impacts are composed of a mix of potential declines and gains in different areas of the economy (the directly regulated sector, upstream and

[144] Labor expenses do, however, contribute toward total costs in the EPA's standard benefit-cost analyses.

downstream sectors, etc.) over time. In light of these difficulties, economic theory provides a constructive framework for analysis.

Microeconomic theory describes how firms adjust input use in response to changes in economic conditions.[145] Labor is one of many inputs to production, along with capital, energy, and materials. In competitive markets, firms choose inputs and outputs to maximize profit as a function of market prices and technological constraints.[146,147]

Berman and Bui (2001) and Morgenstern, Pizer, and Shih (2002), adapt this model to analyze how environmental regulations affect labor demand.[148] They model environmental regulation as effectively requiring certain factors of production, such as pollution abatement capital, at levels that firms would not otherwise choose.

Berman and Bui (2001, pp. 274-75) model two components that drive changes in firm-level labor demand: output effects and substitution effects.[149] Regulation affects the profit-maximizing quantity of output by changing the marginal cost of production. If regulation causes marginal cost to increase, it will place upward pressure on output prices, leading to a decrease in demand, and resulting in a decrease in production. The output effect describes how, holding labor intensity constant, a decrease in production causes a decrease in labor demand. As noted by Berman and Bui, although many assume that regulation increases marginal cost, it need not be the case. A regulation could induce a firm to upgrade to less polluting and more efficient equipment that lowers marginal production costs. In such a case, output could increase for facilities that do not exit the industry. For example, improving the heat rate of a utility boiler increases fuel efficiency, lowering marginal production costs, and thereby potentially increasing

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[145] See Layard and Walters (1978), a standard microeconomic theory textbook, for a discussion, in Chapter 9.

[146] See Hamermesh (1993), Ch. 2, for a derivation of the firm's labor demand function from cost-minimization.

[147] In this framework, labor demand is a function of quantity of output and prices (of both outputs and inputs).

[148] Berman and Bui (2001) and Morgenstern, Pizer, and Shih (2002) use a cost-minimization framework, which is a special case of profit-maximization with fixed output quantities.

[149] The authors also discuss a third component, the impact of regulation on factor prices, but conclude that this effect is unlikely to be important for large competitive factor markets, such as labor and capital. Morgenstern, Pizer and Shih (2002) use a very similar model, but they break the employment effect into three parts: 1) a demand effect; 2) a cost effect; and 3) a factor-shift effect.

the boiler's generation. An unregulated profit-maximizing firm may not have chosen to install such an efficiency-improving technology if the investment cost were too high.

The substitution effect describes how, holding output constant, regulation affects labor-intensity of production. Although stricter environmental regulation may increase use of pollution control equipment and energy to operate that equipment, the impact on labor demand is ambiguous. Equipment inspection requirements, specialized waste handling, or pollution technologies that alter the production process may affect the number of workers necessary to produce a unit of output. Berman and Bui (2001) model the substitution effect as the effect of regulation on pollution control equipment and expenditures required by the regulation and the corresponding change in labor-intensity of production.

In summary, as output and substitution effects may be positive or negative, theory cannot predict the direction of the net effect of regulation on labor demand at the level of the regulated firm. Operating within the bounds of standard economic theory, however, empirical estimation of net employment effects on regulated firms is possible when data and methods of sufficient detail and quality are available. The literature, however, illustrates difficulties with empirical estimation. For example, studies sometimes rely on confidential plant-level employment data from the U.S. Census Bureau, possibly combined with pollution abatement expenditure data that are too dated to be reliably informative. In addition, the most commonly used empirical methods do not permit estimation of net national effects.

The conceptual framework described thus far focused on regulatory effects on plant-level decisions within a regulated industry. Employment impacts at an individual plant do not necessarily represent impacts for the sector as a whole. The approach must be modified when applied at the industry level.

At the industry-level, labor demand is more responsive if: (1) the price elasticity of demand for the product is high, (2) other factors of production can be easily substituted for labor, (3) the supply of other factors is highly elastic, or (4) labor costs are a large share of total production costs.[150] For example, if all firms in an industry are faced with the same regulatory compliance costs and product demand is inelastic, then industry output may not change much, and output of individual firms may change slightly.[151] In this case the output effect may be small, while the substitution effect depends on input substitutability. Suppose, for example, that new equipment for heat rate improvements requires labor to install and operate. In this case the substitution effect may be positive, and with a small output effect, the total effect may be positive. As with potential effects for an individual firm, theory cannot determine the sign or magnitude of industry-level regulatory effects on labor demand. Determining these signs and magnitudes requires additional sector-specific empirical study. For environmental rules, much of the data needed for these empirical studies are not publicly available, would require significant time and resources in order to access confidential U.S. Census data for research, and also would not be necessary for other components of a typical RIA.

In addition to changes to labor demand in the regulated industry, net employment impacts encompass changes in other related sectors. For example, the proposed guidelines may increase demand for pollution control equipment and services. This increased demand may increase revenue and employment in the firms supporting this technology. At the same time, the regulated industry is purchasing the equipment and these costs may impact labor demand at regulated firms. Therefore, it is important to consider the net effect of compliance actions on employment across multiple sectors or industries.

If the U.S. economy is at full employment, even a large-scale environmental regulation is unlikely to have a noticeable impact on aggregate net national employment.[152] Instead, labor would primarily be reallocated from one productive use to another (e.g., from producing electricity or steel to producing high efficiency equipment), and net national employment effects

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[150] See Ehrenberg & Smith, p. 108.

[151] This discussion draws from Berman and Bui (2001), pp. 293.

[152] Full employment is a conceptual target for the economy where everyone who wants to work and is available to do so at prevailing wages is actively employed. The unemployment rate at full employment is not zero.

from environmental regulation would be small and transitory (e.g., as workers move from one job to another).[153]

Affected sectors may experience transitory effects as workers change jobs. Some workers may retrain or relocate in anticipation of new requirements or require time to search for new jobs, while shortages in some sectors or regions could bid up wages to attract workers. These adjustment costs can lead to local labor disruptions. Although the net change in the national workforce is expected to be small, localized reductions in employment may adversely impact individuals and communities just as localized increases may have positive impacts.

If the economy is operating at less than full employment, economic theory does not clearly indicate the direction or magnitude of the net impact of environmental regulation on employment; it could cause either a short-run net increase or short-run net decrease (Schmalansee and Stavins, 2011). An important research question is how to accommodate unemployment as a structural feature in economic models. This feature may be important in assessing large-scale regulatory impacts on employment (Smith 2012).

Environmental regulation may also affect labor supply. In particular, pollution and other environmental risks may impact labor productivity or employees' ability to work.[154] While the theoretical framework for analyzing labor supply effects is analogous to that for labor demand, it is more difficult to study empirically. There is a small emerging literature, described in the next section that uses detailed labor and environmental data to assess these impacts.

To summarize, economic theory provides a framework for analyzing the impacts of environmental regulation on employment. The net employment effect incorporates expected employment changes (both positive and negative) in the regulated sector and elsewhere. Labor demand impacts for regulated firms, and also for the regulated industry, can be decomposed into output and substitution effects which may be either negative or positive. Estimation of net employment effects for regulated sectors is possible when data of sufficient detail and quality are

[153] Arrow et. al. 1996; see discussion on bottom of p. 8. In practice, distributional impacts on individual workers can be important, as discussed in later paragraphs of this section.

[154] E.g. Graff Zivin and Neidell (2012).

available. Finally, economic theory suggests that labor supply effects are also possible. In the next section, we discuss the empirical literature.

10.2 Current State of Knowledge Based on the Peer-Reviewed Literature

The labor economics literature contains an extensive body of peer-reviewed empirical work analyzing various aspects of labor demand, relying on the theoretical framework discussed in the preceding section.[155] This work focuses primarily on effects of employment policies such as labor taxes and minimum wages.[156] In contrast, the peer-reviewed empirical literature specifically estimating employment effects of environmental regulations is more limited.

Empirical studies, such as Berman and Bui (2001), suggest that net employment impacts were not statistically different from zero in the regulated sector. Other research suggests that more highly regulated counties may generate fewer jobs than less regulated ones (Greenstone 2002). Environmental regulations may affect sectors that support pollution reduction earlier than the regulated industry. Rules are usually announced well in advance of their effective dates and then typically provide a period of time for firms to invest in technologies and process changes to meet the new requirements. When a regulation is promulgated, the initial response of firms is often to order pollution control equipment and services to enable compliance when the regulation becomes effective. Estimates of short-term increases in demand for specialized labor within the environmental protection sector have been prepared for several EPA regulations in the past, including the Mercury and Air Toxics Standards (MATS).[157] Overall, the peer-reviewed literature does not contain evidence that environmental regulation has a large impact on net employment (either negative or positive) in the long run across the whole economy.

10.2.1 Regulated Sectors

Berman and Bui (2001) examine how an increase in local air quality regulation affects manufacturing employment in the South Coast Air Quality Management District (SCAQMD), which includes Los Angeles and its suburbs. From 1979 to 1992 the SCAQMD enacted some of

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155 Again, see Hamermesh (1993) for a detailed treatment.

156 See Ehrenberg & Smith (2000), Chapter 4: "Employment Effects: Empirical Estimates" for a concise overview.

157 U.S. EPA (2011b).

the country's most stringent air quality regulations. Using SCAQMD's local air quality regulations, Berman and Bui identify the effect of environmental regulations on net employment in regulated manufacturing industries relative to other plants in the same 4-digit SIC industries but in regions not subject to local regulations.[158] The authors find that "while regulations do impose large costs, they have a limited effect on employment" (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, even when exit and dissuaded entry effects are included" (Berman and Bui, 2001, p. 269).[159]

A small literature examines impacts of environmental regulations on manufacturing employment. Kahn and Mansur (2013) study environmental regulatory impacts on geographic distribution of manufacturing employment, controlling for electricity prices and labor regulation (right to work laws). Their methodology identifies employment impacts by focusing on neighboring counties with different ozone regulations. They find limited evidence that environmental regulations may cause employment to be lower within "county-border-pairs." This result suggests that regulation may cause an effective relocation of labor across a county border, but since one county's loss may be another's gain, such shifts cannot be transformed into an estimate of a national net effect on employment. Moreover this result is sensitive to model specification choices.

10.2.2 Labor Supply Impacts

The empirical literature on environmental regulatory employment impacts focuses primarily on labor demand. However, there is a nascent literature focusing on regulation-induced effects on labor supply.[160] Although this literature is limited by empirical challenges, researchers have found that air quality improvements lead to reductions in lost work days (e.g., Ostro 1987). Limited evidence suggests worker productivity may also improve when pollution is reduced.

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[158] Berman and Bui include over 40 4-digit SIC industries in their sample. They do not estimate the number of jobs created in the environmental protection sector.

[159] Including the employment effect of existing plants and plants dissuaded from opening will increase the estimated impact of regulation on employment.

[160] For a recent review see Graff-Zivin and Neidell (2013).

Graff Zivin and Neidell (2012) used detailed worker-level productivity data from 2009 and 2010, paired with local ozone air quality monitoring data for one large California farm growing multiple crops, with a piece-rate payment structure. Their quasi-experimental structure identifies an effect of daily variation in monitored ozone levels on productivity. They find "ozone levels well below federal air quality standards have a significant impact on productivity: a 10 parts per billion (ppb) decreases in ozone concentrations increases worker productivity by 5.5 percent." (Graff Zivin and Neidell, 2012, p. 3654).[161]

This section has outlined the challenges associated with estimating regulatory effects on both labor demand and supply for specific sectors. These challenges make it difficult to estimate net national employment estimates that would appropriately capture the way in which costs, compliance spending, and environmental benefits propagate through the economy. Quantitative estimates are further complicated by the fact that macroeconomic models often have little sectoral detail and usually assume that the economy is at full employment. The EPA is currently seeking input from an independent expert panel on modeling economy-wide regulatory impacts, including employment effects.[162]

10.3 Employment Related to Installation and Maintenance of NOx Control Equipment

This section discusses employment related to installation of NOx controls on coal and gas-fired electric generating units, industrial boilers, and cement kilns, which are among the highest NOx-emitting source categories in EPA's emissions inventory (see chapter 3 for more detail on emissions). Sections 10.3.1 and 10.3.2 below contain estimates of the number of direct short-term and long-term jobs that would be created by addition of NOx controls at these three categories of emissions sources, for various size units. Because the apportionment of emissions control across emissions sources in this RIA analysis is illustrative and not necessarily representative of the controls that will be required in individual state SIPs, EPA did not estimate

[161] The EPA is not quantifying productivity impacts of reduced pollution in this rulemaking using this study. In light of this recent research, however, the EPA is considering how best to incorporate possible productivity effects in the future.

[162] For further information see: <https://www.federalregister.gov/articles/2014/02/05/2014-02471/draft-supporting-materials-for-the-science-advisory-board-panel-on-the-role-of-economy-wide-modeling>.

short-term or long-term employment that would result from addition of NOx controls at these three source categories at the national level.

10.3.1 Employment Resulting from Addition of NOx Controls at EGUs

Coal-fired EGUs are likely to apply additional NOx controls in response to State Implementation Plans (SIPs) approved pursuant to a revised ozone standard. While many EGUs have already installed and operate various NOx control devices, there are additional existing coal-fired EGUs that could decrease NOx emissions by installing or upgrading their NOx reducing systems. While all existing coal-fired EGUs already have low NOx burners, there are EGUs that could have a selective catalytic reduction (SCR) system installed, or could improve their NOx emissions by replacing an existing selective non-catalytic reduction (SNCR) system with an SCR system. The EPA identified 145 existing coal-fired EGUs, with a total of 51.0 GW of capacity, that (1) are in areas anticipated to need additional NOx reductions under an alternative ozone standard of 65 ppb , and (b) do not already have an SCR emission control system. (For an alternative ozone standard of 70 ppb, there are 15 EGUs so identified, with a total of 7.4 GW of capacity.) While there are currently SNCR systems in use that could be upgraded to an SCR system, the EPA's 2025 baseline analysis[163] estimates that the remaining SNCR systems will already be upgraded by 2025 in response to existing emission control programs.

The EPA used a bottom up engineering analysis using data on labor productivity, engineering estimates of the types of labor needed to manufacture, construct and operate SCRs on EGUs. The EPA's labor estimates include not only labor directly involved with installing SCRs on EGUs and on-site labor used to operate the SCRs once they become operational, but also include the labor requirements in selected major upstream sectors directly involved manufacturing the materials used in SCR systems (steel), as well as the chemicals used to operate an SCR system (ammonia and the catalyst used to in the construction and operation of SCR systems, including such as steel, concrete, or chemicals used to manufacture NOx controls.

[163] The 2025 baseline used in this illustrative analysis incorporates the "state only" implementation option used in the proposed carbon pollution guidelines for existing power plants and emission standards for modified and reconstructed power plants (a.k.a. the proposed Clean Power Plan, June, 2013).

This section presents an illustrative analysis of the direct labor needs to install and operate SCRs at 3 common sizes of coal-fired EGUs: 300 MW, 500 MW and 1000 MW. As discussed below, the illustrative analysis is for a "model plant" of each size, using consistent assumptions about the plant's operation that impact the material and labor needs of an representative plant such as the capacity factor, heat rate, and type of coal. The analysis does not include an estimate of the aggregate total of the labor needed for installing and running SCRs in any particular level of the revised ozone standard, nor does it reflect plant-specific variations in labor needs due to regional differences in prices and labor availability, existing control technology at the plant, etc.

The analysis draws on information from four primary sources:

- Documentation for EPA Base Case v.5.13 Using the Integrated Planning Model. November, 2013
- "ENGINEERING AND ECONOMIC FACTORS AFFECTING THE INSTALLATION OF CONTROL TECHNOLOGIES: An Update". By James E. Staudt, Andover Technology Partners. December, 2011.
- "Regulatory Impact Analysis (RIA) for the final Transport Rule". June 2011
- "Regulatory Impact Analysis for the Proposed Carbon Pollution Guidelines for Existing Power Plants and Emission Standards for Modified and Reconstructed Power Plants". June 2013

10.3.1.1 Existing EGUs Without SCR Systems

The EPA identified 145 existing coal-fired EGU units that are estimated to continue to be in operation in 2025 in the baseline that are located in areas considered likely to be affected by State Implementation Plans developed for a 65 ppb alternative ozone standard. The size distribution of the 145 units is shown in Figure 10-1. The 145 units have a total generating capacity of 51.0 GW and are anticipated to generate 282,000 GWh of electricity in 2025. With the current level of NOx controls installed (or anticipated to be installed by 2025 to meet existing environmental regulations), these 145 units are estimated to emit 290.7 tons of NOx in 2025.

The following key assumptions are used to estimate the amount of labor needed to install and operate individual SCR systems of various sizes.

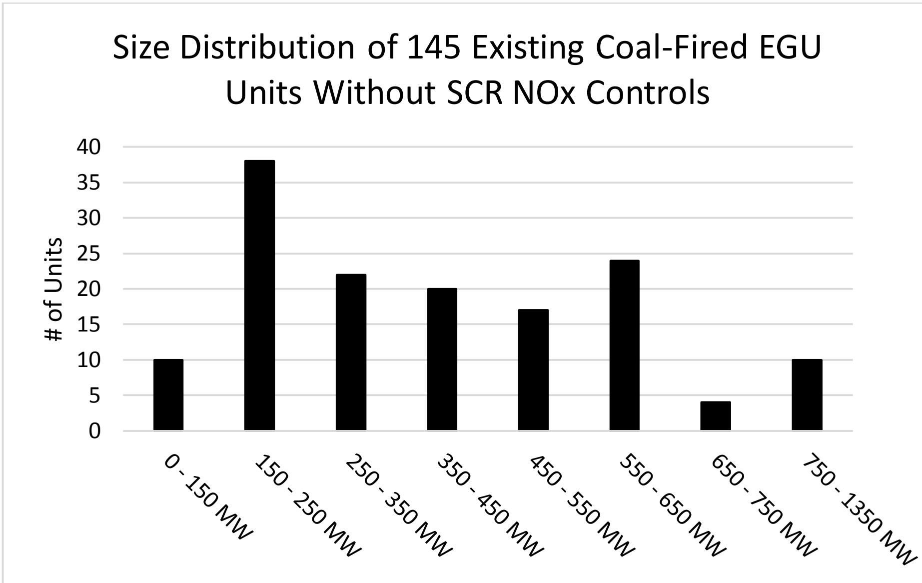

Figure 10-1. Size Distribution of 145 Existing Coal-Fired EGU Units without SCR NOx Controls

10.3.1.2 Labor Estimates for Installing and Operating Individual SCR Systems

All labor estimates in this illustrative analysis are in terms of person-years (i.e., full time equivalents, or FTEs).

The labor involved with manufacturing and installing the SCRs is a one-time labor need, and occurs over a 2 to 3 year construction period; the estimated FTEs during the construction phase are presented as the cumulative amount of labor over the multi-year period. The construction phase labor includes both labor directly involved with installing the SCR on site (including boiler makers, general labor and engineering).

There are three types of annual labor estimated to operate an SCR, and will be needed each year the EGU is in operation. The largest category is on-site labor at the EGU. The estimated amounts of direct labor involved with installing SCR systems is shown in Table 10-1.

Table 10-1. Summary of Direct Labor Impacts for SCR Installation at EGUs

	Plant Size		
	300 MW	**500 MW**	**1000 MW**
Construction Phase (One time, Total Labor over 2-3 Year Period)			
Direct Construction-related Employment	158.7	264.4	528.8
Operation Phase (Annual Operations)			

	Plant Size		
Operation and Maintenance	1.9	2.8	4.6

The key assumptions used in the labor analysis are presented in Table 10-2.

Table 10-2. Key Assumptions in Labor Analysis for EGUs

Assumptions	Key Factor	Source	300 MW	500 MW	1000 MW
Capital Investment to Install SCR	Utility-owned Capital Recovery Rate for Environmental Retrofits (12.1%)	IPM 5/13 Base Case Documentation	$86.1 million	$133 million	$244 million
Result: FTEs to Install an SCR	**1,100 hours/MW**	**Staudt, 2011**	**158.65**	**264.42**	**528.85**
Labor Cost (fixed O&M) per Year		IPM analysis of CPP baseline	$218,000	$310,500	$513,000
Result: FTEs per Year	8.9 FTEs per $1 million of Fixed O&M	CSAPR RIA	1.95	2.76	4.57
Result: Total FTES to Operate an SCR Annually			**1.95**	**2.76**	**4.57**

10.3.2 Assessment of Employment Impacts for Individual Industrial, Commercial, and Institutional (ICI) Boilers and Cement Kilns

Facilities other than electric power generators are likely to apply NOx controls in response to State Implementation Plans (SIPs) approved pursuant to a revised ozone standard. In addition to EGUs, the EPA estimated the amount and types of direct labor that might be used to apply and operate NOX controls for ICI boilers and for cement kilns. As with EGUs, the EPA used a bottom up engineering analysis using data on labor productivity, engineering estimates of the types of labor needed to manufacture, construct and operate NOx controls on ICI boilers and cement kilns. No estimates were made for labor requirements in upstream sectors such as steel, concrete, or chemicals used to manufacture or search as inputs to NOx controls. In addition, the numbers presented in this section are only indicative of the relative number and types of labor that might be used at these two categories of plants, without calculating an estimate of the labor that would be required by them in the aggregate (SC&A, 2014).

10.3.2.1 ICI Boilers

There are a number of control technologies available to reduce NOx emissions from ICI boilers. The EPA anticipates that the most commonly applied control technology for ICI boilers that could require NOx reductions as part of an ozone SIP will be selective catalytic reduction (SCR). The analysis calculates s labor requirements to fabricate, install, and operate different sizes of SCR for coal, oil and natural gas ICI boilers. Estimated total labor costs are a function of total capital costs and boiler size in EPA's Coal Utility Environmental Cost (CUECost) model. Total SCR capital costs of ICI boilers was estimated using the EPA's Control Strategy tool (CoST) model. Labor is estimated to be about 50% of the total capital costs of an SCR. (SC&A, 2014).

Just over 24% of total capital costs are for labor used in SCR fabrication. This percentage was multiplied by the total capital cost, and the resulting dollar amount was converted into full time equivalents (FTE) based on the average annual salary of workers (as outlined in IEC, 2011). The annual compensation came from the Bureau of Labor Statistics (BLS). This salary number was adjusted to account for benefits also based on BLS data. The total fabrication expenditures were divided by the average fabrication labor compensation to estimate the number of full time equivalent workers in SCR fabrication.

The calculation of construction or installation labor is based on previous research on labor required for SCR installation at utility boilers. (Staudt 2011). Based on that, we estimate that 27% of SCR capital costs are spent on installation labor. We applied that percentage to the estimates of the capital costs of SCR for ICI boilers to give us the total labor expenditures, which we then converted to FTE based on average annual compensation provided by BLS.

Operation and Maintenance labor was estimated using the CUECost model. Maintenance and administrative labor for SCR is estimated to be small in relation to fabrication and construction, with the caveat that available information on which to base an estimate is sparse. According to the approach used in the CUECost model, most utility boilers require a full time worker to operate and maintain the equipment. ICI boilers are much smaller and so are likely to require less than one FTE. Table 10-3 below provides summary labor estimates for SCR at varying sized ICI boilers.

Table 10-3. Summary of Direct Labor Impacts for Individual ICI Boilers

Plant Type	Boiler Size (MMBtu/hr)	One-Time Employment Impacts[1] (Annual FTEs)	Recurring Annual Employment Impacts[2] (FTEs per year)
Coal-fired	750	19.5	1.2
	500	15.2	1.1
	400	13.6	1.0
	250	10.7	0.9
Oil-fired	250	9.8	0.9
	150	7.3	0.9
	100	5.5	0.8
	50	3.2	0.8
Natural Gas-fired	250	10.5	0.9
	150	11.0	0.9
	100	8.4	0.9
	50	6.5	0.8

1. Includes Fabrication and Installation Labor
2. Includes Operations, Maintenance, and Administrative Support

10.3.2.2 Cement Kilns

There are a number of technologies that can be used to control NOx emissions at cement kilns. The analysis focused on synthetic non-catalytic reduction (SNCR) as the most likely choice for future NOx controls at cement kilns affected by requirements in ozone SIPs. Although SNCR is not considered an appropriate technology for wet and long dry kilns, most new or recently constructed kilns will likely be preheater and precalciner kilns, and these kilns will likely operate using SNCR as a control technology.

Fabrication capital cost was estimated for an SNCR system for a mid-sized preheater and precalciner kiln (125 to 208 tons of clinker per hour). The percent of capital cost of these systems attributable to labor is 44% based on vendor supplied estimates. (Wojichowski, 2014). This labor cost was converted to FTE using BLS data. A similar methodology was used to estimate installation labor. Labor costs for SNCR installation was estimated by the vendor to be 17% of the capital cost. That was converted to FTE using BLS data. This information is summarized in Table 10-4.

Table 10-4. Estimated Direct Labor Impacts for Individual SNCR Applied to a Mid-Sized Cement Kiln (125-208 tons clinker/hr)

Kiln Type	Preheater / Precalciner
Manufacturing FTE	1.5
Installation FTE	0.9
O&M Annual Recurring FTE	13

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