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

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Overview

In setting primary and secondary national ambient air quality standards (NAAQS), the EPA's responsibility under the law is to establish standards that protect public health and welfare. 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" and public welfare from "any known or anticipated adverse effects." As interpreted by the Agency and the courts, the Act requires the EPA to base the decision for the primary standard 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 Administrator concluded that the current primary standard for ozone does not provide requisite protection to public health with an adequate margin of safety, and that it should be revised to provide increased public health protection. Specifically, the EPA is retaining the indicator (ozone), averaging time (8-hour) and form (annual fourth-highest daily maximum, averaged over 3 years) of the existing primary standard and is revising the level of that standard to 70 ppb. The EPA has also concluded that the current secondary standard for ozone, set at a level of 75 ppb, is not requisite to protect public welfare from known or anticipated adverse effects, and is revising the standard to provide increased protection against vegetation-related effects on public welfare. Specifically, the EPA is retaining the indicator (ozone), averaging time (8-hour) and form (annual fourth-highest daily maximum, averaged over 3 years) of the existing secondary standard and is revising the level of that standard to 70 ppb.[1]

[1] The EPA has concluded that this revision will effectively curtail cumulative seasonal ozone exposures above 17 ppm-hrs in terms of a three-year average seasonal W126 index value, based on the three consecutive month period within the growing season with the maximum index value, with daily exposures cumulated for the 12-hour period from 8:00 am to 8:00 pm.

The EPA performed an illustrative analysis of the potential costs, human health benefits, and welfare benefits of nationally attaining a revised primary ozone standard of 70 ppb and a primary alternative ozone standard level of 65 ppb. Because there are not additional costs and benefits of attaining the secondary standard, the EPA did not need to estimate any incremental costs and benefits associated with attaining a revised secondary standard. Per Executive Orders 12866 and 13563 and the guidelines of OMB Circular A-4, this Regulatory Impact Analysis (RIA) presents the analyses of the revised standard level of 70 ppb and an alternative standard level of 65 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 NAAQS (75 ppb). The 2025 baseline reflects, among other existing regulations, the 2017 and Later Model Year Light-Duty Vehicle Greenhouse Gas Emissions and Corporate Average Fuel Economy Standards, Greenhouse Gas Emissions Standards and Fuel Efficiency Standards for Medium- and Heavy-Duty Engines and Vehicles, the Tier 3 Motor Vehicle Emission and Fuel Standards, the Clean Power Plan, the Mercury and Air Toxics Standards,[2] and the Cross-State Air Pollution Rule, all of which will help many areas move toward attainment of the existing ozone standard (see Appendix 2, Section 2A.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, populations, and premature mortality baseline rates for 2025. We recognize that there are 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. Because of data and resource constraints, we were not able to project emissions and air quality beyond 2025 for California; however, we adjusted baseline air

[2] On June 29, 2015, the United States Supreme Court reversed the D.C. Circuit opinion affirming the Mercury and Air Toxics Standards (MATS). The EPA is reviewing the decision and will determine any appropriate next steps once the review is complete, however, MATS is still currently in effect. The first compliance date was April 2015, and many facilities have installed controls for compliance with MATS. MATS is included in the baseline for this analysis, and the EPA does not believe including MATS substantially alters the results of this analysis.

quality to reflect mobile source emissions reductions for California that would occur between 2025 and 2030.[3] These emissions reductions were the result of mobile source regulations expected to be fully implemented by 2030.

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. Further, Serious nonattainment areas will likely have to attain in late 2026 or early 2027. As such, 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.

While there is uncertainty about the precise timing of emissions reductions and related costs for California, we assume costs associated with the installation of controls occur through the end of 2037 and beginning of 2038. In addition, we estimate benefits for California using projected population demographics and baseline mortality rates 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 estimated using population demographics and baseline mortality rates for 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

[3] At the time of this analysis, there were no future year emissions for California beyond 2030, and projecting emissions beyond 2030 could introduce additional uncertainty.

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 estimates of current and future emissions of relevant precursors (i.e., NOx and VOC) that contribute to the air quality problem and estimates of current and future ozone concentrations (Chapter 2 – Emissions, Air Quality Modeling and Analytic Methodologies); development of illustrative control strategies to attain the revised standard of 70 ppb and an alternative primary standard level of 65 ppb (Chapter 3 – Control Strategies and Emissions Reductions); estimates of the incremental costs of attaining the revised and alternative standard levels (Chapter 4 – Engineering Cost Analysis and Economic Impacts); 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 5 – Qualitative Discussion of Employment Impacts of Air Quality); estimates of the incremental benefits of attaining the revised and alternative standard levels (Chapter 6 – Human Health Benefits Analysis Approach and Results); a qualitative discussion of the welfare benefits of attaining the revised standards (Chapter 7 – Impacts on Public Welfare of Attainment Strategies to Meeting Primary and Secondary Ozone NAAQS); a comparison and discussion of the benefits and costs (Chapter 8 – Comparison of Costs and Benefits); and an analysis of the impacts in the context of the relevant statutory and executive order requirements (Chapter 9 – Statutory and Executive Order Impact Analyses).

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 potential costs and benefits of the illustrative attainment strategies to meet the revised and alternative standard levels. The flowchart below (Figure ES-1) outlines the analytical steps taken to illustrate attainment with the revised and alternative standard levels, and the following discussion describes each of the major steps in the process.

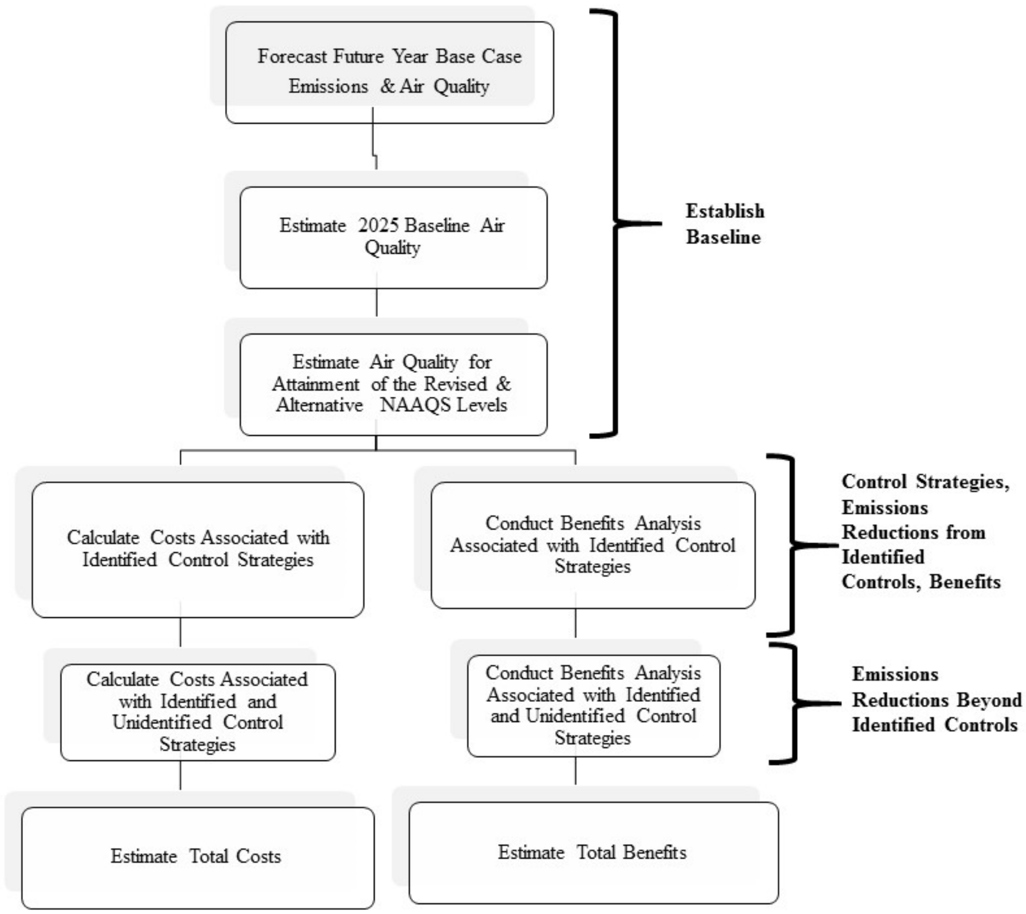

Figure ES-1. Analytical Flowchart for Primary Standards Analyses

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 Appendix 2, Section 2A.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 proposed Clean Power Plan (U.S. EPA, 2014b).[4]

[4] The impact of these forecast changes in NOx emissions between the proposed and final CPP on ozone concentrations in specific locations is uncertain. There is no clear spatial pattern of where emissions are forecast to be higher or lower in the final CPP relative to the proposed CPP. Furthermore, states have flexibility in the form of

We performed a national scale air quality modeling analysis to estimate ozone concentrations for the future base case year of 2025. In addition, we modeled fifteen 2025 emissions sensitivity simulations.[5] The emissions sensitivity simulations were used to develop ozone response factors (ppb/ton) from the modeled response of ozone to changes in NOx and VOC emissions from various sources and locations. These ozone response factors were then used to determine the amount of emissions reductions needed to reach the 2025 baseline and to evaluate the revised and alternative standard levels of 70 and 65 ppb incremental to the baseline. We used the estimated emissions reductions needed to reach the revised and alternative standard levels to analyze the costs and benefits.

ES.1.2 Control Strategies and Emissions Reductions

The EPA used the Control Strategy Tool (CoST) to estimate engineering control costs. We estimated costs for non-electric generating unit point (non-EGU point), nonpoint, and mobile nonroad sources. Some electric generating units (EGUs) run their control equipment part of the year. To estimate the costs for EGUs, we assumed they ran their control equipment all year, and we estimated the costs of additional inputs needed. 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. The 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 a capital recovery factor. Annualized costs represent an equal stream of yearly costs over the period the control technology is expected to operate.

The EPA analyzed illustrative control strategies that areas across the U.S. might employ to attain the revised primary ozone standard level of 70 ppb and an alternative standard level of 65 ppb. The EPA analyzed the impact that additional emissions control technologies and

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their plans that implement the CPP and therefore the specific impact of the CPP on NOx emissions in any state is uncertain.

[5] The approach of using emissions sensitivity simulations to determine the response of ozone at monitor locations to emissions changes in specific regions is similar to the approach used in the November 2014 proposal RIA. However, in the final RIA we conducted sensitivity simulations using ten regions compared to five much larger regions in the proposal RIA.

measures, across numerous sectors, would have on predicted ambient ozone concentrations incremental to the baseline. These control measures, also referred to as identified controls, are based on information available at the time of this analysis and include primarily end-of-pipe control technologies. In addition, to attain the revised and alternative primary standard levels analyzed, some areas needed additional emissions reductions beyond the identified controls, and we refer to these as unidentified controls or measures (see Chapter 3, Sections 3.1 and 3.2 for additional information).[6]

Using the ozone response factors mentioned above, we estimated the emissions reductions over and above the baseline that were needed to meet the revised standard of 70 ppb and an alternative standard level of 65 ppb. Costs of controls incremental to baseline emissions reductions are attributed to the costs of meeting the revised and alternative standard levels. These emissions reductions can come from specific identified controls, as well as unidentified controls in some areas. The baseline shows that by 2025, ozone concentrations would be significantly better than today under current requirements, and depending on the standard level analyzed, some areas in the Eastern, Central, and Western U.S. would need to develop and adopt additional controls to attain the revised and alternative standard levels (see Chapter 3, Section 3.1.3 and Figure 3-5 for additional details on the areas that would need to develop and adopt controls).

ES.1.2.1 Emissions Reductions from Identified Controls in 2025

Figure ES-2 shows the counties projected to exceed the revised and alternative standard levels analyzed for 2025 for areas other than California. For the revised standard of 70 ppb, emissions reductions were required for monitors in the Colorado, Great Lakes, North East, Ohio River Valley and East Texas regions (see Chapter 2, Figure 2-2 for a map of the regions). For the 65 ppb alternative standard level, in addition to the regions listed above, NOx emissions reductions were required in the Arizona-New Mexico, Nevada, and Oklahoma-Arkansas-Louisiana regions. VOC emissions reductions were required in Denver, Houston, Louisville,

[6] In the proposal RIA we discuss emissions reductions resulting from the application of known controls, as well as emissions reductions beyond known controls, using the terminology of “known controls” and “unknown controls.” In the final RIA, we have used slightly different terminology, consistent with past NAAQS RIAs. Here we refer to emissions reductions and controls as either “identified” controls or measures or “unidentified” controls or measures reflecting that unidentified controls or measures can include existing controls or measures for which the EPA does not have sufficient data to accurately estimate their costs.

Chicago and New York City. Tables ES-1 and ES-2 show the emissions reductions from identified controls for the revised and alternative standard levels analyzed. We aggregate results by region – East and West, except California – to present cost and benefits estimates. See Chapter 4, Figure 4.3 for a representation of the East and West regions.

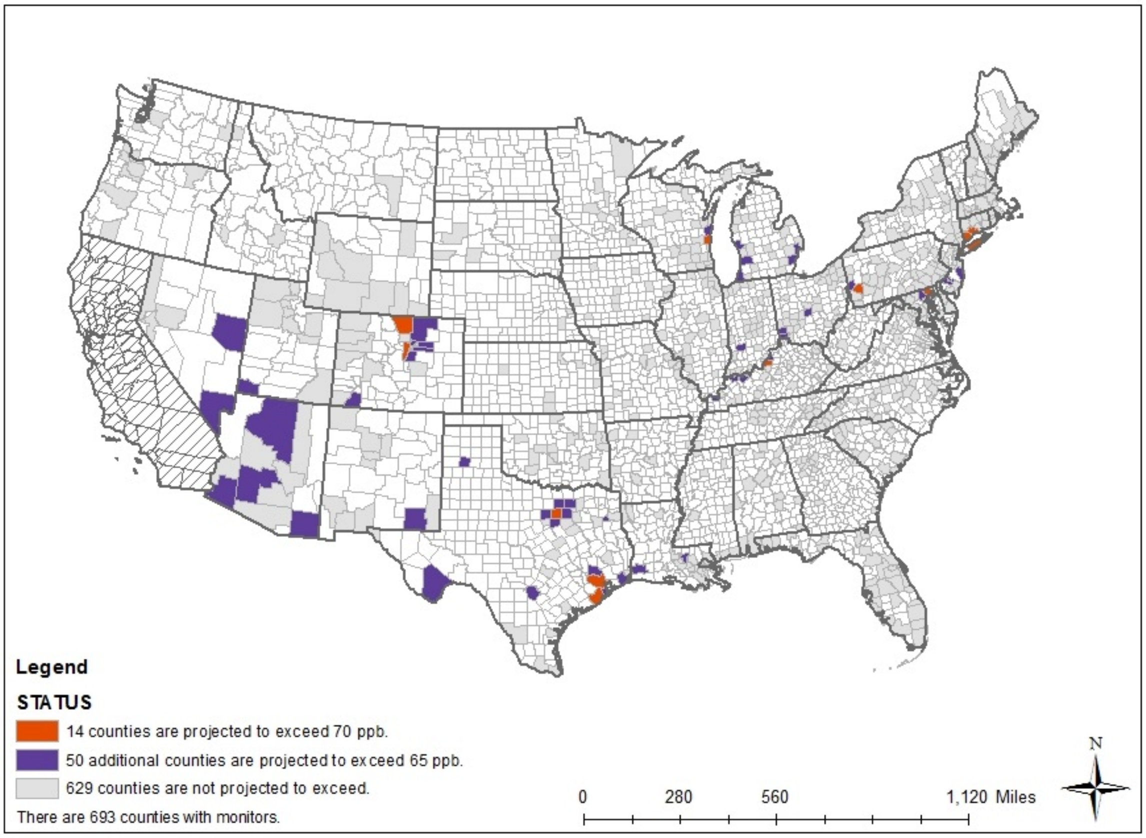

Figure ES-2. Projected Ozone Design Values in the 2025 Baseline Scenario

Table ES-1. Summary of Emissions Reductions by Sector for the Identified Control Strategy for the Revised Standard Level of 70 ppb for 2025, except California (1,000 tons/year)[a]

Geographic Area	Emissions Sector	NO_x	VOC
East	EGU	45	-
	Non-EGU Point	85	1
	Nonpoint	100	19
	Nonroad	3	-
	Onroad	-	-
	Total	**230**	**20**
West	EGU	-	-
	Non-EGU Point	6	-
	Nonpoint	1	-
	Nonroad	-	-
	Onroad	-	-
	Total	**7**	**-**

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

Table ES-2. Summary of Emissions Reductions by Sector for the Identified Control Strategy for Alternative Standard Level of 65 ppb for 2025, except California (1,000 tons/year)[a]

Geographic Area	Emissions Sector	NO_x	VOC
East	EGU	110	-
	Non-EGU Point	220	5
	Nonpoint	160	100
	Nonroad	8	-
	Total	**500**	**100**
West	EGU	0	-
	Non-EGU Point	33	-
	Nonpoint	22	5
	Nonroad	1	-
	Total	**56**	**5**

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

ES.1.2.2 Emissions Reductions beyond Identified Controls in 2025

There were several areas where identified controls did not achieve enough emissions reductions to attain the revised standard level of 70 ppb or alternative standard level of 65 ppb. The EPA then estimated the additional emissions reductions beyond identified controls needed to reach attainment (i.e., unidentified controls). The EPA's application of unidentified control measures does not mean the Agency has concluded that all unidentified control measures are currently not commercially available or do not exist. Unidentified control technologies or measures can include existing controls or measures for which the EPA does not have sufficient data to accurately estimate engineering costs. Likewise, the control measures in the CoST database do not include abatement possibilities from energy efficiency measures, fuel switching, input or process changes, or other abatement strategies that are non-traditional in the sense that they are not the application of an end-of-pipe control. Table ES-3 shows the emissions reductions needed from unidentified controls in 2025 for the U.S., except California, for the revised and alternative standard levels analyzed.

Table ES-3. Summary of Emissions Reductions from the Unidentified Control Strategies for the Revised and Alternative Standard Levels for 2025, except California (1,000 tons/year)[a]

Revised and Alternative Standard Levels	Region	NO_x	VOC
70 ppb	East	47	-
	West	-	-
65 ppb	East	820	-
	West	40	-

[a] Estimates are rounded to two significant figures.

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

Figure ES-3 shows the counties projected to exceed the revised and alternative standard levels analyzed for the post-2025 analysis for California. For the California post-2025 revised and alternative standard level analyses, all identified controls were applied in the baseline, so incremental emissions reductions to demonstrate attainment of the revised and alternative standards were from unidentified controls. Table ES-4 shows the emissions reductions needed from unidentified controls for post-2025 for California for the revised and alternative standard levels analyzed.

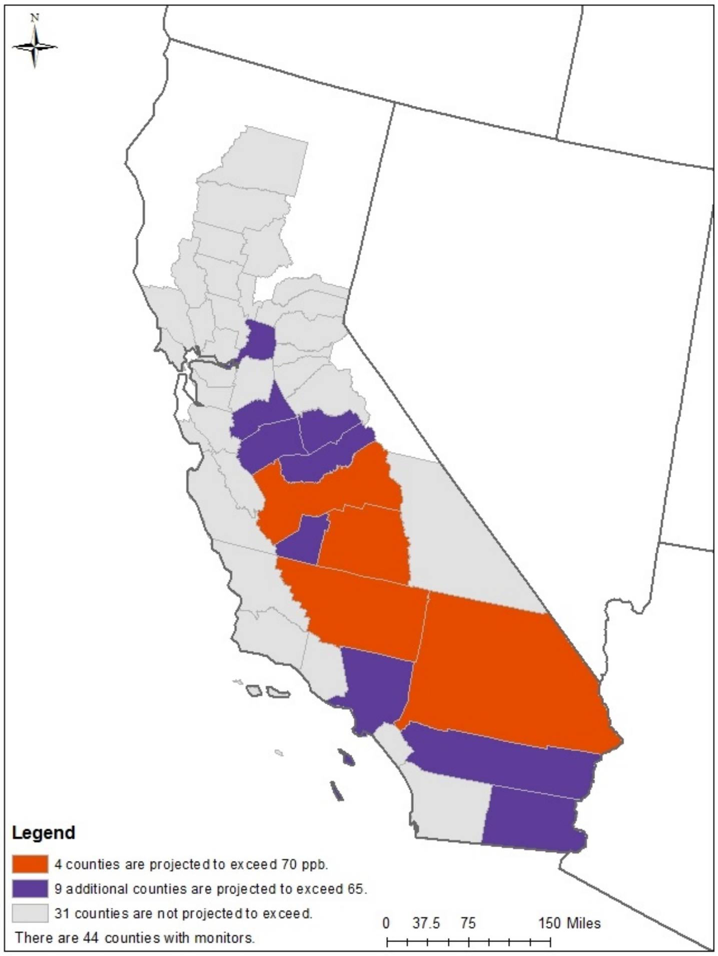

Figure ES-3. Projected Ozone Design Values in the Post-2025 Baseline

Table ES-4. Summary of Emissions Reductions from the Unidentified Control Strategies for the Revised and Alternative Standard Levels for Post-2025 - California (1,000 tons/year)[a]

Revised and Alternative Standard Levels	Region	NO_x	VOC
70 ppb	**CA**	51	-
65 ppb	**CA**	100	-

[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 later assigned dollar values. For this RIA, the health impacts analysis is limited to those health effects that are directly linked to changes in ambient levels of ozone and $PM_{2.5}$ due to reductions in ozone precursor emissions. Emissions reductions of NOx or VOC to attain the ozone standards would simultaneously reduce ambient $PM_{2.5}$ concentrations.

Benefits estimates for ozone were generated using the damage-function approach outlined above wherein changes in ambient ozone concentrations were translated into reductions in the incidence of specific health endpoints (e.g., premature mortality or hospital admissions) using the environmental Benefits Mapping and Analysis Program – Community Edition (BenMAP-CE).

In contrast to ozone, we used a benefit-per-ton approach to estimate $PM_{2.5}$ co-benefits. With this approach, we use the results of previous air quality modeling to derive benefit-per-ton estimates for NOx. These benefit-per-ton estimates provide the monetized human health co-benefits (the sum of premature mortality and premature morbidity) of reducing one ton of a $PM_{2.5}$ precursor (such as NO_X) from a specified source. We then combine these benefit-per-ton estimates with reductions in NOx emissions associated with meeting the revised and alternative standard levels. We acknowledge increased uncertainty associated with the benefit-per-ton approach, relative to using scenario-specific air quality modeling to estimate the $PM_{2.5}$ co-benefits.

In addition to ozone and $PM_{2.5}$ benefits, implementing emissions controls to attain the revised and alternative ozone standard levels would reduce exposure to other ambient pollutants (e.g., NO_2). However, we were not able to quantify the co-benefits of reduced exposure to these pollutants, nor were we able to estimate some anticipated health benefits associated with exposure to ozone and $PM_{2.5}$ due to data and methodology limitations.

ES.1.4 Welfare Benefits of Meeting the Primary and Secondary Standards

Section 302(h) of the Clean Air Act states that effects on welfare include, but are not limited to, "effects on soils, water, crops, vegetation, man-made 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." 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 the RIA for the proposal, we quantified a small portion of the welfare impacts associated with reductions in ozone concentrations to meet the alternative ozone standard levels analyzed. Using a model of commercial agriculture and forest markets, we analyzed the effects on consumers and producers of forest and agricultural products of changes in the W126 index resulting from meeting alternative standards levels. We also assessed the effects of those changes in commercial agricultural and forest yields on carbon sequestration and storage. The analysis provided limited quantitative information on the welfare benefits of meeting alternative secondary standard levels, focusing 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. In this final RIA, we did not update this analysis and refer to the analysis conducted in the proposal RIA (U.S. EPA, 2014c). We did not update the analysis from the proposal RIA because the welfare benefits estimates (i) in the proposal analysis were small, and we anticipated that the estimates in the final analysis would be even smaller, and (ii) are not added to the human health benefits estimates.

ES.2 Results of Benefit-Cost Analysis

Below in Table ES-5, we present the primary costs and benefits estimates for 2025 for all areas except California. We anticipate that benefits and costs will likely begin occurring earlier than 2025, as states begin implementing control measures to show progress towards attainment. In these tables, ranges within the total benefits rows reflect multiple studies upon which the estimates associated with premature mortality were derived. $PM_{2.5}$ co-benefits account for approximately 60 to 70 percent of the estimated benefits, depending on the standard analyzed and on the choice of ozone and PM mortality functions used. Assuming a 7 percent discount rate, for a standard of 70 ppb the total health benefits are comprised of between 29 and 34 percent ozone benefits and between 66 and 71 percent $PM_{2.5}$ co-benefits. Assuming a 7 percent discount rate, for a standard of 65 ppb the total health benefits are comprised of between 29 and 35 percent ozone benefits and between 62 and 70 percent $PM_{2.5}$ co-benefits. In addition for 2025, Table ES-6 presents the numbers of premature deaths avoided for the revised and alternative standard levels analyzed, as well as the other health effects avoided. Table ES-7 provides information on the costs by geographic region for the U.S., except California in 2025, and Table ES-8 provides a regional breakdown of benefits for 2025. See the tables in Chapter 6 for additional characterizations of the monetized benefits.

In the RIA we provide estimates of the costs of emissions reductions to attain the revised and alternative standard levels 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 both control strategies applied within the region and reductions in transport of ozone associated with emissions reductions in other regions.

The net benefits of emissions reductions strategies in a specific region reflect 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 was conducted at the national level, we do not estimate separately the nationwide benefits associated with the emissions reductions occurring in any specific region.[7] 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

[7] 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 analysis for California.

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

Table ES-5. Total Annual Costs and Benefits[a,b] for U.S., except California in 2025 (billions of 2011$, 7% Discount Rate)[c]

	Revised and Alternative Standard Levels	
	70 ppb	**65 ppb**
Total Costs[d]	$1.4	$16
Total Health Benefits	$2.9 to $5.9[e, f]	$15 to $30[e, f]
Net Benefits	**$1.5 to $4.5**	**-$1.0 to $14**

[a] All values are rounded to two significant figures.
[b] Benefits are nationwide benefits of attainment everywhere except California.
[c] The tables in Chapter 6 provide additional characterizations of the monetized benefits, including benefits estimated at a 3 percent discount rate. 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.
[d] The engineering costs in this table are annualized at a 7 percent discount rate to the extent possible. See Chapter 4 for more discussions.
[e] Assuming a 7 percent discount rate, for a standard of 70 ppb the total health benefits are comprised of between 29 and 34 percent ozone benefits and between 66 and 71 percent $PM_{2.5}$ co-benefits. Assuming a 7 percent discount rate, for a standard of 65 ppb the total health benefits are comprised of between 29 and 35 percent ozone benefits and between 62 and 70 percent $PM_{2.5}$ co-benefits.
[f] Excludes additional health and welfare benefits that could not be quantified (see Chapter 6, Section 6.6.3.8).

The guidelines of OMB Circular A-4 require providing comparisons of social costs and social benefits at discount rates of 3 and 7 percent. 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-6. Summary of Total Number of Annual Ozone and PM-Related Premature Mortalities and Premature Morbidity: 2025 National Benefits [a]

	Revised and Alternative Standard Levels	
	70 ppb	**65 ppb**
Ozone-related premature deaths avoided (all ages)	96 to 160	490 to 820
$PM_{2.5}$-related premature deaths avoided (age 30+)	220 to 500	1,100 to 2,500
Other health effects avoided		
Non-fatal heart attacks (age 18-99) (5 studies) [PM]	28 to 260	140 to 1,300
Respiratory hospital admissions (age 0-99)[O3, PM]	250	1,200
Cardiovascular hospital admissions (age 18-99) [PM]	80	400
Asthma emergency department visits (age 0-99) [O3, PM]	630	3,300
Acute bronchitis (age 8-12) [PM]	340	1,700
Asthma exacerbation (age 6-18) [O3, PM]	230,000	1,100,000
Lost work days (age 18-65) [PM]	28,000	140,000
Minor restricted activity days (age 18-65) [O3, PM]	620,000	3,100,000
Upper & lower respiratory symptoms (children 7-14) [PM]	11,000	53,000
School loss days (age 5-17) [O3]	160,000	790,000

[a] Nationwide benefits of attainment everywhere except California. All values are rounded to two significant figures. Additional information on confidence intervals are available in the tables in Chapter 6.

Table ES-7. Summary of Total Control Costs (Identified + Unidentified Control Strategies) by Revised and Alternative Standard Levels for 2025 - U.S., except California (billions of 2011$, 7% Discount Rate)[a]

Revised and Alternative Standards Levels	Geographic Area	Total Control Costs (Identified and Unidentified)
70 ppb	East	1.4
	West	<0.05
	Total	**$1.4**
65 ppb	East	15
	West	<0.75
	Total	**$16**

[a] All values are rounded to two significant figures. Costs are annualized at a 7 percent discount rate to the extent possible. Costs associated with unidentified controls are based on an average cost-per-ton methodology (see Chapter 4, Section 4.3 for more discussion on the average-cost methodology).

Table ES-8. Regional Breakdown of Monetized Ozone-Specific Benefits Results for 2025 (Nationwide Benefits of Attaining the Revised and Alternative Standard Levels Everywhere in the U.S., except California) [a]

	Revised and Alternative Standard Levels	
Region	**70 ppb**	**65 ppb**
East [b]	98%	96%
California	0%	0%
Rest of West	2%	4%

[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 states to the north and east.

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-9. 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-10 presents the numbers of premature deaths avoided for the revised and alternative standard levels analyzed, as well as the other health effects avoided. Table ES-11 provides information on the costs for California for post-2025, and Table ES-12 provides a regional breakdown of benefits for post-2025.

The EPA presents separate costs and benefits results for California because assuming attainment in an earlier year than would be required under the Clean Air Act would likely lead to an overstatement of costs and benefits 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-9. Total Annual Costs and Benefits[a] of the Identified + Unidentified Control Strategies Applied in California, Post-2025 (billions of 2011$, 7% Discount Rate)[b]

	Revised and Alternative Standard Levels	
	70 ppb	**65 ppb**
Total Costs[c]	$0.80	$1.5
Total Health Benefits	$1.2 to $2.1[d]	$2.3 to $4.2[d]
Net Benefits	**$0.4 to $1.3**	**$0.8 to $2.7**

[a] Benefits are nationwide benefits of attainment in California.

[b] The guidelines of OMB Circular A-4 require providing comparisons of social costs and social benefits at discount rates of 3 and 7 percent. The tables in Chapter 6 provide additional characterizations of the monetized benefits, including benefits estimated at a 3 percent discount rate. 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 engineering costs in this table are annualized at a 7 percent discount rate to the extent possible. See Chapter 4 for more discussions.

[d] Excludes additional health and welfare benefits that could not be quantified (see Chapter 6, Section 6.6.3.8).

Table ES-10. Summary of Total Number of Annual Ozone and PM-Related Premature Mortalities and Premature Morbidity: Post-2025[a]

	Revised and Alternative Standard Levels	
	70 ppb	**65 ppb**
Ozone-related premature deaths avoided (all ages)	72 to 120	150 to 240
$PM_{2.5}$-related premature deaths avoided (age 30+)	43 to 98	84 to 190
Other health effects avoided		
Non-fatal heart attacks (age 18-99) (5 studies) [PM]	6 to 51	11 to 100
Respiratory hospital admissions (age 0-99)[O3, PM]	150	300
Cardiovascular hospital admissions (age 18-99) [PM]	16	31
Asthma emergency department visits (age 0-99) [O3, PM]	380	760
Acute bronchitis (age 8-12) [PM]	64	130
Asthma exacerbation (age 6-18) [O3, PM]	160,000	330,000
Lost work days (age 18-65) [PM]	5,300	10,000
Minor restricted activity days (age 18-65) [O3, PM]	360,000	720,000
Upper & lower respiratory symptoms (children 7-14) [PM]	2,000	3,900
School loss days (age 5-17) [O3]	120,000	240,000

[a] Nationwide benefits of attainment in California. All values are rounded to two significant figures. Additional information on confidence intervals are available in the tables in Chapter 6.

Table ES-11. Summary of Total Control Costs (Identified + Unidentified Control Strategies) by Revised and Alternative Standards for Post-2025 - California (billions of 2011$, 7% Discount Rate)[a]

Revised and Alternative Standard Level	Geographic Area	Total Control Costs (Identified and Unidentified)
70 ppb	California	$0.80
65 ppb	California	$1.5

[a] All values are rounded to two significant figures. Costs are annualized at a 7 percent discount rate to the extent possible. Costs associated with unidentified controls are based on an average cost-per-ton methodology.

Table ES-12. Regional Breakdown of Monetized Ozone-Specific Benefits Results for Post-2025 (Nationwide Benefits of Attaining Revised and Alternative Standards just in California)[a]

	Revised and Alternative Standard Levels	
Region	**70 ppb**	**65 ppb**
East [b]	3%	2%
California	90%	91%
Rest of West	7%	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 states to the north and east.

ES.3 Improvements between the Proposal and Final RIAs

In the regulatory impact analyses for both the proposed and final ozone NAAQS, there were two geographic areas outside of California where the majority of emissions reductions were needed to meet the revised standard level of 70 ppb – Texas and the Northeast. In analyzing 70 ppb in the final RIA, there were approximately 50 percent fewer emissions reductions needed in these two geographic areas. For an alternative standard of 65 ppb in the final RIA, emissions reductions needed nationwide were approximately 20 percent lower than at proposal. The primary reason for the difference in emissions reductions estimated for attainment is that in the final RIA we conducted more geographically-refined air quality sensitivity modeling to develop improved ozone response factors (see Chapter 2, Section 2.2.2 for a more detailed discussion of the air quality modeling) and focused the emissions reduction strategies on geographic areas closer to the monitors with the highest design values (see Chapter 3, Section 3.1.1 for a more detailed discussion of the emissions reduction strategies). The improvements in air quality modeling and emissions reduction strategies account for about 80 percent of the difference in needed emissions reductions between the proposal and final RIAs.

In Texas and the Northeast, the updated response factors and more focused emissions reduction strategies resulted in larger changes in ozone concentrations in response to more geographically focused emissions reductions. In east Texas, the ppb/ton ozone response factors used in the final RIA were 2 to 3 times more responsive than the factors used in the proposal RIA at controlling monitors in Houston and Dallas. In the Northeast, the ppb/ton ozone response factors used in the final RIA were 2.5 times more responsive than the factors used in the proposal RIA at the controlling monitor on Long Island, NY.

A secondary reason for the difference is that between the proposal and final RIAs we updated emissions inventories, models and model inputs for the base year of 2011. See Appendix 2, Section 2A.1.3 for additional discussion of the updated emissions inventories, models and model inputs. When projected to 2025, these changes in inventories, models and inputs had compounding effects for year 2025, and in some areas resulted in lower projected base case design values for 2025. The updated emissions inventories, models, and model inputs account for about 20 percent of the difference in needed emissions reductions between the proposal and final RIAs.

These differences in the estimates of emissions reductions needed to attain the revised and alternative standard levels affect the estimates for the costs and benefits in this RIA. For a revised standard of 70 ppb, the costs were 60 percent lower than at proposal and the benefits were 55 percent lower than at proposal. The percent decrease in costs is slightly more than the percent decrease in emissions reductions because a larger number of lower cost identified controls were available to bring areas into attainment with 70 ppb.[8] The percent decrease in benefits is similar to the percent decrease in emissions reductions. For an alternative standard level of 65 ppb, the costs were less than three percent more than those estimated at proposal and the benefits were 22 percent lower than at proposal. The percent change in costs was less than the percent decrease in emissions reductions because in the final analysis we applied identified controls in smaller geographic areas, resulting in fewer identified controls available within those

[8] In the final RIA, outside of California all areas were projected to meet the current standard of 75 ppb. As such, no identified controls were used to bring areas into attainment with 75 ppb. In the proposal RIA, some of these lower cost controls were used to bring areas into attainment with 75 ppb, making them unavailable for application in the analysis of 70 ppb.

areas and an increase in higher cost unidentified controls being applied to bring areas into attainment with 65 ppb. The percent decrease in benefits is similar to the percent decrease in emissions reductions.

ES.4 Uncertainty

Despite uncertainties inherent in any complex, quantitative analysis, the underlying tools and models (CoST and BenMAP) have been peer-reviewed and the analytical methods are consistent with standard economic practice. For a detailed discussion on uncertainty associated with developing illustrative control strategies to attain the alternative standard levels, see Chapter 3, Section 3.4. For a description of the key assumptions and uncertainties related to ozone benefits, see Chapter 6, Section 6.5, and for an additional qualitative discussion of sources of uncertainty associated with both ozone-related benefits and $PM_{2.5}$-related co-benefits, see Appendix 6A. For a discussion of the limitations and uncertainties in the engineering cost analyses, see Chapter 4, Section 4.7. For a general discussion about key factors that could impact how air quality changes over time, see Chapter 8, Section 8.3.

ES.5 References

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CHAPTER 1: INTRODUCTION AND BACKGROUND

Introduction

The Environmental Protection Agency (EPA) initiated the current ozone National Ambient Air Quality Standards (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. In addition, as required by the Clean Air Act (CAA), the documents were peer-reviewed by the Clean Air Scientific Advisory Committee (CASAC), the Administrator's independent advisory committee established by the CAA. The final documents for this review reflect the EPA staff's consideration of the comments and recommendations made by the CASAC and the public on draft versions of these documents.

The EPA has concluded that the current primary standard for ozone, set at a level of 75 ppb, is not requisite to protect public health with an adequate margin of safety, and is revising the standard to provide increased public health protection. Specifically, the EPA is retaining the indicator (ozone), averaging time (8-hour) and form (annual fourth-highest daily maximum, averaged over 3 years) of the existing primary standard and is revising the level of that standard to 70 ppb. The EPA is making 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.

The EPA has also concluded that the current secondary standard for ozone, set at a level of 75 ppb, is not requisite to protect public welfare from known or anticipated adverse effects, and is revising the standard to provide increased protection against vegetation-related effects on public welfare. Specifically, the EPA is retaining the indicator (ozone), averaging time (8-hour) and form (annual fourth-highest daily maximum, averaged over 3 years) of the existing secondary standard and is revising the level of that standard to 70 ppb. The EPA has concluded that this revision will effectively curtail cumulative seasonal ozone exposures above 17 ppm-hrs, in terms of a three-year average seasonal W126 index value, based on the three consecutive month period within the growing season with the maximum index value, with daily exposures cumulated for the 12-hour period from 8:00 am to 8:00 pm. Thus, the EPA has concluded that this revision will provide the requisite protection against known or anticipated adverse effects to the public welfare.

This Regulatory Impact Analysis (RIA) analyzes the human health benefits and costs and welfare cobenefits of the revised standard of 70 ppb as well as a more stringent alternative level of 65 ppb. 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. As interpreted by the Agency and the courts, the CAA 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 the implementation process, as they decide what timelines, strategies, and policies are appropriate for their circumstances. 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 part of setting the standards.

This chapter summarizes provides a brief background on NAAQS, the need for NAAQS, and an overview of this RIA, including a discussion of its design. The EPA prepared this RIA both to provide the public with information on 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 National Ambient Air Quality Standards

Sections 108 and 109 of the CAA govern the establishment and revision of the 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 CAA 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 Role of Executive Orders in the Regulatory Impact Analysis

While this RIA is separate from the NAAQS decision-making process, several statutes and executive orders still apply to any public documentation. The analyses required by these statutes and executive orders are presented in detail in Chapter 9, and below we briefly discuss requirements of Orders 12866 and 13563 and the guidelines of the Office of Management and Budget (OMB) Circular A-4 (U.S. OMB, 2003).

In accordance with Executive Orders 12866 and 13563 and the guidelines of OMB Circular A-4, the RIA analyzes the benefits and costs associated with emissions controls to attain the revised 8-hour ozone standard of 70 ppb in ambient air, incremental to a baseline of attaining the existing standard (8-hour ozone standard of 75 ppb).[9] OMB Circular A-4 requires analysis of one potential alternative standard level more stringent than the revised standard and one less stringent than the revised standard. In this RIA, we analyze a more stringent alternative standard level of 65 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). Further, as discussed in the notice of final rulemaking, the available scientific evidence and quantitative risk and exposure information on the health effects of ozone exposure provide strong support for a revised standard of 70 ppb, but do not identify a bright line for identifying any specific standard level between 70 and 75 ppb for analysis in the RIA. As such, we did not analyze a standard between 70 and 75 ppb in this RIA.

1.1.3 Illustrative Nature of the Analysis

The control strategies presented in this RIA are an illustration of one possible set of control strategies states might choose to implement to meet the revised standards. States—not the EPA—will implement the revised NAAQS and will ultimately determine appropriate emissions control strategies and measures. State Implementation Plans (SIPs) will likely vary from the EPA's estimates provided in this analysis due to differences in the data and assumptions that states use to develop these plans. Because states are ultimately responsible for implementing strategies to meet the revised standards, the control strategies in this RIA are considered hypothetical. The hypothetical strategies were constructed with the understanding that there are

[9] On April 30, 2012 the EPA issued final designations for the 2008 ozone NAAQS. After final designations, areas have up to three years to submit attainment SIPs. Because of the timing of these SIP submittals, the EPA does not have the most current information on control measures and emissions reductions needed to meet the current standard of 75 ppb. To account for potential emissions reductions associated with meeting the current standard, we estimate these emissions reductions in defining the baseline.

inherent uncertainties in projecting emissions and control applications. Additional important uncertainties and limitations are documented in the relevant portions of the RIA.

The EPA's national program rules require technology application or emissions limits for a specific set of sources or source groups. In contrast, a NAAQS establishes a standard level and requires states to identify and secure emissions reductions to meet the standard level from *any* set of sources or source groups. To avoid double counting the impacts of NAAQS and other national program rules, the EPA includes federal regulations and enforcement actions in its baseline for this analysis (See Section 1.3.1 for additional discussion of the baseline). The benefits and costs of the revised standards will not be realized until specific control measures are mandated by SIPs or other federal regulations.

1.2 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 a market failure. The major types of market failure include: externality, market power, and inadequate or asymmetric information. Correcting market failures is one reason for regulation, but it is not the only reason. Other possible justifications include improving the function of government, removing distributional unfairness, or promoting privacy and personal freedom.

Environmental problems are classic examples of externalities -- uncompensated benefits or costs imposed on another party as a results of one's actions. For example, the smoke from a factory may adversely affect the health of local residents and soil the property in nearby neighborhoods. If bargaining was costless and all property rights were well defined, people would eliminate externalities through bargaining without the need for government regulation.

From an economics perspective, setting an air quality standard is a straightforward remedy to address an externality in which firms emit pollutants, resulting in health and environmental problems without compensation for those incurring the problems. Setting a standard with a reasonable margin of safety attempts to place the cost of control on those who emit the pollutants and lessens the impact on those who suffer the health and environmental problems from higher levels of pollution. For additional discussion on the ozone air quality problem, see Chapter 2 of the Policy Assessment for the Review of the Ozone National Ambient Air Quality Standards (US EPA, 2014).

1.3 Overview and Design of the RIA

The RIA evaluates the costs and benefits of hypothetical national control strategies to attain the revised ozone standard of 70 ppb and an alternative ozone standard level of 65 ppb.

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

The RIA is intended to evaluate the overall potential costs and benefits of reaching attainment with the revised and alternative ozone standard levels. To develop and evaluate control strategies for attaining a more stringent primary standard, it is important to estimate ozone levels in the future after attaining the current NAAQS of 75 ppb, and taking into account projections of future air quality reflecting on-the-books Federal regulations, substantial federal regulatory proposals, enforcement actions, state regulations, population and where possible, economic growth. Establishing this baseline for the analysis then allows us to estimate the incremental costs and benefits of attaining the revised and alternative standard levels.

Attaining 75 ppb reflects emissions reductions (i) already achieved as a result of national regulations, (ii) 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 current ozone standard), and (iii) from additional controls that the EPA estimates need to be included to attain the current standard. Additional 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 for this analysis, a baseline that reflects attainment of 75 ppb. First, national ozone concentrations were projected to the analysis year (2025) based on forecasts of population and where possible, 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 estimated additional emissions reductions needed to meet the current standard of 75 ppb and make adjustments for the proposed Clean Power Plan.

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 6.2, 2011 Emissions Modeling Platform (US EPA, 2015). If the national rules reflected in the baseline result in changes in ozone concentrations or actual emissions reductions that are lower or higher than those estimated, the costs and benefits estimated in this final RIA would be higher or lower, respectively.

- Carbon Pollution Emission Guidelines for Existing Stationary Sources: Electric Utility Generating Units (Proposed Rule) (U.S. EPA, 2014a)
- Tier 3 Motor Vehicle Emission and Fuel Standards (U.S. EPA, 2014c)
- 2017 and Later Model Year Light-Duty Vehicle Greenhouse Gas Emissions and Corporate Average Fuel Economy Standards (U.S. EPA, 2012)
- Cross State Air Pollution Rule (CSAPR) (U.S. EPA, 2011)
- Mercury and Air Toxics Standards (U.S. EPA, 2011a)[10]
- Greenhouse Gas Emissions Standards and Fuel Efficiency Standards for Medium- and Heavy-Duty Engines and Vehicles (U.S. EPA, 2011d)[11]
- C3 Oceangoing Vessels (U.S. EPA, 2010)
- Reciprocating Internal Combustion Engines (RICE) NESHAPs (U.S. EPA, 2010a)
- Regulation of Fuels and Fuel Additives: Modifications to Renewable Fuel Standard Program (RFS2) (U.S. EPA, 2010b)
- Light-Duty Vehicle Greenhouse Gas Emission Standards and Corporate Average Fuel Economy Standards; Final Rule for Model-Year 2012-2016 (U.S. EPA, 2010c)
- Hospital/Medical/Infectious Waste Incinerators: New Source Performance Standards and Emission Guidelines: Final Rule Amendments (U.S. EPA, 2009)

[10] On June 29, 2015, the United States Supreme Court reversed the D.C. Circuit opinion affirming the Mercury and Air Toxics Standards (MATS). The EPA is reviewing the decision and will determine any appropriate next steps once the review is complete, however, MATS is still currently in effect. The first compliance date was April 2015, and many facilities have installed controls for compliance with MATS. MATS is included in the baseline for this analysis, and the EPA does not believe including MATS substantially alters the results of this analysis.

[11] This rule is Phase 1 of the Heavy Duty Greenhouse Gas Standards for New Vehicles and Engines (76 FR 57106, September 15, 2011) and is included in the 2025 base case. Phase 2 of the Heavy Duty Greenhouse Gas Standards for New Vehicles and Engines (80 FR 40138, July 13, 2015) is not included in the 2025 base case because the rulemaking was not finalized in time to include in this analysis. If the emissions reductions from Phase 1 were not included in the baseline in this analysis, the estimated costs and benefits of achieving the revised and alternative standards analyzed would be higher because more emissions reductions would be needed.

- 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, 2005a)
- NO_x Emission Standard for New Commercial Aircraft Engines (U.S. EPA, 2005)

To define the baseline in the ozone NAAQS final RIA, we adjusted the 2025 final ozone NAAQS base case air quality to reflect the proposed Clean Power Plan (CPP) using the Option 1 State illustrative compliance approach from the CPP proposal RIA. We recognize that the difference in forecast NO_x emissions from the electricity sector between the CPP proposal and final likely has some effect on baseline ozone concentrations, and therefore on estimated NO_x emissions reductions needed to meet the ozone standards analyzed in the NAAQS final RIA.

The power sector modeling for the final CPP reflected updated inputs including lower costs for new renewable energy resources and changes in the composition of electric generating resources relative to the baseline used for the proposed CPP. These updated inputs resulted in changes in the baseline level and spatial distribution of NO_x emissions in the final CPP. In addition, in the final CPP the CO_2 emissions goals for states and compliance timing changed from the proposal, which further changed the level and spatial distribution of NO_x emissions. The net effect of these changes is that total forecast annual NO_x emissions in 2025 for the electricity sector were between 13,000 and 51,000 tons lower under the final CPP than under the proposed CPP.

The impact of these forecast changes in NO_x emissions on ozone concentrations in specific locations is uncertain. There is no clear spatial pattern of where emissions are forecast to be higher or lower in the final CPP relative to the proposed CPP. Furthermore, states have flexibility in the form of their plans that implement the CPP and therefore the specific impact of the CPP on NO_x emissions in any state is uncertain. Finally, because no air quality modeling was done for the final CPP, we are not able to implement the same approach to reflect the impact of the final CPP on ozone air quality in the NAAQS baseline that we used to account for the proposed CPP in the baseline.

We recognize that not accounting for the final CPP in the baseline introduces additional uncertainty into the NAAQS final RIA. However, in the final CPP EPA recommends that states take a multipollutant planning approach that recognizes co-pollutant impacts of CO_2 compliance decisions and takes into account local air quality impacts. Given the flexibility that states have in addressing both their CO_2 and air quality requirements, EPA expects that states will design strategies to meet both the CPP and NAAQS in the most cost-effective manner[12], and thus costs for the combined set of actions will likely differ from the combined costs provided in the separate RIAs.

The baseline for this analysis does not assume emissions controls that might be implemented to meet the current $PM_{2.5}$, NO_2, or SO_2 NAAQS. For the current $PM_{2.5}$ and SO_2 NAAQS, the Agency has not issued final designations and does not have information on what areas would need emissions controls; for the current NO_2 NAAQS there are no nonattainment areas. 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.[13] More importantly, all control strategies analyzed in NAAQS RIAs are hypothetical.

[12] "…the EPA believes that the Clean Power Plan provides an opportunity for states to consider strategies for meeting future CAA planning obligations as they develop their plans under this rulemaking. Multi-pollutant strategies that incorporate criteria pollutant reductions over the planning horizons specific to particular states, jointly with strategies for reducing CO_2 emissions from affected EGUs needed to meet Clean Power Plan requirements over the time horizon of this rule, may accomplish greater environmental results with lower long-term costs." Page 1333 of the *Carbon Pollution Emission Guidelines for Existing Stationary Sources: Electric Utility Generating Units*, currently available at the following link: http://www2.epa.gov/sites/production/files/2015-08/documents/cpp-final-rule.pdf. In the future, please refer to the official version in a forthcoming FR publication, which will appear on the Government Printing Office's FDSys website (http://gpo.gov/fdsys/search/home.action) and on Regulations.gov (http://www.regulations.gov) in Docket No. EPA–HQ–OAR–2013-0602.

[13] There were no additional NOx controls applied in the 2012 $PM_{2.5}$ NAAQS RIA, and therefore there would be little to no impact on the controls selected in 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 controls on a $/ug basis, states may choose to adopt different control options. These options could

1.3.2 Establishing the Baseline for Evaluation of Revised and Alternative Standards

The RIA evaluates, to the extent possible, the costs and benefits of attaining the revised and alternative ozone standards incremental to attaining the current 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 attain the revised standard by 2025. As such, we developed our projected baselines for emissions, air quality, and populations and present the primary costs and benefits estimates for 2025.

The selection of 2025 as the analysis year in the RIA does not predict or prejudge attainment dates that will ultimately be assigned to individual areas under the CAA. The CAA contains a variety of potential attainment dates and flexibility to move to later dates (up to 20 years), provided that the date is as expeditious as practicable. 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 provides 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.

The EPA recognizes that areas designated nonattainment for the revised ozone NAAQS and classified as Marginal or Moderate will likely incur some costs prior to the 2025 analysis year. The Agency, however, anticipates that on-the-books federal emissions control measures[14] will be sufficient to bring the majority of these areas into attainment by 2025. Areas designated

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include NOx controls, and it is difficult to determine the impact on costs and benefits for this RIA because it depends highly on the control measures that would be chosen and the costs of these measures.

[14] These federal control measures are listed above in section 1.3.1.

nonattainment and classified as Marginal are required to develop emission inventories, emission statements, and produce a CAA section 110 infrastructure SIP. These areas are not required to develop any control measures aside from the federal emissions control measures reflected in the baseline. As a result, the Agency anticipates that costs in these Marginal areas will be minimal. In addition to the federal control measures and the requirements for Marginal nonattainment areas, states with nonattainment areas designated as Moderate are required by the CAA to develop state implementation plans (SIPs) demonstrating attainment by no later than the assigned attainment date. The CAA also requires these states to address Reasonably Available Control Technologies (RACT) for sources in the Moderate nonattainment area, which could lead to additional point source controls in an area beyond the federal emissions control measures. Additionally, the CAA requires some Moderate areas with larger populations to implement basic vehicle inspection and maintenance (I/M) in the area. Should these federal programs and CAA required programs prove inadequate for the area to attain the revised standard by the attainment date, the state would need to identify additional emissions control measures in its SIP to meet attainment requirements.

In addition, in estimating the incremental costs and benefits of the revised and alternative standards, we recognize that there are areas that are not required to meet the existing ozone standard by 2025 -- the CAA 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.[15,16] Because of data and resource constraints, 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

[15] 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 by December 31, 2032 and nonattainment areas classified as Extreme will likely have to attain by December 31, 2037.

[16] In this RIA before deciding to continue to analyze California beyond the future analysis year of 2025, we reviewed California's NOx and VOC emissions within existing nonattainment areas. The vast majority of these emissions come from emissions sources located in existing nonattainment areas that would likely have to attain the final standard sometime between 2032 and 2037. As a result, we concluded that analyzing California separately and after 2025 continued to be an appropriate analytical decision.

expected to be fully implemented by 2030.[17] 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 difference in 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.

1.3.3 Cost Analysis Approach

The EPA estimated total costs under partial and full attainment of the revised and alternative ozone standard levels analyzed. These cost estimates reflect only engineering costs, which generally includes the costs of purchasing, installing, and operating the referenced control technologies. The technologies and control strategies selected for analysis illustrate 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 identified controls. Costs for full attainment include estimates for the costs associated with the additional emissions reductions that are needed beyond identified

[17] At the time of this analysis, there were no future year emissions for California beyond 2030, and projecting emissions beyond 2030 could introduce additional uncertainty.

controls. The EPA recognizes that the portion of the cost estimates from emissions reductions beyond identified controls reflects substantial uncertainty about which sectors and which technologies might become available for cost-effective application in the future.

1.3.4 Human Health Benefits

The EPA estimated human health (i.e., mortality and morbidity effects) under both partial and full attainment of the two alternative ozone standard levels analyzed. 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, which has been used in many recent RIAs (e.g., U.S. EPA, 2011a, 2011c) and The Benefits and Costs of the Clean Air Act 1990 to 2020 (U.S. EPA, 2011b). 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 6). In addition, unquantified health benefits are also discussed in Chapter 6.

1.3.5 Welfare Benefits of Meeting the Primary and Secondary Standards

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. Welfare co-benefits are discussed further in Chapter 7.

1.4 Updates between the Proposal and Final RIAs

For NAAQS RIAs, the Agency always reviews the underlying data used and makes methodological and model improvements both between proposal and final analyses and between different NAAQS analyses. For this final RIA, we made updates to the emissions inventory based on public comments and input from the states, updated the oil and gas sector emissions projections based on input from the states, and used updated versions of IPM and the onroad mobile source model.[18] For a detailed discussion of these emissions inventory, model, and model input updates, see Appendix 2, Section 2A.1.3. In addition, based on the analyses in the proposal RIA, in this final NAAQS RIA the EPA decided to conduct more refined air quality modeling to assess emissions changes closer to monitors in certain areas, specifically Texas and the Northeast.

The net effects of the emissions inventory, model, and model input updates are changes in projected 2025 ozone air quality design values (DVs)[19] in many areas. These new projected DVs were higher than previously modeled for the proposal RIA in some locations and lower in others. The new projections show lower 2025 DVs in Central Texas from Houston to Dallas, the El Paso area (NM and TX) and Big Bend, Texas, and several states in the central U.S., including Oklahoma, Kansas, Missouri, Arkansas, Mississippi, Tennessee, and southern Kentucky. The new projections also show higher 2025 DVs in Denver, Las Vegas, Phoenix, Charlotte, the upper Midwest, and parts of the New York/New Jersey areas. See Appendix 2A, Section 2A.4 for detailed information on the updated DVs.

We also conducted additional air quality modeling runs to provide more spatially resolved air quality response factors, allowing us to more appropriately represent the effectiveness of emissions reductions from sources closer to receptor monitors compared to the regional response factors used for the November 2014 proposal RIA (see Chapter 2, Section 2.2.2 for a discussion of the additional air quality modeling). In the final RIA, there were approximately 50 percent fewer emissions reductions needed in Texas and the Northeast to reach a revised standard of 70 ppb. For an alternative standard of 65 ppb in the final RIA, emissions reductions needed nationwide were approximately 20 percent lower than at proposal.

The primary reasons for the difference in emissions reductions estimated in the final RIA are the more spatially resolved air quality modeling and resulting improved ozone response

[18] Based on the timing associated with both preparing an updated 2025 base case and completing the analyses in this final RIA, we used the IPM v5.14 base case because the IPM v5.15 base case was not available.
[19] The DV is calculated as the 3-year average of the annual 4th highest daily maximum 8-hour ozone concentration in parts per billion, with decimal digits truncated. The DV is a metric that is compared to the standard level to determine whether a monitor is violating the NAAQS. The ozone DV is described in more detail in Chapter 2, Section 2.2.

factors, as well as the focus of the emissions reduction strategies on geographic areas closer to the monitors with the highest design values (see Chapter 3, Section 3.1.1 for a more detailed discussion of the emissions reduction strategies). The improvements in air quality modeling and emissions reduction strategies account for about 80 percent of the difference in estimated needed emissions reductions between the proposal and final RIAs.

For example, in analyzing the revised standard of 70 ppb, in Texas and the Northeast the updated response factors and more focused emissions reduction strategies resulted in larger changes in ozone concentrations in response to more geographically focused emissions reductions. In east Texas, the air quality response factors used in the final RIA were 2 to 3 times more responsive than the factors used in the proposal RIA at controlling monitors in Houston and Dallas. In the Northeast, the air quality response factors used in the final RIA were 2.5 times more responsive than the factors used in the proposal RIA at the controlling monitor on Long Island, NY.

The updates made to the emissions inventories, models, and model inputs for the base year of 2011 account for the remaining 20 percent of the difference in estimated emissions reductions needed between the proposal and final RIAs. When projected to 2025, these changes in inventories, models and inputs had compounding effects for year 2025, and in some areas resulted in lower projected base case DVs for 2025.

For additional information on how the revised emissions reduction estimates affect the cost estimates, see Chapter 4, Section 4.6. For additional information on how the revised emissions reduction estimates affect the benefits estimates, see Chapter 6, Section 6.1.

1.5 Organization of the Regulatory Impact Analysis

This RIA is organized into the following remaining chapters:

- *Chapter 2*: *Emissions, Air Quality Modeling and Analytic Methodologies*. 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 benefits and costs.
- *Chapter 3: Control Strategies and Emissions Reductions*. The 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 4: Engineering Cost Analysis and Economic Impacts.* The chapter summarizes the data sources and methodology used to estimate the engineering costs of partial and full attainment of the three alternative standard levels analyzed.
- *Chapter 5: Qualitative Discussion of Employment Impacts of Air Quality.* The chapter provides a discussion of some possible types of employment impacts of reducing emissions of ozone precursors.
- *Chapter 6: Human Health Benefits Analysis Approach and Results*. The chapter quantifies the health-related benefits of the ozone-related air quality improvements associated with the three alternative standard levels analyzed.
- *Chapter 7*: *Impacts on Public Welfare of Attainment Strategies to Meet the Primary and Secondary Ozone NAAQS.* The chapter includes a discussion of the welfare-related benefits of meeting alternative primary and secondary ozone standards and a limited quantitative analysis for effects associated with changes in yields of commercial forests and agriculture, and associated changes in carbon sequestration and storage.
- *Chapter 8: Comparison of Benefits and Costs*. The chapter compares estimates of the total benefits with total costs and summarizes the net benefits of the three alternative standards analyzed.
- *Chapter 9*: *Statutory and Executive Order Impact Analyses*. The chapter summarizes the Statutory and Executive Order impact analyses.

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CHAPTER 2: EMISSIONS, AIR QUALITY MODELING AND ANALYTIC METHODOLOGIES

Overview

This regulatory impacts analysis (RIA) evaluates the costs as well as the health and environmental benefits associated with complying with the revised (70 ppb) and alternative (65 ppb) 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 modeling conducted with emissions projected to the year 2025 including all current on-the-books federal regulations.[20] As described in section 2.1, the emissions inputs for the 2011 base year and 2025 base case simulations were updated between the November 2014 proposal RIA (EPA, 2014a) and this final analysis. These updates were made in response to comments provided by states and newly available emissions models and projection information. In the following sections, the 2025 base case from the November 2014 proposal RIA will be referred to as the "proposal 2025 base case" while the updated 2025 base case will be referred to as the "final 2025 base case". In addition, a series of emissions sensitivity[21] modeling runs were conducted to determine the response of ozone to changes in 2025 emissions. These sensitivity runs were used to develop ozone response factors (ppb/ton) that represent the modeled response of ozone to changes in NO_x and VOC emissions from various sources and locations.[22]

The following scenarios were developed based on applying the ozone response factors to the final 2025 base case ozone concentrations: (1) the baseline scenario (a scenario that includes

[20] Emissions reductions to attain the 2012 $PM_{2.5}$ NAAQS are not included in the proposal or final 2025 base case because the scenarios modeled in the 2012 $PM_{2.5}$ NAAQS RIA did not reflect any NOx emissions reductions (US EPA, 2012).
[21] Sensitivity refers to modeling simulations designed to capture the response of ozone concentrations to changes in emissions.
[22] All emissions sensitivity model runs were created with reductions incremental to the proposal 2025 base case scenario.

attainment of the current standard of 75 ppb)[23] and (2) the revised standard level scenario and an alternative standard level scenario that both represent incremental emissions reductions beyond the baseline to meet levels of 70 and 65 ppb respectively.[24] For each scenario we calculated emissions reductions necessary to meet the target standard level and resulting ozone concentrations at ozone monitoring locations. We used the emissions reductions as inputs in the estimation of control strategies (Chapter 3) and costs (Chapter 4) associated with attaining the revised and alternative ozone standard levels. The emissions reductions were also used to estimate changes in health-related ozone concentration metrics under each scenario allowing us to calculate the health-related benefits that would result from the reductions in emission and ozone concentrations associated with meeting various standard levels (Chapter 6). Figure 2-1 below outlines these general steps and Table 2-1 lists all of the scenarios discussed above with their respective definitions.

[23] As described in chapter 1, section 1.3.2, we use a "2025 baseline scenario" for areas of the contiguous U.S. outside of California and a "post-2025 baseline scenario" for California due to the later attainment dates for some areas in that state.

[24] For the revised standard and the alternative standard we present both a scenario which represents only the portion of emissions reductions that come from identified controls (identified control strategies) and one that represents total emissions reductions necessary to attain the respective standard levels (identified + unidentified control strategies).

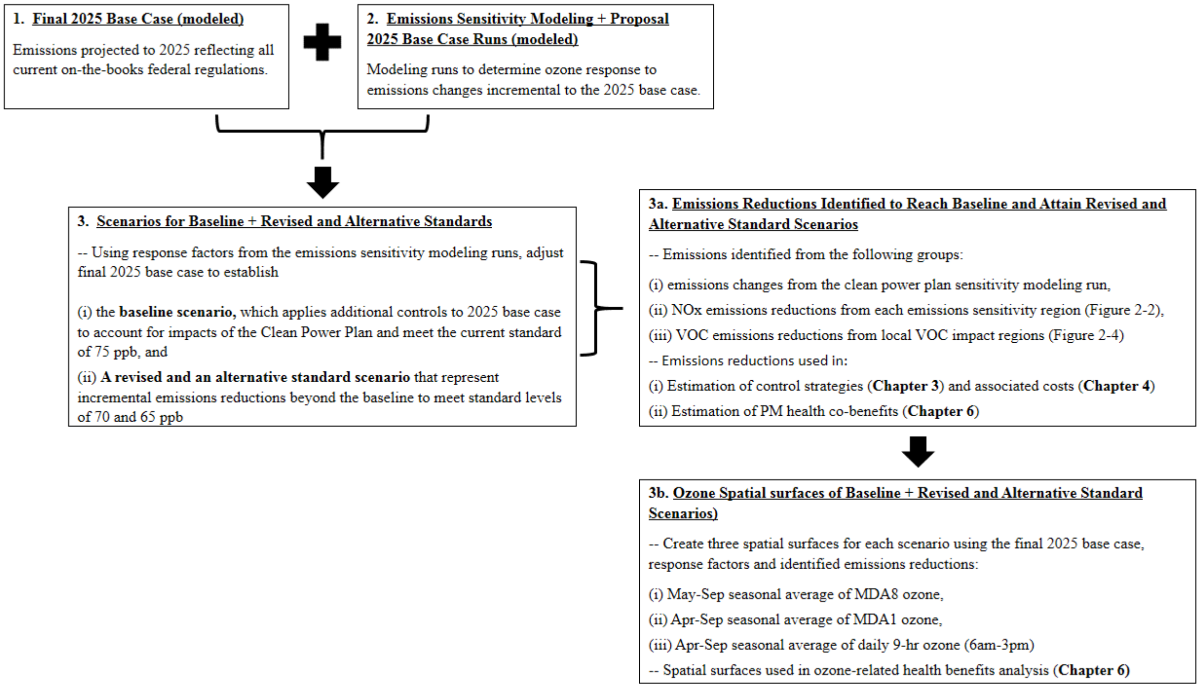

Figure 2-1. Process to Determine Emissions Reductions Needed to Meet Baseline and Alternative Standards Analyzed

Table 2-1. Terms Describing Different Scenarios Discussed in This Analysis

Scenario name	Definition
Base year	Photochemical model simulations for 2011 using best estimates or actual meteorology, emissions and resulting ozone concentrations
2025 base case	Modeling conducted with emissions projected to the year 2025 including all current on-the-books federal regulations and using 2011 meteorology
Proposal 2025 base case	The 2025 base case from the November 2014 proposal RIA
Final 2025 base case	The updated 2025 base case that includes improvements to 2011 emissions and 2025 emissions projections described in section 2.1
2025 baseline	2025 ozone concentrations from the final 2025 base case that have been adjusted to account for potential impacts from the proposed Clean Power Plan.[25] Costs and benefits of revised and alternative standard levels for all areas of the contiguous U.S. outside of California are calculated incremental to this scenario.
Post-2025 baseline	2025 ozone concentrations from the final 2025 base case that have been adjusted to account for potential impacts from the Clean Power Plan plus additional emissions reductions in California to attain of the current (75 ppb) ozone standard sometime after 2025. Costs and benefits of revised and alternative standard levels for California are calculated incremental to this scenario.
Revised standard	Emissions reductions and resulting ozone concentrations incremental to the baseline scenario needed to reach attainment of the 70 ppb ozone standard.
Alternative standard	Emissions reductions and resulting ozone concentrations incremental to the baseline scenario that would be needed to reach attainment of a 65 ppb ozone standard.
Emissions reductions from identified control strategies	The portion of emissions reductions and resulting ozone concentrations that come from identified emissions controls described in chapter 3
Emissions reductions from identified + unidentified control strategies	Total emissions reductions and resulting ozone concentrations that are applied to reach either the revised or alternative ozone standard

The remainder of the chapter is organized as follows: Section 2.1 describes the 2025 base case emissions and air quality modeling simulation; Section 2.2 describes how we project ozone levels into the future including the methodology for constructing the baseline, revised standard, and alternative standard scenarios (this methodology is applied in chapter 3 sections 3.1 and 3.2); and Section 2.3 describes the creation of spatial surfaces that serve as inputs to health benefits calculations discussed in Chapter 6.

2.1 Emissions and Air Quality Modeling Platform

The 2011-based modeling platform was used to provide emissions, meteorology and other inputs to the 2011 and 2025 air quality model simulations. This platform was chosen

[25] No additional reductions to meet the current (75 ppb) standard are applied since no areas outside of California are projected to violate the current standard once ozone adjustments for the Clean Power Plan are made.

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 6.11, Environ, 2014) for photochemical model simulations performed for the RIA. CAMx requires a variety of input files that contain information pertaining to the modeling domain and simulation period. These files include gridded, hourly emissions estimates and meteorological data, and initial and boundary conditions. Separate emissions inventories were prepared for the final 2011 base year, the final 2025 base case, the proposal 2025 base case and the 2025 emissions sensitivity simulations. 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. 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.

Information on the components of the 2011-based modeling platform, including information on the 2011 base year and 2025 base case emission inventories, and the model evaluation methodology and results are provided in Appendix 2A. Additional details on the final 2011 base year and 2025 base case emissions inventories can also be found in the Technical Support Document (TSD): Preparation of Emissions Inventories for the Version 6.2, 2011 Emissions Modeling Platform (US EPA, 2015). Section 4 of the TSD summarizes the control and growth assumptions by source type that were used to create the U.S. final 2025 base case emissions inventory and includes a table of those assumptions for each major source sector.

Section 2.4 of this document summarizes the changes to the emissions inventories used in the final modeling as compared to the November 2014 proposal modeling.

2.2 Projecting Ozone Levels into the Future

In this section we present the methods used to create the future baseline and the two scenarios that demonstrate attainment of the revised and alternative NAAQS levels analyzed in this RIA. First, in section 2.2.1, we describe the procedures for projecting ozone "design values" into the future. In section 2.2.2, we present the development of 15 emissions sensitivity simulations and in section 2.2.3 we show how to calculate ppb/ton ozone response factors from these sensitivity simulations. Next, in section 2.2.4, we describe the approach for using this information to construct the baseline, revised standard and alternative standard scenarios. The implementation of these methods using the 2025 base case ozone levels together with the ozone response factors and the resulting emissions scenarios and associated ozone levels is presented in Chapter 3. Finally, in section 2.2.5 we discuss a small subset of monitoring sites that were not included in the quantitative analysis.

2.2.1 Methods for Calculating Future Year Ozone Design Values

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 being analyzed. 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 finalized for the new NAAQS in Appendix U to 40 CFR Part 50 – Interpretation of the Primary and Secondary National Ambient Air Quality Standards for Ozone. For the purpose of this analysis, ozone DVs were derived from data reported in EPA's air quality system (AQS) for the years 2009-2013. The base period 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, 2014c) because it tends to minimize the year-to-year meteorologically-driven variability in ozone concentrations given that the future year meteorology is unknown.

For sites with fewer than five years of valid monitoring data available, the current year DV was calculated using a minimum of 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.

Future year ozone design values were calculated at monitor locations using the Model Attainment Test Software program (Abt Associates, 2014). This program 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 2-1 describes the recommended model attainment test in its simplest form, as applied for monitoring site *i*:

$$(\mathrm{DVF})_{\mathrm{i}} = (RRF)_i \times (DVB)_i \qquad \text{Equation 2-1}$$

$\mathrm{DVF_i}$ 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 ozone in the vicinity of the monitor that is simulated on high ozone days. The recently released draft version of EPA's ozone and $PM_{2.5}$ photochemical modeling guidance (US EPA, 2014c) 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 is calculated based on the 10 highest days in the base year modeling in the vicinity of the monitor location when the base 8-hr daily maximum ozone values were greater than or equal to 60 ppb for that day.[26] 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

[26] 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, 2014c). 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.

was greater than or equal to 60 ppb, as long as there were at least 5 days that met that criteria. At monitor locations with fewer than 5 days with ozone greater than or equal to 60 ppb, no RRF or DVF was calculated for the site and the monitor in question was not included in this analysis.

2.2.2 Emissions Sensitivity Simulations

A total of fifteen emissions sensitivity modeling runs were conducted to determine ozone response to reductions of NO_x and VOC emissions in different areas. (See Table 2-2 for a list of the sensitivity runs). The sensitivity modeling provides an efficient and flexible approach that allowed us to evaluate ozone responses from multiple source regions and several levels of emissions reductions simultaneously. All emissions sensitivity simulations included emissions reductions incremental to the proposal 2025 base case.[27] Ozone response factors (ppb/ton) were created by comparing changes in projected ozone levels between the proposal 2025 base case and the individual emission sensitivity simulations. These response factors were then applied to the final 2025 base case design values. There were three types of sensitivity runs, each of which is described in more detail below: (1) explicit emissions control cases; (2) across-the-board reductions in anthropogenic emissions in different areas; and (3) combination cases that included both explicit emissions controls and across-the-board reductions.

Table 2-2. List of Emissions Sensitivity Modeling Runs Modeled in CAMx to Determine Ozone Response Factors

Emissions Sensitivity Simulation	Region	Pollutant	Emissions Change	Types
1	National	All	Clean Power Plan	Explicit control
2	National	VOC	50% VOC cut	Across-the-board
3	California	NOx	CA explicit emissions control	Explicit control
4	N. California	NOx	Sensitivity 3 + 50% NOx cut in N. CA	Combination
5	N. California	NOx	Sensitivity3 + 90% NOx cut in N. CA	Combination
6	S. California	NOx	Sensitivity 3 + 50% NOx cut in S. CA	Combination
7	S. California	NOx	Sensitivity 3 + 90% NOx cut in S. CA	Combination

[27] Modeling incremental changes from the proposal 2025 base case provided consistency with sensitivity simulations performed for the proposal and allowed us to leverage a subset of sensitivity simulations created as part of that proposal. This was necessary due to timing and resource constraints. Since the sensitivity simulations are used to create relative ppb/ton response factors, it is appropriate to apply changes derived from these sensitivities to the final 2025 base case modeling since atmospheric chemistry regimes are not likely to have changed substantially between the proposal and final 2025 base case simulations.

8	Nevada	NOx	50% NOx cut	Across-the-board
9	Arizona/New Mexico	NOx	50% NOx cut	Across-the-board
10	Colorado	NOx	50% NOx cut	Across-the-board
11	E. Texas	NOx	50% NOx cut	Across-the-board
12	Oklahoma/Arkansas/Louisiana	NOx	50% NOx cut	Across-the-board
13	Great Lakes	NOx	50% NOx cut	Across-the-board
14	Ohio River Valley	NOx	50% NOx cut	Across-the-board
15	Northeast Corridor	NOx	50% NOx cut	Across-the-board

Explicit Emissions Controls: Two explicit emissions control sensitivity modeling runs were conducted. These emissions control sensitivity runs are referred to as "explicit emissions control" runs because they represent the impact of sets of specific controls rather than sensitivities to all anthropogenic emissions. First, we modeled one possible representation of implementing 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 Clean Power Plan sensitivity). Emissions for this simulation are described in the regulatory impact analysis for that proposed rule (EPA, 2014d). Second, we conducted an additional emissions control sensitivity run that included NOx emissions reductions from controls applied to specific sources in California. Based on analysis conducted for the November 2014 proposal RIA (EPA, 2014a) and projected design values (DVs)[28] from the final 2025 base case, it was determined that California was the only region for which all identified controls would be exhausted before reaching the baseline. Therefore, we created a sensitivity run in which all identified NO_x emissions controls below $15,000/ton were applied in California. The explicit controls were only applied in a 200 km buffer area around counties in California projected to violate 70 ppb in the proposal 2025 base case. The EPA's Control Strategy Tool (CoST) (EPA, 2014e) was used to determine the potential reductions in this area. NO_x controls were identified for all nonpoint, non-EGU point, and nonroad sources. This emissions sensitivity was created as part of the analysis for the November 2014 proposal RIA (EPA, 2014a). The assumptions about which sources were available for controls in California are the same as those described in Chapter 3, with the exception that for proposal, only controls with a cost under $15,000 per ton were considered. In

[28] 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 2.2.

the final rule we have identified additional controls in California (i.e. controls with a cost between $15,000 per ton and $19,000 per ton: the thresholds applied for proposal and final RIA respectively) which were not included in the California explicit emissions control case that we modeled but which were accounted for using ppb/ton response factors from combination emissions sensitivities described below.

Across-the-board Emissions Reductions: We performed across-the-board sensitivity modeling for areas of the U.S. projected to contain monitors with ozone design values greater than 65 ppb in the proposal 2025 base case. We created 8 regions that contain these monitoring sites, as shown in Figure 2-2. The boundaries of these regions were generally defined in terms of the borders of a single state or a small group of adjacent states. In addition, we also used the two "buffer regions" (one in East Texas and the other in the Northeastern U.S.) that were created in the analysis for the November 2014 proposal RIA for areas with 2025 baseline DVs above 70 ppb and were not updated using the final 2025 base case modeling. These buffers around counties projected to violate 70 ppb allowed us to target reductions in locations close to the highest ozone monitors, an approach that is likely to be most effective at reducing ozone concentrations for these relatively isolated violations.[29] The two buffer regions were determined based on 200 km buffers around all monitors projected to be above 70 ppb in the proposal 2025 base case. In Texas, the buffer region was restricted to counties within state boundaries. In the Northeast, the buffer was restricted to a subset of the states/counties that are currently under the jurisdiction of the Ozone Transport Commission (OTC), a multistate region that already has interstate cooperation for air quality planning. The Texas and Northeast buffer areas are shown in Figure 2-3.[30] Unlike in California, it was not clear that all identified controls would be required in any one region to meet the 65 and/or 70 ppb standard levels. Therefore, in these two regions we generated more general emissions response factors using an across-the-board 50%

[29] Note that counties projected to violate the alternative 65 ppb standard are more broadly distributed throughout the U.S. and less isolated in nature. Therefore it may be less important to differentiate between impacts from very local emissions within 200 km of a violating county compared to impacts from emissions across a statewide or multistate region in designing control strategies for those areas.
[30] The 200 km buffers are shaded in orange and counties that contained one or more monitors projected to be above 70 ppb in the proposal 2025 base case modeling are shaded in blue.

reduction in U.S. anthropogenic NOx emissions. We also performed a VOC sensitivity run with a 50% cut in anthropogenic VOC emissions across the 48 contiguous states.

Combination Emissions Sensitivities: We conducted four additional emissions sensitivity modeling runs that combined the explicit emissions controls with across-the-board reductions in California. Based on a previous EPA analysis (EPA, 2014a; EPA, 2014f) we identified California as the region most likely to need NOx reductions beyond 50% to reach the revised and alternative standard levels. Therefore, we modeled both a 50% and a 90% NOx emissions reduction in California to capture nonlinearities in ozone response to large NOx emissions changes. The 50% and 90% NOx reductions were applied in Northern and Southern California separately recognizing that the topography in California effectively isolates the air shed in the San Joaquin Valley from the southernmost portion of the state which has the effect of limiting the impact of emissions from Southern California on ozone in Northern California and vice versa. The geographic delineation of Northern and Southern California for these emissions sensitivity simulations is shown in Figure 2-2. In all four California emissions sensitivities, the 50% and 90% NOx reductions were applied on top of the California explicit controls sensitivity run (sensitivity simulation #3).

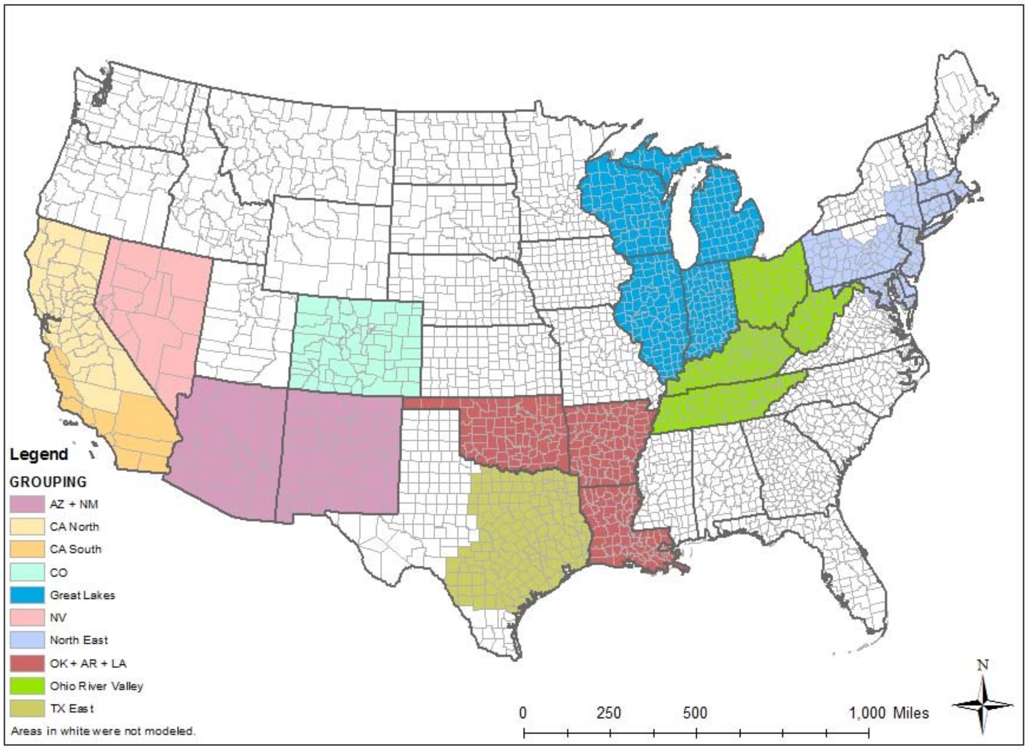

Figure 2-2. Across-the-Board Emissions Reduction and Combination Sensitivity Regions[31]

[31] Combination Sensitivities were used for the two California regions whereas, Across-the-Board Sensitivities were used in all other regions.

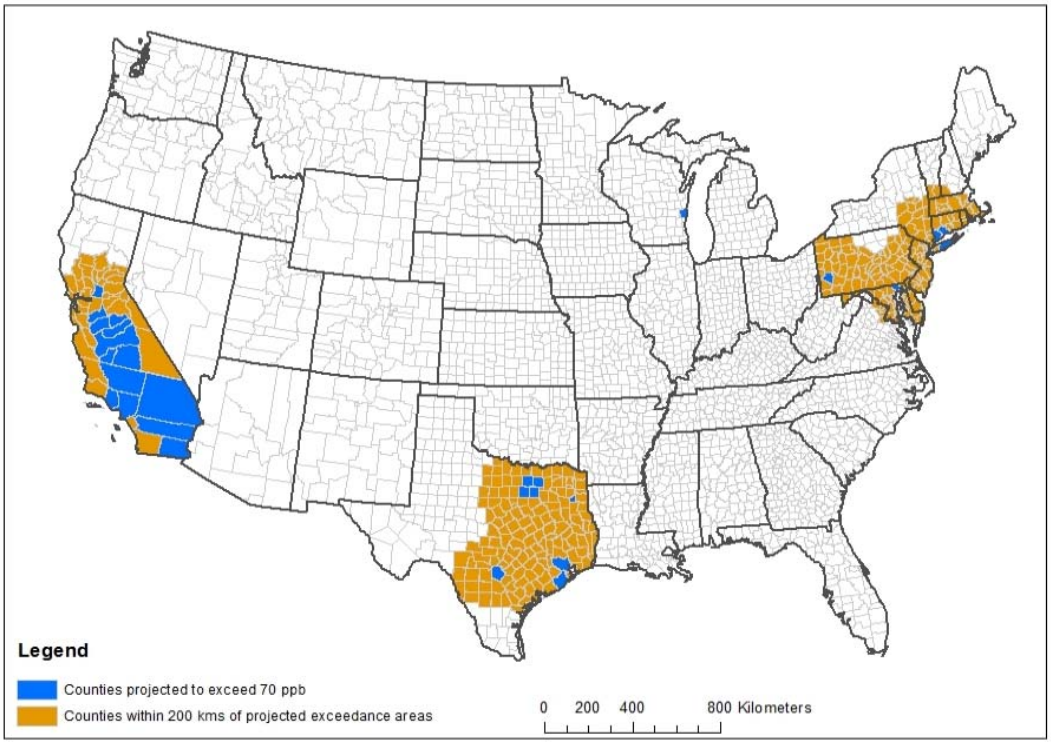

Figure 2-3. Map of 200 km Buffer Regions in California, East Texas and the Northeast Created as Part of the Analysis for the November 2014 Proposal RIA[32]

2.2.3 Determining Ozone Response Factors from Emissions Sensitivity Simulations

Section 2.2.1 describes, in general terms, how the 2025 projections for ozone DVs were computed. This procedure was followed for the proposal and final 2025 base case modeling and for each of the fifteen emissions sensitivity modeling simulations. Using the projected DVs and corresponding emissions changes, a unique ozone response factor (ppb/ton) was calculated for each emissions sensitivity at each ozone monitor using equation 2-2:

[32] The California buffer was used to determine the area over which explicit controls were applied in the California explicit control sensitivity simulation (sensitivity simulation #3). The Texas and Northeast buffers were used to delineate the areas over which across-the-board anthropogenic NOx emissions reductions were applied in sensitivity simulations #11 and #15 respectively.

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

In equation 2-2, $R_{i,j}$ represents the ozone response at monitor j to emissions changes between the 2025 proposal base case and the sensitivity simulation i; $DV_{i,j}$ represents the DV at monitor j for emissions sensitivity i; $DV_{2025base,j}$ represents the DV at monitor j in the proposal 2025 base case; and ΔE_i represents the difference in NO_x or VOC emissions (tons) between the proposal 2025 base case and emissions sensitivity run i.

In California where emissions reductions in four sensitivity runs (i) were incremental to emissions reductions in another run (k), the following equation was used:

$$R_{i,j} = \frac{DV_{i,j} - DV_{k,j}}{\Delta E_{ik}} \qquad \text{Equation 2-3}$$

in which ΔE_{ik} represents the difference in NO_x emissions (tons) between the emissions run k and emissions run i. For emissions sensitivity simulations #4 and #6 (50% NOx reductions), k represented emissions sensitivity #3 (California explicit control). For emissions sensitivity simulations #5 and #7 (90% NOx reductions), k represented emissions sensitivities #4 and #6 respectively.

For the VOC emissions sensitivity run, we determined it was appropriate to compute response factors for smaller geographic areas than were modeled in the emissions sensitivity simulations shown in Figure 2-2. Past work has shown that impacts of anthropogenic VOC emissions on ozone DVs in the U.S. tend to be much more localized than reductions in NOx (Jin et al., 2008). Consistent with past analyses (US EPA, 2008) we made the simplifying assumption that VOC reductions do not affect ozone at distances more than 100 km from the emissions source. Consequently, we created a series of VOC impact regions in 7 areas (Figure 2-4) for which our modeling showed that ozone is responsive to VOC emissions reductions and which had the highest ozone DVs in the NOx sensitivity regions: New York City, Chicago, Louisville, Houston, Denver, and Northern and Southern California.[33] VOC impact regions were

[33] The following additional local VOC areas were also explored but were found not to be helpful in reaching the revised or alternative NAAQS levels in this analysis: Dallas, Detroit, Pittsburgh, and Baltimore. This may be due to the construct of the attainment scenarios analyzed and does not mean that VOC controls would not be effective in these areas under alternative assumptions about regional NOx controls.

delineated by creating a 100km buffer around counties containing monitors violating 60 ppb in the proposal 2025 base case modeling. In addition, VOC impact regions were constrained by state boundaries except in cases where a current nonattainment area straddled multiple states (e.g., 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 affect locations in Wisconsin, Illinois, Indiana, and Michigan that border Lake Michigan (Dye et al., 1995). For California, the VOC impact regions were delineated identically to the Northern and Southern California regions used in the NOx emissions sensitivity runs except that the Northern California region did not extend beyond the 200 km buffer shown in Figure 2-3. To create the ozone response factors to VOC for each monitoring site within a VOC impact region, 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 emissions sensitivity modeling run.

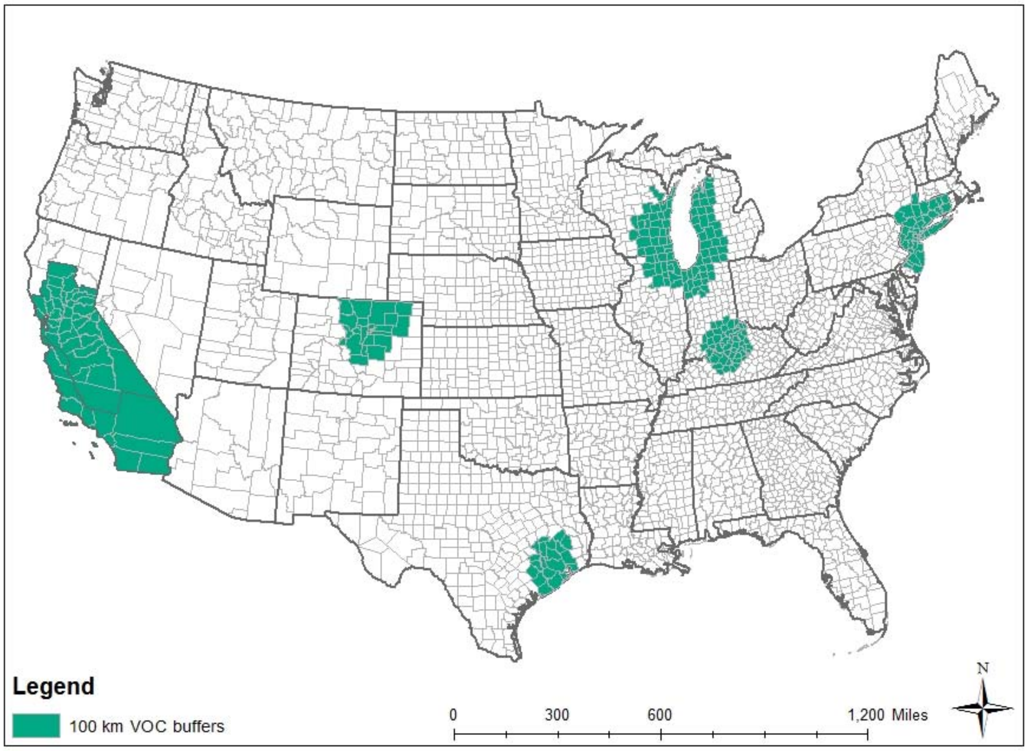

Figure 2-4. Map of VOC Impact Regions

2.2.4 Combining Response from Multiple Sensitivity Runs to Determine Tons of Emissions Reductions to Meet Various NAAQS Levels

Ozone DVs were calculated for the baseline scenario as well as for the revised and alternative standards using Equation 2-4 in which $DV_{2025,j}$ is the ozone DV at monitor j in the final 2025 base case, $R_{n,j}$ is the ozone response factor for sensitivity n at monitor j, and ΔE_n is the tons of emissions reductions from region n being applied to reach the desired standard level:

$$DV_j = DV_{2025,j} + \left(R_{1,j} \times \Delta E_1\right) + \left(R_{2,j} \times \Delta E_2\right) + \left(R_{3,j} \times \Delta E_3\right) + \cdots \qquad \text{Equation 2-4}$$

For the baseline as well as the two alternative standards analyzed, we determine the least amount of emissions reductions (tons) needed in each region (ΔE_n) to bring the ozone DVs at all monitors down to the particular standard level being analyzed. Note that California was analyzed independent of the rest of the country due to the later attainment dates in many California counties. Therefore, in determining the necessary emissions reductions, we did not account for any impacts of California reductions on other areas of the U.S. and vice versa. The application of equation 2-4 to determine emissions reductions necessary to meet the various standard levels at U.S. locations outside of California is presented in chapter 3, section 3.2.

Because California included multiple incremental sensitivity simulations, Equation 2-4 had to be slightly modified for calculating DV changes to emissions reductions in that state. The modeled impacts from multiple California sensitivity simulations were combined in a linear manner to estimate the overall impacts. For example, at any monitor in California we could use the following equation to determine the DVs that would result from a 75% reduction in Northern California emissions beyond the explicit emissions control sensitivity simulation:

$$DV_{75\%CA,j} = DV_{2025,j} + \left(R_{CA_{explicitcontrol},j} \times \Delta E_{CA_{explicitcontrol}}\right) + \left(R_{NCA50NOx,j} \times \Delta E_{50NOx}\right) + \left(R_{NCA90NOX,j} \times \Delta E_{75NOX}\right) \quad \text{Equation 2-5}$$

In equation 2-5, $DV_{2025,j}$ represents the projected DV from the final 2025 base case at monitor j, $\Delta E_{NE\_explicitcontrol}$ represents the difference in NO_x emissions between the proposal 2025 base case and the 2025 California explicit emissions control sensitivity; ΔE_{50NOx} represents the difference in NO_x emissions between the 2025 California explicit emissions control sensitivity and the combined California explicit emissions control with 50% Northern California NOx cuts sensitivity; and ΔE_{75NOX} represents the additional emissions reductions needed to reach a 75% NOx cut in Northern California above and beyond the emissions reductions in the combined California explicit emissions control run with 50% Northern California NOx cuts run. Note that in this equation, emission reductions in Northern California impact monitors (j) in both the Northern and Southern California regions. Similar to the methods applied in other regions, we determine the smallest amount of emissions reductions (tons) in northern and southern California regions necessary to decrease all ozone DVs in each region to the standard level being analyzed. The application of equation 2-5 to determine emissions reductions necessary to meet the various standard levels in California is presented in chapter 3, section 3.3.

While ozone responses can be nonlinear and vary by emissions source type and location, in this analysis we make several simplifying assumptions. First, we assume that every ton of NO_x or VOC reduced within a region results in the same ozone response regardless of where the emissions reductions come from within the region because we do not have any information on the differential ozone response from emissions changes at different locations within the region. However, the somewhat smaller emissions sensitivity regions used in this analysis compared to the November 2014 proposal RIA provide a more spatially resolved representation of the ozone response to emissions changes and thus reduces but does not eliminate this uncertainty. Second, we assume that NO_x and VOC responses are additive. Third, we assume that the responses from multiple regions are additive. Fourth, 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 California where we have multiple levels of emissions reductions, we assume linearity within each simulation, but we are able to capture discrete shifts in ozone response based on the multiple sensitivity simulations (i.e., one response for explicit emissions control run reductions, another response level up to 50% NO_x emissions reductions beyond the explicit emissions control run, and a third level of response between 50% and 90% NO_x emissions reductions beyond the explicit emissions control run). Finally, outside of California, the ozone response to NO_x reductions greater than 50% is based on an extrapolation beyond the modeled emissions reductions. However, only East Texas and the Northeast require NOx reductions greater than 50% in the 65 ppb scenario and in both cases the NOx reductions are not substantially greater than 50% (52% and 56% respectively), so we expect that the ozone response from the 50% sensitivity is appropriate for extrapolation to 52% and 56% with only a small amount of additional uncertainty.

2.2.5 Monitoring Sites Excluded from Quantitative Analysis

There were 1,225 ozone monitors with complete ozone data for at least one DV period covering the years 2009-2013. We included 1,165, or 95% of these sites in the analysis to determine the tons of emissions reductions necessary for each of the three scenarios (i.e., the baseline and two alternative standard level scenarios). However, there were three types of sites that were excluded from this analysis. First, we did not analyze the baseline or attainment levels at each of the 41 sites that did not have a valid projected final 2025 base case DV because there were fewer than 5 modeled days above 60 ppb in the 2011 CAMx simulation, as required in the EPA SIP modeling guidance (US EPA, 2014c). It is unlikely that these sites would have any substantial impact on costs and benefits because the reason that projections could not be made is that they have no more than 4 modeled days above 60 ppb. Only one of these 41 sites (site 311079991 in Knox County, NE) has a base year DV greater than 65 ppb. These sites are listed in Appendix 2A.

Second, seven 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 here based on summertime conditions to a wintertime ozone event, which is driven by different characterizations 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 2A 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 12 relatively remote, rural sites in the Western U.S. with projected baseline DVs between 66 and 69 ppb that showed limited response to the NO_x and VOC emissions sensitivities we modeled. Air agencies responsible for attainment at 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)
- 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. These sites were initially identified in the November 2014 RIA proposal (EPA, 2014a) and more detailed discussion of their characteristics were provided in Appendix 3A of that document. Only the subset of those sites with DVs greater than 65 ppb in the 2025 baseline scenario are excluded in this analysis since sites projected to have DVs at or below 65 ppb would not incur any additional costs or benefits.

2.3 Creating Spatial Surfaces for BenMap

The emissions reductions for attainment of the current, revised, and an alternative NAAQS level determined in chapter 3 were used to create spatial fields of ozone concentrations (i.e., spatial surfaces) for input into the calculation of health benefits associated with attainment of each NAAQS level, incremental to the baseline. The spatial surfaces used to calculate ozone-related health benefits with the BenMap tool (Chapter 6) are described below.

Health benefits associated with meeting different ozone standard levels were calculated based on the following three ozone metrics, as described in more detail in Chapter 6: May-Sep seasonal mean of 8-hr daily maximum ozone, Apr-Sep seasonal mean of 1-hr daily maximum ozone, and May-Sep seasonal mean of 9-hr daily average ozone (6am-3pm). For each metric, spatial fields (i.e., gridded surfaces) were created for a total of 8 scenarios, including:

- 2025 baseline
- post-2025 baseline
- 2025 70 ppb identified control strategies
- 2025 70 ppb identified + unidentified control strategies
- post-2025 70 ppb identified + unidentified control strategies
- 2025 65 ppb identified control strategies
- 2025 65 ppb identified + unidentified control strategies

- post-2025 65 ppb identified + unidentified control strategies

The surfaces created for the 2025 scenarios represent attainment at all contiguous U.S. monitors outside of California, while the surfaces for the post-2025 scenarios represent all contiguous U.S. monitors including those in California meeting the standard being evaluated. The effects due only to California meeting the standard are isolated in Chapter 6 through a series of BenMap simulations using these surfaces and varying assumptions about population demographics. In addition, for the 2025 scenarios we include "identified control" and "identified + unidentified control" strategies in which the identified control strategies only include ozone changes resulting from emissions reductions from identified control measures, while the identified + unidentified controls strategies include ozone changes resulting from all emissions reductions necessary to attain the standard from both identified controls and unidentified measures.

The ozone surfaces were created using the following steps, which are described in more detail below and depicted in Figure 2-5.

- Step 1: Create spatial fields of gridded ozone concentrations for each of the three seasonal metrics using the model-predicted hourly ozone concentrations.
- Step 2: Create spatial fields of gridded ozone response factors for each seasonal metric.
- Step 3: Create spatial field of gridded ozone concentrations for baseline, revised standard, and alternative standard scenarios and each seasonal ozone metric
- Step 4: Create 2011 enhanced Voronoi Neighbor Averaging (eVNA) fused surface of 2011 modeled and 2010-2012 observed values for each seasonal ozone metric
- Step 5: Create eVNA fused modeled/monitored surface for each attainment scenario and each seasonal ozone metric

<u>Step 1: Create spatial fields of seasonal ozone metrics for each model simulation</u>

- Inputs: Hourly gridded model concentrations for final 2011 base year, proposal and final 2025 base cases, and the fifteen 2025 emissions sensitivity simulations detailed in Section 2.2.2
- Outputs: Seasonal ozone metrics for 2011, proposal and final 2025 base cases, and fifteen 2025 emissions sensitivity simulations (18 total spatial fields for each metric)

Step 2: Create spatial fields of ppb/ton ozone response factors

- Inputs: Seasonal ozone metrics for proposal 2025 base case and fifteen 2025 emissions sensitivity simulations **(from Step 1)**; Amount of emissions reductions (tons) modeled in each emissions sensitivity
- Outputs: Gridded ozone response factor (ppb/ton) for each seasonal ozone metric from each emissions sensitivity simulation
- Methods:
  - Calculate the change in the seasonal ozone metrics between each emissions sensitivity simulation (i) and the proposal 2025 base case. This step results in 15 spatial fields of gridded ozone changes (ΔO_3) for each seasonal ozone metric.
  - Divide each of the spatial fields of ozone changes by the tons of emissions reductions applied in that emissions sensitivity simulation compared to the proposal 2025 base case: ($\frac{\Delta O_3}{\Delta E}$). This step results in 15 spatial fields of gridded ozone response factors (ppb/ton) for each seasonal ozone metric.

Step 3: Create spatial field seasonal ozone metrics for baseline, revised standard, and alternative standard scenarios

- Inputs: Gridded ozone response factor for each seasonal ozone metric from each emissions sensitivity simulation **(from Step 2)**; Amount of emissions reductions from each region (from Appendix 3A); Gridded ozone surface for each seasonal metric from the final 2025 base case **(from Step 1)**.
- Outputs: Gridded seasonal ozone metrics for each attainment scenario
- Methods:
  - The gridded ozone response factors from Step 2 were multiplied by the relevant tons of emissions reductions for each sensitivity and then summed to create a gridded field representing the scenario using question (Equation 2-6)

$$O3_{xy,s,m} = O3_{xy,2025,m} + \left(R_{xy,1,m} \times \Delta E_{1,s}\right) + \left(R_{xy,2,m} \times \Delta E_{2,s}\right) + \left(R_{xy,2,m} \times \Delta E_{3,s}\right) + \cdots \qquad \text{Equation 2-6}$$

In equation 2-6, $ozone_{xy,s,m}$ 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 final 2025 base case simulation at grid cell *x,y* aggregated to metric *m*. $R_{xy,1,m}$ represents the ozone 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 that were found to be necessary for scenario *s*.

Identified control strategy ozone surfaces at each standard level were created by only including ΔE values for emissions coming from identified controls as described in Chapter 3. Post-2025 surfaces include all emissions reductions outside of California that we estimate in 2025 plus additional reductions in California which would occur after 2025.[34]

Step 4: Create 2011 enhanced Voronoi Neighbor Averaging (eVNA) fused surface of 2011 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) (**from Step 1)**
- Outputs: 2011 fused modeled/monitored surfaces for each seasonal ozone metric
- Methods: 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: Create eVNA fused modeled/monitored surfaces for baseline, revised standard, and alternative standard scenarios

[34] A small error was discovered in the post-2025 surfaces in that the baseline surface included 202,000 tons of NOx emission reductions in California rather than the actual 206,000 tons of NOx emissions reductions applied to reach 75 ppb in California. This error was carried through to the 70 ppb and 65 ppb surfaces for the post-2025 scenarios so the incremental changes in ozone between the baseline and alternative NAAQS level surfaces should not be significantly impacted.

- Inputs: 2011 fused model/observed surfaces for each seasonal ozone metric (**from Step 4)**; modeled seasonal ozone metrics (gridded fields) for 2011 (**from Step 1)** and each attainment scenario (**from Step 3**).
- Outputs: Fused modeled/monitored surface for each attainment scenario and each seasonal ozone metric
- Methods: The 24 model-based surfaces (i.e., 8 scenarios and 3 metrics) were used as inputs in the MATS tool along with the gridded 2011 base year and 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 base year 2011 model field. This RRF field was then multiplied by the 2011 eVNA field to create a gridded eVNA field for each scenario.

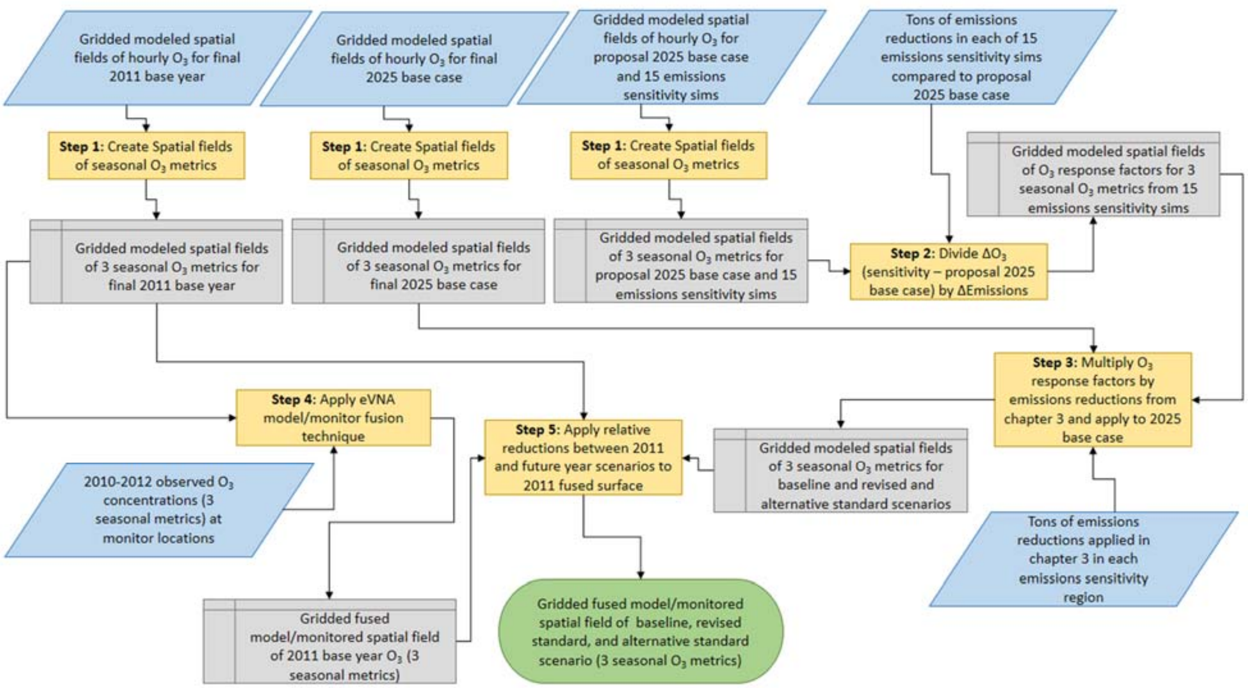

Figure 2-5. Process Used to Create Spatial Surfaces for BenMap

2.4 Improvements in Emissions and Air Quality for the Final RIA

2.4.1 Improvement in Emissions

Between proposal and the final rule, improvements were implemented in both the base (2011) and future year (2025) emissions scenarios. The proposal emissions are documented in the Version 6.1, 2011 Emissions Modeling Platform (US EPA, 2014b) TSD. Many improvements to the inventories resulted from the Federal Register notices for the 2011 and 2018 Emissions Modeling platforms released in November 2013 and June 2014. Comments on these notices were received from states, industry, and other organizations. Although the 2025 emissions were not specifically released for comment, improved methodologies and data were also applied to the updated 2025 emissions wherever possible. For example, many improvements were made on the National Electric Energy Data System database that is a key input in the preparation of future year EGU inventories; state agencies and regional planning organizations provided specific growth and control factors for stationary sources; and improvements were made to the modeling of onroad mobile sources in the base and future years.

Most updates to the 2011 emissions are reflected in the 2011 National Emissions Inventory (NEI) version 2. These updates included 1) the use of the 2014 version of the Motor Vehicle Emissions Simulator (MOVES2014) for onroad mobile source emissions, along with many upgrades to the input databases used by MOVES; 2) updated oil and gas emissions based on the Oil and Gas Emissions Estimate Tool version 2.0; 3) version 3.6.1 of the Biogenic Emission Inventory System along with improved land use data; 4) many updates to point and nonpoint source emissions submitted directly into the Emission Inventory System (EIS) by states; 5) improved temporal allocation of electric generating unit (EGU) and onroad mobile source emissions; 6) upgraded VOC speciation to be consistent with the most recent chemical mechanism available in CAMx (i.e., CB6); and 7) improved spatial surrogates for heavy-duty trucks, buses, and other types of vehicles. In addition, Canadian emissions were upgraded to the latest available data from Environment Canada for the year 2010 and Mexican emissions were upgraded to use the 2008 Inventario Nacional de Emisiones de Mexico, whereas for the proposal modeling the Mexican emissions had been based on those developed for 1999. The cumulative national impact of the changes to 2011 emissions between the proposal and final RIA resulted in a 1% increase in NOx emissions and no change in VOC emissions, although local changes were larger.

Improvements to the 2025 emissions included 1) using the Integrated Planning Model (IPM) version 5.14 with associated input databases and a representation of the Cross-State Air Pollution Rule (CSAPR); 2) using MOVES2014 to represent emission reductions from the Tier 3 Final rulemaking and recent light and heavy duty greenhouse gas mobile source rules; 3) use of

Annual Energy Outlook (AEO) 2014 for projections of vehicle miles traveled, oil and gas growth, and growth in other categories; and 4) improved representation of growth and controls for non-EGU stationary source emissions. The cumulative national impact of the changes to the 2025 emissions between the proposal and final RIAs resulted in a 2% reduction in NOx emissions and a 1% increase in VOC emissions, although localized changes were larger. For more information on the improvements to the 2025 emissions, see Appendix 2A and the Emissions Modeling TSD.

The net effects of the emissions inventory, model, and model input updates are changes in projected 2025 ozone air quality design values (DVs) in many areas. These new projected DVs were higher than previously modeled for the proposal RIA in some locations and lower in others. The new projections show lower 2025 DVs in Central Texas from Houston to Dallas, the El Paso area (NM and TX) and Big Bend, Texas, and several states in the central U.S., including Oklahoma, Kansas, Missouri, Arkansas, Mississippi, Tennessee, and southern Kentucky. The new projections also show higher 2025 DVs in Denver, Las Vegas, Phoenix, Charlotte, the upper Midwest, and some parts of the New York/New Jersey areas. See Appendix 2A, Section 2A.4 for detailed information on the updated DVs.

2.4.2 Improvements in Air Quality Modeling

In this final RIA, we used emissions sensitivity simulations to determine the response of ozone at monitor locations to emissions changes in specific regions, similar to the approach used in the November 2014 proposal RIA (EPA, 2014a). However, when we reviewed the analysis for the November 2014 proposal RIA we determined that in certain locations (e.g., Texas and the Northeast) where violations of the 70 ppb scenario were limited to fairly localized areas, the analysis could be improved by using more geographically refined ozone response factors. In addition, we determined that smaller regions would also provide more refined ozone responses across the rest of the U.S. As a result, in this final RIA we designed 10 smaller regions to determine ozone response factors (see Figure 2-2), compared to the 5 larger regions used in the proposal RIA (see Figure 3-3 in EPA, 2014a). This more geographically refined resolution allows us to more accurately represent the increased effectiveness of emissions reductions closer to monitor locations compared to emissions reductions from sources that are further away. For example, in the proposal RIA, we analyzed one large Southwest region and made no differentiation between the impacts of emissions from Nevada, Utah, New Mexico, Arizona, or Colorado on the monitors in Denver. In the final RIA, the smaller regions allow us to differentiate the impact of NOx emissions reductions in Colorado on ozone concentrations in Denver compared to NOx emissions reductions in Arizona and New Mexico on ozone in Denver. Similarly, in the final RIA we differentiate the impacts of east Texas emissions on ozone at Dallas and Houston monitors from impacts of emissions in west Texas, Louisiana, Oklahoma, Mississippi, Arkansas, Kansas and Missouri on those same monitors (in the proposal RIA, we used one large central U.S. region that did not differentiate these impacts).

In Texas and the Northeast, the improved response factors resulted in larger changes in ozone concentrations in response to the more geographically focused emissions reductions. For example, in east Texas, emissions reductions were 2 to 3 times more effective at reducing ozone concentrations at controlling monitors in Houston and Dallas than equivalent regional emissions reductions used in the proposal. In the Northeast, local emissions reductions were 2.5 times more effective at reducing ozone concentrations at the controlling monitor on Long Island, NY than the equivalent regional emissions reductions used in the proposal. The more geographically refined modeling and improved ozone response factors resulted in fewer emissions reductions needed to meet a revised standard of 70 ppb and an alternative standard level of 65 ppb. For additional discussion on how these improved response factors affect emissions reductions needed to reach a revised standard of 70 ppb and an alternative standard level of 65 ppb, see Chapter 3, Section 3.3. For additional discussion on how the improved response factors and reduced emissions reductions impact cost estimates, see Chapter 4, Section 4.6, and for additional discussion on how this impacts benefits estimates, see Chapter 6, Section 6.1.

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APPENDIX 2A: ADDITIONAL AIR QUALITY ANALYSIS AND RESULTS

2A.1 2011 Emissions and Air Quality Modeling Platform

2A.1.1 Photochemical Model Description and Modeling Domain

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 2A-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.

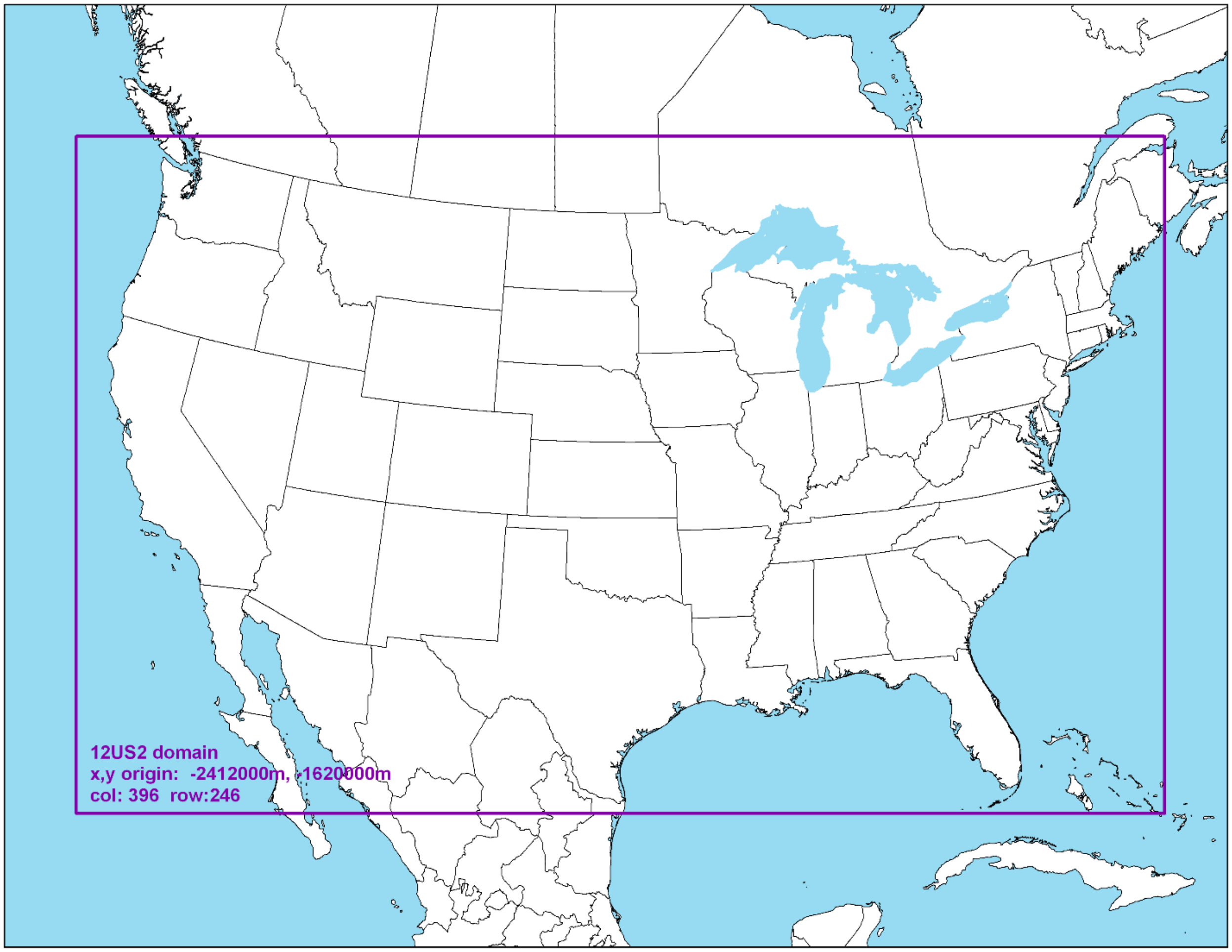

Figure 2A-1. Map of the CAMx Modeling Domain Used for Ozone NAAQS RIA

2A.1.2 Meteorological Inputs, Initial Conditions, and Boundary Conditions

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.

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 standard version 8-03-02 (Yantosca and Carouge, 2010) 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.

2A.1.3 2025 Base Case Emissions Inputs

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 were preprocessed into CAMx-ready inputs using the Sparse Matrix Operator Kernel Emissions (SMOKE) modeling system (Houyoux et al., 2000).

The 2025 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 IPM[35] version 5.14 (http://epa.gov/powersectormodeling/psmodel514.html). 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. Improvements to the National Electric Energy Data System database, a key input in the preparation of future year EGU inventories, were implemented as a result of updated information becoming available and based on comments submitted in response the January 2014 Federal Register notice. 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 CSAPR issued July 6, 2011.[36]

Projections for most stationary emissions sources other than EGUs (i.e., non-EGUs) were developed by using the EPA Control Strategy Tool (CoST) to create post-controls future year inventories. CoST is described in chapter 4 (section 4.1.1) and 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 received by EPA in response to the January 2014 Federal

[35] IPM is a multiregional, dynamic, deterministic linear programming model of the U.S. electric power sector.
[36] An emissions modeling sensitivity run described in Section 2.2.2 also includes a representation of EPA's proposed carbon pollution guidelines under section 111(d) of the Clean Air Act (CAA).

Register Notice, along with emissions reductions due to national and local rules, control programs, plant closures, consent decrees and settlements. Some improvements made based on comments included the use of growth and control factors provided by states and by regional organizations on behalf of states. Reductions to criteria air pollutant (CAP) emissions from stationary engines resulting as cobenefits to the Reciprocating Internal Combustion Engines (RICE) National Emission Standard for Hazardous Air Pollutants (NESHAP) are included. Reductions due to the New Source Performance Standards (NSPS) VOC controls for oil and gas sources, and the NSPS for process heaters, internal combustion engines, and natural gas turbines are also included.

Regional projection factors for point and nonpoint oil and gas emissions were developed using Annual Energy Outlook (AEO) 2014 projections from year 2011 to year 2025 **(**http://www.eia.gov/forecasts/aeo/**)**. Projected emissions for corn ethanol, cellulosic 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.

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. Fugitive dust for paved and unpaved roads take growth in VMT and population into account. 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. Impacts from the New Source Performance Standards (NSPS) for wood burning devices are also included.

Projection factors for the remaining nonpoint sources such as stationary source fuel combustion, industrial processes, solvent utilization, and waste disposal, reflect emissions reductions due to control programs along with comments on the growth and control of these sources as a result of the January 2014 Federal Register notice and information gathered from prior rulemakings and outreach to states on emission inventories. Future year portable fuel container (PFC) inventories reflect the impact of the final Mobile Source Air Toxics (MSAT2).

The MOVES2014-based 2025 onroad mobile source 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 2017 and Later Model Year Light-Duty Vehicle Greenhouse Gas Emissions and Corporate Average Fuel Economy Standards (LD GHG), the Renewable Fuel Standard (RFS2), the Mobile Source Air Toxics Rule, the Light Duty Green House Gas/Corporate Average Fuel Efficiency (CAFE) standards for 2012-2016, the Greenhouse Gas Emissions Standards and Fuel Efficiency Standards for Medium- and Heavy-Duty Engines and Vehicles, the Light-Duty Vehicle Tier 2 Rule, and the Heavy-Duty Diesel Rule. 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 emissions sectors because that information was not included in the provided inventories. The input databases for MOVES, the methods for projecting activity data, and the emissions estimation methods implemented with MOVES were improved from those used in the proposal modeling with some improvements based on comments received via the January 2014 Federal Register notice.

The nonroad mobile 2025 emissions, including railroads and commercial marine vessel emissions also include all national control programs. These control programs include the Clean Air Nonroad Diesel Rule – Tier 4, the Nonroad Spark Ignition rules, and the Locomotive-Marine Engine rule. For ocean-going vessels (Class 3 marine), 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. 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.

All modeled 2011 and 2025 emissions cases use the 2010 Canada emissions data. Note that 2010 is the latest year for which Environment 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, emissions compiled from the Inventario Nacional de Emisiones de Mexico, 2008 were used for 2011, as that was the latest complete inventory available. For 2025, projected emissions for the year 2025 based on the 2008 inventory were used (ERG, 2014). Offshore oil platform emissions for the United States represent the year 2011 and are consistent with those in the 2011 National Emissions Inventory, version 2. Biogenic and fire emissions were held constant for all emissions cases and were based on 2011-specific data. Table 2A-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 2A-1. 2011 and 2025 Base Case NOx and VOC Emissions by Sector (thousand tons)

Sector	2011 NO_x	2025 NO_x	2011 VOC	2025 VOC
EGU-point	2,000	1,400	36	42
NonEGU-point	1,200	1,200	800	830
Point oil and gas	500	460	160	190
Wild and Prescribed Fires	330	330	4,700	4,700
Nonpoint oil and gas	650	720	2,600	3,500
Residential wood combustion	34	35	440	410
Other nonpoint	760	790	3,700	3,500
Nonroad	1,600	800	2,000	1,200
Onroad	5,700	1,700	2,700	910
C3 Commercial marine vessel (CMV)	130	100	5	9
Locomotive and C1/C2 CMV	1,100	680	48	24
Biogenics	1,000	1,000	41,000	41,000
TOTAL	**15,000**	**9,300**	**58,000**	**56,000**

2A.1.4 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. The purpose of this evaluation is to examine the ability of the 2011 air quality modeling platform to represent the magnitude and spatial and temporal variability of measured (i.e., observed) ozone concentrations within the modeling domain. The evaluation presented here is based on model simulations using the v2 version of the 2011 emissions platform (i.e., case name 2011eh_cb6v2_v6_11g, also called the "final RIA 2011 base year" in chapter 2)[37]. The model evaluation for ozone focuses on 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) (Figures 2A-2a and 2A-2b).

Included in the evaluation are statistical measures of model performance based upon model-predicted versus observed concentrations that were paired in space and time. Model performance statistics were calculated for several spatial scales and temporal periods. Statistics were calculated for individual monitoring sites and for each of nine climate regions of the 12-km U.S. modeling domain. The regions include the Northeast, Ohio Valley, Upper Midwest,

[37] For an evaluation of the proposal RIA 2011 base year modeling, please see appendix 3-A of EPA, 2014d.

Southeast, South, Southwest, Northern Rockies, Northwest and West[38,39], which are defined based upon the states contained within the National Oceanic and Atmospheric Administration (NOAA) climate regions (Figure 2A-3)[40] as were originally identified in Karl and Koss (1984).

For maximum daily average 8-hour (MDA8) ozone, model performance statistics were created for each climate region for the May through September ozone season.[41] In addition to the performance statistics, we prepared several graphical presentations of model performance for MDA8 ozone. These graphical presentations include:

(1) density scatter plots of observed AQS data and predicted MDA8 ozone concentrations for May through September;

(2) regional maps that show the mean bias and error as well as normalized mean bias and error calculated for MDA8 $\geq$ 60 ppb for May through September at individual AQS and CASTNet monitoring sites;

(3) bar and whisker plots that show the distribution of the predicted and observed MDA8 ozone concentrations by month (May through September) and by region and by network; and

(4) time series plots (May through September) of observed and predicted MDA8 ozone concentrations for 12 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 evaluation of the ozone predictions in the 2011 CAMx modeling platform, we have selected the mean bias, mean

[38] The nine climate regions are defined by States where: Northeast includes CT, DE, ME, MA, MD, NH, NJ, NY, PA, RI, and VT; Ohio Valley includes IL, IN, KY, MO, OH, TN, and WV; Upper Midwest includes IA, MI, MN, and WI; Southeast includes AL, FL, GA, NC, SC, and VA; South includes AR, KS, LA, MS, OK, and TX; Southwest includes AZ, CO, NM, and UT; Northern Rockies includes MT, NE, ND, SD, WY; Northwest includes ID, OR, and WA; and West includes CA and NV.
[39] Note most monitoring sites in the West region are located in California (see Figures 2A-2a and 2A-2b), therefore statistics for the West will be mostly representative of California ozone air quality.
[40] NOAA, National Centers for Environmental Information scientists have identified nine climatically consistent regions within the contiguous U.S., http://www.ncdc.noaa.gov/monitoring-references/maps/us-climate-regions.php.
[41] In calculating the ozone season statistics we limited the data to those observed and predicted pairs with observations that are greater than or equal 60 ppb in order to focus on concentrations at the upper portion of the distribution of values.

error, normalized mean bias, and normalized mean error to characterize model performance, statistics which are consistent with the recommendations in Simon et al. (2012) and the draft photochemical modeling guidance (US EPA, 2014c). As noted above, we calculated the performance statistics by climate region for the period of 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 percentage units and is defined as:

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 percentage units and is defined as:

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

As described in more detail below, 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., 2012; US EPA, 2005; US EPA, 2009; US EPA, 2011). These other modeling studies represent a wide range of modeling analyses that 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 are within the range found in other recent peer-reviewed and regulatory applications. The model performance results, as described in this document, demonstrate that the predictions from the 2011 modeling platform closely replicate the corresponding observed concentrations in terms of the magnitude, temporal fluctuations, and spatial differences for 8-hour daily maximum ozone.

The density scatter plots of MDA8 ozone are provided Figure 2A-4. 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 2A-2. 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 2A-5 through 2A-13. Spatial plots of the mean bias and error as well as the normalized mean bias and error for individual monitors are shown in Figures 2A-14 through 2A-17. The statistics shown in these two sets of 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 12 representative high ozone monitoring sites are provided in Figure 2A-18, (a) through (l). These sites are listed in Table 2A-3.

The density scatter plots in Figure 2A-4 provide a qualitative comparison of model-predicted and observed MDA8 ozone concentrations. In these plots the intensity of the colors indicates the density of individual observed/predicted paired values. The greatest number of individual paired values is denoted by the core area in white. The plots indicate that the predictions correspond closely to the observations in that a large number of observed/predicted paired values lie along or close to the 1:1 line shown on each plot. Overall, performance is best for observed values ≥ 60. The model tends to over-predict the observed values to some extent particularly at low and mid-range concentrations generally < 60 ppb in each of the regions. This feature is most evident in the South and Southeast states. In the West, high concentrations are under-predicted and low and mid-range concentrations are over-predicted. Observed and predicted values are in close agreement in the Southwest and Northwest regions.

As indicated by the statistics in Table 2A-2, 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 at AQS sites in the Western region and at rural CASTNet sites in the South, Southwest and Western regions 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 2A-5 through 2A-13. The distribution of predicted concentrations tends to be close to that of the observed data at the 25th percentile, median and 75th percentile values for each region, although there is a small persistent overestimation bias for these metrics in the Northeast, Southeast, and Ohio Valley regions, and under-prediction at CASTNet sites in the West and Southwest[42]. The CAMx model, as applied here, also has a tendency to under-predict the highest observational concentrations at both the AQS and CASTNet network sites.

Figures 2A-14 through 2A-17 show the spatial variability in bias and error at monitor locations. Mean bias, as seen from Figure 2A-14, is less than 5 ppb at many sites across the East with over-prediction of 5 to 10 ppb at some sites from the Southeast into the Northeast. Elsewhere, mean bias is generally in the range of -5 to -10 ppb. Figure 2A-15 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 Northeast and generally under predict in the Southwest, Northern Rockies, Northwest and West. Model performance in the Ohio Valley and Upper Midwest states shows both under and over predictions.

[42] The over-prediction at CASTNet sites in the Northwest may not be representative of performance in rural areas of this region because there are so few observed and predicted data pairs in this region.

Model error, as seen from Figure 2A-16, is 10 ppb or less at most of the sites across the modeling domain. Figure 2A-17 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 (i.e., greater than 15 percent) is evident at sites in several areas most notably along portions of the Northeast 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 12 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 because they are high ozone sites in those urban areas with the highest projected ozone levels in the 2025 base case simulation. The results, as shown in Figures 2A-18 (a) through (l), 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 the day-to-day variability in the observations: Alleghany County, PA; 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; Queens County, NY; Suffolk County, NY; Sheboygan County, WI. Note that at the site in Brazoria County, TX there is an extended period from mid-July to mid-August with very low observed ozone concentrations, mostly in the range of 30 to 40 ppb. The model predicted values during this period in the range of 40 to 60 ppb which is not quite as low as the observed values. The sites in Douglas County, CO and Harford County, MD closely track the day-to-day variability in the observed MDA8 values, but some days are over predicted while other days are under predicted to some extent. Finally, the daily modeled ozone at the two California sites evaluated correlates well with observations but has a persistent low bias. Looking across all 12 sites indicates that the modeling platform is able to capture the site to site differences in the short-term variability of ozone concentrations.

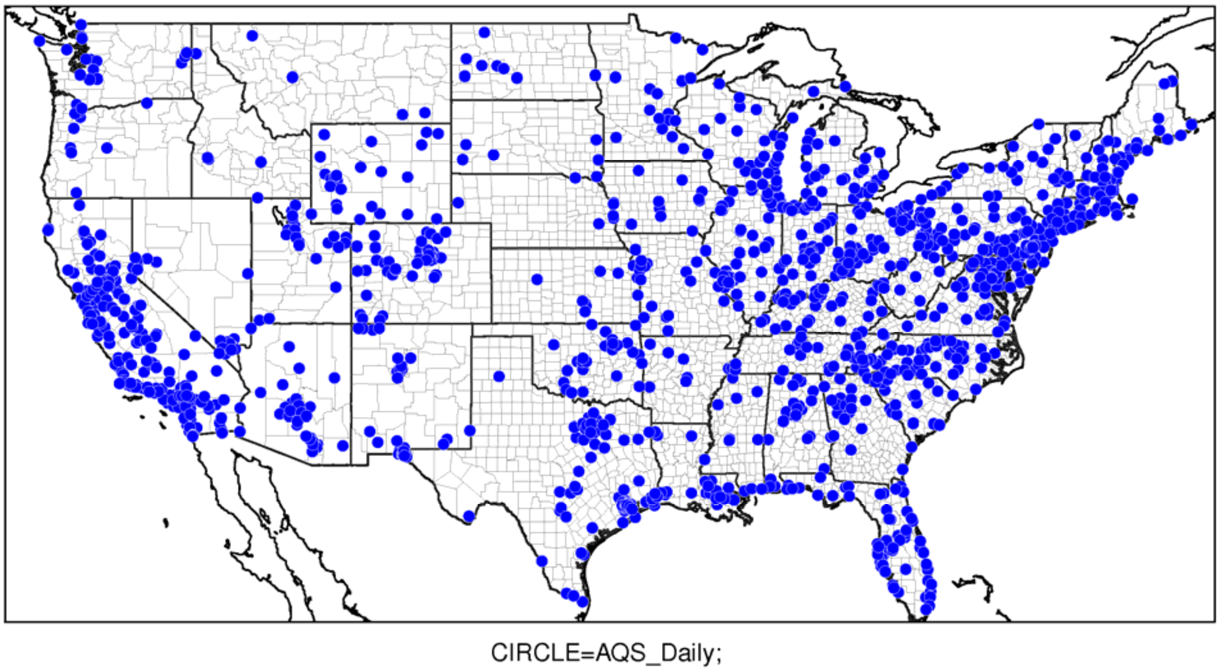

Figure 2A-2a. AQS Ozone Monitoring Sites

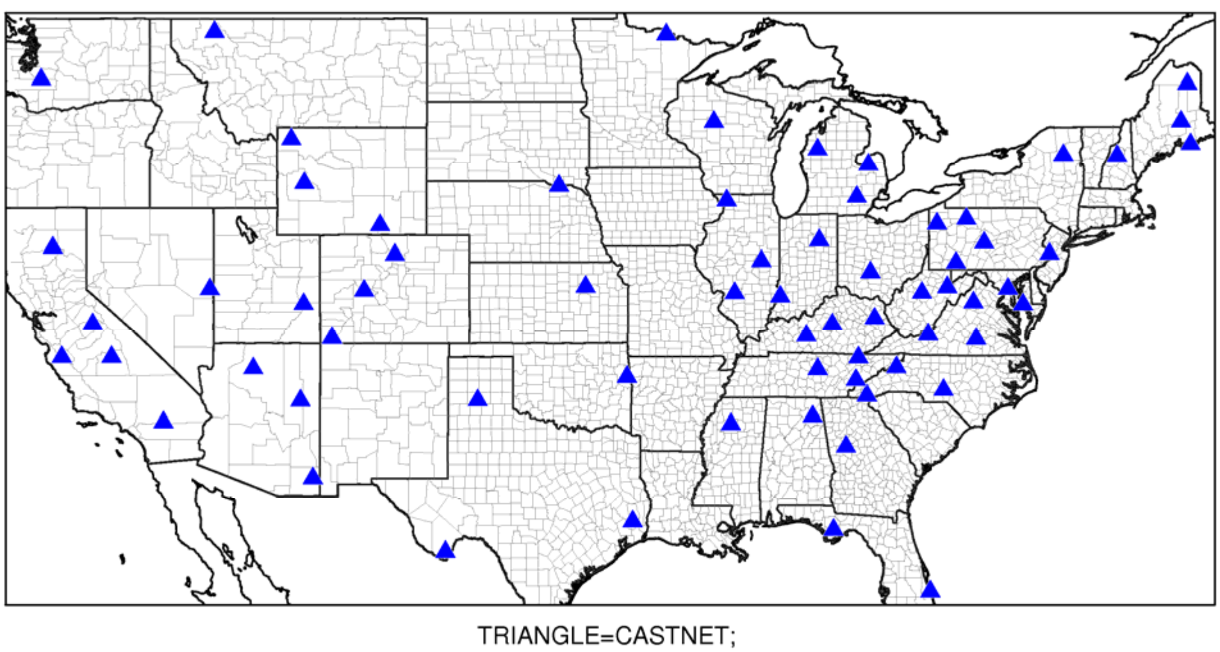

Figure 2A-2b. CASTNet Ozone Monitoring Sites

U.S. Climate Regions

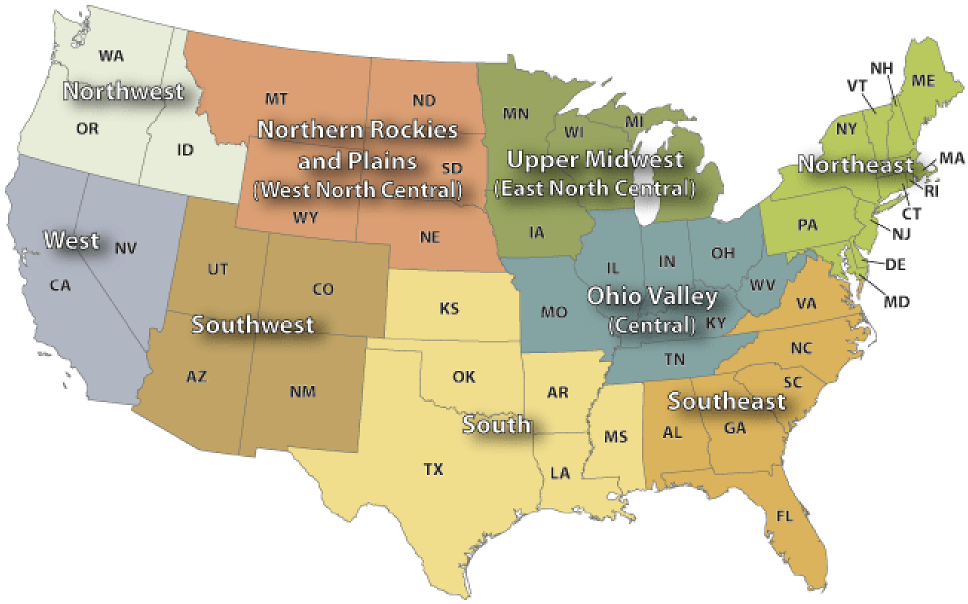

Figure 2A-3. NOAA Nine Climate Regions (source: http://www.ncdc.noaa.gov/monitoring-references/maps/us-climate-regions.php#references)

Table 2A-2. MDA8 Ozone Performance Statistics Greater than or Equal to 60 Ppb for May through September by Climate Region, by Network

Network	Climate region	No. of Obs	MB	ME	NMB (%)	NME (%)
AQS	Northeast	3,998	2.2	7.4	3.2	10.8
	Ohio Valley	6,325	0.3	7.6	0.4	11.3
	Upper Midwest	1,162	-3.0	7.5	-4.4	11.0
	Southeast	37,280	-2.5	8.1	-3.6	11.9
	South	5,694	-3.7	8.1	-5.4	11.7
	Southwest	6,033	-5.2	7.9	-7.8	12.0
	Northern Rockies	380	-5.9	7.4	-9.4	11.7
	Northwest	79	-5.4	8.1	-8.5	12.6
	West	8,665	-7.3	9.5	-10.3	13.5
CASTNet	Northeast	264	2.3	6.1	3.4	9.1
	Ohio Valley	107	-2.3	6.2	-3.4	9.4
	Upper Midwest	38	-3.9	5.9	-5.8	8.8
	Southeast	2,068	-5.0	8.2	-7.5	12.1
	South	215	-7.1	8.0	-10.7	12.0
	Southwest	382	-7.7	8.6	-11.7	13.1
	Northern Rockies	110	-7.8	8.1	-12.2	12.8
	Northwest	-	-	-	-	-
	West	425	-12.1	12.5	-16.6	17.1

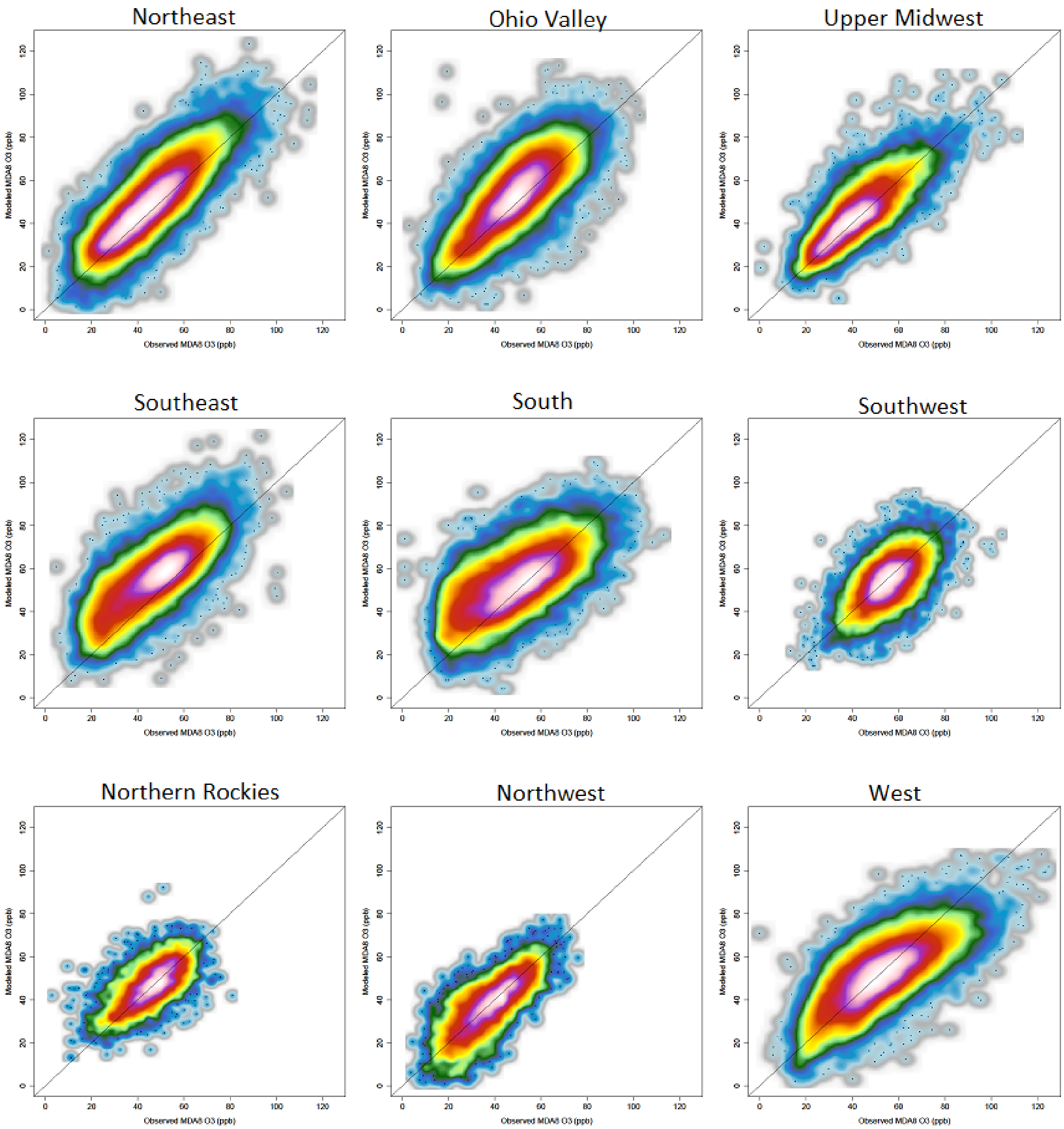

Figure 2A-4. Density Scatter Plots of Observed/Predicted MDA8 Ozone for the Northeast, Ohio River Valley, Upper Midwest, Southeast, South, Southwest, Northern Rockies, Northwest and West Regions

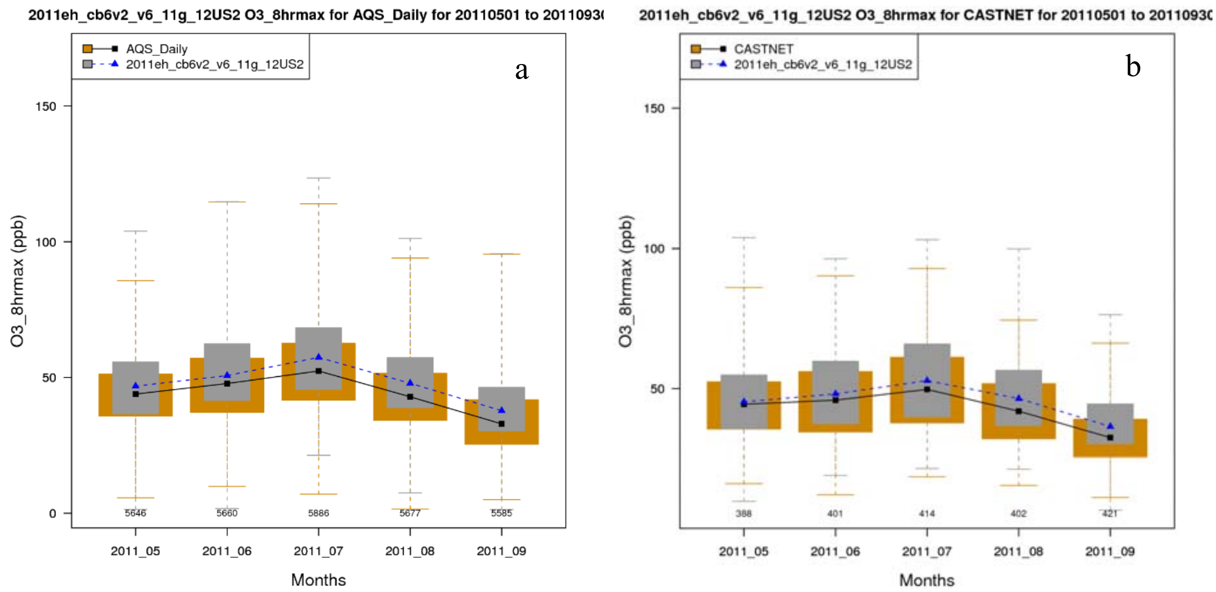

Figure 2A-5. Distribution of Observed and Predicted MDA8 Ozone by Month for the Period May through September for the Northeast Region, (a) AQS Network and (b) CASTNet Network. [symbol = median; top/bottom of box = 75th/25th percentiles; top/bottom line = max/min values]

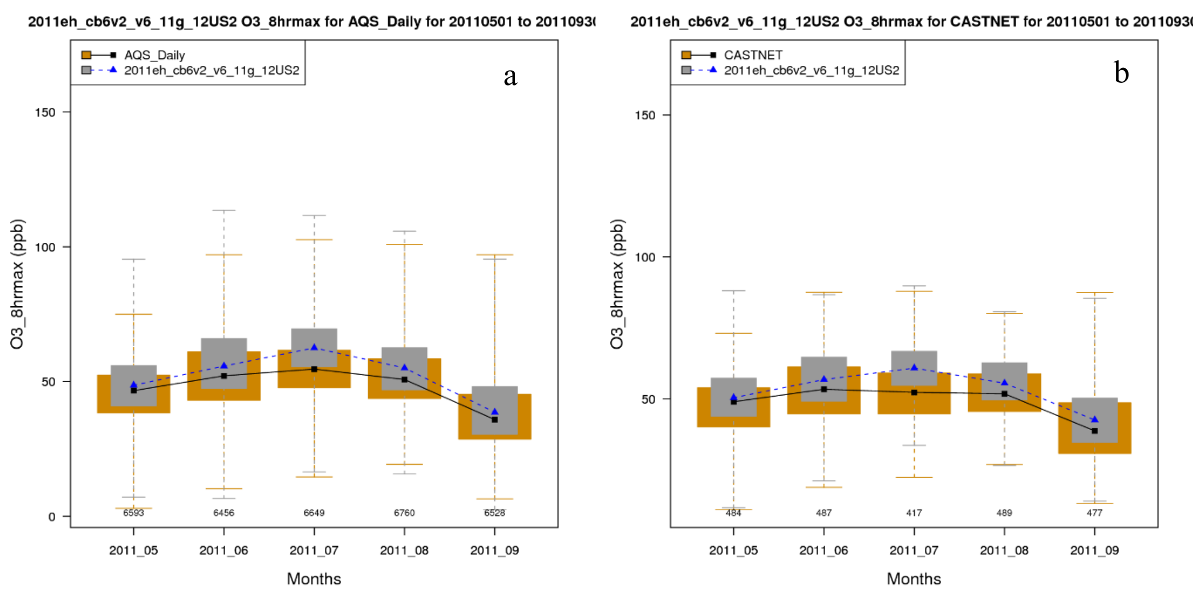

Figure 2A-6. Distribution of Observed and Predicted MDA8 Ozone by Month for the Period May through September for the Ohio Valley Region, (a) AQS Network and (b) CASTNet Network

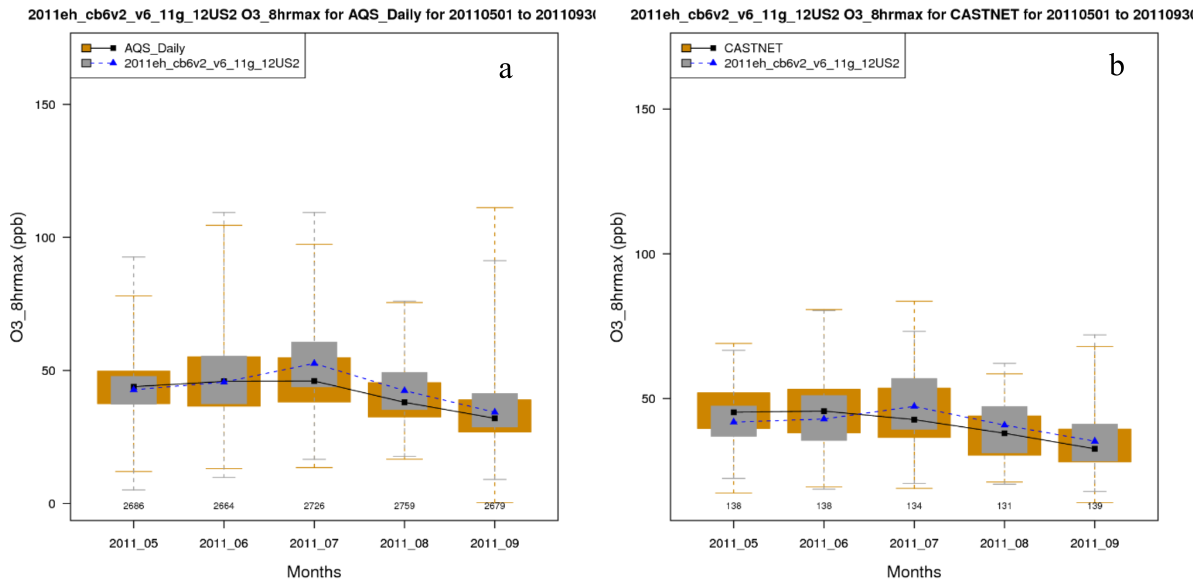

Figure 2A-7. Distribution of Observed and Predicted MDA8 Ozone by Month for the Period May through September for the Upper Midwest Region, (a) AQS Network and (b) CASTNet Network

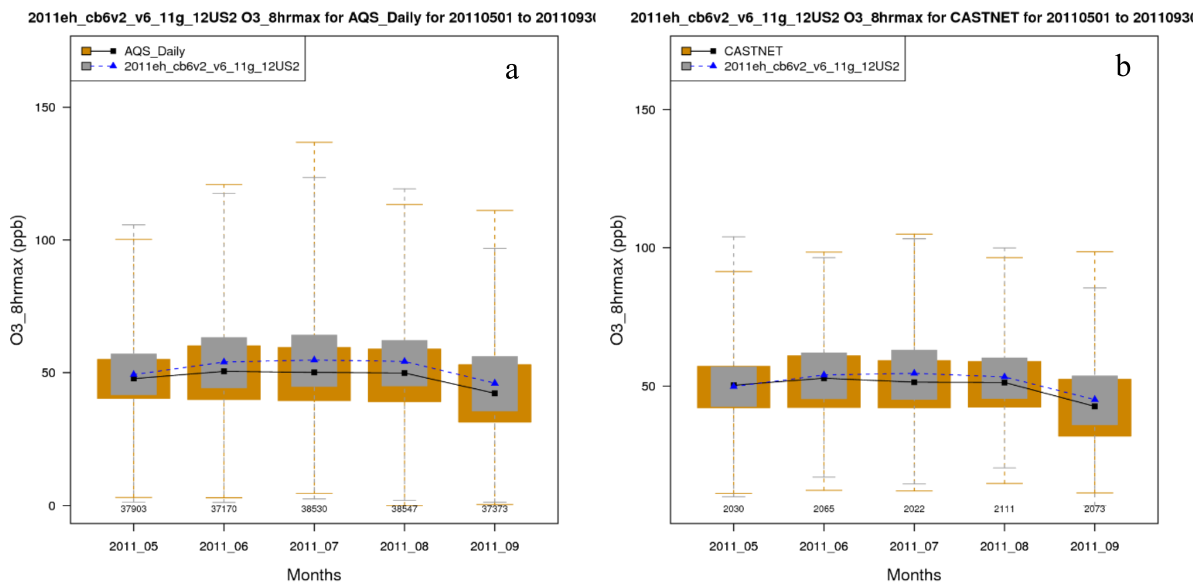

Figure 2A-8. Distribution of Observed and Predicted MDA8 Ozone by Month for the Period May through September for the Southeast Region, (a) AQS Network and (b) CASTNet Network

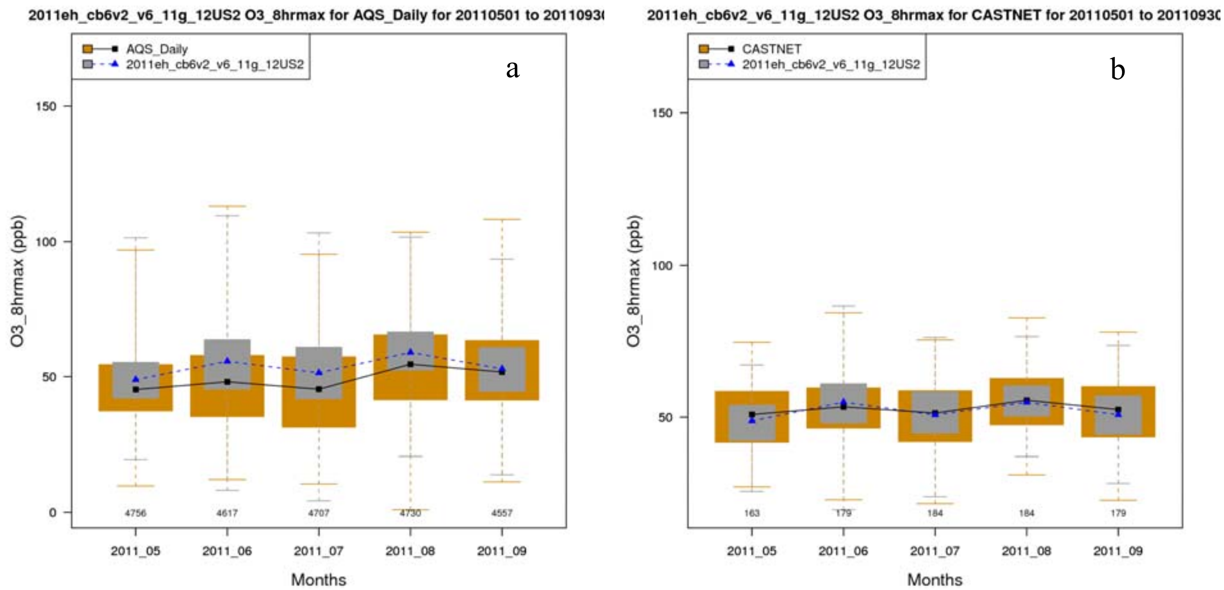

Figure 2A-9. Distribution of Observed and Predicted MDA8 Ozone by Month for the Period May through September for the South Region, (a) AQS Network and (b) CASTNet Network

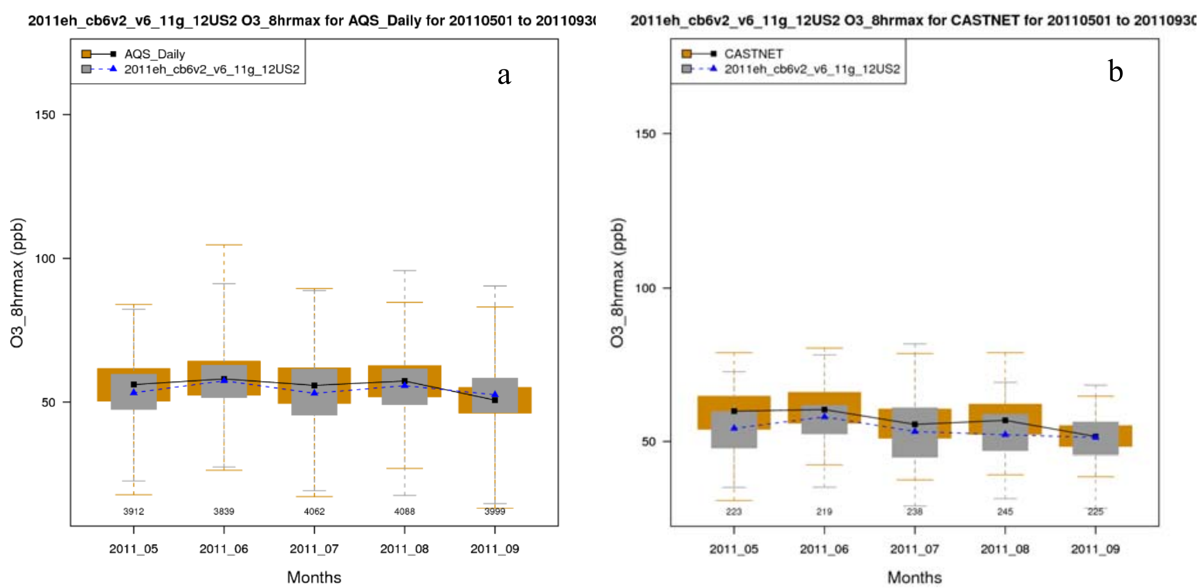

Figure 2A-10. Distribution of Observed and Predicted MDA8 Ozone by Month for the Period May through September for the Southwest Region, (a) AQS Network and (b) CASTNet Network

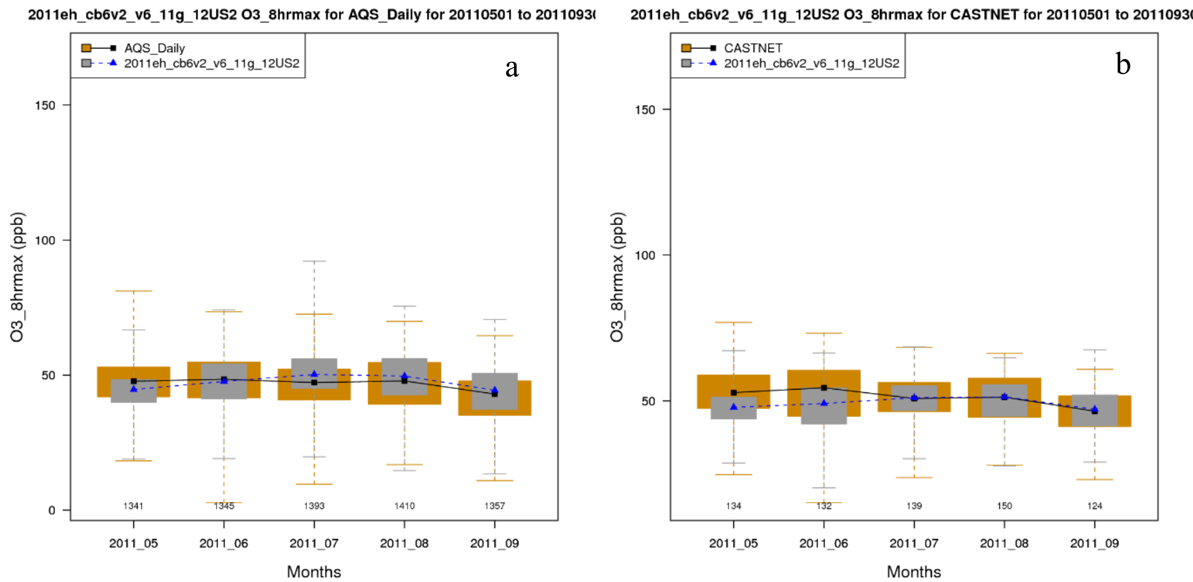

Figure 2A-11. Distribution of Observed and Predicted MDA8 Ozone by Month for the Period May through September for the Northern Rockies Region, (a) AQS Network and (b) CASTNet Network

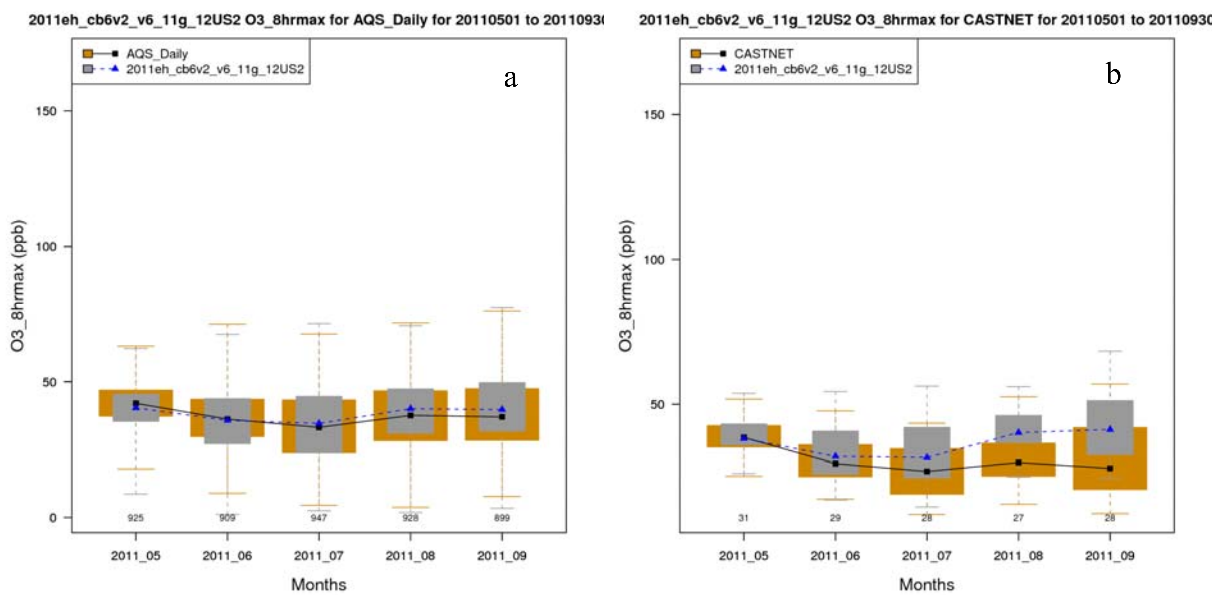

Figure 2A-12. Distribution of Observed and Predicted MDA8 Ozone by Month for the Period May through September for the Northwest Region, (a) AQS Network and (b) CASTNet Network

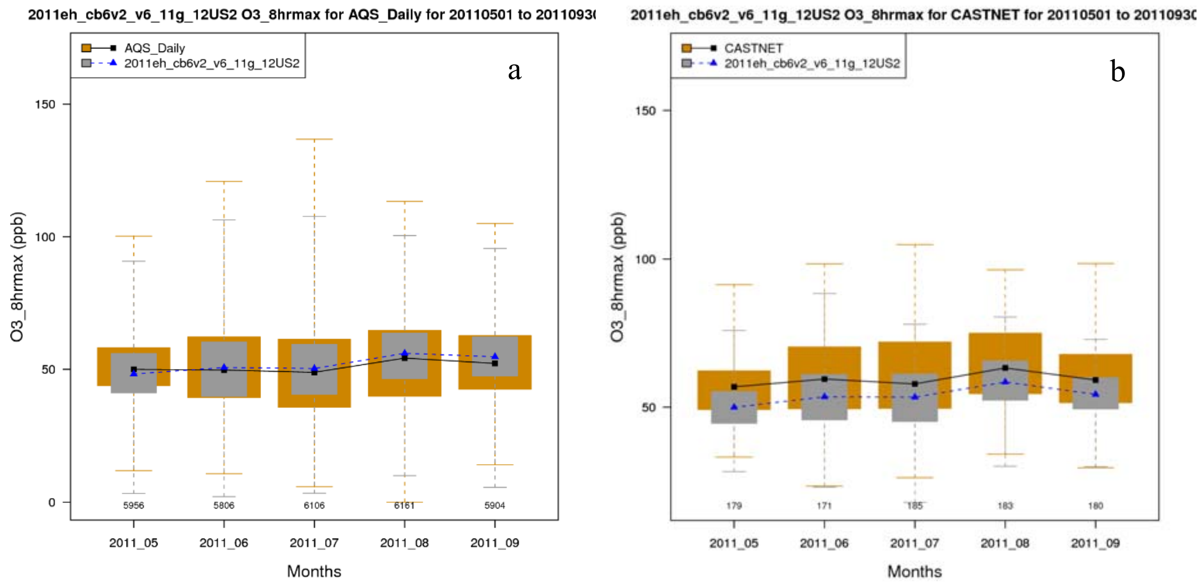

Figure 2A-13. Distribution of Observed and Predicted MDA8 Ozone by Month for the Period May through September for the West Region, (a) AQS Network and (b) CASTNet Network

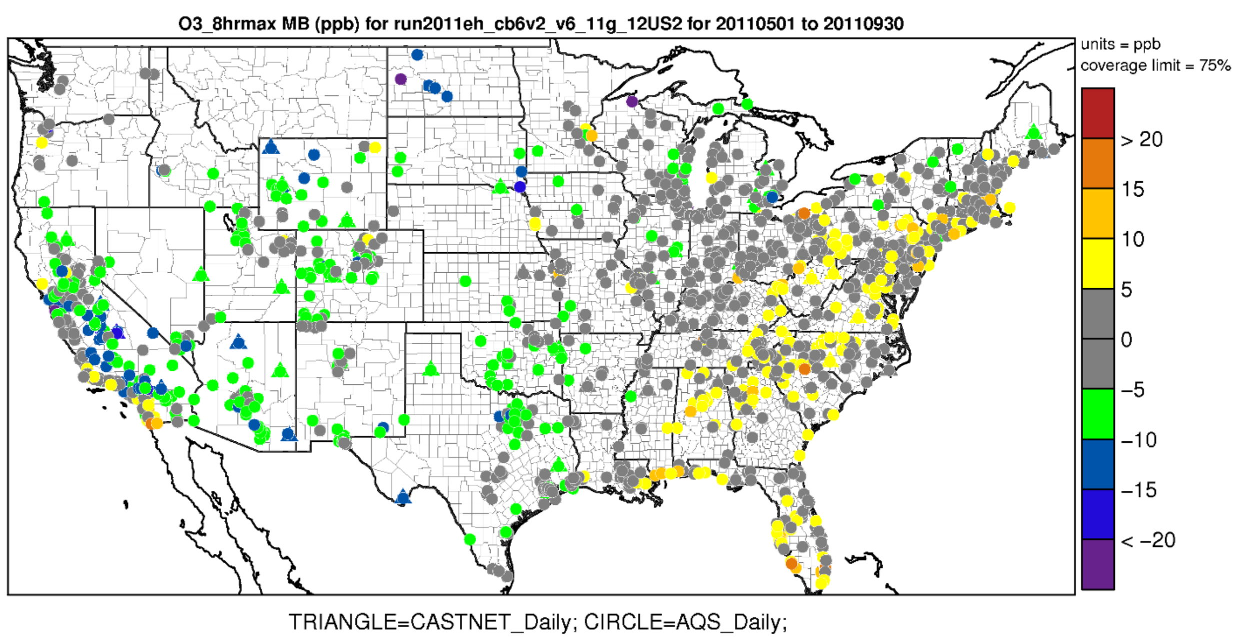

Figure 2A-14. Mean Bias (ppb) of MDA8 Ozone Greater than or Equal to 60 ppb over the Period May-September 2011 at AQS and CASTNet Monitoring

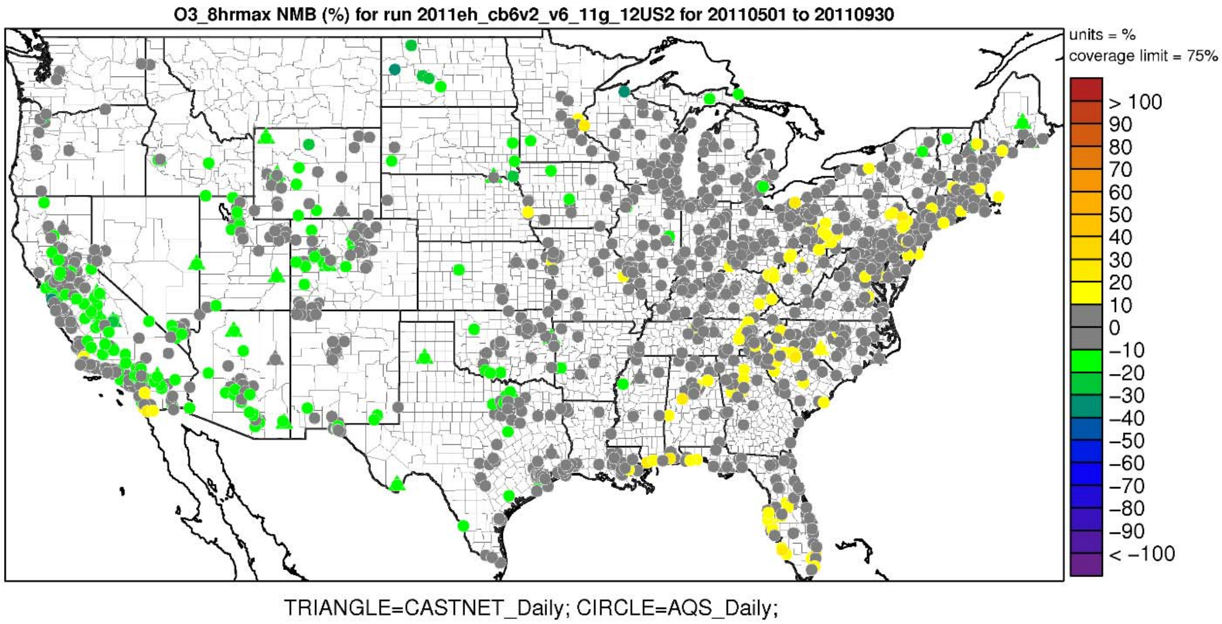

Figure 2A-15. Normalized Mean Bias (%) of MDA8 Ozone Greater than or Equal to 60 ppb over the Period May-September 2011 at AQS and CASTNet Monitoring Sites

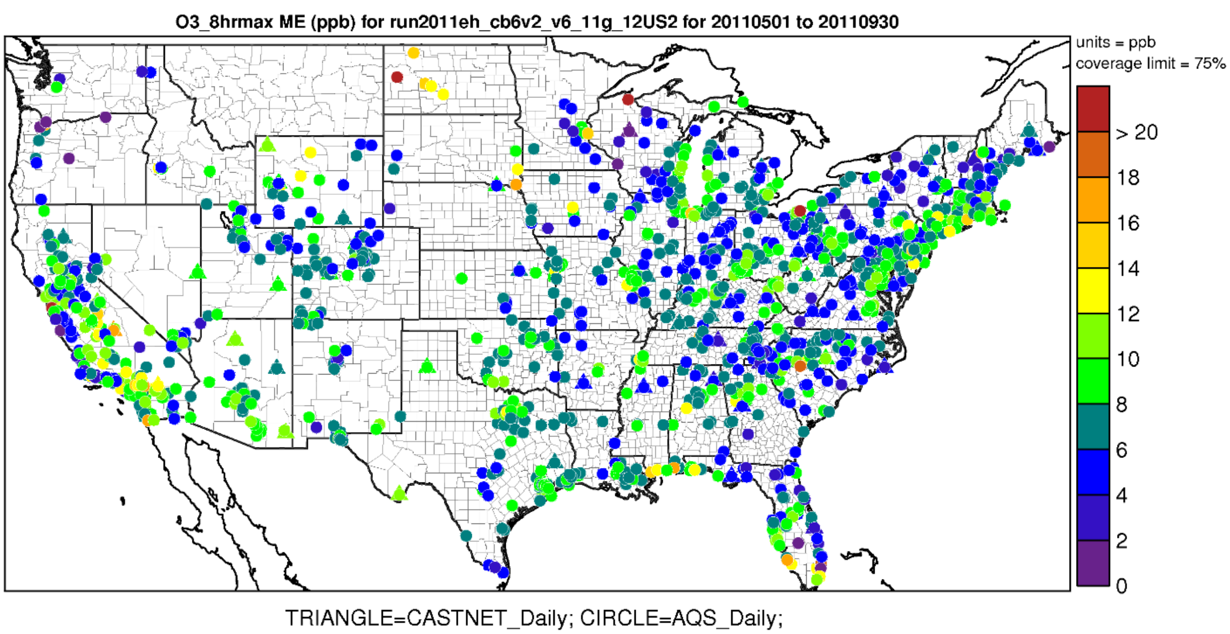

Figure 2A-16. Mean Error (ppb) of MDA8 Ozone Greater than or Equal to 60 ppb over the Period May-September 2011 at AQS and CASTNet Monitoring Sites

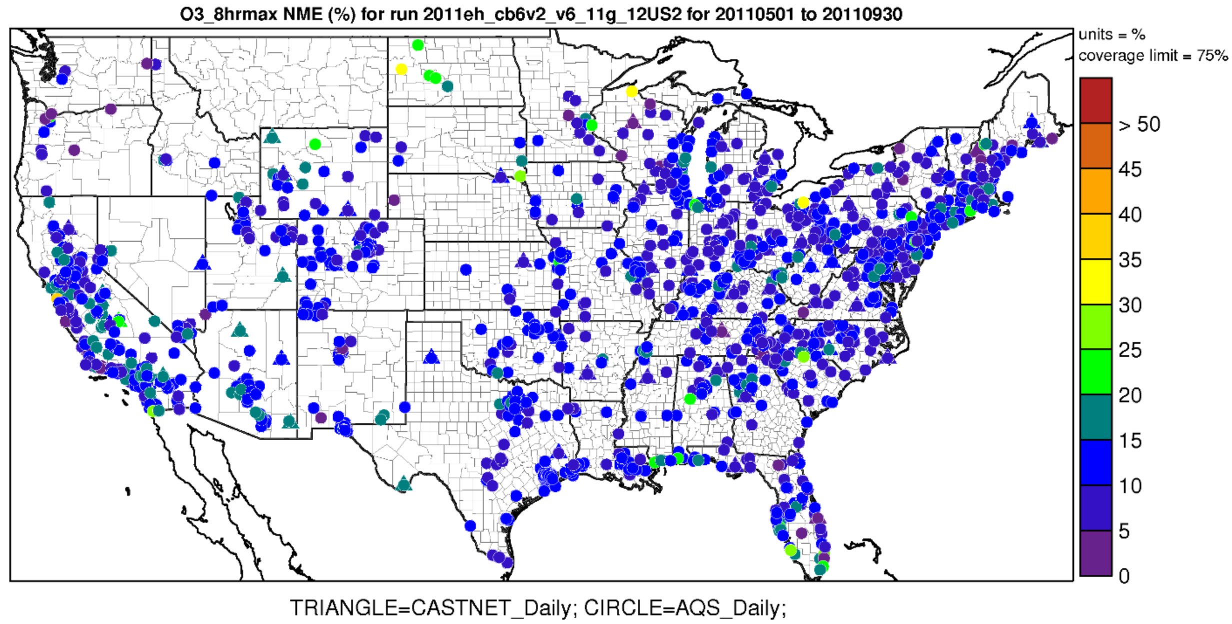

Figure 2A-17. Normalized Mean Error (%) of MDA8 Ozone Greater than or Equal to 60 ppb over the Period May-September 2011 at AQS and CASTNet Monitoring Sites

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

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

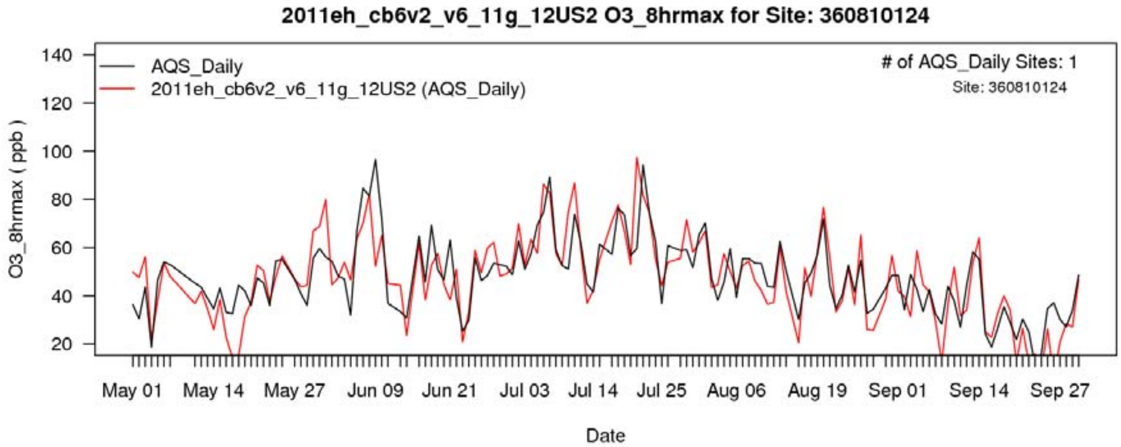

Figure 2A-18a. Time Series of Observed (black) and Predicted (red) MDA8 Ozone for May through September 2011 at Site 360810124 in Queens, New York

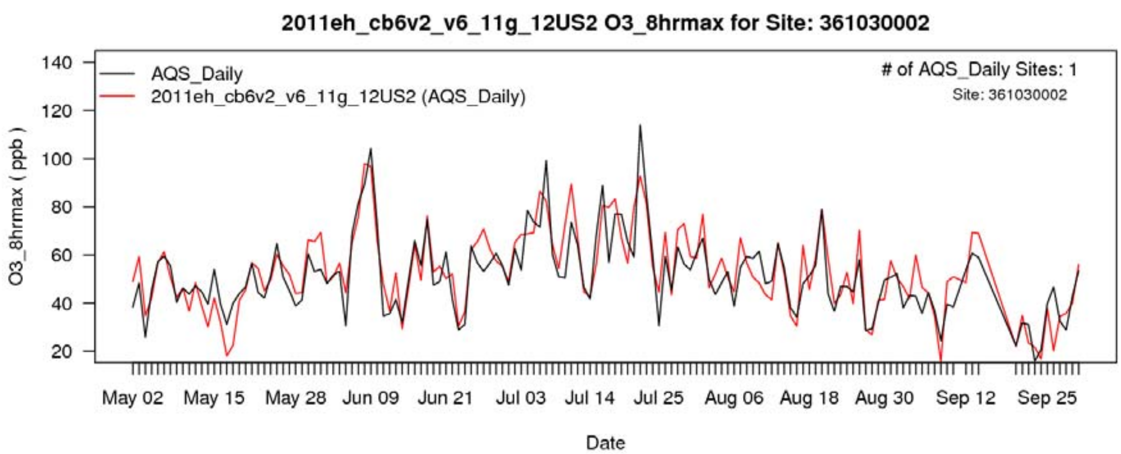

Figure 2A-18b. Time Series of Observed (black) and Predicted (red) MDA8 Ozone for May through September 2011 at Site 361030002 in Suffolk County, New York

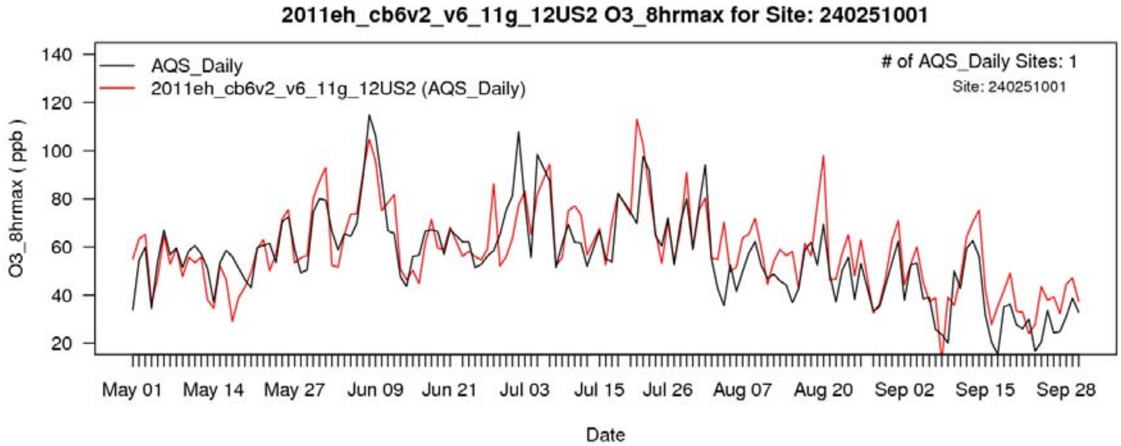

Figure 2A-18c. Time Series of Observed (black) and Predicted (red) MDA8 Ozone for May through September 2011 at Site 240251001 in Harford Co., Maryland

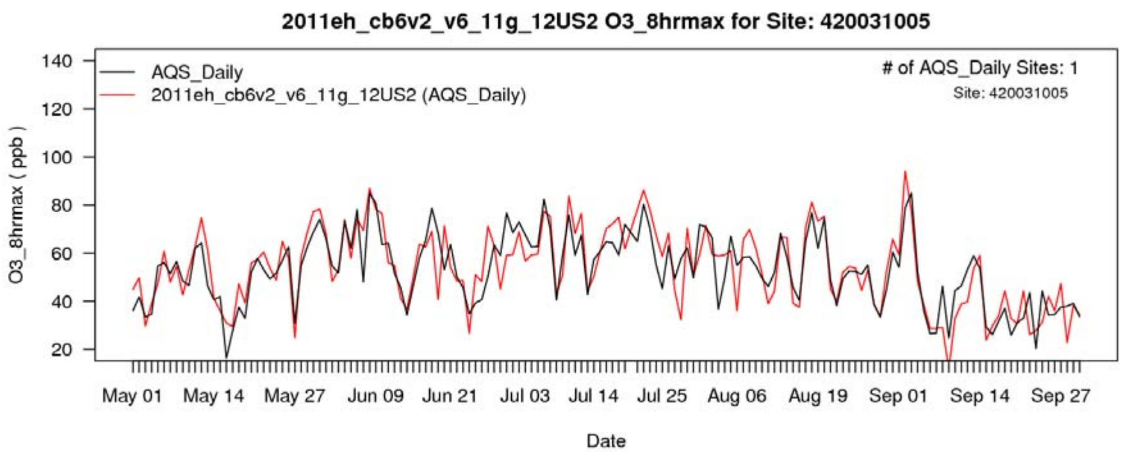

Figure 2A-18d. Time Series of Observed (black) and Predicted (red) MDA8 Ozone for May through September 2011 at Site 420031005 in Allegheny Co., Pennsylvania

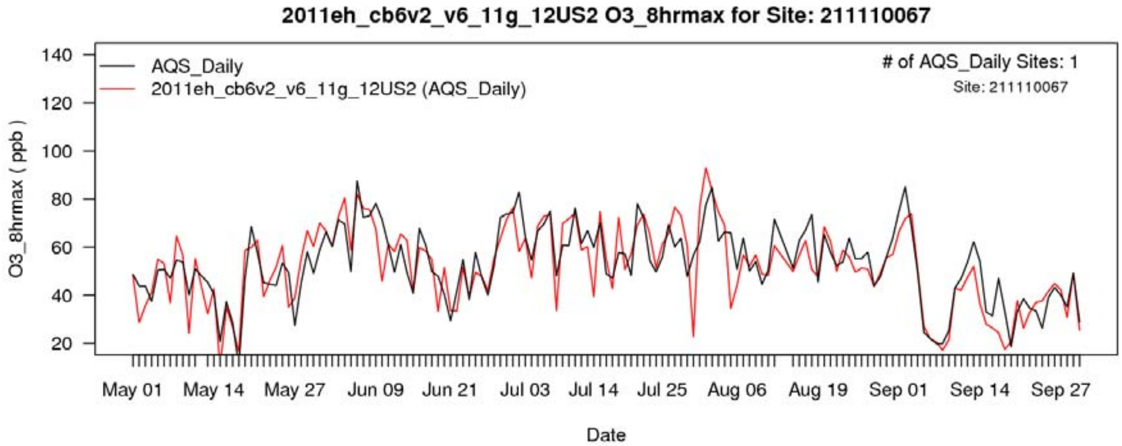

Figure 2A-18e. Time Series of Observed (black) and Predicted (red) MDA8 Ozone for May through September 2011 at Site 211110067 in Jefferson Co., Kentucky

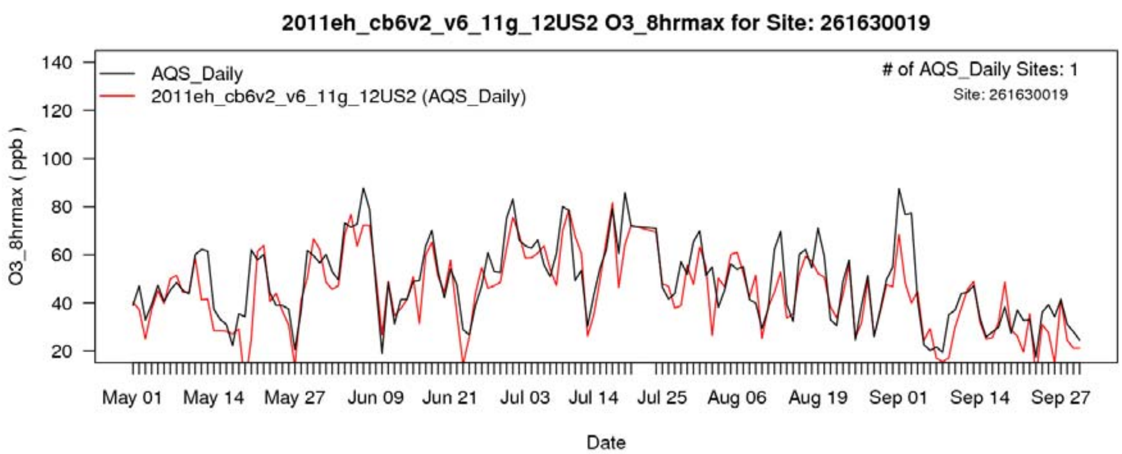

Figure 2A-18f. Time Series of Observed (black) and Predicted (red) MDA8 Ozone for May through September 2011 at Site 261630019 in Wayne Co., Michigan

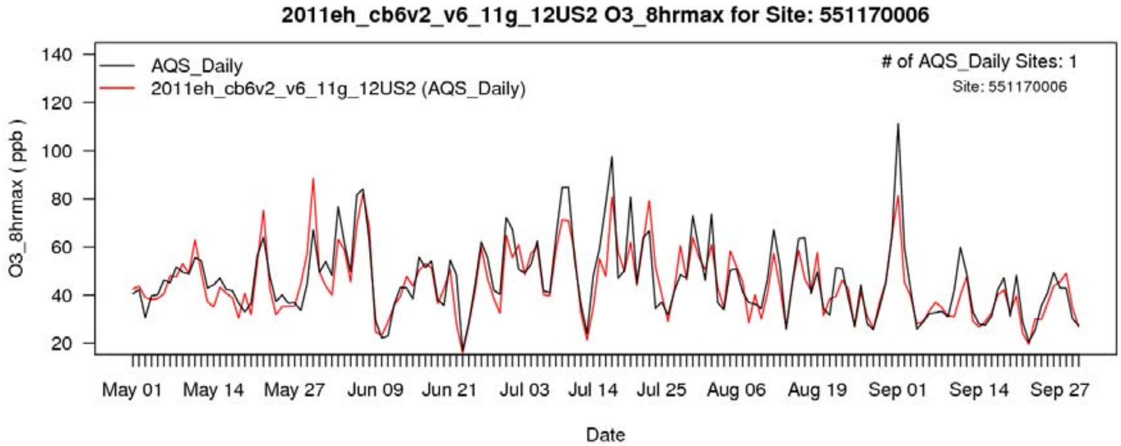

Figure 2A-18g. Time Series of Observed (black) and Predicted (red) MDA8 Ozone for May through September 2011 at Site 551170006 in Sheboygan Co., Wisconsin

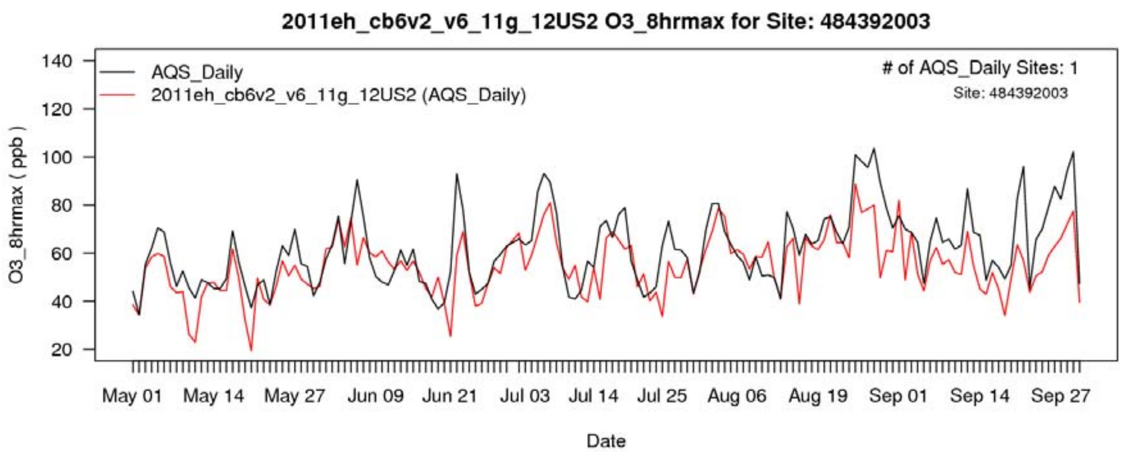

Figure 2A-18h. Time Series of Observed (black) and Predicted (red) MDA8 Ozone for May through September 2011 at Site 484392003 in Tarrant Co., Texas

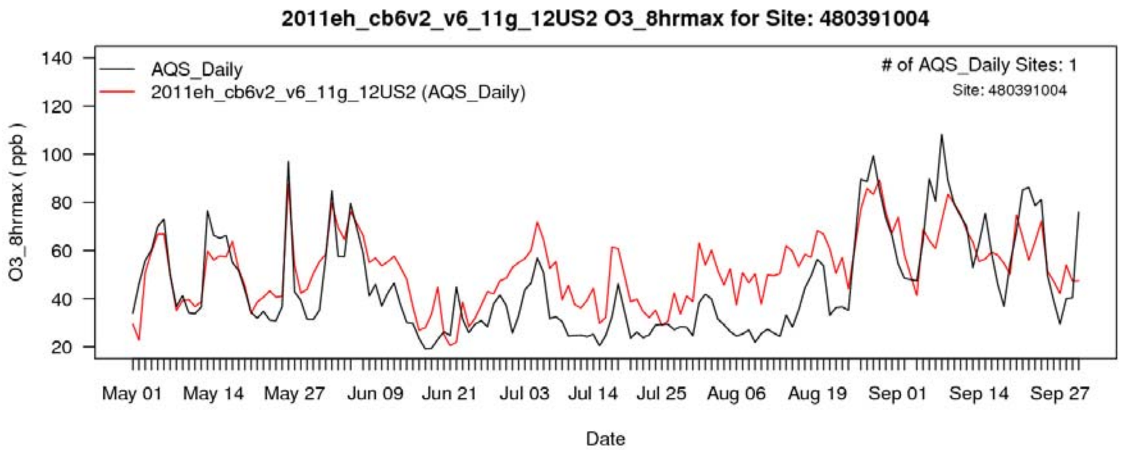

Figure 2A-18i. Time Series of Observed (black) and Predicted (red) MDA8 Ozone for May through September 2011 at Site 480391004 in Brazoria Co., Texas

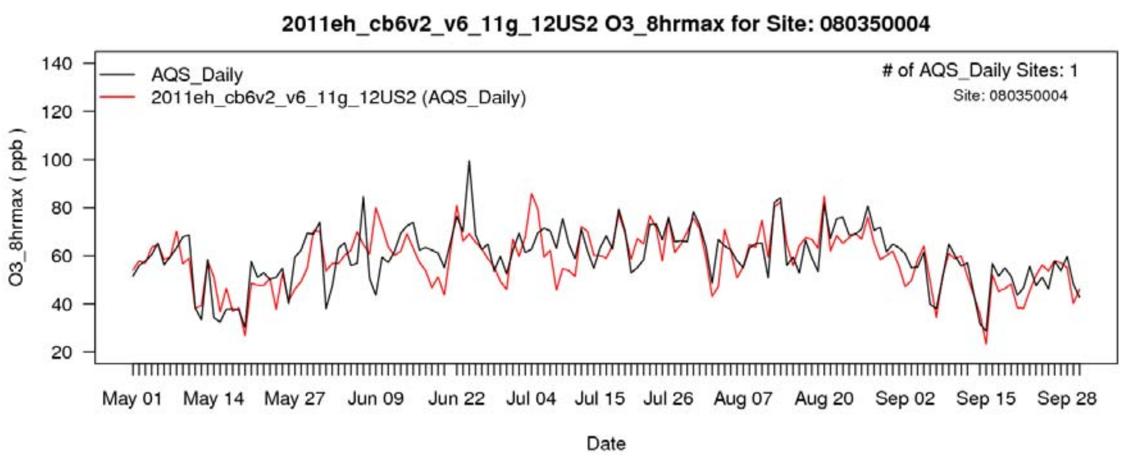

Figure 2A-18j. Time Series of Observed (black) and Predicted (red) MDA8 Ozone for May through September 2011 at Site 80350004 in Douglas Co., Colorado

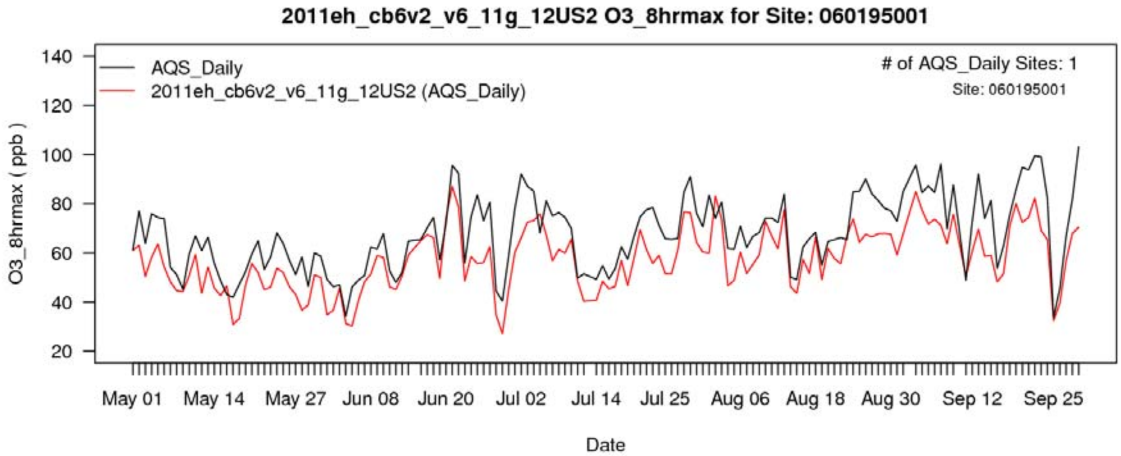

Figure 2A-18k. Time Series of Observed (black) and Predicted (red) MDA8 Ozone for May through September 2011 at Site 60195001 in Fresno Co., California

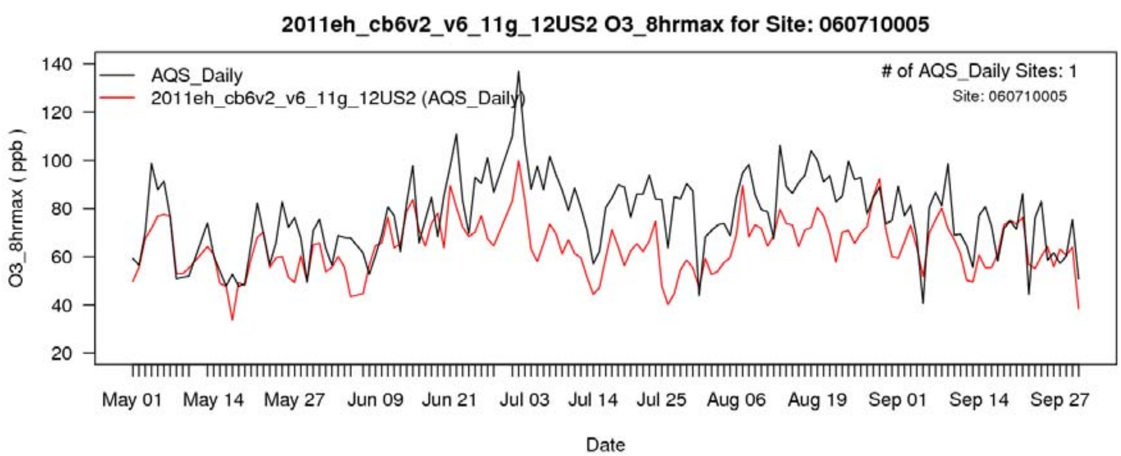

Figure 2A-18l. Time Series of Observed (black) and Predicted (red) MDA8 Ozone for May through September 2011 at Site 60710005 in San Bernardino Co., California

2A.2 VOC Impact Regions

As described in Chapter 2, we defined VOC impact regions for the following urban areas: New York City, Chicago, Louisville, Houston, Denver, Northern California and Southern California.[43] 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 2A-19 shows the impact of 50% U.S. 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.

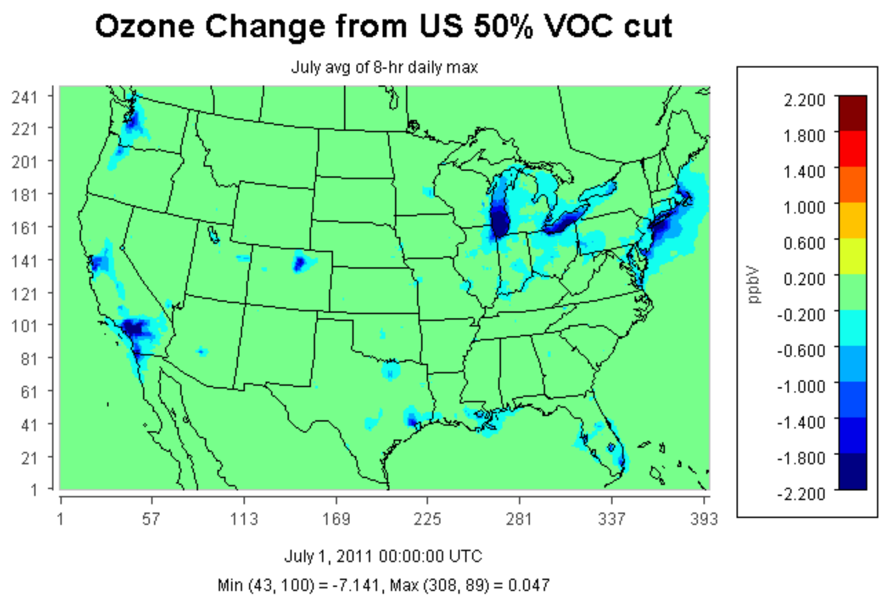

Figure 2A-19. Change in July Average of 8-hr Daily Maximum Ozone Concentration (ppb) Due to 50% Cut in U.S. Anthropogenic VOC Emissions

2A.3 Monitors Excluded from the Quantitative Analysis

There were 1,225 ozone monitors with complete ozone data for at least one DV period covering the years 2009-2013. Of those sites, we quantitatively analyzed 1,165 in this analysis. As discussed in Chapter 2, 60 sites were excluded from the quantitative analysis of emissions

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[43] Other local VOC areas that had similar levels of ozone response to the 50% VOC reduction were also explored but were found not to be helpful in reaching alternative NAAQS levels in this analysis: Dallas, Detroit, Pittsburgh, and Baltimore. This may be due to the construct of the attainment scenarios explored here and does not mean that VOC controls might not be effective in these areas under alternate assumptions about regional NOx controls.

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.

2A.3.1 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 (US EPA, 2014c). A list of the 41 sites in this category is given in Table 2A-4.

Table 2A-4. Monitors without Projections due to Insufficient High Modeling Days to Meet EPA Guidance for Projecting Design Values

Site ID	Lat	Long	State	County
60010009	37.74307	-122.17	California	Alameda
60010011	37.81478	-122.282	California	Alameda
60131004	37.9604	-122.357	California	Contra Costa
60231004	40.77694	-124.178	California	Humboldt
60450008	39.14566	-123.203	California	Mendocino
60750005	37.76595	-122.399	California	San Francisco
60811001	37.48293	-122.203	California	San Mateo
60932001	41.72689	-122.634	California	Siskiyou
160230101	43.46056	-113.562	Idaho	Butte
230031100	46.69643	-68.033	Maine	Aroostook
260330901	46.49361	-84.3642	Michigan	Chippewa
270052013	46.85181	-95.8463	Minnesota	Becker
270177416	46.70527	-92.5238	Minnesota	Carlton
270750005	47.94862	-91.4956	Minnesota	Lake
270834210	44.4438	-95.8179	Minnesota	Lyon
271370034	48.41333	-92.8306	Minnesota	Saint Louis
300298001	48.51017	-113.997	Montana	Flathead
300490004	46.8505	-111.987	Montana	Lewis and Clark
311079991	42.8292	-97.854	Nebraska	Knox
380070002	46.8943	-103.379	North Dakota	Billings
380130004	48.64193	-102.402	North Dakota	Burke
380150003	46.82543	-100.768	North Dakota	Burleigh
380171004	46.93375	-96.8554	North Dakota	Cass
380250003	47.3132	-102.527	North Dakota	Dunn
380530002	47.5812	-103.3	North Dakota	McKenzie
380570004	47.29861	-101.767	North Dakota	Mercer
380650002	47.18583	-101.428	North Dakota	Oliver
410170122	44.0219	-121.26	Oregon	Deschutes

Site ID	Lat	Long	State	County
410290201	42.22989	-122.788	Oregon	Jackson
410591003	45.82897	-119.263	Oregon	Umatilla
460110003	44.3486	-96.8073	South Dakota	Brookings
530090013	48.29786	-124.625	Washington	Clallam
530330080	47.56824	-122.309	Washington	King
530530012	46.7841	-121.74	Washington	Pierce
530531010	46.75833	-122.124	Washington	Pierce
530570020	48.39779	-122.505	Washington	Skagit
530730005	48.95074	-122.554	Washington	Whatcom
550030010	46.602	-90.656	Wisconsin	Ashland
551250001	46.052	-89.653	Wisconsin	Vilas
560390008	43.67083	-110.599	Wyoming	Teton
560391011	44.55972	-110.401	Wyoming	Teton

2A.3.2 Winter Ozone

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 conducive to 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 2A-5.

Table 2A-5. 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).

2A.3.3 Monitoring Sites in Rural/Remote Areas of the West and Southwest

As mentioned in Chapter 2, model-predicted ozone concentrations at 12 sites in rural/remote areas in the West and Southwest were excluded from the quantitative analysis. These 12 sites are a subset of 26 sites identified in the November 2014 RIA proposal (US EPA, 2014d). The original 26 sites had two common characteristics. First, they had small modeled responses 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. All of these 12 monitoring sites have 2025 baseline concentrations below 70 ppb. Therefore, no emissions reductions would be required for these sites to meet a primary standard of 70 ppb in 2025. Of the 26 sites identified in the RIA proposal, only the sites with 2025 baseline DVs (or post-2025 baseline DVs for CA) above 65 ppb are excluded from this analysis. More details on these 12 sites are provided in Table 2A-6. We have qualitatively characterized the predominant ozone influence for each site in Table 2A-6. 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 2A-20 shows the location of all sites listed in Table 2A-7 and for demonstrative purposes assigns each site to a category based on the predominant source of ozone in that location. The table and figure indicate that all 12 sites have 2025 or post-2025 baseline design values below 70 ppb as mentioned above. Of the 12 sites, 5 sites are characterized as border sites, 5 sites are characterized as being strongly influenced by California emissions, and 2 sites are influenced by other ozone sources.

Table 2A-6. Monitors with Limited Response to Regional NOx and National VOC Emissions Reductions in the 2025 and Post-2025 Baselines

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
Yuma Supersite	40278011	Arizona	Yuma	51	SLAMS	Mexican border + California	75	66
El Centro-9th st	60251003	California	Imperial	-	SLAMS	California + Mexican Border	81	68
Yosemite NP	60430003	California	Mariposa	5265	CASTNET	California + Other sources	77	67
Sequoia and Kings Canyon NP	61070006	California	Tulare	1890	Non-EPA Federal (NPS)	California + Other sources	81	69
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
BLM land near Carlsbad	350151005	New Mexico	Eddy	780	SLAMS	Central region + Southwest region + Mexican border	70	67
Big Bend NP	480430101	Texas	Brewster	1052	CASTNET	Mexican border	70	68
BLM Land/Carlsbad	483819991	Texas	Randall	780	SLAMS	Central region + Mexican border + Other sources	73	66
Zion NP	490530130	Utah	Washington	1213	Non-EPA Federal (NPS)	California + Other sources	71	66

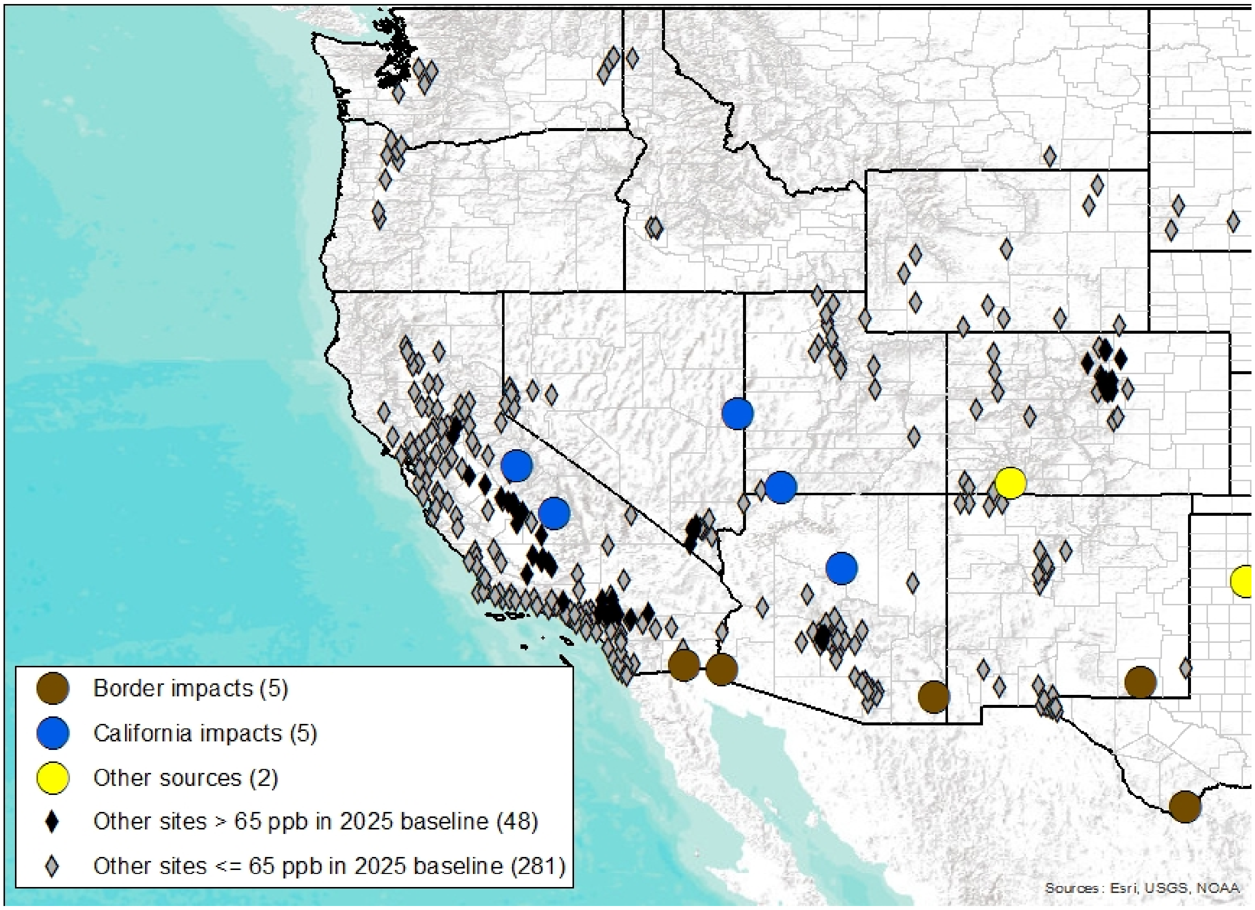

Figure 2A-20. Location of Sites Identified in Table 2A-6

In Figure 2A-20, 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 2A-6 are shown as small diamonds. Gray diamonds represent sites that had DVs less than or equal to 65 ppb in the 2025 baseline (or post-2025 baseline for California sites). Black diamonds represent sites that had DVs greater than 65 ppb in the 2025 baseline (or post-2025 baseline for California sites).

2A.4 Design Values for All Monitors Included in the Quantitative Analysis

In addition to other information, Tables 2A-7 and 2A-8 provide baseline design values corresponding to the information presented in the maps in Figures 3-5 and 3-11. Note that some counties contain more than one monitor and the highest monitor is used for the map.

Table 2A-7. Design Values (ppb) for California Monitors

Site ID	Lat	Long	State	County	O3 DV for Scenario: Base Case	Baseline	70	65
60010007	37.68753	-121.784	California	Alameda	67	62	58	54
60012001	37.65446	-122.032	California	Alameda	54	50	48	45
60050002	38.33991	-120.764	California	Amador	60	55	51	47
60070007	39.71404	-121.619	California	Butte	61	56	52	48
60070008	39.76154	-121.842	California	Butte	53	49	46	43
60090001	38.20185	-120.682	California	Calaveras	63	57	54	50
60111002	39.20294	-122.018	California	Colusa	53	49	46	43
60130002	37.93601	-122.026	California	Contra Costa	66	61	57	53
60131002	38.00631	-121.642	California	Contra Costa	64	58	55	51
60170010	38.72528	-120.822	California	El Dorado	66	60	55	50
60170012	38.81161	-120.033	California	El Dorado	62	59	57	56
60170020	38.89094	-121.003	California	El Dorado	66	60	55	50
60190007	36.70551	-119.742	California	Fresno	82	74	70	65
60190011	36.78532	-119.774	California	Fresno	81	73	69	64
60190242	36.84139	-119.874	California	Fresno	81	75	70	66
60192009	36.63423	-120.382	California	Fresno	64	59	55	52
60194001	36.5975	-119.504	California	Fresno	77	69	65	60
60195001	36.81911	-119.717	California	Fresno	83	75	70	65
60210003	39.53376	-122.192	California	Glenn	56	52	49	46
60250005	32.67619	-115.484	California	Imperial	71	63	62	61
60254003	33.0325	-115.624	California	Imperial	66	56	54	53
60254004	33.21361	-115.545	California	Imperial	64	53	52	50
60270101	36.50861	-116.848	California	Inyo	67	64	63	63
60290007	35.34609	-118.852	California	Kern	80	74	69	64
60290008	35.05444	-119.404	California	Kern	75	69	65	61
60290011	35.05055	-118.147	California	Kern	71	62	60	57
60290014	35.35609	-119.041	California	Kern	77	70	66	61
60290232	35.43887	-119.017	California	Kern	76	70	66	61
60295002	35.23668	-118.789	California	Kern	74	67	63	59
60296001	35.50359	-119.273	California	Kern	74	68	65	61
60311004	36.3144	-119.645	California	Kings	74	68	63	59
60333001	39.0327	-122.922	California	Lake	50	47	44	42

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
60370002	34.1365	-117.924	California	Los Angeles	75	56	52	48
60370016	34.14435	-117.85	California	Los Angeles	88	65	61	56
60370113	34.05111	-118.456	California	Los Angeles	62	49	46	42
60371002	34.17605	-118.317	California	Los Angeles	72	53	49	45
60371103	34.06659	-118.227	California	Los Angeles	61	46	42	39
60371201	34.19925	-118.533	California	Los Angeles	83	64	60	56
60371302	33.90139	-118.205	California	Los Angeles	58	54	52	50
60371602	34.01194	-118.07	California	Los Angeles	63	52	48	44
60371701	34.06703	-117.751	California	Los Angeles	79	61	57	52
60372005	34.1326	-118.127	California	Los Angeles	73	54	50	46
60374002	33.82376	-118.189	California	Los Angeles	57	52	50	49
60376012	34.38344	-118.528	California	Los Angeles	89	66	61	57
60379033	34.67139	-118.131	California	Los Angeles	80	62	59	55
60390004	36.86667	-120.01	California	Madera	70	64	61	57
60392010	36.95326	-120.034	California	Madera	74	68	64	60
60410001	37.97231	-122.52	California	Marin	47	44	42	39
60430006	37.54993	-119.845	California	Mariposa	65	61	58	55
60470003	37.2816	-120.435	California	Merced	72	66	62	58
60530002	36.49577	-121.732	California	Monterey	50	44	42	40
60530008	36.20929	-121.126	California	Monterey	50	44	42	40
60531003	36.69676	-121.637	California	Monterey	46	40	38	36
60550003	38.31094	-122.296	California	Napa	53	49	46	43
60570005	39.23433	-121.057	California	Nevada	62	57	53	49
60570007	39.31656	-120.845	California	Nevada	60	55	51	47
60590007	33.83062	-117.938	California	Orange	62	51	48	46
60591003	33.67464	-117.926	California	Orange	60	50	47	45
60592022	33.63003	-117.676	California	Orange	62	46	43	41
60595001	33.92513	-117.953	California	Orange	68	55	52	49
60610003	38.93568	-121.1	California	Placer	67	60	55	50
60610004	39.10028	-120.953	California	Placer	60	54	50	47
60610006	38.74573	-121.266	California	Placer	70	64	58	53
60650004	34.007	-117.521	California	Riverside	78	61	57	54
60650008	33.7411	-115.821	California	Riverside	56	47	46	45
60650009	33.44787	-117.089	California	Riverside	60	45	43	41
60650012	33.92086	-116.858	California	Riverside	87	65	61	57
60650016	33.58333	-117.083	California	Riverside	64	48	45	42
60651016	33.945	-116.83	California	Riverside	88	66	62	58
60652002	33.70853	-116.215	California	Riverside	74	60	58	55
60655001	33.85275	-116.541	California	Riverside	81	63	60	57
60656001	33.78942	-117.228	California	Riverside	80	59	55	52

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
60658001	33.99958	-117.416	California	Riverside	88	67	63	58
60658005	33.99564	-117.493	California	Riverside	84	64	60	56
60659001	33.67649	-117.331	California	Riverside	75	55	52	49
60659003	33.61241	-114.603	California	Riverside	60	54	53	52
60670002	38.71209	-121.381	California	Sacramento	66	60	55	50
60670006	38.61378	-121.368	California	Sacramento	67	61	56	51
60670010	38.55823	-121.493	California	Sacramento	62	57	52	48
60670011	38.30259	-121.421	California	Sacramento	63	57	53	49
60670012	38.6833	-121.164	California	Sacramento	76	69	63	58
60670014	38.65078	-121.507	California	Sacramento	60	55	51	47
60675003	38.49448	-121.211	California	Sacramento	72	66	60	55
60690002	36.8441	-121.362	California	San Benito	54	47	45	43
60690003	36.48522	-121.157	California	San Benito	62	55	52	50
60710001	34.89501	-117.024	California	San Bernardino	70	58	56	53
60710005	34.2431	-117.272	California	San Bernardino	100	75	70	65
60710012	34.42613	-117.564	California	San Bernardino	86	66	63	59
60710306	34.51001	-117.331	California	San Bernardino	77	61	57	54
60711004	34.10374	-117.629	California	San Bernardino	91	69	65	60
60711234	35.76387	-117.397	California	San Bernardino	64	61	60	59
60712002	34.10002	-117.492	California	San Bernardino	97	74	69	64
60714001	34.41807	-117.286	California	San Bernardino	89	68	64	59
60714003	34.05977	-117.147	California	San Bernardino	96	72	67	62
60719002	34.07139	-116.391	California	San Bernardino	82	66	63	61
60719004	34.10688	-117.274	California	San Bernardino	91	67	63	58
60730001	32.63123	-117.059	California	San Diego	59	52	51	50
60730003	32.79119	-116.942	California	San Diego	63	50	48	46
60730006	32.83646	-117.129	California	San Diego	63	51	49	47
60731001	32.95212	-117.264	California	San Diego	58	49	47	46
60731002	33.12771	-117.075	California	San Diego	58	45	43	41
60731006	32.84224	-116.768	California	San Diego	71	55	53	51
60731008	33.21703	-117.396	California	San Diego	57	44	42	41
60731010	32.70149	-117.15	California	San Diego	54	48	47	46
60731016	32.84547	-117.124	California	San Diego	59	48	46	44
60731201	33.36259	-117.09	California	San Diego	59	45	43	41
60732007	32.55216	-116.938	California	San Diego	54	48	47	46
60771002	37.95074	-121.269	California	San Joaquin	59	54	50	46
60773005	37.6825	-121.441	California	San Joaquin	71	65	61	57
60790005	35.63163	-120.691	California	San Luis Obispo	56	51	49	47
60792006	35.25658	-120.67	California	San Luis Obispo	47	42	41	39
60793001	35.36631	-120.843	California	San Luis Obispo	47	42	41	40

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
60794002	35.03146	-120.501	California	San Luis Obispo	51	45	43	41
60798001	35.49158	-120.668	California	San Luis Obispo	54	49	47	45
60798005	35.64368	-120.231	California	San Luis Obispo	68	62	59	56
60798006	35.35472	-120.04	California	San Luis Obispo	65	60	57	54
60830008	34.46245	-120.026	California	Santa Barbara	52	46	45	44
60830011	34.42778	-119.691	California	Santa Barbara	50	43	42	41
60831008	34.94915	-120.438	California	Santa Barbara	43	38	36	35
60831013	34.72556	-120.428	California	Santa Barbara	55	48	46	45
60831014	34.54166	-119.791	California	Santa Barbara	59	51	50	48
60831018	34.52744	-120.197	California	Santa Barbara	49	45	44	43
60831021	34.40278	-119.458	California	Santa Barbara	59	52	50	49
60831025	34.48974	-120.047	California	Santa Barbara	61	54	53	52
60832004	34.63782	-120.458	California	Santa Barbara	47	42	41	40
60832011	34.44551	-119.828	California	Santa Barbara	50	44	43	42
60833001	34.60582	-120.075	California	Santa Barbara	53	46	45	43
60834003	34.59611	-120.63	California	Santa Barbara	54	49	48	46
60850002	36.99957	-121.575	California	Santa Clara	60	54	50	47
60850005	37.3485	-121.895	California	Santa Clara	58	54	51	48
60851001	37.22686	-121.98	California	Santa Clara	61	56	53	49
60852006	37.07938	-121.6	California	Santa Clara	63	58	54	51
60852009	37.31844	-122.07	California	Santa Clara	58	54	51	47
60870007	36.98392	-121.989	California	Santa Cruz	48	44	41	39
60890004	40.54958	-122.38	California	Shasta	51	46	43	40
60890007	40.45291	-122.299	California	Shasta	57	52	48	45
60890009	40.68925	-122.402	California	Shasta	59	54	50	47
60893003	40.53681	-121.574	California	Shasta	58	55	53	51
60950004	38.10251	-122.238	California	Solano	53	49	47	43
60950005	38.22707	-122.076	California	Solano	57	52	49	45
60953003	38.35837	-121.95	California	Solano	58	54	50	47
60970003	38.4435	-122.71	California	Sonoma	39	37	35	33
60990005	37.64158	-120.995	California	Stanislaus	67	61	57	53
60990006	37.48798	-120.837	California	Stanislaus	77	70	65	60
61010003	39.13877	-121.619	California	Sutter	54	50	46	43
61010004	39.20557	-121.82	California	Sutter	63	58	54	51
61030004	40.26208	-122.094	California	Tehama	64	59	55	52
61030005	40.17583	-122.237	California	Tehama	62	57	54	51
61070009	36.48944	-118.829	California	Tulare	79	73	69	65
61072002	36.33218	-119.291	California	Tulare	71	65	61	57
61072010	36.03183	-119.055	California	Tulare	76	70	66	62
61090005	37.98158	-120.38	California	Tuolumne	61	57	53	50

Site ID	Lat	Long	State	County	O3 DV for Scenario: Base Case	Baseline	70	65
61110007	34.20824	-118.869	California	Ventura	65	51	48	46
61110009	34.40285	-118.81	California	Ventura	65	51	49	46
61111004	34.44657	-119.23	California	Ventura	67	58	56	54
61112002	34.27574	-118.685	California	Ventura	73	57	54	51
61113001	34.25324	-119.143	California	Ventura	55	46	44	42
61130004	38.53445	-121.773	California	Yolo	58	53	50	47
61131003	38.66121	-121.733	California	Yolo	60	55	51	48

Table 2A-8. Design Values (ppb) for Continental U.S. Monitors outside of California

Site ID	Lat	Long	State	County	O3 DV for Scenario: Base Case	Baseline	70	65
10030010	30.498	-87.8814	Alabama	Baldwin	53	52	52	51
10331002	34.75878	-87.6506	Alabama	Colbert	47	45	45	43
10499991	34.2888	-85.9698	Alabama	DeKalb	51	50	50	47
10510001	32.49857	-86.1366	Alabama	Elmore	50	48	48	47
10550011	33.90404	-86.0539	Alabama	Etowah	47	46	46	45
10690004	31.19066	-85.4231	Alabama	Houston	50	49	49	48
10730023	33.55306	-86.815	Alabama	Jefferson	55	54	54	53
10731003	33.48556	-86.915	Alabama	Jefferson	56	55	54	54
10731005	33.33111	-87.0036	Alabama	Jefferson	57	55	55	54
10731009	33.45972	-87.3056	Alabama	Jefferson	56	55	55	54
10731010	33.54528	-86.5492	Alabama	Jefferson	56	55	54	54
10732006	33.38639	-86.8167	Alabama	Jefferson	56	55	55	54
10735002	33.70472	-86.6692	Alabama	Jefferson	54	53	53	52
10735003	33.80167	-86.9425	Alabama	Jefferson	55	54	53	53
10736002	33.57833	-86.7739	Alabama	Jefferson	59	57	57	56
10890014	34.68767	-86.5864	Alabama	Madison	54	53	53	51
10890022	34.77273	-86.7562	Alabama	Madison	51	50	50	48
10970003	30.76994	-88.0875	Alabama	Mobile	53	51	51	51
10972005	30.47467	-88.1411	Alabama	Mobile	53	52	52	51
11011002	32.40712	-86.2564	Alabama	Montgomery	50	49	49	48
11030011	34.51874	-86.9769	Alabama	Morgan	54	54	54	52
11130002	32.46797	-85.0838	Alabama	Russell	50	49	49	48
11170004	33.31732	-86.8251	Alabama	Shelby	55	54	54	53
11190002	32.36401	-88.2019	Alabama	Sumter	50	47	47	46
11250010	33.0896	-87.4597	Alabama	Tuscaloosa	46	45	45	44
40051008	35.20611	-111.653	Arizona	Coconino	63	63	63	63
40070010	33.6547	-111.107	Arizona	Gila	63	63	63	61

Site ID	Lat	Long	State	County	O3 DV for Scenario: Base Case	Baseline	70	65
40128000	34.2319	-113.58	Arizona	La Paz	65	65	65	65
40130019	33.48385	-112.143	Arizona	Maricopa	67	67	67	65
40131004	33.56033	-112.066	Arizona	Maricopa	69	68	68	65
40131010	33.45223	-111.733	Arizona	Maricopa	59	59	59	56
40132001	33.57454	-112.192	Arizona	Maricopa	65	65	65	62
40132005	33.70633	-111.856	Arizona	Maricopa	65	65	65	62
40133002	33.45793	-112.046	Arizona	Maricopa	65	64	64	62
40133003	33.47968	-111.917	Arizona	Maricopa	66	65	65	63
40134003	33.40316	-112.075	Arizona	Maricopa	67	67	67	64
40134004	33.29898	-111.884	Arizona	Maricopa	63	62	62	60
40134005	33.4124	-111.935	Arizona	Maricopa	60	60	60	58
40134008	33.82169	-112.017	Arizona	Maricopa	64	64	64	61
40134010	33.63713	-112.342	Arizona	Maricopa	60	60	60	57
40134011	33.37005	-112.621	Arizona	Maricopa	57	57	57	55
40137003	33.29023	-112.161	Arizona	Maricopa	61	61	61	59
40137020	33.48824	-111.856	Arizona	Maricopa	64	63	63	61
40137021	33.50799	-111.755	Arizona	Maricopa	66	65	65	63
40137022	33.47461	-111.806	Arizona	Maricopa	63	62	62	60
40137024	33.50813	-111.839	Arizona	Maricopa	63	63	63	61
40139508	33.9828	-111.799	Arizona	Maricopa	61	61	61	59
40139702	33.54549	-111.609	Arizona	Maricopa	64	63	63	61
40139704	33.61103	-111.725	Arizona	Maricopa	64	63	63	61
40139706	33.71881	-111.672	Arizona	Maricopa	63	63	63	60
40139997	33.50383	-112.096	Arizona	Maricopa	67	67	67	64
40170119	34.8225	-109.892	Arizona	Navajo	61	59	59	58
40190021	32.17454	-110.737	Arizona	Pima	61	58	58	57
40191011	32.20441	-110.878	Arizona	Pima	57	55	55	53
40191018	32.42526	-111.064	Arizona	Pima	59	58	58	56
40191020	32.04767	-110.774	Arizona	Pima	60	56	56	54
40191028	32.29515	-110.982	Arizona	Pima	57	55	55	54
40191030	31.87952	-110.996	Arizona	Pima	59	56	56	55
40191032	32.173	-110.98	Arizona	Pima	57	54	54	53
40191034	32.38082	-111.127	Arizona	Pima	56	55	54	53
40213001	33.4214	-111.544	Arizona	Pinal	62	62	62	59
40213003	32.95436	-111.762	Arizona	Pinal	59	59	59	57
40213007	32.50831	-111.308	Arizona	Pinal	61	60	60	59
40217001	33.08009	-111.74	Arizona	Pinal	61	61	61	59
40218001	33.29347	-111.286	Arizona	Pinal	65	64	64	62
40258033	34.5467	-112.476	Arizona	Yavapai	63	63	63	63
50199991	34.1795	-93.0988	Arkansas	Clark	55	53	52	49

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
50350005	35.19729	-90.1931	Arkansas	Crittenden	61	60	60	54
51010002	35.83273	-93.2083	Arkansas	Newton	55	53	53	52
51130003	34.45441	-94.1433	Arkansas	Polk	64	62	60	57
51190007	34.75619	-92.2813	Arkansas	Pulaski	53	50	49	48
51191002	34.83572	-92.2606	Arkansas	Pulaski	56	52	52	50
51191008	34.68134	-92.3287	Arkansas	Pulaski	56	53	52	51
51430005	36.1797	-94.1168	Arkansas	Washington	60	58	57	56
80013001	39.83812	-104.95	Colorado	Adams	66	66	65	60
80050002	39.56789	-104.957	Colorado	Arapahoe	70	70	69	64
80050006	39.63852	-104.569	Colorado	Arapahoe	64	64	63	59
80130011	39.95721	-105.238	Colorado	Boulder	65	65	64	60
80310014	39.75176	-105.031	Colorado	Denver	63	63	62	58
80310025	39.70401	-104.998	Colorado	Denver	62	62	61	57
80350004	39.53449	-105.07	Colorado	Douglas	70	70	69	64
80410013	38.95834	-104.817	Colorado	El Paso	64	64	63	61
80410016	38.8531	-104.901	Colorado	El Paso	65	65	65	63
80450012	39.54182	-107.784	Colorado	Garfield	63	63	63	60
80519991	38.9564	-106.986	Colorado	Gunnison	64	64	64	63
80590002	39.80033	-105.1	Colorado	Jefferson	62	62	62	57
80590005	39.63878	-105.139	Colorado	Jefferson	66	67	66	61
80590006	39.9128	-105.189	Colorado	Jefferson	71	71	70	65
80590011	39.74372	-105.178	Colorado	Jefferson	71	71	70	65
80590013	39.54152	-105.298	Colorado	Jefferson	63	63	62	58
80677001	37.13678	-107.629	Colorado	La Plata	64	63	63	63
80677003	37.10258	-107.87	Colorado	La Plata	62	62	62	61
80690007	40.2772	-105.546	Colorado	Larimer	66	66	66	61
80690011	40.59254	-105.141	Colorado	Larimer	71	71	70	65
80690012	40.6421	-105.275	Colorado	Larimer	64	64	63	59
80691004	40.57747	-105.079	Colorado	Larimer	64	64	63	58
80770020	39.13058	-108.314	Colorado	Mesa	64	64	64	62
80810002	40.50695	-107.891	Colorado	Moffat	60	59	59	58
80830006	37.35005	-108.592	Colorado	Montezuma	61	61	61	60
80830101	37.19833	-108.49	Colorado	Montezuma	60	60	60	59
81030005	40.03889	-107.848	Colorado	Rio Blanco	60	60	59	58
81230009	40.38637	-104.737	Colorado	Weld	70	70	69	64
90010017	41.00361	-73.585	Connecticut	Fairfield	70	70	67	59
90011123	41.39917	-73.4431	Connecticut	Fairfield	65	65	62	54
90013007	41.1525	-73.1031	Connecticut	Fairfield	71	70	68	60
90019003	41.11833	-73.3367	Connecticut	Fairfield	73	72	70	62
90031003	41.78472	-72.6317	Connecticut	Hartford	61	61	58	51

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
90050005	41.82134	-73.2973	Connecticut	Litchfield	57	56	54	47
90070007	41.55222	-72.63	Connecticut	Middlesex	65	64	61	53
90090027	41.3014	-72.9029	Connecticut	New Haven	63	63	61	54
90099002	41.26083	-72.55	Connecticut	New Haven	71	71	68	60
90110124	41.35362	-72.0788	Connecticut	New London	66	65	63	56
90131001	41.97639	-72.3881	Connecticut	Tolland	62	62	59	51
90159991	41.8402	-72.01	Connecticut	Windham	57	56	54	47
100010002	38.98475	-75.5552	Delaware	Kent	59	58	56	49
100031007	39.55111	-75.7308	Delaware	New Castle	60	59	57	49
100031010	39.81722	-75.5639	Delaware	New Castle	61	59	57	49
100031013	39.77389	-75.4964	Delaware	New Castle	62	61	58	50
100032004	39.73944	-75.5581	Delaware	New Castle	60	59	56	49
100051002	38.64448	-75.6127	Delaware	Sussex	61	60	58	51
100051003	38.7792	-75.1627	Delaware	Sussex	64	63	61	55
110010041	38.89722	-76.9528	District Of Columbia	District of Columbia	58	57	55	46
110010043	38.92185	-77.0132	District Of Columbia	District of Columbia	62	61	58	49
120013011	29.54472	-82.2961	Florida	Alachua	50	50	50	49
120030002	30.20111	-82.4411	Florida	Baker	52	51	50	50
120050006	30.13043	-85.7315	Florida	Bay	52	51	50	50
120090007	28.05361	-80.6286	Florida	Brevard	53	52	52	52
120094001	28.31056	-80.6156	Florida	Brevard	54	53	53	53
120110033	26.07354	-80.3385	Florida	Broward	52	51	51	51
120112003	26.29203	-80.0965	Florida	Broward	50	50	50	50
120118002	26.087	-80.111	Florida	Broward	53	53	53	53
120210004	26.27	-81.711	Florida	Collier	49	48	48	48
120230002	30.17806	-82.6192	Florida	Columbia	52	51	51	50
120310077	30.47773	-81.5873	Florida	Duval	52	50	50	49
120310100	30.261	-81.454	Florida	Duval	53	51	51	50
120310106	30.37822	-81.8409	Florida	Duval	52	50	50	50
120330004	30.52537	-87.2036	Florida	Escambia	56	53	52	52
120330018	30.36805	-87.271	Florida	Escambia	58	55	55	54
120550003	27.18889	-81.3406	Florida	Highlands	53	52	51	51
120570081	27.74003	-82.4651	Florida	Hillsborough	60	57	57	57
120571035	27.92806	-82.4547	Florida	Hillsborough	56	54	54	54
120571065	27.89222	-82.5386	Florida	Hillsborough	60	58	58	58
120573002	27.96565	-82.2304	Florida	Hillsborough	56	55	55	54
120590004	30.84861	-85.6039	Florida	Holmes	49	48	48	46
120619991	27.8492	-80.4554	Florida	Indian River	54	53	53	53
120690002	28.525	-81.7233	Florida	Lake	54	52	52	52

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
120712002	26.54786	-81.98	Florida	Lee	52	51	51	51
120713002	26.44889	-81.9394	Florida	Lee	50	49	48	48
120730012	30.43972	-84.3464	Florida	Leon	48	48	48	47
120730013	30.48444	-84.1994	Florida	Leon	48	48	48	47
120813002	27.63278	-82.5461	Florida	Manatee	53	51	51	51
120814012	27.48056	-82.6189	Florida	Manatee	53	52	51	51
120814013	27.44944	-82.5222	Florida	Manatee	51	49	49	49
120830003	29.17028	-82.1008	Florida	Marion	52	51	51	51
120830004	29.1925	-82.1733	Florida	Marion	50	49	49	49
120850007	27.17246	-80.2407	Florida	Martin	51	50	50	50
120860027	25.73338	-80.1618	Florida	Miami-Dade	58	58	58	58
120860029	25.58638	-80.3268	Florida	Miami-Dade	56	56	56	56
120910002	30.42653	-86.6662	Florida	Okaloosa	52	50	50	49
120950008	28.45417	-81.3814	Florida	Orange	58	56	56	56
120952002	28.59639	-81.3625	Florida	Orange	59	58	58	58
120972002	28.34722	-81.6367	Florida	Osceola	52	51	51	51
120990009	26.73083	-80.2339	Florida	Palm Beach	55	54	54	54
120990020	26.59123	-80.0609	Florida	Palm Beach	54	53	53	53
121010005	28.33194	-82.3058	Florida	Pasco	53	51	51	51
121012001	28.195	-82.7581	Florida	Pasco	54	53	53	52
121030004	27.94639	-82.7319	Florida	Pinellas	55	54	54	54
121030018	27.78587	-82.7399	Florida	Pinellas	55	53	53	53
121035002	28.09	-82.7008	Florida	Pinellas	53	52	52	52
121056005	27.93944	-82.0003	Florida	Polk	54	52	52	51
121056006	28.02889	-81.9722	Florida	Polk	55	53	52	52
121130015	30.39413	-87.008	Florida	Santa Rosa	56	54	53	53
121151005	27.30694	-82.5706	Florida	Sarasota	57	56	56	55
121151006	27.35028	-82.48	Florida	Sarasota	55	53	53	53
121152002	27.08919	-82.3626	Florida	Sarasota	54	52	52	52
121171002	28.74611	-81.3106	Florida	Seminole	55	53	53	52
121272001	29.10889	-80.9939	Florida	Volusia	47	45	45	45
121275002	29.20667	-81.0525	Florida	Volusia	51	49	49	49
121290001	30.0925	-84.1611	Florida	Wakulla	53	52	51	51
130210012	32.80541	-83.5435	Georgia	Bibb	53	49	49	47
130510021	32.06923	-81.0488	Georgia	Chatham	51	50	50	49
130550001	34.47429	-85.408	Georgia	Chattooga	51	49	49	47
130590002	33.91807	-83.3445	Georgia	Clarke	51	50	50	49
130670003	34.01548	-84.6074	Georgia	Cobb	55	54	54	53
130730001	33.58214	-82.1312	Georgia	Columbia	51	51	50	49
130770002	33.40404	-84.746	Georgia	Coweta	49	48	48	47

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
130850001	34.37632	-84.0598	Georgia	Dawson	49	48	48	47
130890002	33.68797	-84.2905	Georgia	DeKalb	56	55	55	54
130970004	33.74366	-84.7792	Georgia	Douglas	52	51	51	50
131210055	33.72019	-84.3571	Georgia	Fulton	59	58	58	57
131270006	31.16974	-81.4959	Georgia	Glynn	48	47	47	47
131350002	33.96127	-84.069	Georgia	Gwinnett	55	54	54	53
131510002	33.43358	-84.1617	Georgia	Henry	59	58	58	57
132130003	34.7852	-84.6264	Georgia	Murray	52	51	51	48
132150008	32.5213	-84.9448	Georgia	Muscogee	50	50	49	49
132230003	33.9285	-85.0453	Georgia	Paulding	53	51	51	49
132319991	33.1787	-84.4052	Georgia	Pike	52	51	51	50
132450091	33.43335	-82.0222	Georgia	Richmond	53	52	51	50
132470001	33.59108	-84.0653	Georgia	Rockdale	55	54	54	53
132611001	31.9543	-84.0811	Georgia	Sumter	53	52	52	51
160010010	43.6007	-116.348	Idaho	Ada	60	59	59	59
160010017	43.5776	-116.178	Idaho	Ada	60	60	60	60
160010019	43.63459	-116.234	Idaho	Ada	54	53	53	53
160550003	47.78891	-116.805	Idaho	Kootenai	47	47	47	47
170010007	39.91541	-91.3359	Illinois	Adams	57	56	55	54
170190007	40.24491	-88.1885	Illinois	Champaign	60	59	59	56
170191001	40.05224	-88.3725	Illinois	Champaign	61	60	60	57
170230001	39.21086	-87.6683	Illinois	Clark	58	58	58	53
170310001	41.67099	-87.7325	Illinois	Cook	64	63	63	58
170310032	41.75583	-87.5454	Illinois	Cook	57	57	57	57
170310064	41.79079	-87.6016	Illinois	Cook	53	52	52	52
170310076	41.7514	-87.7135	Illinois	Cook	63	63	62	58
170311003	41.98433	-87.792	Illinois	Cook	49	49	49	50
170311601	41.66812	-87.9906	Illinois	Cook	62	62	61	57
170314002	41.85524	-87.7525	Illinois	Cook	52	51	51	51
170314007	42.06029	-87.8632	Illinois	Cook	48	48	48	49
170314201	42.14	-87.7992	Illinois	Cook	56	55	55	57
170317002	42.06186	-87.6742	Illinois	Cook	54	54	54	56
170436001	41.81305	-88.0728	Illinois	DuPage	58	58	57	53
170491001	39.06716	-88.5489	Illinois	Effingham	58	58	57	54
170650002	38.08216	-88.6249	Illinois	Hamilton	64	65	65	61
170831001	39.11054	-90.3241	Illinois	Jersey	61	60	60	59
170859991	42.2869	-89.9997	Illinois	Jo Daviess	58	57	56	55
170890005	42.04915	-88.273	Illinois	Kane	63	62	62	58
170971007	42.46757	-87.81	Illinois	Lake	57	57	57	58
171110001	42.22144	-88.2422	Illinois	McHenry	61	60	60	55

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
171132003	40.51874	-88.9969	Illinois	McLean	59	57	56	53
171150013	39.86683	-88.9256	Illinois	Macon	59	59	58	56
171170002	39.39608	-89.8097	Illinois	Macoupin	57	56	55	54
171190008	38.89019	-90.148	Illinois	Madison	62	61	61	59
171191009	38.72657	-89.96	Illinois	Madison	63	62	62	60
171193007	38.86067	-90.1059	Illinois	Madison	62	61	60	59
171199991	38.869	-89.6228	Illinois	Madison	60	59	59	58
171430024	40.68742	-89.6069	Illinois	Peoria	53	50	50	47
171431001	40.7455	-89.5859	Illinois	Peoria	61	58	57	54
171570001	38.17628	-89.7885	Illinois	Randolph	58	57	57	55
171613002	41.51473	-90.5174	Illinois	Rock Island	50	49	49	47
171630010	38.61203	-90.1605	Illinois	Saint Clair	62	61	61	59
171670014	39.83152	-89.6409	Illinois	Sangamon	59	58	57	56
171971011	41.22154	-88.191	Illinois	Will	55	55	54	50
172012001	42.33498	-89.0378	Illinois	Winnebago	58	57	57	53
180030002	41.22142	-85.0168	Indiana	Allen	57	56	56	53
180030004	41.09497	-85.1018	Indiana	Allen	58	57	57	53
180110001	39.99748	-86.3952	Indiana	Boone	60	60	60	55
180150002	40.54046	-86.553	Indiana	Carroll	58	58	57	54
180190008	38.39383	-85.6642	Indiana	Clark	65	65	64	58
180350010	40.30002	-85.2454	Indiana	Delaware	56	56	55	52
180390007	41.71805	-85.8306	Indiana	Elkhart	56	55	55	50
180431004	38.30806	-85.8342	Indiana	Floyd	65	65	64	58
180550001	38.98558	-86.9901	Indiana	Greene	68	68	67	62
180570006	40.0683	-85.9925	Indiana	Hamilton	59	58	58	54
180590003	39.93504	-85.8405	Indiana	Hancock	55	55	54	50
180630004	39.759	-86.3971	Indiana	Hendricks	57	56	56	52
180690002	40.96071	-85.3798	Indiana	Huntington	55	54	54	50
180710001	38.92084	-86.0805	Indiana	Jackson	57	57	57	51
180810002	39.41724	-86.1524	Indiana	Johnson	58	58	58	53
180839991	38.7408	-87.4853	Indiana	Knox	65	65	64	59
180890022	41.60668	-87.3047	Indiana	Lake	55	55	55	53
180890030	41.6814	-87.4947	Indiana	Lake	57	56	56	54
180892008	41.63946	-87.4936	Indiana	Lake	57	56	56	54
180910005	41.71702	-86.9077	Indiana	LaPorte	66	65	65	61
180910010	41.6291	-86.6846	Indiana	LaPorte	59	59	59	55
180950010	40.00255	-85.6569	Indiana	Madison	55	55	54	50
180970050	39.85892	-86.0213	Indiana	Marion	60	59	59	54
180970057	39.74902	-86.1863	Indiana	Marion	59	58	58	53
180970073	39.78949	-86.0609	Indiana	Marion	60	60	59	55

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
180970078	39.8111	-86.1145	Indiana	Marion	59	59	58	54
181090005	39.57563	-86.4779	Indiana	Morgan	56	56	56	51
181230009	38.11316	-86.6036	Indiana	Perry	65	65	65	59
181270024	41.61756	-87.1992	Indiana	Porter	57	57	57	55
181270026	41.51029	-87.0385	Indiana	Porter	55	54	54	51
181290003	38.00529	-87.7184	Indiana	Posey	61	61	61	56
181410010	41.5517	-86.3706	Indiana	St. Joseph	52	52	51	48
181410015	41.69669	-86.2147	Indiana	St. Joseph	58	57	57	52
181411007	41.7426	-86.1105	Indiana	St. Joseph	53	53	52	48
181450001	39.61342	-85.8706	Indiana	Shelby	62	61	61	56
181630013	38.11395	-87.537	Indiana	Vanderburgh	63	63	62	58
181630021	38.01325	-87.5779	Indiana	Vanderburgh	63	63	62	58
181670018	39.48615	-87.4014	Indiana	Vigo	55	54	54	50
181670024	39.56056	-87.3131	Indiana	Vigo	55	55	54	50
181699991	40.816	-85.6611	Indiana	Wabash	61	61	61	57
181730008	38.052	-87.2783	Indiana	Warrick	63	63	63	58
181730009	38.1945	-87.3414	Indiana	Warrick	61	61	60	55
181730011	37.95451	-87.3219	Indiana	Warrick	64	64	63	58
190170011	42.74306	-92.5131	Iowa	Bremer	53	52	52	51
190450021	41.875	-90.1776	Iowa	Clinton	57	56	55	53
190850007	41.83226	-95.9282	Iowa	Harrison	55	54	54	53
190851101	41.78026	-95.9484	Iowa	Harrison	56	55	55	54
191130028	41.91056	-91.6519	Iowa	Linn	55	54	54	53
191130033	42.28101	-91.5269	Iowa	Linn	53	53	53	52
191130040	41.97677	-91.6877	Iowa	Linn	53	53	53	52
191370002	40.96911	-95.045	Iowa	Montgomery	56	55	55	54
191471002	43.1237	-94.6935	Iowa	Palo Alto	57	56	55	55
191530030	41.60316	-93.6431	Iowa	Polk	49	48	48	47
191630014	41.69917	-90.5219	Iowa	Scott	55	54	53	52
191630015	41.53001	-90.5876	Iowa	Scott	56	56	55	53
191690011	41.88287	-93.6878	Iowa	Story	50	49	49	48
191770006	40.69508	-92.0063	Iowa	Van Buren	55	54	53	51
191810022	41.28553	-93.584	Iowa	Warren	53	52	51	50
200910010	38.83858	-94.7464	Kansas	Johnson	61	60	60	59
201030003	39.32739	-94.951	Kansas	Leavenworth	58	57	57	56
201070002	38.13588	-94.732	Kansas	Linn	59	58	58	57
201619991	39.1021	-96.6096	Kansas	Riley	62	61	61	60
201730001	37.78139	-97.3372	Kansas	Sedgwick	55	54	54	53
201730010	37.70207	-97.3148	Kansas	Sedgwick	64	63	62	61
201730018	37.89751	-97.4921	Kansas	Sedgwick	62	61	61	60

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
201770013	39.02427	-95.7113	Kansas	Shawnee	62	61	61	61
201910002	37.47689	-97.3664	Kansas	Sumner	65	64	64	63
201950001	38.77008	-99.7634	Kansas	Trego	65	65	65	64
202090021	39.11722	-94.6356	Kansas	Wyandotte	55	54	54	53
210130002	36.60843	-83.7369	Kentucky	Bell	50	49	49	45
210150003	38.91833	-84.8526	Kentucky	Boone	59	58	57	51
210190017	38.45934	-82.6404	Kentucky	Boyd	58	58	57	50
210290006	37.98629	-85.7119	Kentucky	Bullitt	62	62	61	56
210373002	39.02188	-84.4745	Kentucky	Campbell	66	66	65	58
210430500	38.23887	-82.9881	Kentucky	Carter	56	56	55	49
210470006	36.91171	-87.3233	Kentucky	Christian	53	53	52	49
210590005	37.78078	-87.0753	Kentucky	Daviess	67	67	67	61
210610501	37.13194	-86.1478	Kentucky	Edmonson	57	57	57	53
210670012	38.06503	-84.4976	Kentucky	Fayette	58	58	58	52
210890007	38.54814	-82.7312	Kentucky	Greenup	59	59	58	51
210910012	37.93829	-86.8972	Kentucky	Hancock	66	66	66	60
210930006	37.70561	-85.8526	Kentucky	Hardin	59	59	58	53
211010014	37.8712	-87.4638	Kentucky	Henderson	68	68	68	63
211110027	38.13784	-85.5765	Kentucky	Jefferson	66	65	65	59
211110051	38.06091	-85.898	Kentucky	Jefferson	68	68	67	61
211110067	38.22876	-85.6545	Kentucky	Jefferson	71	71	70	63
211130001	37.89147	-84.5883	Kentucky	Jessamine	56	57	57	52
211390003	37.15539	-88.394	Kentucky	Livingston	61	65	65	60
211451024	37.05822	-88.5725	Kentucky	McCracken	64	69	68	64
211759991	37.9214	-83.0662	Kentucky	Morgan	57	56	56	49
211850004	38.4002	-85.4443	Kentucky	Oldham	68	68	67	60
211930003	37.28329	-83.2093	Kentucky	Perry	56	56	55	49
211950002	37.4826	-82.5353	Kentucky	Pike	56	56	55	48
211990003	37.09798	-84.6115	Kentucky	Pulaski	51	51	50	46
212130004	36.70861	-86.5663	Kentucky	Simpson	53	53	53	48
212218001	36.78389	-87.8519	Kentucky	Trigg	56	57	56	51
212219991	36.7841	-87.8499	Kentucky	Trigg	57	58	57	52
212270008	37.03544	-86.2506	Kentucky	Warren	51	50	50	46
212299991	37.7046	-85.0485	Kentucky	Washington	57	57	57	52
220050004	30.23389	-90.9683	Louisiana	Ascension	63	62	62	61
220150008	32.53626	-93.7489	Louisiana	Bossier	66	64	62	59
220170001	32.67639	-93.8597	Louisiana	Caddo	64	62	60	56
220190002	30.14333	-93.3719	Louisiana	Calcasieu	66	66	65	64
220190008	30.26167	-93.2842	Louisiana	Calcasieu	60	60	59	58
220190009	30.22778	-93.5783	Louisiana	Calcasieu	63	62	60	57

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
220330003	30.41976	-91.182	Louisiana	East Baton Rouge	67	67	67	65
220330009	30.46198	-91.1792	Louisiana	East Baton Rouge	64	63	63	62
220330013	30.70092	-91.0561	Louisiana	East Baton Rouge	60	59	59	58
220470009	30.22056	-91.3161	Louisiana	Iberville	62	62	61	60
220470012	30.20699	-91.1299	Louisiana	Iberville	65	65	64	63
220511001	30.04357	-90.2751	Louisiana	Jefferson	64	64	63	63
220550007	30.2175	-92.0514	Louisiana	Lafayette	60	60	59	58
220570004	29.76389	-90.7652	Louisiana	Lafourche	62	61	61	60
220630002	30.3125	-90.8125	Louisiana	Livingston	62	62	62	61
220710012	29.99444	-90.1028	Louisiana	Orleans	60	59	58	58
220730004	32.50971	-92.0461	Louisiana	Ouachita	55	55	55	54
220770001	30.68174	-91.3662	Louisiana	Pointe Coupee	63	62	62	61
220870004	29.93961	-89.9239	Louisiana	St. Bernard	59	58	58	57
220890003	29.98417	-90.4106	Louisiana	St. Charles	61	60	60	59
220930002	29.99444	-90.82	Louisiana	St. James	58	58	58	57
220950002	30.05833	-90.6083	Louisiana	St. John the Baptist	63	62	62	61
221030002	30.4293	-90.1997	Louisiana	St. Tammany	63	62	62	61
221210001	30.50064	-91.2136	Louisiana	West Baton Rouge	59	59	59	58
230010014	43.97462	-70.1246	Maine	Androscoggin	50	49	48	44
230052003	43.56104	-70.2073	Maine	Cumberland	57	57	55	50
230090102	44.3517	-68.227	Maine	Hancock	58	57	56	52
230090103	44.37705	-68.2609	Maine	Hancock	55	54	53	49
230112005	44.23062	-69.785	Maine	Kennebec	50	50	49	45
230130004	43.91796	-69.2606	Maine	Knox	55	55	53	49
230173001	44.25092	-70.8606	Maine	Oxford	46	45	45	42
230194008	44.73598	-68.6708	Maine	Penobscot	47	46	45	42
230230006	44.005	-69.8278	Maine	Sagadahoc	49	49	47	43
230290019	44.53191	-67.5959	Maine	Washington	49	49	48	44
230290032	44.96363	-67.0607	Maine	Washington	46	46	45	42
230310038	43.65676	-70.6291	Maine	York	49	48	47	43
230310040	43.58889	-70.8773	Maine	York	52	51	50	46
230312002	43.34317	-70.471	Maine	York	60	59	57	52
240030014	38.9025	-76.6531	Maryland	Anne Arundel	64	63	61	51
240051007	39.46202	-76.6313	Maryland	Baltimore	65	63	61	53
240053001	39.31083	-76.4744	Maryland	Baltimore	67	66	63	53
240090011	38.53672	-76.6172	Maryland	Calvert	63	63	60	51
240130001	39.44417	-77.0417	Maryland	Carroll	61	60	59	51

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
240150003	39.70111	-75.86	Maryland	Cecil	66	65	62	54
240170010	38.50417	-76.8119	Maryland	Charles	61	60	58	50
240199991	38.445	-76.1114	Maryland	Dorchester	61	60	58	52
240210037	39.42276	-77.3752	Maryland	Frederick	62	62	60	52
240230002	39.70595	-79.012	Maryland	Garrett	59	59	58	52
240251001	39.41	-76.2967	Maryland	Harford	74	73	70	59
240259001	39.56333	-76.2039	Maryland	Harford	63	61	59	49
240290002	39.3052	-75.7972	Maryland	Kent	62	61	58	50
240313001	39.11444	-77.1069	Maryland	Montgomery	60	59	57	49
240330030	39.05528	-76.8783	Maryland	Prince George's	61	60	58	49
240338003	38.81194	-76.7442	Maryland	Prince George's	63	62	60	50
240339991	39.0284	-76.8171	Maryland	Prince George's	62	61	58	50
240430009	39.56558	-77.7216	Maryland	Washington	60	59	58	52
245100054	39.32889	-76.5525	Maryland	Baltimore (City)	62	62	59	50
250010002	41.9758	-70.0236	Massachusetts	Barnstable	59	59	57	51
250034002	42.63668	-73.1674	Massachusetts	Berkshire	57	57	55	50
250051002	41.63328	-70.8792	Massachusetts	Bristol	59	59	57	50
250070001	41.33047	-70.7852	Massachusetts	Dukes	64	64	62	54
250092006	42.47464	-70.9708	Massachusetts	Essex	58	57	56	52
250094005	42.81441	-70.8178	Massachusetts	Essex	57	56	55	50
250095005	42.77084	-71.1023	Massachusetts	Essex	56	56	54	49
250130008	42.19438	-72.5551	Massachusetts	Hampden	59	59	56	49
250150103	42.40058	-72.5231	Massachusetts	Hampshire	52	52	50	44
250154002	42.29849	-72.3341	Massachusetts	Hampshire	57	56	54	47
250170009	42.62668	-71.3621	Massachusetts	Middlesex	55	54	52	46
250171102	42.41357	-71.4828	Massachusetts	Middlesex	54	53	51	45
250213003	42.21177	-71.114	Massachusetts	Norfolk	59	59	57	52
250250041	42.31737	-70.9684	Massachusetts	Suffolk	56	55	54	50
250250042	42.3295	-71.0826	Massachusetts	Suffolk	49	49	48	44
250270015	42.27432	-71.8755	Massachusetts	Worcester	55	55	53	47
250270024	42.0997	-71.6194	Massachusetts	Worcester	55	54	53	47
260050003	42.76779	-86.1486	Michigan	Allegan	70	69	69	63
260190003	44.61694	-86.1094	Michigan	Benzie	62	61	61	56
260210014	42.19779	-86.3097	Michigan	Berrien	68	68	67	62
260270003	41.89557	-86.0016	Michigan	Cass	63	62	62	57
260370001	42.79834	-84.3938	Michigan	Clinton	57	56	55	51
260490021	43.04722	-83.6702	Michigan	Genesee	61	60	60	56
260492001	43.16834	-83.4615	Michigan	Genesee	60	60	59	55
260630007	43.83639	-82.6429	Michigan	Huron	61	61	60	57
260650012	42.73862	-84.5346	Michigan	Ingham	57	56	56	52

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
260770008	42.27807	-85.5419	Michigan	Kalamazoo	61	60	59	55
260810020	42.98417	-85.6713	Michigan	Kent	60	59	59	54
260810022	43.17667	-85.4166	Michigan	Kent	59	58	58	53
260910007	41.99557	-83.9466	Michigan	Lenawee	60	60	59	55
260990009	42.73139	-82.7935	Michigan	Macomb	67	67	66	62
260991003	42.51334	-83.006	Michigan	Macomb	69	68	68	63
261010922	44.307	-86.2426	Michigan	Manistee	61	60	60	55
261050007	43.95333	-86.2944	Michigan	Mason	62	61	60	56
261130001	44.31056	-84.8919	Michigan	Missaukee	58	57	57	53
261210039	43.27806	-86.3111	Michigan	Muskegon	66	66	65	60
261250001	42.46306	-83.1832	Michigan	Oakland	66	65	65	60
261390005	42.89445	-85.8527	Michigan	Ottawa	63	62	62	57
261470005	42.95334	-82.4562	Michigan	St. Clair	65	65	65	60
261530001	46.28888	-85.9502	Michigan	Schoolcraft	60	60	59	55
261579991	43.6138	-83.3591	Michigan	Tuscola	58	57	57	52
261610008	42.24057	-83.5996	Michigan	Washtenaw	62	62	62	58
261619991	42.4165	-83.902	Michigan	Washtenaw	60	60	59	55
261630001	42.22862	-83.2082	Michigan	Wayne	61	61	61	57
261630019	42.43084	-83.0001	Michigan	Wayne	70	70	69	65
261659991	44.1809	-85.739	Michigan	Wexford	56	55	55	51
270031001	45.40184	-93.2031	Minnesota	Anoka	53	53	53	52
270031002	45.13768	-93.2076	Minnesota	Anoka	57	56	56	56
270353204	46.39674	-94.1303	Minnesota	Crow Wing	51	49	49	49
270495302	44.47375	-93.0126	Minnesota	Goodhue	53	53	53	52
270953051	46.2053	-93.7595	Minnesota	Mille Lacs	48	47	47	47
271095008	43.99691	-92.4504	Minnesota	Olmsted	53	53	53	52
271377550	46.81826	-92.0894	Minnesota	Saint Louis	42	41	41	40
271390505	44.79144	-93.5125	Minnesota	Scott	54	53	53	53
271453052	45.54984	-94.1335	Minnesota	Stearns	53	50	50	50
271636015	45.11728	-92.8553	Minnesota	Washington	52	52	51	51
271713201	45.20916	-93.6692	Minnesota	Wright	55	52	52	52
280010004	31.56075	-91.3904	Mississippi	Adams	55	54	54	53
280110001	33.74606	-90.723	Mississippi	Bolivar	61	60	60	59
280330002	34.82166	-89.9878	Mississippi	DeSoto	57	55	55	51
280450003	30.30083	-89.3959	Mississippi	Hancock	53	50	50	49
280470008	30.39037	-89.0498	Mississippi	Harrison	56	51	51	50
280490010	32.38573	-90.1412	Mississippi	Hinds	49	48	48	47
280590006	30.37829	-88.5339	Mississippi	Jackson	59	58	58	57
280750003	32.36457	-88.7315	Mississippi	Lauderdale	50	49	49	47
280810005	34.26492	-88.7662	Mississippi	Lee	51	50	50	48

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
281619991	34.0026	-89.799	Mississippi	Yalobusha	52	51	51	49
290030001	39.9544	-94.849	Missouri	Andrew	60	59	58	57
290190011	39.0786	-92.3152	Missouri	Boone	56	56	56	55
290270002	38.70608	-92.0931	Missouri	Callaway	56	55	55	54
290370003	38.75976	-94.58	Missouri	Cass	57	57	57	56
290390001	37.69	-94.035	Missouri	Cedar	61	60	59	57
290470003	39.40745	-94.2654	Missouri	Clay	63	62	62	61
290470005	39.30309	-94.3766	Missouri	Clay	62	61	61	60
290470006	39.33191	-94.5808	Missouri	Clay	64	63	63	62
290490001	39.5306	-94.556	Missouri	Clinton	64	63	63	62
290770036	37.25614	-93.2999	Missouri	Greene	56	55	55	54
290770042	37.31951	-93.2046	Missouri	Greene	58	57	57	56
290970004	37.2385	-94.4247	Missouri	Jasper	65	62	61	58
290990019	38.44863	-90.3985	Missouri	Jefferson	64	63	63	62
291130003	39.0447	-90.8647	Missouri	Lincoln	63	62	62	61
291370001	39.47514	-91.7891	Missouri	Monroe	58	57	57	55
291570001	37.70264	-89.6986	Missouri	Perry	62	62	62	59
291831002	38.87255	-90.2265	Missouri	Saint Charles	66	65	65	64
291831004	38.8994	-90.4492	Missouri	Saint Charles	65	64	64	62
291860005	37.90084	-90.4239	Missouri	Sainte Genevieve	61	60	60	59
291890005	38.4902	-90.7052	Missouri	Saint Louis	59	59	58	57
291890014	38.7109	-90.4759	Missouri	Saint Louis	65	64	64	63
292130004	36.70773	-93.222	Missouri	Taney	59	57	57	55
295100085	38.6565	-90.1986	Missouri	St. Louis City	63	63	62	61
300870001	45.36615	-106.49	Montana	Rosebud	53	52	52	52
310550019	41.24749	-95.9731	Nebraska	Douglas	58	57	57	57
310550028	41.20796	-95.9459	Nebraska	Douglas	51	50	50	50
310550035	41.30676	-95.961	Nebraska	Douglas	54	53	53	53
311090016	40.98472	-96.6772	Nebraska	Lancaster	47	47	46	46
320010002	39.47247	-118.784	Nevada	Churchill	52	52	52	52
320030022	36.39101	-114.907	Nevada	Clark	63	62	62	60
320030023	36.80791	-114.061	Nevada	Clark	57	57	57	56
320030043	36.10637	-115.253	Nevada	Clark	69	69	69	65
320030071	36.16975	-115.263	Nevada	Clark	69	69	69	65
320030073	36.17342	-115.333	Nevada	Clark	68	68	68	65
320030075	36.27058	-115.238	Nevada	Clark	68	67	67	64
320030538	36.14296	-115.056	Nevada	Clark	63	63	63	60
320030540	36.1419	-115.079	Nevada	Clark	63	63	63	60
320030601	35.97813	-114.846	Nevada	Clark	65	65	65	63
320031019	35.78567	-115.357	Nevada	Clark	67	67	67	65

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
320032002	36.19126	-115.123	Nevada	Clark	63	63	63	60
320190006	39.60279	-119.248	Nevada	Lyon	60	60	60	60
320310016	39.52508	-119.808	Nevada	Washoe	59	58	58	58
320310020	39.46922	-119.775	Nevada	Washoe	59	59	59	59
320310025	39.39984	-119.74	Nevada	Washoe	59	59	59	58
320311005	39.54092	-119.747	Nevada	Washoe	59	59	59	59
320312002	39.25041	-119.957	Nevada	Washoe	53	53	53	53
320312009	39.64526	-119.84	Nevada	Washoe	59	59	59	58
325100002	39.16725	-119.732	Nevada	Carson City	59	59	59	59
330012004	43.56611	-71.4964	New Hampshire	Belknap	51	51	50	46
330050007	42.93047	-72.2724	New Hampshire	Cheshire	50	50	48	44
330074001	44.27017	-71.3038	New Hampshire	Coos	58	57	57	54
330074002	44.30817	-71.2177	New Hampshire	Coos	51	51	50	48
330090010	43.62961	-72.3096	New Hampshire	Grafton	50	49	48	45
330111011	42.71866	-71.5224	New Hampshire	Hillsborough	54	53	51	46
330115001	42.86175	-71.8784	New Hampshire	Hillsborough	57	56	55	49
330131007	43.2185	-71.5145	New Hampshire	Merrimack	52	52	50	46
330150014	43.07533	-70.748	New Hampshire	Rockingham	54	53	52	47
330150016	43.04528	-70.7138	New Hampshire	Rockingham	54	54	52	48
330150018	42.86254	-71.3802	New Hampshire	Rockingham	55	55	53	47
340010006	39.46487	-74.4487	New Jersey	Atlantic	60	59	57	50
340030006	40.87044	-73.992	New Jersey	Bergen	64	63	61	53
340071001	39.68425	-74.8615	New Jersey	Camden	68	67	64	55
340110007	39.42227	-75.0252	New Jersey	Cumberland	59	57	55	47
340130003	40.72099	-74.1929	New Jersey	Essex	65	64	61	53
340150002	39.80034	-75.2121	New Jersey	Gloucester	69	68	65	56
340170006	40.67025	-74.1261	New Jersey	Hudson	64	63	61	53
340190001	40.51526	-74.8067	New Jersey	Hunterdon	63	62	60	52
340210005	40.28309	-74.7426	New Jersey	Mercer	64	63	61	53
340219991	40.3125	-74.8729	New Jersey	Mercer	62	61	59	51
340230011	40.46218	-74.4294	New Jersey	Middlesex	66	65	62	53
340250005	40.27765	-74.0051	New Jersey	Monmouth	67	65	63	54
340273001	40.78763	-74.6763	New Jersey	Morris	63	62	60	52
340290006	40.06483	-74.4441	New Jersey	Ocean	67	66	63	54
340315001	41.05862	-74.2555	New Jersey	Passaic	62	61	59	52
340410007	40.92458	-75.0678	New Jersey	Warren	52	52	50	44
350010023	35.1343	-106.585	New Mexico	Bernalillo	58	58	58	57
350010024	35.0631	-106.579	New Mexico	Bernalillo	59	59	59	58
350010027	35.1539	-106.697	New Mexico	Bernalillo	62	62	62	61
350010029	35.01708	-106.657	New Mexico	Bernalillo	59	59	59	58

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
350010032	35.06407	-106.762	New Mexico	Bernalillo	58	57	57	56
350011012	35.1852	-106.508	New Mexico	Bernalillo	63	62	62	61
350011013	35.19324	-106.614	New Mexico	Bernalillo	60	60	60	58
350130008	31.93056	-106.631	New Mexico	Dona Ana	57	56	56	56
350130017	31.79583	-106.558	New Mexico	Dona Ana	58	58	58	57
350130020	32.04111	-106.409	New Mexico	Dona Ana	59	58	58	58
350130021	31.79611	-106.584	New Mexico	Dona Ana	62	61	61	61
350130022	31.78778	-106.683	New Mexico	Dona Ana	62	61	61	61
350130023	32.3175	-106.768	New Mexico	Dona Ana	57	57	56	56
350171003	32.69194	-108.124	New Mexico	Grant	61	61	61	60
350250008	32.72666	-103.123	New Mexico	Lea	61	61	61	60
350290003	32.2558	-107.723	New Mexico	Luna	58	57	57	56
350431001	35.29944	-106.548	New Mexico	Sandoval	55	55	55	54
350439004	35.61528	-106.724	New Mexico	Sandoval	58	58	58	58
350450009	36.74222	-107.977	New Mexico	San Juan	57	57	57	56
350450018	36.80973	-107.652	New Mexico	San Juan	62	62	62	61
350451005	36.79667	-108.473	New Mexico	San Juan	56	55	55	54
350451233	36.8071	-108.695	New Mexico	San Juan	56	55	55	54
350490021	35.61975	-106.08	New Mexico	Santa Fe	60	59	59	59
350610008	34.8147	-106.74	New Mexico	Valencia	58	58	58	56
360010012	42.68075	-73.7573	New York	Albany	57	56	54	49
360050133	40.8679	-73.8781	New York	Bronx	67	66	64	57
360130006	42.49963	-79.3188	New York	Chautauqua	61	61	60	55
360130011	42.29071	-79.5896	New York	Chautauqua	61	61	60	55
360150003	42.11096	-76.8022	New York	Chemung	57	57	56	53
360270007	41.78555	-73.7414	New York	Dutchess	58	57	55	48
360290002	42.99328	-78.7715	New York	Erie	61	61	60	56
360310002	44.36608	-73.9031	New York	Essex	57	57	56	54
360310003	44.39308	-73.8589	New York	Essex	57	57	56	53
360410005	43.44957	-74.5163	New York	Hamilton	57	56	55	52
360430005	43.68578	-74.9854	New York	Herkimer	56	55	54	52
360450002	44.08747	-75.9732	New York	Jefferson	62	62	61	60
360530006	42.73046	-75.7844	New York	Madison	55	54	53	50
360551007	43.14618	-77.5482	New York	Monroe	60	59	59	57
360610135	40.81976	-73.9483	New York	New York	65	64	63	58
360631006	43.22386	-78.4789	New York	Niagara	64	64	64	61
360650004	43.30268	-75.7198	New York	Oneida	53	52	52	49
360671015	43.05235	-76.0592	New York	Onondaga	59	59	58	56
360715001	41.52375	-74.2153	New York	Orange	56	55	53	46
360750003	43.28428	-76.4632	New York	Oswego	58	58	58	56

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
360790005	41.45589	-73.7098	New York	Putnam	57	56	54	47
360810124	40.73614	-73.8215	New York	Queens	72	71	70	65
360830004	42.78189	-73.4636	New York	Rensselaer	57	56	55	49
360850067	40.59664	-74.1253	New York	Richmond	73	72	70	63
360870005	41.18208	-74.0282	New York	Rockland	62	61	59	51
360910004	43.01209	-73.6489	New York	Saratoga	56	55	54	49
360930003	42.79901	-73.9389	New York	Schenectady	54	53	52	47
361010003	42.09142	-77.2098	New York	Steuben	57	56	55	52
361030002	40.74529	-73.4192	New York	Suffolk	74	73	70	62
361030004	40.96078	-72.7124	New York	Suffolk	67	66	63	54
361030009	40.82799	-73.0575	New York	Suffolk	71	70	68	59
361099991	42.4006	-76.6538	New York	Tompkins	58	58	57	54
361111005	42.14403	-74.4943	New York	Ulster	58	58	56	51
361173001	43.23086	-77.1714	New York	Wayne	56	56	56	54
361192004	41.05192	-73.7637	New York	Westchester	65	64	62	54
370030004	35.929	-81.1898	North Carolina	Alexander	52	51	50	49
370110002	35.97222	-81.9331	North Carolina	Avery	50	49	49	46
370119991	36.1058	-82.0454	North Carolina	Avery	50	49	49	44
370210030	35.5001	-82.5999	North Carolina	Buncombe	52	51	51	49
370270003	35.93583	-81.5303	North Carolina	Caldwell	51	50	50	49
370319991	34.8848	-76.6203	North Carolina	Carteret	51	50	50	48
370330001	36.30703	-79.4674	North Carolina	Caswell	55	54	54	53
370370004	35.75722	-79.1597	North Carolina	Chatham	49	48	47	46
370510008	35.15869	-78.728	North Carolina	Cumberland	54	53	53	51
370511003	34.96889	-78.9625	North Carolina	Cumberland	55	54	54	52
370590003	35.89707	-80.5573	North Carolina	Davie	54	53	53	51
370630015	36.03294	-78.9054	North Carolina	Durham	52	51	51	50
370650099	35.98833	-77.5828	North Carolina	Edgecombe	56	54	54	52
370670022	36.11056	-80.2267	North Carolina	Forsyth	58	57	57	55
370670028	36.20306	-80.2158	North Carolina	Forsyth	54	53	53	51
370670030	36.026	-80.342	North Carolina	Forsyth	56	55	55	53
370671008	36.05083	-80.1439	North Carolina	Forsyth	55	55	55	53
370690001	36.09619	-78.4637	North Carolina	Franklin	52	52	51	50
370750001	35.25793	-83.7956	North Carolina	Graham	54	54	54	51
370770001	36.14111	-78.7681	North Carolina	Granville	56	54	54	53
370810013	36.10071	-79.8105	North Carolina	Guilford	57	56	56	55
370870008	35.50716	-82.9634	North Carolina	Haywood	50	49	49	47
370870035	35.37917	-82.7925	North Carolina	Haywood	55	54	54	52
370870036	35.59	-83.0775	North Carolina	Haywood	55	54	54	52
370990005	35.52444	-83.2361	North Carolina	Jackson	55	54	54	51

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
371010002	35.59083	-78.4619	North Carolina	Johnston	55	54	54	52
371070004	35.23146	-77.5688	North Carolina	Lenoir	54	52	52	50
371090004	35.43856	-81.2768	North Carolina	Lincoln	57	55	55	53
371139991	35.0608	-83.4306	North Carolina	Macon	50	49	49	48
371170001	35.81069	-76.8978	North Carolina	Martin	51	51	50	48
371190041	35.2401	-80.7857	North Carolina	Mecklenburg	65	64	64	63
371191005	35.11316	-80.9195	North Carolina	Mecklenburg	60	60	60	59
371191009	35.34722	-80.695	North Carolina	Mecklenburg	62	61	61	60
371239991	35.2632	-79.8365	North Carolina	Montgomery	50	49	49	48
371290002	34.36417	-77.8386	North Carolina	New Hanover	50	49	49	48
371450003	36.30697	-79.092	North Carolina	Person	63	56	56	54
371470006	35.63861	-77.3581	North Carolina	Pitt	55	54	54	52
371570099	36.30889	-79.8592	North Carolina	Rockingham	57	56	56	55
371590021	35.55187	-80.395	North Carolina	Rowan	57	56	55	54
371590022	35.53448	-80.6676	North Carolina	Rowan	57	57	56	55
371730002	35.43551	-83.4437	North Carolina	Swain	49	49	48	46
371790003	34.97389	-80.5408	North Carolina	Union	54	53	53	52
371830014	35.85611	-78.5742	North Carolina	Wake	54	53	53	52
371830016	35.59694	-78.7925	North Carolina	Wake	57	56	56	54
371990004	35.76541	-82.2649	North Carolina	Yancey	54	53	53	51
390030009	40.77094	-84.0539	Ohio	Allen	61	61	60	56
390071001	41.9597	-80.5728	Ohio	Ashtabula	62	62	61	55
390090004	39.30798	-82.1182	Ohio	Athens	58	58	57	52
390170004	39.38338	-84.5444	Ohio	Butler	67	66	66	59
390170018	39.52948	-84.3934	Ohio	Butler	67	67	66	59
390179991	39.5327	-84.7286	Ohio	Butler	65	64	64	58
390230001	40.00103	-83.8046	Ohio	Clark	61	61	60	55
390230003	39.85567	-83.9977	Ohio	Clark	61	60	60	54
390250022	39.0828	-84.1441	Ohio	Clermont	65	65	64	57
390271002	39.43004	-83.7885	Ohio	Clinton	63	63	62	56
390350034	41.55523	-81.5753	Ohio	Cuyahoga	58	57	57	56
390350060	41.49212	-81.6784	Ohio	Cuyahoga	52	52	52	52
390350064	41.36189	-81.8646	Ohio	Cuyahoga	56	56	56	55
390355002	41.53734	-81.4588	Ohio	Cuyahoga	57	57	57	56
390410002	40.35669	-83.064	Ohio	Delaware	60	59	59	54
390479991	39.6359	-83.2605	Ohio	Fayette	58	57	57	52
390490029	40.0845	-82.8155	Ohio	Franklin	67	66	65	59
390490037	39.96523	-82.9555	Ohio	Franklin	61	61	60	54
390490081	40.0877	-82.9598	Ohio	Franklin	58	58	57	52
390550004	41.51505	-81.2499	Ohio	Geauga	60	60	59	54

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
390570006	39.66575	-83.9429	Ohio	Greene	59	59	58	52
390610006	39.2787	-84.3661	Ohio	Hamilton	70	70	69	62
390610010	39.21494	-84.6909	Ohio	Hamilton	66	65	65	58
390610040	39.12886	-84.504	Ohio	Hamilton	68	67	67	60
390810017	40.36644	-80.6156	Ohio	Jefferson	61	60	59	54
390830002	40.31003	-82.6917	Ohio	Knox	60	59	59	53
390850003	41.67301	-81.4225	Ohio	Lake	58	58	58	57
390850007	41.72681	-81.2422	Ohio	Lake	53	53	53	52
390870011	38.62901	-82.4589	Ohio	Lawrence	54	54	53	47
390870012	38.50811	-82.6593	Ohio	Lawrence	59	59	58	51
390890005	40.02604	-82.433	Ohio	Licking	59	58	58	52
390930018	41.42088	-82.0957	Ohio	Lorain	54	54	54	54
390950024	41.64407	-83.5463	Ohio	Lucas	55	55	55	53
390950027	41.49417	-83.7189	Ohio	Lucas	59	58	58	54
390950034	41.67521	-83.3069	Ohio	Lucas	61	61	60	58
390970007	39.78819	-83.4761	Ohio	Madison	60	59	59	53
390990013	41.09614	-80.6589	Ohio	Mahoning	58	58	57	51
391030004	41.0604	-81.9239	Ohio	Medina	57	57	57	52
391090005	40.08455	-84.1141	Ohio	Miami	59	59	58	53
391130037	39.78563	-84.1344	Ohio	Montgomery	63	62	61	55
391219991	39.9428	-81.3373	Ohio	Noble	52	52	51	47
391331001	41.18247	-81.3305	Ohio	Portage	56	56	55	50
391351001	39.83562	-84.7205	Ohio	Preble	59	59	58	54
391510016	40.82805	-81.3783	Ohio	Stark	62	61	61	55
391510022	40.71278	-81.5983	Ohio	Stark	58	57	57	52
391514005	40.9314	-81.1235	Ohio	Stark	59	58	58	53
391530020	41.10649	-81.5035	Ohio	Summit	60	59	59	53
391550009	41.45424	-80.591	Ohio	Trumbull	57	56	56	51
391550011	41.24046	-80.6626	Ohio	Trumbull	62	62	61	55
391650007	39.42689	-84.2008	Ohio	Warren	64	64	63	57
391670004	39.43212	-81.4604	Ohio	Washington	60	60	59	53
391730003	41.37769	-83.6111	Ohio	Wood	61	60	60	56
400019009	35.75074	-94.6697	Oklahoma	Adair	64	61	60	58
400159008	35.11194	-98.2528	Oklahoma	Caddo	63	61	60	59
400170101	35.47922	-97.7515	Oklahoma	Canadian	62	61	61	59
400219002	35.85408	-94.986	Oklahoma	Cherokee	65	61	60	59
400270049	35.32011	-97.4841	Oklahoma	Cleveland	63	62	61	59
400310651	34.63298	-98.4288	Oklahoma	Comanche	64	64	63	60
400370144	36.10548	-96.3612	Oklahoma	Creek	62	59	59	58
400430860	36.15841	-98.932	Oklahoma	Dewey	65	65	64	63

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
400719010	36.95622	-97.0314	Oklahoma	Kay	63	62	61	60
400871073	35.15965	-97.4738	Oklahoma	McClain	62	61	60	59
400892001	34.477	-94.656	Oklahoma	McCurtain	61	59	58	56
400979014	36.22841	-95.2499	Oklahoma	Mayes	67	63	62	60
401090033	35.47704	-97.4943	Oklahoma	Oklahoma	65	64	64	62
401090096	35.4778	-97.303	Oklahoma	Oklahoma	63	63	62	61
401091037	35.61413	-97.4751	Oklahoma	Oklahoma	66	65	64	63
401159004	36.92222	-94.8389	Oklahoma	Ottawa	63	61	60	58
401210415	34.90227	-95.7844	Oklahoma	Pittsburg	66	64	63	60
401359021	35.40814	-94.5244	Oklahoma	Sequoyah	62	60	59	57
401430137	36.35744	-95.9992	Oklahoma	Tulsa	65	63	62	61
401430174	35.95371	-96.005	Oklahoma	Tulsa	64	60	60	59
401430178	36.1338	-95.7645	Oklahoma	Tulsa	65	62	62	60
401431127	36.2049	-95.9765	Oklahoma	Tulsa	66	63	62	61
410050004	45.25928	-122.588	Oregon	Clackamas	54	54	54	54
410090004	45.76853	-122.772	Oregon	Columbia	45	45	45	45
410390060	44.02631	-123.084	Oregon	Lane	48	48	48	48
410391007	43.8345	-123.035	Oregon	Lane	49	49	49	49
410470004	44.81029	-122.915	Oregon	Marion	49	49	49	49
410510080	45.49664	-122.603	Oregon	Multnomah	51	51	51	51
410671004	45.40245	-122.854	Oregon	Washington	50	50	50	50
420010002	39.93	-77.25	Pennsylvania	Adams	58	56	55	49
420019991	39.9231	-77.3078	Pennsylvania	Adams	59	58	56	50
420030008	40.46542	-79.9608	Pennsylvania	Allegheny	67	67	65	59
420030010	40.44558	-80.0162	Pennsylvania	Allegheny	65	64	63	57
420030067	40.37564	-80.1699	Pennsylvania	Allegheny	65	64	63	57
420031005	40.61395	-79.7294	Pennsylvania	Allegheny	71	71	69	62
420050001	40.81418	-79.5648	Pennsylvania	Armstrong	65	64	63	56
420070002	40.56252	-80.5039	Pennsylvania	Beaver	62	62	61	58
420070005	40.68472	-80.3597	Pennsylvania	Beaver	66	66	65	59
420070014	40.7478	-80.3164	Pennsylvania	Beaver	65	64	63	58
420110006	40.51408	-75.7897	Pennsylvania	Berks	59	57	55	48
420110011	40.38335	-75.9686	Pennsylvania	Berks	63	61	59	52
420130801	40.53528	-78.3708	Pennsylvania	Blair	66	64	62	55
420170012	40.10722	-74.8822	Pennsylvania	Bucks	66	65	62	54
420210011	40.30972	-78.915	Pennsylvania	Cambria	61	60	58	52
420270100	40.81139	-77.877	Pennsylvania	Centre	63	62	61	54
420279991	40.7208	-77.9319	Pennsylvania	Centre	65	64	62	55
420290100	39.83446	-75.7682	Pennsylvania	Chester	62	59	57	49
420334000	41.1175	-78.5262	Pennsylvania	Clearfield	63	63	61	54

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
420430401	40.24699	-76.847	Pennsylvania	Dauphin	59	57	55	49
420431100	40.27222	-76.6814	Pennsylvania	Dauphin	63	60	58	50
420450002	39.83556	-75.3725	Pennsylvania	Delaware	61	60	58	50
420479991	41.598	-78.7674	Pennsylvania	Elk	55	55	54	48
420490003	42.14175	-80.0386	Pennsylvania	Erie	60	60	59	54
420550001	39.96111	-77.4756	Pennsylvania	Franklin	56	55	54	48
420590002	39.80933	-80.2657	Pennsylvania	Greene	58	57	56	50
420630004	40.56333	-78.92	Pennsylvania	Indiana	66	65	63	57
420690101	41.47912	-75.5782	Pennsylvania	Lackawanna	61	60	58	52
420692006	41.44278	-75.6231	Pennsylvania	Lackawanna	59	58	56	50
420710007	40.04667	-76.2833	Pennsylvania	Lancaster	66	61	58	51
420710012	40.04383	-76.1124	Pennsylvania	Lancaster	65	61	59	51
420730015	40.99585	-80.3464	Pennsylvania	Lawrence	61	60	59	53
420750100	40.33733	-76.3834	Pennsylvania	Lebanon	63	61	59	52
420770004	40.61194	-75.4325	Pennsylvania	Lehigh	62	61	59	52
420791100	41.20917	-76.0033	Pennsylvania	Luzerne	55	54	52	46
420791101	41.26556	-75.8464	Pennsylvania	Luzerne	55	54	52	46
420810100	41.2508	-76.9238	Pennsylvania	Lycoming	58	56	55	50
420850100	41.21501	-80.4848	Pennsylvania	Mercer	62	61	60	54
420859991	41.4271	-80.1451	Pennsylvania	Mercer	55	54	54	49
420890002	41.08306	-75.3233	Pennsylvania	Monroe	54	53	51	45
420910013	40.11222	-75.3092	Pennsylvania	Montgomery	63	61	59	51
420950025	40.62806	-75.3411	Pennsylvania	Northampton	61	59	57	50
420958000	40.69222	-75.2372	Pennsylvania	Northampton	56	55	53	47
420990301	40.45694	-77.1656	Pennsylvania	Perry	59	58	57	51
421010004	40.00889	-75.0978	Pennsylvania	Philadelphia	55	54	52	45
421010024	40.0764	-75.0115	Pennsylvania	Philadelphia	69	68	65	56
421011002	40.03599	-75.0024	Pennsylvania	Philadelphia	66	65	63	54
421119991	39.9878	-79.2515	Pennsylvania	Somerset	54	53	52	46
421174000	41.64472	-76.9392	Pennsylvania	Tioga	59	58	57	52
421250005	40.14667	-79.9022	Pennsylvania	Washington	60	60	58	52
421250200	40.17056	-80.2614	Pennsylvania	Washington	60	59	58	52
421255001	40.44528	-80.4208	Pennsylvania	Washington	62	61	60	55
421290006	40.42808	-79.6928	Pennsylvania	Westmoreland	62	62	60	54
421290008	40.30469	-79.5057	Pennsylvania	Westmoreland	60	60	58	51
421330008	39.96528	-76.6994	Pennsylvania	York	62	57	55	48
421330011	39.86097	-76.4621	Pennsylvania	York	62	58	56	48
440030002	41.61524	-71.72	Rhode Island	Kent	60	59	57	50
440071010	41.84157	-71.3608	Rhode Island	Providence	59	59	57	50
440090007	41.49511	-71.4237	Rhode Island	Washington	63	63	60	53

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
450010001	34.32532	-82.3864	South Carolina	Abbeville	47	46	46	44
450030003	33.34223	-81.7887	South Carolina	Aiken	49	48	48	47
450070005	34.62324	-82.5321	South Carolina	Anderson	53	52	52	51
450150002	32.98725	-79.9367	South Carolina	Berkeley	49	49	49	48
450190046	32.94102	-79.6572	South Carolina	Charleston	51	50	50	49
450250001	34.61537	-80.1988	South Carolina	Chesterfield	50	50	49	48
450290002	33.00787	-80.965	South Carolina	Colleton	48	47	47	45
450310003	34.2857	-79.7449	South Carolina	Darlington	53	52	52	50
450370001	33.73996	-81.8536	South Carolina	Edgefield	46	45	45	44
450450016	34.75185	-82.2567	South Carolina	Greenville	52	51	51	50
450451003	35.0574	-82.3729	South Carolina	Greenville	50	49	49	48
450770002	34.65361	-82.8387	South Carolina	Pickens	54	53	53	51
450770003	34.85154	-82.7446	South Carolina	Pickens	50	50	49	48
450790007	34.09396	-80.9623	South Carolina	Richland	51	50	50	49
450790021	33.81468	-80.7811	South Carolina	Richland	46	45	44	43
450791001	34.13126	-80.8683	South Carolina	Richland	54	53	53	52
450830009	34.98871	-82.0758	South Carolina	Spartanburg	56	55	55	54
450910006	34.93582	-81.2284	South Carolina	York	50	49	49	48
460330132	43.5578	-103.484	South Dakota	Custer	58	58	57	57
460710001	43.74561	-101.941	South Dakota	Jackson	52	52	52	51
460930001	44.15564	-103.316	South Dakota	Meade	53	53	52	52
460990008	43.54792	-96.7008	South Dakota	Minnehaha	56	56	55	55
461270003	42.88021	-96.7853	South Dakota	Union	54	53	53	52
470010101	35.96522	-84.2232	Tennessee	Anderson	55	55	54	48
470090101	35.63149	-83.9435	Tennessee	Blount	59	59	58	52
470090102	35.60306	-83.7836	Tennessee	Blount	51	50	50	45
470259991	36.47	-83.8268	Tennessee	Claiborne	48	47	47	43
470370011	36.205	-86.7447	Tennessee	Davidson	52	52	51	46
470370026	36.15074	-86.6233	Tennessee	Davidson	56	55	55	49
470419991	36.0388	-85.7331	Tennessee	DeKalb	54	54	53	49
470651011	35.23348	-85.1816	Tennessee	Hamilton	55	55	54	50
470654003	35.10264	-85.1622	Tennessee	Hamilton	56	55	54	50
470890002	36.10563	-83.6021	Tennessee	Jefferson	58	57	56	51
470930021	36.08551	-83.7648	Tennessee	Knox	53	53	52	47
470931020	36.01919	-83.8738	Tennessee	Knox	55	54	54	48
471050109	35.72089	-84.3422	Tennessee	Loudon	57	56	55	49
471210104	35.28938	-84.9461	Tennessee	Meigs	55	54	54	50
471490101	35.73288	-86.5989	Tennessee	Rutherford	53	53	52	47
471550101	35.69667	-83.6097	Tennessee	Sevier	57	57	56	52
471550102	35.56278	-83.4981	Tennessee	Sevier	57	56	56	52

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
471570021	35.2175	-90.0197	Tennessee	Shelby	60	59	58	52
471570075	35.1517	-89.8502	Tennessee	Shelby	61	60	59	53
471571004	35.37815	-89.8345	Tennessee	Shelby	58	57	56	51
471632002	36.54144	-82.4248	Tennessee	Sullivan	60	60	59	53
471632003	36.58211	-82.4857	Tennessee	Sullivan	59	59	59	53
471650007	36.29756	-86.6531	Tennessee	Sumner	60	60	59	54
471650101	36.45398	-86.5641	Tennessee	Sumner	57	56	56	51
471870106	35.95153	-87.137	Tennessee	Williamson	55	55	54	48
471890103	36.06083	-86.2863	Tennessee	Wilson	57	57	56	52
480271047	31.088	-97.6797	Texas	Bell	63	62	60	56
480290032	29.51509	-98.6202	Texas	Bexar	67	66	63	59
480290052	29.63206	-98.5649	Texas	Bexar	68	67	64	60
480290059	29.27538	-98.3117	Texas	Bexar	60	59	57	53
480391004	29.52044	-95.3925	Texas	Brazoria	76	75	70	64
480391016	29.04376	-95.4729	Texas	Brazoria	64	63	61	57
480610006	25.8925	-97.4938	Texas	Cameron	57	57	56	54
480850005	33.13242	-96.7864	Texas	Collin	69	68	64	58
481130069	32.81995	-96.8601	Texas	Dallas	69	68	64	58
481130075	32.91921	-96.8085	Texas	Dallas	70	69	65	59
481130087	32.67645	-96.8721	Texas	Dallas	69	68	64	58
481210034	33.21906	-97.1963	Texas	Denton	71	70	66	60
481211032	33.41064	-96.9446	Texas	Denton	70	69	65	59
481390016	32.48208	-97.0269	Texas	Ellis	66	65	62	57
481391044	32.17543	-96.8702	Texas	Ellis	61	60	57	53
481410029	31.78577	-106.324	Texas	El Paso	54	54	54	54
481410037	31.76829	-106.501	Texas	El Paso	62	62	62	61
481410044	31.7657	-106.455	Texas	El Paso	60	60	60	60
481410055	31.74674	-106.403	Texas	El Paso	58	58	58	58
481410057	31.6675	-106.288	Texas	El Paso	57	57	57	57
481410058	31.89391	-106.426	Texas	El Paso	61	60	60	60
481671034	29.25447	-94.8613	Texas	Galveston	70	69	67	64
481830001	32.37868	-94.7118	Texas	Gregg	70	66	62	55
482010024	29.90104	-95.3261	Texas	Harris	71	70	66	60
482010026	29.80271	-95.1255	Texas	Harris	70	69	66	61
482010029	30.03953	-95.6739	Texas	Harris	69	68	64	59
482010046	29.82809	-95.2841	Texas	Harris	67	66	62	57
482010047	29.83472	-95.4892	Texas	Harris	67	66	62	55
482010051	29.62361	-95.4736	Texas	Harris	69	68	64	58
482010055	29.69574	-95.4993	Texas	Harris	70	69	65	58
482010062	29.62583	-95.2675	Texas	Harris	68	67	63	57

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
482010066	29.72472	-95.5036	Texas	Harris	66	66	62	55
482010070	29.73513	-95.3156	Texas	Harris	67	66	62	56
482010075	29.75278	-95.3503	Texas	Harris	68	67	63	57
482010416	29.68639	-95.2947	Texas	Harris	69	68	64	58
482011015	29.76165	-95.0814	Texas	Harris	67	66	63	59
482011034	29.76797	-95.2206	Texas	Harris	73	72	67	61
482011035	29.73373	-95.2576	Texas	Harris	70	69	65	59
482011039	29.67003	-95.1285	Texas	Harris	75	74	70	65
482011050	29.58305	-95.0155	Texas	Harris	72	71	68	64
482030002	32.66899	-94.1675	Texas	Harrison	63	61	59	55
482150043	26.22623	-98.2911	Texas	Hidalgo	56	55	55	53
482151048	26.13108	-97.9373	Texas	Hidalgo	55	54	53	52
482210001	32.44231	-97.8035	Texas	Hood	65	64	61	56
482311006	33.15308	-96.1156	Texas	Hunt	61	60	57	52
482450009	30.03644	-94.0711	Texas	Jefferson	64	63	60	56
482450011	29.8975	-93.9911	Texas	Jefferson	65	64	62	58
482450022	29.86395	-94.3178	Texas	Jefferson	62	61	58	54
482450101	29.728	-93.894	Texas	Jefferson	69	69	67	64
482450102	29.9425	-94.0006	Texas	Jefferson	62	61	58	55
482450628	29.865	-93.955	Texas	Jefferson	63	62	60	57
482451035	29.97892	-94.0109	Texas	Jefferson	63	62	60	56
482510003	32.35359	-97.4367	Texas	Johnson	68	67	64	59
482570005	32.56495	-96.3177	Texas	Kaufman	62	60	57	53
483091037	31.65307	-97.0707	Texas	McLennan	64	63	60	56
483390078	30.3503	-95.4251	Texas	Montgomery	66	66	62	57
483491051	32.03194	-96.3991	Texas	Navarro	63	61	58	54
483550025	27.76534	-97.4342	Texas	Nueces	64	63	62	59
483550026	27.83241	-97.5554	Texas	Nueces	64	63	61	59
483611001	30.08526	-93.7613	Texas	Orange	64	63	61	57
483611100	30.19417	-93.8669	Texas	Orange	61	59	57	53
483670081	32.86878	-97.9059	Texas	Parker	68	67	64	59
483739991	30.7017	-94.6742	Texas	Polk	60	60	58	55
483970001	32.93652	-96.4592	Texas	Rockwall	66	65	62	56
484230007	32.34401	-95.4158	Texas	Smith	65	62	60	55
484390075	32.98789	-97.4772	Texas	Tarrant	70	69	66	60
484391002	32.80582	-97.3566	Texas	Tarrant	69	68	65	59
484392003	32.9225	-97.2821	Texas	Tarrant	74	73	69	62
484393009	32.98426	-97.0637	Texas	Tarrant	73	72	68	61
484393011	32.65637	-97.0886	Texas	Tarrant	69	68	65	59
484530014	30.35442	-97.7603	Texas	Travis	64	63	60	56

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
484530020	30.48317	-97.8723	Texas	Travis	61	60	58	54
484690003	28.83617	-97.0055	Texas	Victoria	62	60	58	54
484790016	27.51127	-99.5203	Texas	Webb	59	59	58	56
490030003	41.49271	-112.019	Utah	Box Elder	61	60	60	60
490037001	41.94595	-112.233	Utah	Box Elder	61	60	60	60
490050004	41.73111	-111.838	Utah	Cache	59	59	59	58
490071003	39.60996	-110.801	Utah	Carbon	65	62	62	62
490110004	40.90297	-111.884	Utah	Davis	62	61	61	61
490131001	40.20865	-110.841	Utah	Duchesne	63	63	63	62
490352004	40.73639	-112.21	Utah	Salt Lake	66	65	65	65
490353006	40.73639	-111.872	Utah	Salt Lake	66	65	65	64
490370101	38.45861	-109.821	Utah	San Juan	64	64	64	64
490450003	40.54331	-112.3	Utah	Tooele	65	65	65	64
490490002	40.25361	-111.663	Utah	Utah	63	63	63	62
490495008	40.43028	-111.804	Utah	Utah	60	59	59	59
490495010	40.13634	-111.661	Utah	Utah	63	63	63	62
490530006	37.129	-113.637	Utah	Washington	62	62	62	62
490570002	41.20632	-111.976	Utah	Weber	65	64	64	64
490571003	41.30361	-111.988	Utah	Weber	65	65	65	64
500030004	42.88759	-73.2498	Vermont	Bennington	53	53	52	47
500070007	44.52839	-72.8688	Vermont	Chittenden	53	52	52	50
510030001	38.07657	-78.504	Virginia	Albemarle	54	54	53	49
510130020	38.8577	-77.0592	Virginia	Arlington	63	63	60	51
510330001	38.20087	-77.3774	Virginia	Caroline	56	55	53	47
510360002	37.34438	-77.2593	Virginia	Charles	61	59	58	56
510410004	37.35748	-77.5936	Virginia	Chesterfield	58	56	55	51
510590030	38.77335	-77.1047	Virginia	Fairfax	63	62	59	50
510610002	38.47367	-77.7677	Virginia	Fauquier	50	49	48	43
510690010	39.28102	-78.0816	Virginia	Frederick	55	54	53	48
510719991	37.3297	-80.5578	Virginia	Giles	48	48	47	44
510850003	37.60613	-77.2188	Virginia	Hanover	59	57	56	54
510870014	37.55652	-77.4003	Virginia	Henrico	61	58	58	55
511071005	39.02473	-77.4893	Virginia	Loudoun	60	59	57	49
511130003	38.52199	-78.4358	Virginia	Madison	60	59	58	54
511390004	38.66373	-78.5044	Virginia	Page	56	55	55	50
511479991	37.1655	-78.3069	Virginia	Prince Edward	54	50	50	48
511530009	38.85287	-77.6346	Virginia	Prince William	58	58	56	49
511611004	37.28342	-79.8845	Virginia	Roanoke	54	53	53	51
511630003	37.62668	-79.5126	Virginia	Rockbridge	52	51	51	48
511650003	38.47753	-78.8195	Virginia	Rockingham	55	55	54	50

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
511790001	38.48123	-77.3704	Virginia	Stafford	54	53	51	43
511970002	36.89117	-81.2542	Virginia	Wythe	54	53	53	49
515100009	38.8104	-77.0444	Virginia	Alexandria City	62	61	59	50
516500008	37.10373	-76.387	Virginia	Hampton City	60	59	59	57
518000004	36.90118	-76.4381	Virginia	Suffolk City	60	59	59	57
518000005	36.66525	-76.7308	Virginia	Suffolk City	56	55	54	52
530110011	45.61667	-122.517	Washington	Clark	50	50	50	50
530330010	47.5525	-122.065	Washington	King	50	50	50	50
530330017	47.49022	-121.773	Washington	King	49	49	49	49
530330023	47.1411	-121.938	Washington	King	55	55	55	55
530630001	47.41645	-117.53	Washington	Spokane	51	51	51	51
530630021	47.67248	-117.365	Washington	Spokane	51	51	51	51
530630046	47.82728	-117.274	Washington	Spokane	50	50	50	50
530670005	46.95256	-122.595	Washington	Thurston	48	47	47	47
540030003	39.44801	-77.9641	West Virginia	Berkeley	56	56	54	49
540110006	38.42413	-82.4259	West Virginia	Cabell	58	57	57	50
540219991	38.8795	-80.8477	West Virginia	Gilmer	52	52	51	45
540250003	37.90853	-80.6326	West Virginia	Greenbrier	54	53	53	48
540291004	40.42154	-80.5807	West Virginia	Hancock	63	63	62	57
540390010	38.3456	-81.6283	West Virginia	Kanawha	64	64	63	55
540610003	39.64937	-79.9209	West Virginia	Monongalia	63	62	61	54
540690010	40.11488	-80.701	West Virginia	Ohio	61	60	59	53
540939991	39.0905	-79.6617	West Virginia	Tucker	56	55	55	49
541071002	39.32353	-81.5524	West Virginia	Wood	57	57	56	50
550090026	44.53098	-87.908	Wisconsin	Brown	57	56	55	52
550210015	43.3156	-89.1089	Wisconsin	Columbia	57	56	55	52
550250041	43.10084	-89.3573	Wisconsin	Dane	56	56	55	52
550270001	43.46611	-88.6211	Wisconsin	Dodge	62	61	61	57
550290004	45.237	-86.993	Wisconsin	Door	64	63	63	58
550350014	44.761	-91.143	Wisconsin	Eau Claire	51	51	50	49
550390006	43.6874	-88.422	Wisconsin	Fond du Lac	61	60	60	56
550410007	45.563	-88.8088	Wisconsin	Forest	53	53	52	50
550550002	43.002	-88.8186	Wisconsin	Jefferson	58	58	57	54
550590019	42.50472	-87.8093	Wisconsin	Kenosha	59	58	59	59
550610002	44.44312	-87.5052	Wisconsin	Kewaunee	63	62	62	57
550630012	43.7775	-91.2269	Wisconsin	La Crosse	53	52	52	51
550710007	44.13862	-87.6161	Wisconsin	Manitowoc	66	66	65	60
550730012	44.70735	-89.7718	Wisconsin	Marathon	53	52	52	49
550790010	43.01667	-87.9333	Wisconsin	Milwaukee	56	56	55	53
550790026	43.06098	-87.9135	Wisconsin	Milwaukee	60	60	60	57

Site ID	Lat	Long	State	County	O3 DV for Scenario:			
					Base Case	Baseline	70	65
550790085	43.181	-87.9	Wisconsin	Milwaukee	64	64	63	59
550870009	44.30738	-88.3951	Wisconsin	Outagamie	60	59	59	56
550890008	43.343	-87.92	Wisconsin	Ozaukee	66	65	65	61
550890009	43.49806	-87.81	Wisconsin	Ozaukee	62	61	61	57
551010017	42.7139	-87.7986	Wisconsin	Racine	57	57	57	56
551050024	42.50908	-89.0628	Wisconsin	Rock	60	60	59	55
551110007	43.4351	-89.6797	Wisconsin	Sauk	55	53	53	50
551170006	43.679	-87.716	Wisconsin	Sheboygan	71	71	70	65
551199991	45.2066	-90.5969	Wisconsin	Taylor	53	53	52	51
551270005	42.58001	-88.499	Wisconsin	Walworth	60	60	59	56
551330027	43.02008	-88.2151	Wisconsin	Waukesha	58	57	57	53
560019991	41.3642	-106.24	Wyoming	Albany	65	65	65	64
560050123	44.6522	-105.29	Wyoming	Campbell	60	59	59	59
560050456	44.14696	-105.53	Wyoming	Campbell	59	59	59	59
560070100	41.38694	-107.617	Wyoming	Carbon	60	59	59	58
560130232	43.08167	-107.549	Wyoming	Fremont	60	59	59	59
560210100	41.18223	-104.778	Wyoming	Laramie	63	62	62	60
560350700	42.48636	-110.099	Wyoming	Sublette	60	60	60	60
560359991	42.9288	-109.788	Wyoming	Sublette	62	62	62	62
560370077	41.158	-108.619	Wyoming	Sweetwater	59	58	58	58
560370200	41.67745	-108.025	Wyoming	Sweetwater	57	56	56	56
560370300	41.75056	-109.788	Wyoming	Sweetwater	60	60	60	60
560410101	41.3731	-111.042	Wyoming	Uinta	58	58	58	58

2A.5 References

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CHAPTER 3 : CONTROL STRATEGIES AND EMISSIONS REDUCTIONS

Overview

To estimate the costs and benefits of alternative ozone standard levels, the EPA has analyzed hypothetical control strategies that areas across the country might employ to attain the revised ozone standard level of 70 ppb and a more stringent alternative standard of 65 ppb. The future year for analyzing the incremental costs and benefits of meeting a revised ozone standard is 2025.[44] This analysis year was chosen because most areas of the U.S. will be required to meet a revised ozone standard by 2025. California was analyzed independently from the rest of the U.S. because of the potential for longer compliance timelines in many areas. Consequently, we created two baseline scenarios, a 2025 baseline for all areas outside of California and a post-2025 baseline for California.

This chapter documents the (i) emissions control measures EPA applied to illustrate attainment with the revised ozone standard of 70 ppb and the alternative standard of 65 ppb and (ii) projected emissions reductions associated with the measures. The chapter is organized into five sections. Section 3.1 provides a summary of the steps that we took to determine necessary emissions reductions to create the 2025 baseline and the control strategies to reach the revised standard level of 70 ppb and an alternative standard of 65 ppb in the continental U.S. outside of California. Section 3.2 describes the steps we took to determine necessary emissions reductions to create the post 2025-baseline and the control strategy to reach 70 ppb and 65 ppb for California. In Section 3.3 we discuss key differences between the results from the analysis conducted for the proposal RIA and this analysis. In Section 3.4 we list the key limitations and uncertainties associated with the control strategy analysis. And finally, Section 3.5 includes the references for the chapter.

To conduct the control strategy analyses, we first require information on total emissions reductions needed to simulate attainment. For that purpose we need (i) projected future design value and design value (DV) targets for each area, (ii) the sensitivity of ozone DVs to the NOx

[44] Please see Chapter 1, Section 1.3.2 for a detailed discussion of the potential nonattainment designations and their timing.

and VOC emissions reductions, and (iii) available NOx and VOC reductions from identified controls (as will be described in section 3.1.1). Second, to find an illustrative control strategy to achieve the emissions reductions needed, we need information about available identified controls[45] for specific sources and associated emissions reductions. More details on air quality modeling and information about projected future DVs, DV targets, and ppb/ton ozone response factors are provided in Chapter 2. In this chapter we calculate the necessary emissions reductions and describe the creation of hypothetical control strategies for the post-2025 baseline and for the revised and alternative standard levels analyzed.

3.1 The 2025 Control Strategy Scenarios

To create the baseline, we projected 2025 ozone DVs for the base case scenario as described in Section 2.2 of Chapter 2. We adjusted the 2025 base case for all areas of the U.S. to account for emissions reductions from the Clean Power Plan in creating the 2025 baseline. In addition, because in the final 2025 base case projections no monitors outside of California were projected to violate the current standard of 75 ppb, no additional controls were applied to create the 2025 baseline.

3.1.1 Approach for the Revised Standard of 70 ppb and Alternative Standard of 65 ppb

The control strategies applied to illustrate attainment of the revised and alternative standards analyzed involved several steps. We applied regional and local ppb/ton ozone response factors to estimate resulting ozone DVs at air quality monitor locations to find the target emissions levels. Then we applied controls to reach those targets levels. These steps are described in this section.

As described in Chapter 2, we performed a series of photochemical modeling simulations to determine the response of ozone DVs at monitor locations to emissions reductions in specific

[45] In the proposal RIA we discuss emissions reductions resulting from the application of known controls, as well as emissions reductions beyond known controls, using the terminology of “known controls” and “unknown controls.” In the final RIA, we have used slightly different terminology, consistent with past NAAQS RIAs. Here we refer to emissions reductions and controls as either “identified” controls or measures or “unidentified” controls or measures reflecting that unidentified controls or measures can include existing controls or measures for which the EPA does not have sufficient data to accurately estimate their costs.

regions (NOx emissions reductions) and urban areas (VOC emissions reductions). We estimated the necessary emissions reductions sequentially, one region at a time. For each air quality sensitivity region (see Chapter 2, Figure 2-2 for a map of the sensitivity regions), we determined the amount of emissions reductions necessary for all monitors within the region to meet the standard level analyzed.

To implement this approach, we ranked the monitors in descending order by baseline DV, and the region that included the monitor with the highest projected baseline DV (East Texas) was analyzed first. We estimated the emissions reductions to decrease ozone concentrations to the level needed for that region. We then estimated the impact that those emissions reductions would have on all other remaining regions. After emissions reductions were estimated for each region, the remaining monitors were re-ordered based on the resulting DVs and the next region with the highest baseline DV was targeted for emissions reductions, if needed, and the impact of its reductions were estimated for the remaining regions. We repeated this process until all regions had been analyzed. For each region analyzed, we determined (i) the quantity of emissions reductions from available identified NOx & VOC controls, and (ii) the impact of these controls on ozone concentrations. If additional decreases in ozone concentrations were needed in the region being analyzed, additional emissions reductions would have to come from unidentified controls. Figure 3-1 shows a summary of this process. A numeric example of the calculation methodology is provided in Appendix 3A. In addition, ozone DVs at all evaluated monitors are provided for each scenario in Appendix 2A, Section 2A.4.

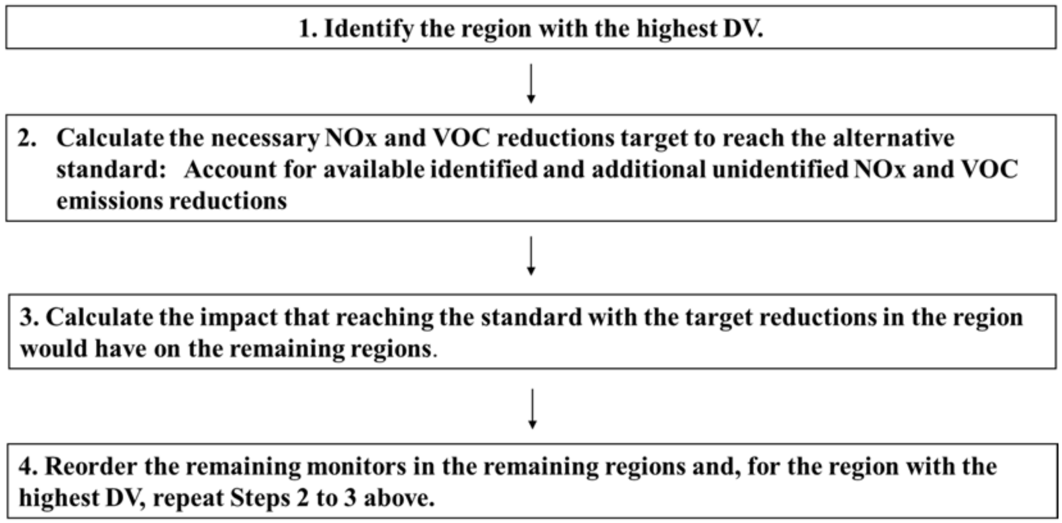

Figure 3-1. Process to Find Needed Reductions to Reach the Revised and Alternative Standards

Because emissions reductions in NOx and VOC have different resulting air quality impacts on ozone and because different combinations of reductions from these pollutants could potentially render the same reduction in ozone, it is important to know for each region, a-priori, the total potential available reductions of these two pollutants from identified controls. To find these potentially available tons of NOx and VOC emissions reductions, we ran a maximum emissions reductions run using CoST (Control Strategy Tool) (a description of CoST, its algorithms, and the control strategy applied to obtain the necessary reductions follows in section 3.1.2), and applied the reductions from these controls as part of the process described above to obtain the total needed emissions reductions. First, we estimated the available NOx and VOC emissions reductions from identified controls to determine the reductions in ozone concentrations. In this analysis, identified VOC controls are generally more expensive than identified NOx controls and are only effective at reducing ozone in a limited number of locations. For completeness, we applied the more expensive identified VOC controls in these locations before applying any unidentified controls. Then we estimated any additional NOx emissions reductions needed from unidentified controls to achieve the target reduction. We did not apply any unidentified VOC controls.[46] States will likely pursue the most effective controls

[46] Past air quality modeling experience has indicated that in most areas NOx emissions reductions are more effective at reducing ozone concentrations at the monitor with the highest DV.

for reducing ozone concentrations. In this analysis, overall NOx controls are more effective at reducing high ozone concentrations, so we applied unidentified NOx controls.

To define the geographic areas within which we would obtain NOX emissions reductions, we created a 200 kilometer buffer around each county with a DV projected to exceed the standard level being analyzed, but we limited the buffer to within the borders of the state containing the exceeding county. The area outside the buffer but within the air quality modeling sensitivity region was also identified. To define the geographic areas within which we would obtain VOC emissions reductions, we created a 100 kilometer buffer around the county with the projected monitor exceedance in areas where the modeling showed that ozone concentrations are responsive to VOC emissions reductions. Figures 3-2 and 3-3 are maps displaying the NOX and VOC buffers respectively. We used these buffers in estimating available emissions reductions to target the application of identified controls as close to the projected exceeding monitors as possible, within each region.

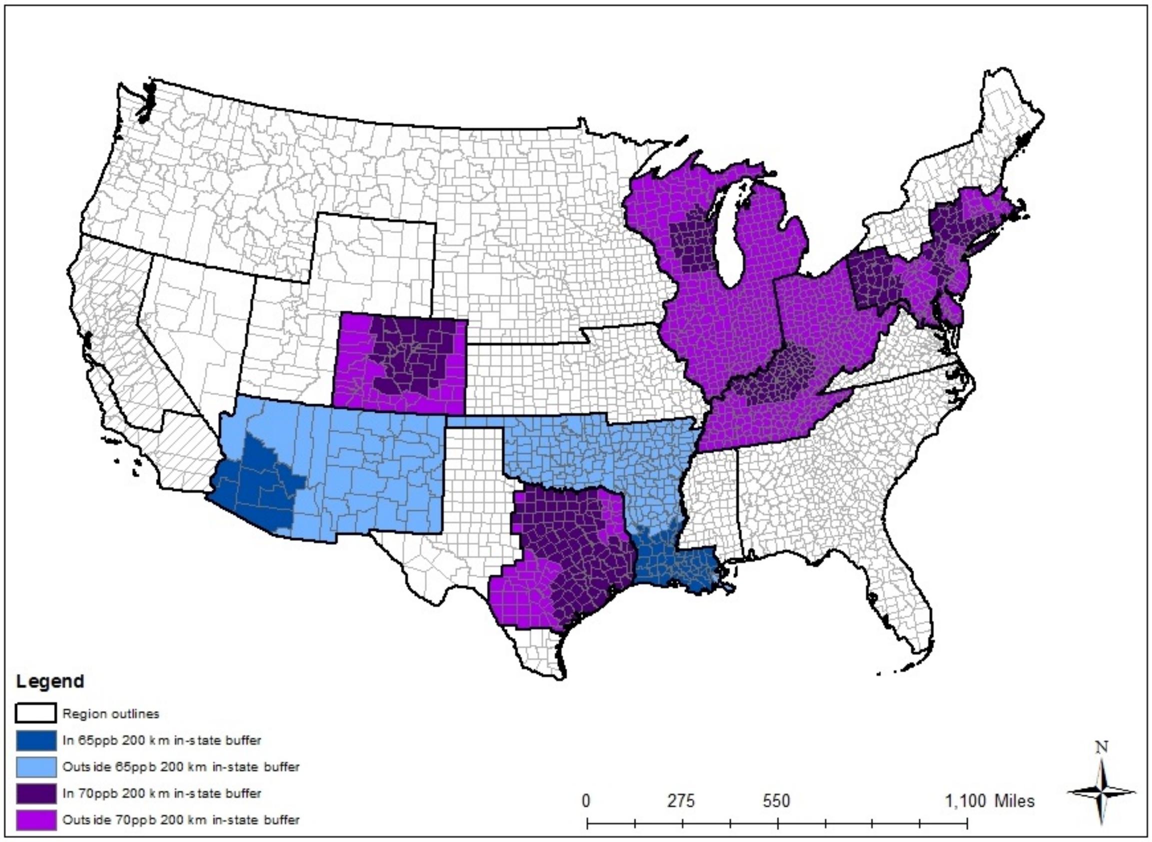

Figure 3-2. Buffers of 200 km for NOx Emissions Reductions around Projected Exceedance Areas

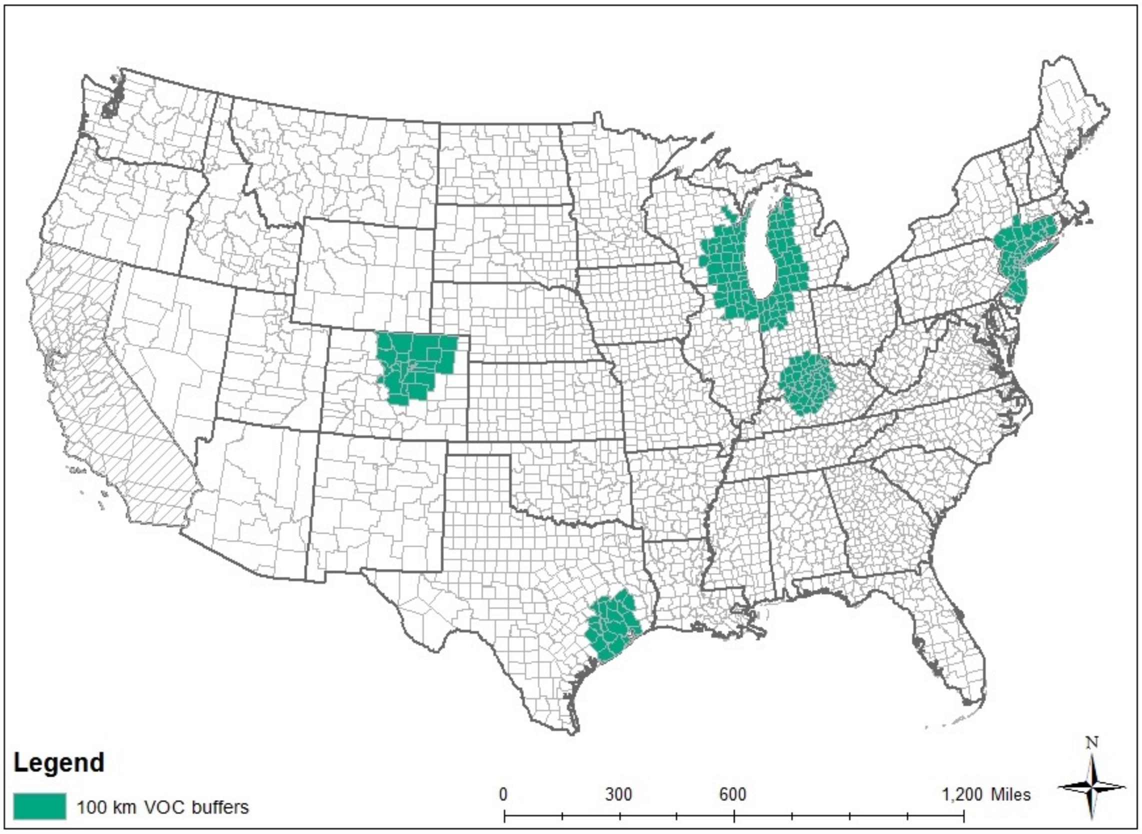

Figure 3-3. Buffers of 100 km for VOC Emissions Reductions around Projected Exceedance Areas

Once we completed the process of estimating the necessary emissions reductions to meet the revised standard of 70 ppb and alternative standard of 65 ppb for each region, we applied control strategies to simulate attainment with them. For each air quality sensitivity region containing a monitor projected to exceed either the revised or the alternative standard, we applied NO_X controls to simulate attainment with the respective standard. If these controls did not bring the area into attainment and VOC reductions were needed, then we applied a control strategy within the 100 km buffer to reach the VOC target reductions. If the quantity of emissions reductions needed were greater than the available emissions reductions from NO_X and VOC controls within the buffer, additional identified controls were applied within the remaining air quality sensitivity region outside the buffer. If further emission reductions were needed within the region then we assumed that unidentified controls would be used for that region to meet the standard analyzed. Figure 3-4 illustrates this process.

1. Determine if needed NOx reductions in the 200 km buffer are less than available reductions from identified controls. If yes: apply the least cost algorithm to get the control strategy. If no: If VOC reductions are also needed go to step 2. If VOC reductions are not needed then go to step 3.

↓

2. If VOC reductions are also needed then apply the maximum emissions reductions algorithm to get the control strategy (all available VOC reductions are applied when VOC reductions are needed).

↓

3. Determine if needed NOx reductions within the 200 km buffer are greater than available within-buffer reductions from identified controls. If yes: apply maximum emissions reductions inside the buffer and least cost outside the buffer within the region. If no: go to step 4.

↓

4. Determine if needed NOx emissions reductions in the region are greater than available reductions from identified controls within the region, then apply maximum emissions reductions in the region plus unidentified controls.

Figure 3-4. Process to Estimate the Control Strategies for the Revised and Alternative Standards

3.1.2 Identified Control Measures

Control measures applied to meet the revised and alternative standards were identified for four emissions sectors: Electric Generating Units (EGUs), Non-Electric Generating Unit Point Sources (Non-EGUs), Nonpoint (Area) Sources, and Nonroad Mobile Sources. Onroad mobile source controls were not applied because they are largely addressed in existing rules such as the Tier 3 rule (U.S. EPA, 2014a). Controls applied for the revised and alternative standard analyses are listed in Table 3-1.

The control measures we applied were identified using the EPA's Control Strategy Tool (CoST) (U.S. EPA, 2014b), the NONROAD Model (U.S. EPA, 2005) and the Integrated Planning Model (IPM) (U.S. EPA, 2015).[47] CoST models emissions reductions and engineering costs associated with control strategies applied to non-EGUs, area, and mobile sources of air pollutant emissions by matching control measures to emissions sources using algorithms such as

[47] For the final RIA, an updated version of IPM was used. As a result of the updated version of IPM, after accounting for emissions reductions from the proposed Clean Power Plan we applied fewer controls to EGU sources than we applied in the proposal RIA.

"maximum emissions reduction", "least cost", and "apply measures in series". For this control strategy analysis, we applied both the maximum emissions reduction (when all available reductions were needed) and least cost algorithms[48] (when not all available reductions were needed). These controls are described further in Appendix 3A.

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 non-EGU 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. VOC controls applied included surface coating, solvents, and fuel storage tanks.

To more accurately depict available controls, the EPA employed a decision rule in which controls were not applied to any non-EGU point or nonpoint sources with less than 25 tons/year of emissions per pollutant for NO_X and 10 tons/year for VOC. This decision rule is more inclusive of sources than the decision rule employed in the previous Ozone and $PM_{2.5}$ NAAQS RIAs where we applied a minimum of 50 tons/year for each pollutant. The reason for not applying controls to sources below these levels is that many of these sources likely already have controls in place that may not be reflected in the emissions inventory inputs, and we don't believe it is cost effective to apply an additional control device (see Chapter 4, Section 4.1.1 for a brief discussion of the emissions inventory inputs). Furthermore, controls were not applied if their cost per ton exceeded $19,000/ton for NOx or $33,000/ton for VOC (see Chapter 4, Section 4.1.1 for a discussion about these cutoff values). In addition, we only apply controls that replace existing controls if replacement controls are at least 10% more effective than the existing control. This is because we assume that replacement below that level would not be cost effective.

[48] A maximum emissions reductions run in CoST will yield the same result as a least cost control strategy run with 100% control.

Table 3-1. Identified Controls Applied for the Revised and Alternative Standard Analyses Strategies

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	
	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
	Biosolid Injection Technology	Reduced Solvent Utilization
	Episodic Burn Ban	Gas Recovery in Landfills
EGU	SCR and SNCR	
Nonroad	Diesel Retrofits & Engine Rebuilds	

3.1.3 Results

Figure 3-5 shows the counties projected to exceed the revised standard and alternative standard analyzed for the 2025 baseline for areas other than California. For the 70 ppb control strategy, NO_X emissions reductions were required for monitors in the following regions: Colorado, Great Lakes, North East, Ohio River Valley and East Texas (see Chapter 2, Figure 2-2 for a depiction of the sensitivity regions). VOC reductions were required in Houston (see Chapter 2, Figure 2-4 for a depiction of the VOC impact regions). For the 65 ppb alternative standard, in addition to the regions listed above, NOx reductions were also applied in the Arizona-New Mexico, Nevada, and Oklahoma-Arkansas-Louisiana regions. VOC reductions were required in Denver, Houston, Louisville, Chicago and New York City.[49] It is important to note that for both the 70 ppb revised standard as well as the 65 ppb alternative standard when VOC reductions were needed all available reductions from identified controls were applied using the maximum emissions reductions algorithm. In all of these areas, we used all of the available identified VOC controls, and for remaining reductions in ozone concentrations we applied unidentified NOx controls. Summaries of the emissions reductions are presented by region and source category in Appendix 3A.

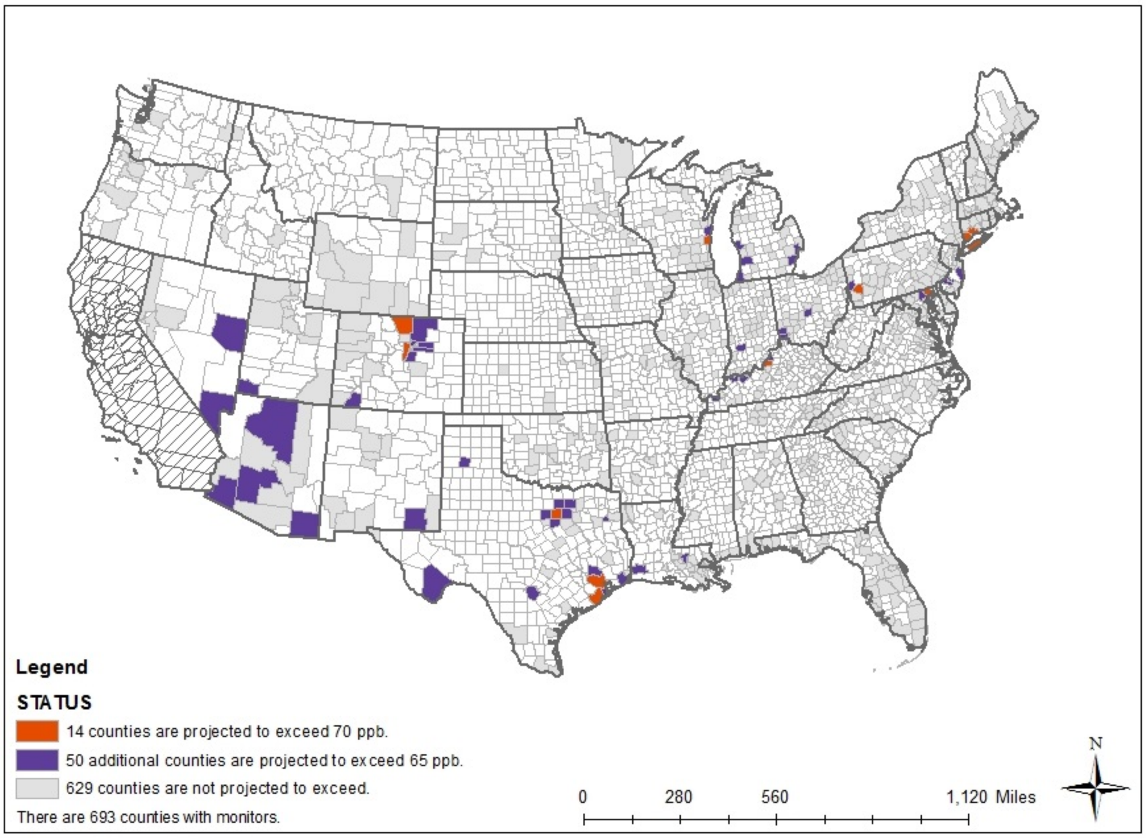

Figure 3-5. Projected Ozone Design Values in the 2025 Baseline Scenario

Table 3-2 shows the number of exceeding counties and the number of neighboring counties to which controls were applied for the revised and alternative standards analyzed. Figure 3-6 shows counties where NOx controls were applied for the revised and alternative

[49] These five urban areas were determined to have ozone that was sensitive to reductions of VOC emissions in some locations and were the areas with the highest ozone DVs in their respective regions. See Chapter 2, section 2.3 and Appendix 2A.

standards, and Figure 3-7 depicts counties where VOC controls were applied for the revised and alternative standards analyzed. For a complete list of geographic areas for the revised and alternative standards analyzed see Appendix 3A.

Table 3-2. Number of Counties with Exceedances and Number of Additional Counties Where Reductions Were Applied for the 2025 Revised and Alternative Standards Analyses - U.S., except California

Revised and Alternative Standards	Number of Counties with Exceedances	Number of Additional Counties Where Reductions Were Applied
70 ppb	14	663
65 ppb	50	1,170

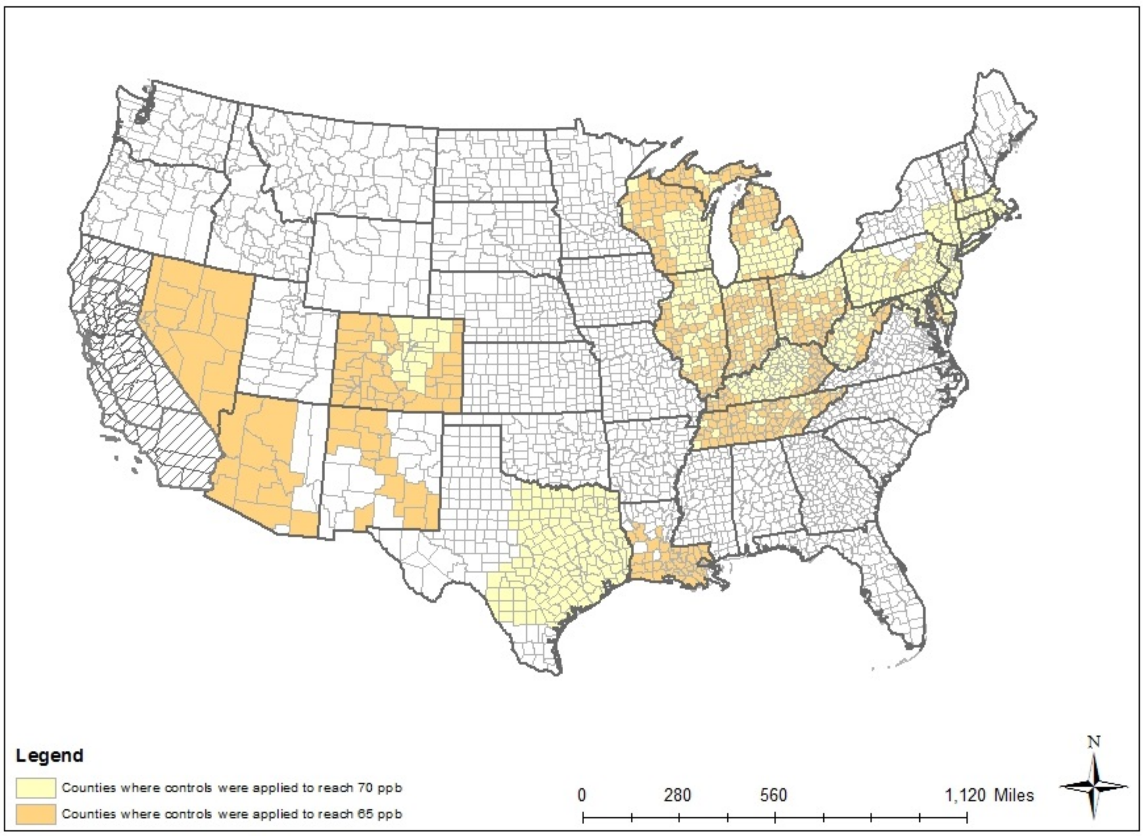

Figure 3-6. Counties Where NO_X Emissions Reductions Were Applied to Simulate Attainment with the Revised and Alternative Ozone Standards in the 2025 Analysis

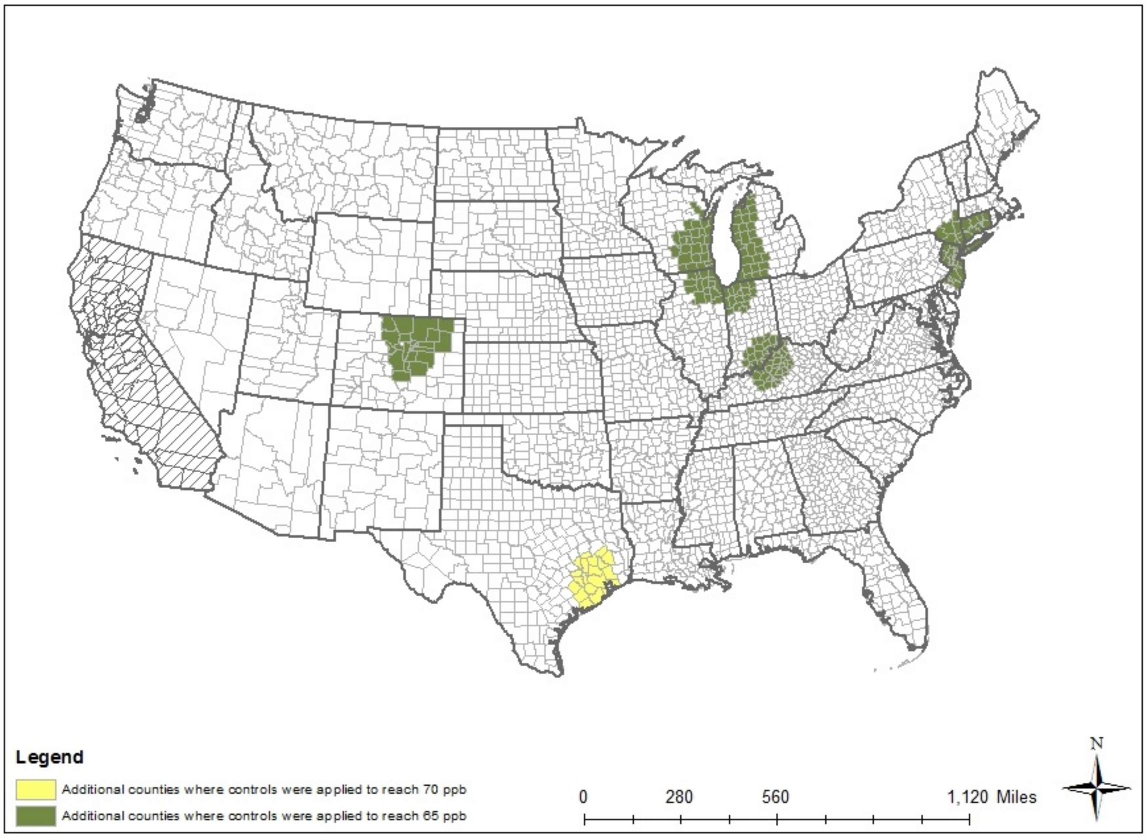

Figure 3-7. Counties Where VOC Emissions Reductions Were Applied to Simulate Attainment with the Revised and Alternative Ozone Standards in the 2025 Analyses

Table 3-3 shows the modeled 2011 and 2025 base case NOx and VOC emissions by sector (this table is also Table 2A-1 in Appendix 2A). Additional details on the emissions by state are given in the emissions modeling TSD. Tables 3-4 and 3-5 show the emissions reductions from identified controls for the revised and alternative standard analyzed. The largest emission reductions were in the non-EGU point source and nonpoint source sectors. For details regarding emissions reductions by control measure see Appendix 3.A

Table 3-3. 2011 and 2025 Base Case NOx and VOC Emissions by Sector (1000 tons)

Sector	2011 NO_x	2025 NO_x	2011 VOC	2025 VOC
EGU-point	2,000	1,400	36	42
NonEGU-point	1,200	1,200	800	830
Point oil and gas	500	460	160	190
Wild and Prescribed Fires	330	330	4,700	4,700
Nonpoint oil and gas	650	720	2,600	3,500
Residential wood combustion	34	35	440	410
Other nonpoint	760	790	3,700	3,500
Nonroad	1,600	800	2,000	1,200
Onroad	5,700	1,700	2,700	910
C3 Commercial marine vessel (CMV)	130	100	5	9
Locomotive and C1/C2 CMV	1,100	680	48	24
Biogenics	1,000	1,000	41,000	41,000
TOTAL	**15,000**	**9,300**	**58,000**	**56,000**

Table 3-4. Summary of Emissions Reductions by Sector for the Identified Control Strategies Applied for the Revised 70 ppb Ozone Standard in 2025, except California (1,000 tons/year)[a]

Geographic Area	Emissions Sector	NO_x	VOC
East	EGU	45	-
	Non-EGU Point	85	1
	Nonpoint	100	19
	Nonroad	3	-
	Onroad	-	-
	Total	**230**	**20**
West	EGU	-	-
	Non-EGU Point	6	-
	Nonpoint	1	-
	Nonroad	-	-
	Onroad	-	-
	Total	**7**	**-**

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

Table 3-5. Summary of Emissions Reductions by Sector for the Identified Control Strategies for the Alternative 65 ppb Ozone Standard in 2025 - except California (1,000 tons/year)[a]

Geographic Area	Emissions Sector	NO_x	VOC
East	EGU	110	-
	Non-EGU Point	220	5
	Nonpoint	160	100
	Nonroad	8	-
	Total	**500**	**100**
West	EGU	0	-
	Non-EGU Point	33	-
	Nonpoint	22	5
	Nonroad	1	-
	Total	**56**	**5**

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

As mentioned previously, there were several areas where identified controls did not achieve enough emissions reductions to meet the revised and alternative standards of 70 and 65 ppb. Texas East was the only area where identified controls were not enough to get the needed emissions reductions for 70 ppb. Great Lakes, Colorado, Texas East, Ohio River Valley, Northeast and Nevada were the areas where identified controls were not enough to get the needed emissions reductions for 65 ppb. See Chapter 2, Figure 2-2 for a map showing these areas. To complete the analysis, the EPA then assumed that the remaining reductions needed to meet the standard would be obtained from unidentified controls. Table 3-6 shows the emissions reductions needed from unidentified controls in 2025 for the U.S., except California, for the revised and alternative standards analyzed.

Table 3-6. Summary of Emissions Reductions for the Revised and Alternative Standards for the Unidentified Control Strategies for 2025 - except California (1,000 tons/year)[a]

Revised and Alternative Standards	Region	NO_x	VOC
70 ppb[b]	East	47	-
	West	-	-
65 ppb[c]	East	820	-
	West	40	-

[a] Estimates are rounded to two significant figures.
[b] Unidentified controls for the revised standard of 70 ppb are needed in the Texas East (see Chapter 2, Figure 2-2 for a description of these regions).
[c] Unidentified controls for the 65 ppb alternative standard are needed in Nevada, Colorado, Texas East, Great Lakes, Ohio River Valley and North East (see Chapter 2, Figure 2-2 for a description of these regions).

Table 3-7 summarizes the total (identified and unidentified) emissions reductions needed to meet the revised and alternative standard levels in 2025 for the East and West, except California (see Chapter 4, Figure 4-3 for a map depicting the East and West regions). In the East for 2025, the unidentified NOx emissions reductions needed as percentage of the total reductions increases from 17 percent to 62 percent as the standard level analyzed decreases from 70 ppb to 65 ppb. In the West, unidentified NOX emissions reductions are only needed for the 65 ppb alternative standard and account for 42 percent of the total reductions needed. No unidentified VOC reductions are needed in the East or West for the 70 ppb and 65 ppb standard levels.

Table 3-7. Summary of Emissions Reductions from the Identified + Unidentified Control Strategies by Alternative Standard Levels in 2025, Except California (1,000 tons/year)[a]

		Alternative Standard	
Geographic Area	**Emissions Reductions**	**70 ppb**	**65 ppb**
East	NOx Identified	230	500
	NOx Unidentified	50	820
	% NOx Unidentified	**17%**	**62%**
	VOC Identified	20	100
	VOC Unidentified	0	0
	% VOC Unidentified	**0%**	**0%**
West	NOx Identified	7	56
	NOx Unidentified	0	40
	% NOx Unidentified	**0%**	**42%**
	VOC Identified	0	5
	VOC Unidentified	0	0
	% VOC Unidentified	**0%**	**0%**

[a] Estimates are rounded to two significant figures.

3.2 The Post-2025 Scenario for California

The post-2025 baseline and alternative standard level scenarios for California were created using similar methods to those described above in Section 3.1. However, in contrast to the rest of the U.S., substantial emissions reductions were needed in California to meet the current standard of 75 ppb. All identified controls were used to meet the current standard in this process, so the revised and alternative standards analyzed in California relied entirely on unidentified measures.

3.2.1 Creation of the Post-2025 Baseline Scenario for California

The final 2025 base case projections predict several areas of California would have ozone DVs above the current standard level of 75 ppb. Therefore, we estimated emissions reductions in the following order to construct the post-2025 baseline scenario for California: (1) emissions changes from the Clean Power Plan, (2) mobile source emissions changes between 2025 and 2030, (3) identified controls of NO_x emissions from nonpoint, non-EGU point, and nonroad sources, (4) identified controls of VOC emissions, and (5) additional NO_x reductions beyond identified controls (i.e., unidentified controls). All controls applied to these sources were above and beyond reductions from on-the-books regulations that were included in the final 2025 base case modeling. The following paragraphs and Figure 3-8 outline these steps.

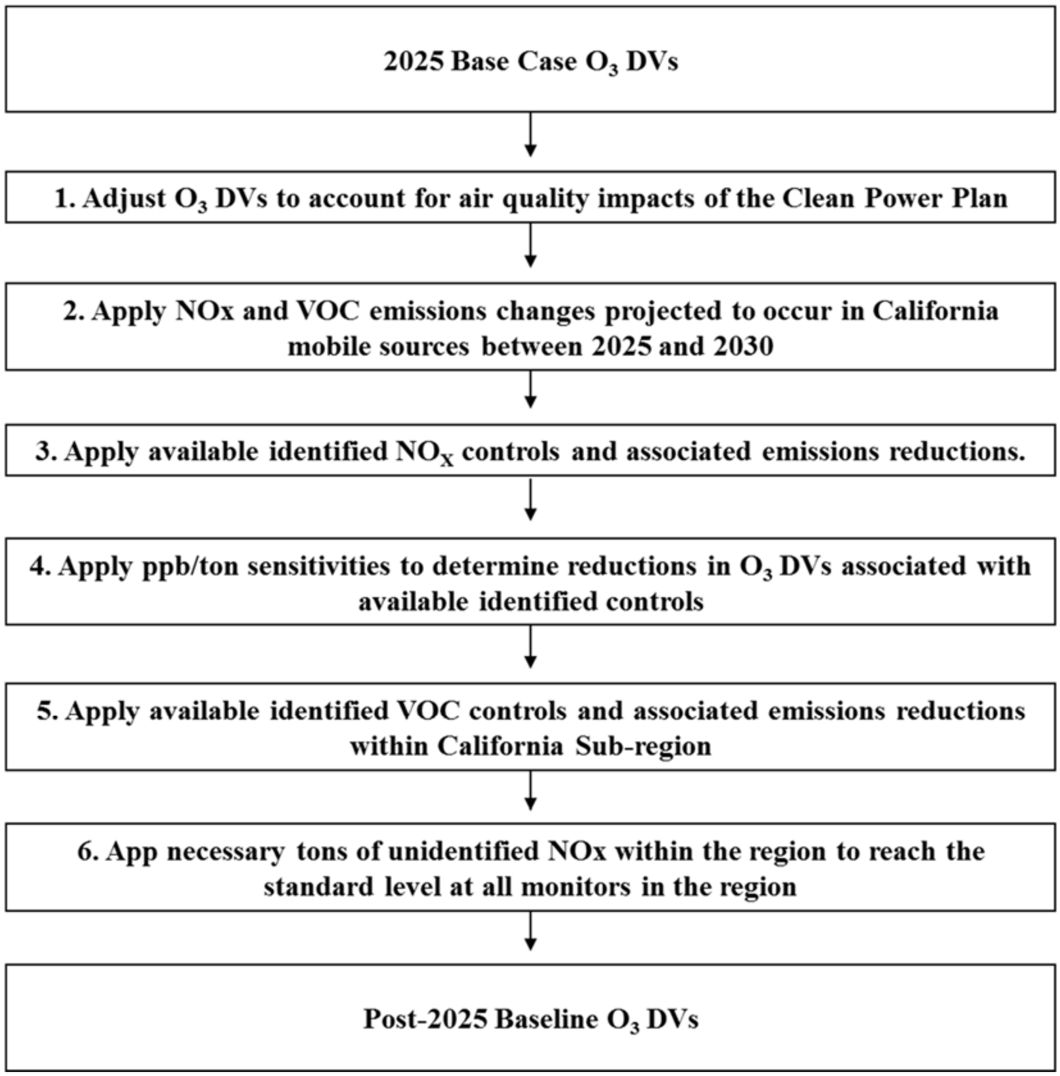

Figure 3-8. Steps to Create the Post-2025 Baseline for California

To create the post-2025 baseline, in Step 1 we accounted for emissions reductions from the Clean Power Plan.[50] In Step 2 we applied the 2025 to 2030 mobile source emissions reductions because many locations in California will likely have attainment dates farther into the future than 2025. Although emissions projections years beyond 2025 were not available, California provided emissions projections in the year 2030 of both VOC and NO_x 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

[50] We adjusted the 2025 base case to reflect emissions reductions from the Clean Power Plan to create the post-2025 baseline.

these mobile source changes resulted in: (1) VOC emissions that were 1% less than those modeled in the California base case in both the Northern and Southern California sub-regions (see Chapter 2, Figure 2-2 for a depiction of the California sub-regions), and (2) NO_x emissions that were 4% less than those modeled in the California base 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 post-2025 baseline scenario in California using the response ratios developed from the air quality sensitivity simulations as described in Chapter 2. In Step 3 available NOx reductions from identified control measures were applied from three sectors:[51] Non-Electric Generating Unit Point Sources (Non-EGUs), Nonpoint (Area) Sources, and Nonroad Mobile Sources. No controls for EGUs within the parameters of size (25 tpy) and dollar per ton control costs less than $19,000 per ton were available for California. Table 3-1 above also includes identified controls that we applied in California. In Step 4, the ppb/ton from the sensitivities were applied to determine ozone reductions. Then, in Step 5, VOC controls were applied in the California counties indicated in Figure 3-10.

In Step 6, we used additional reductions (assumed to come from unidentified NOX controls) in Southern California and associated regional ppb/ton response factors from the Southern California combined sensitivity simulations to reduce DVs at Southern California monitors to reach the current standard of 75 ppb. As described in Chapter 2 and shown in the example calculation in Appendix 3-A, we applied emissions responses derived from multiple emissions sensitivity simulations to capture the nonlinear response of large emissions reductions in Southern California. Similarly, we used unidentified reductions and associated ppb/ton response factors from Northern California to reduce DVs at Northern California monitors. Since the highest projected DVs occurred in Southern California, we first quantified necessary emissions reductions from unidentified NOx reduction measures to reduce all Southern California monitors to 75 ppb or lower. We then recalculated the resulting Northern California DVs before determining how many additional emissions reductions from unidentified NOx

[51] 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 because states will ultimately determine controls as part of the SIP process.

control measures in Northern California would be necessary to bring all Northern California monitors into attainment with the current standard of 75 ppb. Summaries of the emissions reductions are presented for the post-2025 baseline in Appendix 3A. The resulting ozone DVs at all evaluated monitors are also provided in Appendix 2A, Section 2A.4.

The post-2025 baseline for this analysis presents one scenario of future year air quality based upon specific control measures, additional emissions reductions beyond identified 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 approach relying on the identified federal measures and other strategies that states may employ. California may ultimately employ other strategies and/or other federal rules may be adopted that would also help in achieving attainment with the current standard.

3.2.2 Approach for Revised Standard of 70 ppb and Alternative Standard of 65 ppb for California

We created the post-2025 70 ppb and 65 ppb scenarios by applying emissions reductions incrementally to the post-2025 baseline. As mentioned above, all identified measures in California were exhausted in reaching the post-2025 baseline. We started with the post-2025 baseline and then applied NOx from unidentified controls to meet the revised and alternative standard levels. As with the baseline, we first identified the NOx reductions in Southern California that would be required to bring Southern California monitors down to the revised and alternative standard levels. We then recalculated the Northern California DVs that would result from the Southern California emissions reductions and applied additional Northern California unidentified NOx emissions reductions to bring all Northern California monitors down to the revised and alternative standard levels. Also, as was done for the baseline, we applied ppb/ton response levels that were derived from multiple emissions sensitivities to capture nonlinear responses of ozone to large emissions reductions in California (see example calculation in Appendix 3-A).

3.2.3 Results for California

Nine counties in California were projected to exceed the current ozone standard of 75 ppb in the post-2025 baseline scenario (see Figure 3-9). Figure 3-10 shows areas where identified control measures were applied to bring ozone DVs in those counties into attainment with the current standard and establish the baseline. Table 3-8 includes a summary of NOx and VOC emissions reductions needed to demonstrate attainment of the current ozone standard of 75 ppb.

Figure 3-9. Counties Projected to Exceed 75 ppb in the Post-2025 Baseline Scenario

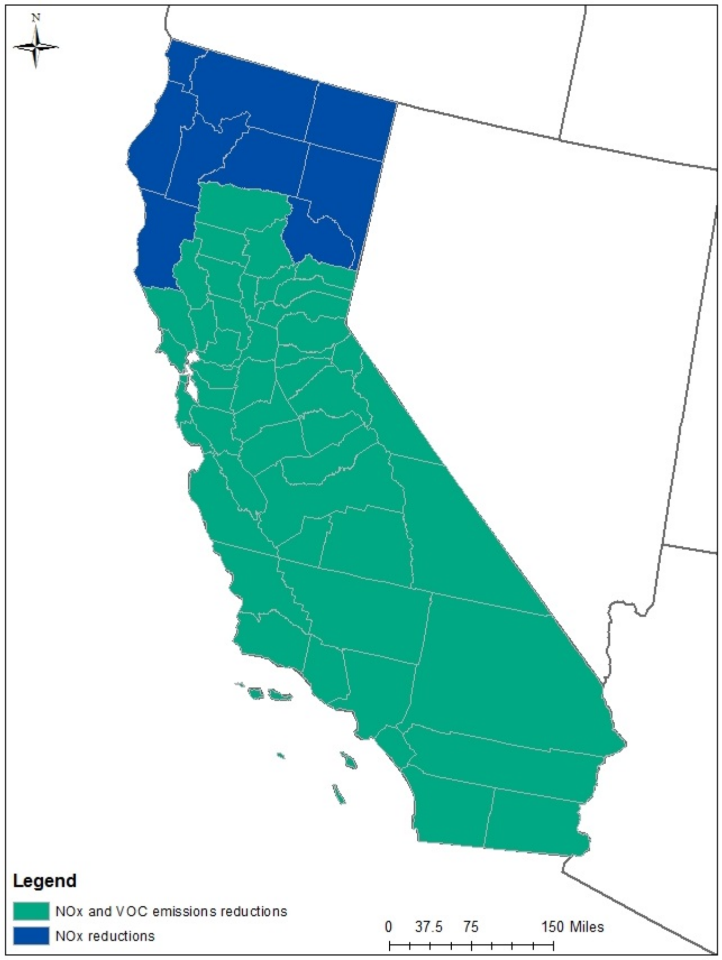

Figure 3-10. Counties Where Emissions Reductions Were Applied to Demonstrate Attainment with the Current Standard

Table 3-8. Summary of Emissions Reductions (Identified + Unidentified Controls) Applied to Demonstrate Attainment in California for the Post-2025 Baseline (1,000 tons/year)[a]

	Emissions Sector	**NO_x**	**VOC**
Identified Controls	EGU	-	-
	Non-EGU Point	14	1
	Nonpoint	14	54
	Nonroad	4	-
	Onroad	-	-
	Total	**32**	**55**
Unidentified Controls	All	160	-
	Total	**190**	**55**
Percent Unidentified		**84%**	**0%**

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

Figure 3-11 shows the California counties projected to exceed the revised and alternative standards analyzed for the post-2025 baseline analysis. Table 3-9 shows the emissions reductions needed from unidentified controls to meet the revised standard level of 70 ppb and alternative standard level of 65 ppb in those counties for the post-2025 analysis. Table 3-10 highlights that there were no identified NOx emissions reductions available for meeting the revised and alternative standard levels for post-2025 California and that 100 percent of the NOx emissions reductions needed were unidentified controls.

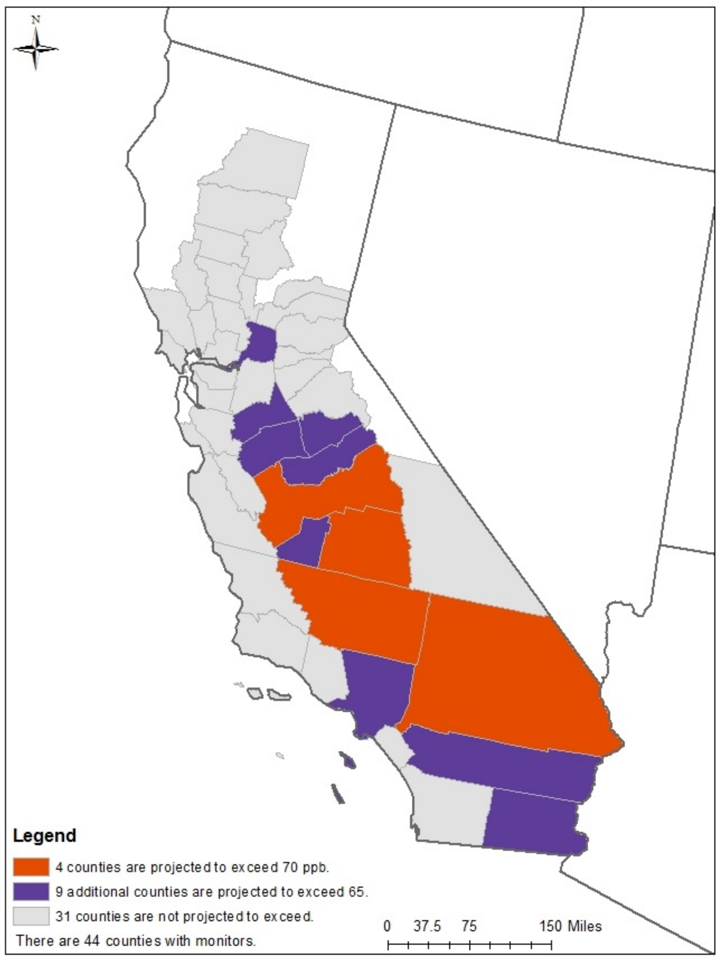

Figure 3-11. Projected Ozone Design Values in the Post-2025 Baseline Scenario

Table 3-9. Summary of Emissions Reductions from Unidentified Control Strategy for the Revised and Alternative Standard Levels for Post-2025 - California (1,000 tons/year)[a]

Alternative Standard	Region	NO_x	VOC
70 ppb	CA	51	-
65 ppb	CA	100	-

[a] Estimates are rounded to two significant figures.

Table 3-10. Summary of Emissions Reductions from the Identified + Unidentified Control Strategy by the Revised and Alternative Standard Levels for Post-2025 - California (1,000 tons/year)[a]

		Alternative Standard	
Geographic Area	**Emissions Reductions**	**70 ppb**	**65 ppb**
California	NOx Identified	0	0
	NOx Unidentified	51	100
	% NOx Unidentified	**100%**	**100%**
	VOC Identified	0	0
	VOC Unidentified	0	0
	% VOC Unidentified	**0%**	**0%**

[a] Estimates are rounded to two significant figures.

3.3 Improvements and Refinements since the Proposal RIA

In the regulatory impact analyses for both the ozone NAAQS proposal and final, there were two geographic areas outside of California where the majority of emissions reductions were needed to meet an alternative standard level of 70 ppb – Texas and the Northeast. In analyzing the revised standard of 70 ppb for the final RIA, there were approximately 50 percent fewer emissions reductions needed in these two areas. For an alternative standard of 65 ppb, emissions reductions needed nationwide were approximately 20 percent lower than at proposal.

The primary reason for the difference in emissions reductions needed for both 70 and 65 ppb is that in the final RIA we conducted more geographically-refined air quality sensitivity modeling to develop improved response factors (i.e., changes in ozone concentrations in response to emissions reductions). More detailed air quality modeling and improved response factors account for 80 percent of the difference in needed emissions reductions between proposal and final. See Chapter 2, Section 2.4.2 for a discussion of the air quality modeling.

For the analysis of the revised standard of 70 ppb, in Texas and the Northeast, the improved and refined response factors and more geographically focused emissions reductions strategies resulted in larger changes in ozone concentrations. In east Texas, the ppb/ton response factors used in the final RIA were 2 to 3 times more responsive than the factors used in the proposal RIA at controlling monitors in Houston and Dallas. In the Northeast, the ppb/ton response factors used in the final RIA were 2.5 times more responsive than the factors used in the proposal RIA at the controlling monitor on Long Island, NY.

A secondary reason for the difference is that between the proposal and final RIAs we updated models and model inputs for the base year of 2011. See Appendix 2, Section 2A.1.3 for additional discussion of the updated models and model inputs. When projected to 2025, these changes in models and inputs had compounding effects for year 2025, and in some areas resulted in lower projected base case design values for 2025. In these areas, the difference between the base case design values and a standard of 70 ppb was smaller, thus requiring fewer emissions reductions to attain the 70 ppb revised standard.

Note that the more spatially refined emissions sensitivity modeling had more impact on the results at 70 ppb than it did on the results at 65 ppb due to the more localized nature of projected exceedances at 70 ppb. For example, as described above, the new sensitivity regions showed that emissions reductions in eastern Texas would have a larger impact on ozone in Houston and Dallas than the same emissions reductions would have if they were spread over the central U.S. states used in the proposal RIA. Conversely, these same east Texas emissions reductions would have less impact on violating monitors in Louisiana or Oklahoma. Therefore, for the 65 ppb scenario, additional local controls were necessary in Louisiana and Oklahoma.

As a consequence of the use of more geographically refined sensitivity regions, emissions reductions control strategies were also applied in geographic areas closer to the monitors of projected exceedances. For example, in the proposal RIA, the Central region included Texas, Oklahoma, Kansas, Missouri, Arkansas, Louisiana and Mississippi, meaning that controls could be applied anywhere in those states after identified controls had been exhausted within the 200 km buffer. But in the final RIA, the only geographic area where we applied controls was East Texas. Thus, once identified control measures were exhausted there, we had to obtain remaining reductions from unidentified control measures. While the total amount of emissions needed to meet the 65 ppb alternative standard is lower than it was in the proposal RIA, the fraction of emissions reductions from identified controls was smaller.

3.4 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 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 there is significant flexibility for states to determine which measures to apply to comply with the standard.
- **Sequential processing of regional emission reductions:** Because this method prioritizes emissions reductions in the regions with the highest ozone values first but then does not go back and re-evaluate the amount of reduction in the higher priority region after emissions reductions have been applied in lower-priority regions, there is the potential to reduce a greater quantity of emissions at monitors in the higher priority regions. For instance, in the 65 ppb scenario, in the Northeast, the monitor which required the largest emissions reductions to reach 65 ppb was located in Queens, NY. After identifying necessary emissions reductions in the Northeast region, that monitor had a projected DV of 65.996 ppb (which truncates to 65 ppb). Additional reductions from lower priority regions such as the Ohio River Valley and the Great Lakes, brought the DV at that site down to 65.002 ppb. In theory, fewer tons of emissions reductions could then have been applied in the Northeast to reach a DV less than 66 ppb. However, if emissions reductions in the Northeast were rolled back, then necessary reductions in all lower priority regions would need to be recalculated and consequently the degree to which the Northeast emissions reductions were rolled back would also need to be recalculated. This could be quantified either in an iterative process or through a linear programming model that found a least cost solution based on all response factors and associated costs. Neither of these options were available for this analysis, but it should be noted that this likely leads to some overestimate in our

calculation of tons of emissions reductions necessary to meet the 70 and 65 ppb standard levels and in the resulting costs and benefits.

- **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:** As we focus on the advances that might be expected in existing pollution control technologies, we recognize that the control measures applied do not reflect potential effects of technological change that may be available in future years. 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 or costs. 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 may change 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 (see Chapter 4, Section 4.2 for additional discussion of the Phase 2 Heavy Duty Greenhouse Gas Standards for New Vehicles and Engines). These regulations could reduce the current baseline levels of emissions.

3.5 References

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APPENDIX 3A: CONTROL STRATEGIES AND EMISSIONS REDUCTIONS

Overview

Chapter 3 describes the approach that EPA used in applying control measures to demonstrate attainment of alternative ozone standard levels of 70. This Appendix contains more detailed information about the control strategy analyses, including numerical examples of the calculation methods for changes in ozone DVs, the control measures that were applied and the geographic areas in which they were applied.

3A.1 Target Emissions Reductions Needed to Create the Baseline, Post-2025 Baseline and Alternatives

Tables 3A-1 to 3A-3 depict emissions reductions required in each region to reach the alternative standard level scenarios for the U.S. except California, and the post-2025 Baseline and alternative standard levels for California. These emissions reductions were determined using the methodology described in Chapter 3, Sections 3.1 and 3.2 and illustrated in the numerical example in section 3A.2 of this Appendix. Sector-specific controls used for these reductions are discussed in more detail in Chapter 3. These emissions reductions were used to create the ozone surfaces described in Chapter 2, Section 2.4.

Table 3A-1. Emissions Reductions Applied Beyond the Baseline Scenario to Create the 70 ppb Scenario

	Emissions reductions (thousand tons) applied from		
	NOx reductions from identified controls	**VOC reductions from identified controls**	**Additional NOx reductions from unidentified measures**
Northeast	111	-	-
Ohio River Valley	27	-	-
Great Lakes	18	-	-
East Texas	123	20 (Houston)	-
Colorado	7	-	-
N. California	Exhausted in baseline scenario	Exhausted in baseline scenario	35
S. California	Exhausted in baseline scenario	Exhausted in baseline scenario	16

Table 3A-2. Emissions Reductions Applied Beyond the Baseline Scenario to Create the 65 ppb Scenario

	Emissions reductions (thousand tons) applied from		
	NOx reductions from identified controls	**VOC reductions from identified controls**	**Additional NOx reductions from unidentified measures**
Northeast	163	41 (NY area)	285
Ohio River Valley	169	7 (Louisville area)	112
Great Lakes	197	39 (Chicago area)	56
OK/AR/LA	24	-	-
E. Texas	123	20 (Houston area)	188
AZ/NM	29	-	-
Colorado	36	5 (Denver area)	20
Nevada	10	-	-
N. California	Exhausted in baseline scenario	Exhausted in baseline scenario	65.5
S. California	Exhausted in baseline scenario	Exhausted in baseline scenario	32

Table 3A-3. Emissions Reductions Applied to Create the Post-2025 Baseline Scenario*

	Emissions reductions (thousand tons) applied from			
	2025-2030 California mobile source changes	**NOx reductions from identified controls**	**VOC reductions from identified controls**	**Additional NOx reductions from unidentified measures**
N. California	8 (NOx) 3 (VOC)	16	27	24
S. California	6 (NOx) 3 (VOC)	16	29	136

*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.

3A.2 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 Sections 3.1 and 3.2. For each monitor, numerical examples are given for calculating the emissions reductions necessary to attain the current standard of 75 ppb (i.e., the baseline scenario) as well as the 70 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 70.9 would meet an alternative standard level of 70. 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 2-5 from Chapter 2.

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

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

$$\begin{aligned} DV_{60195001,baseline} &= \underbrace{83.4}_{DV_{60195001,2025}} + \overbrace{\underbrace{-0.7}_{\Delta DV_{60195001,111d}}}^{NOx+VOC} + \overbrace{\left(\underbrace{-5.0 \times 10^{-5}}_{R_{60195001,CAcontrl}} \times \underbrace{32{,}000}_{\Delta E_{CAcontrl}}\right)}^{NOx} \\ &+ \overbrace{\left(\underbrace{-1.4 \times 10^{-4}}_{R_{60195001,CAcontrol+50NOx,NCA}} \times \underbrace{8000 + 24{,}000}_{\Delta E_{mobile,2030}+\Delta E}\right)}^{NOx} \\ &+ \overbrace{\left(\underbrace{-9.9 \times 10^{-6}}_{R_{60195001,VOC\_50,NCA}} \times \underbrace{3000 + 27{,}000}_{\Delta E_{mobile,2030}+\Delta E}\right)}^{VOC} \\ &+ \left(\underbrace{-2.3 \times 10^{-6}}_{R_{60195001,CAcontrol+50NOx,SCA}} \times \underbrace{6000 + 100{,}000}_{\Delta E_{mobile,2030}+\Delta E}\right) \\ &+ \left(\underbrace{-2.6 \times 10^{-6}}_{R_{60195001,CAcontrol+90NOx,SCA}} \times \underbrace{36{,}000}_{\Delta E}\right) = 75.9\ ppb \end{aligned}$$

Example 2. Fresno California monitor 60195001 (65 ppb scenario):

$$DV_{j,65} = \underbrace{75.9}_{DV_{j,baseline}} + \overbrace{\left(\underbrace{-1.4 \times 10^{-4}}_{R_{60195001,CAcontrol+50NOx,NCA}} \times \underbrace{35{,}000}_{\Delta E} \right)}^{NOx\ up\ to\ 50\%\ of\ CA\ modeled\ control\ sensitivity} + \left(\underbrace{-2.6 \times 10^{-6}}_{R_{60195001,CAcontrol+90NOx,SCA}} \times \underbrace{16{,}000}_{\Delta E} \right) = 70.9\ ppb$$

Example 3. Dallas monitor 484392003 (baseline):

$$DV_{484392003,baseline} = \underbrace{74.3}_{DV_{484392003,2025}} + \overbrace{\underbrace{-1.0}_{\Delta DV_{484392003,111d}}}^{NOx+VOC} = 73.3\ ppb$$

Example 4. Dallas monitor 484392003 (65 ppb scenario):

$$DV_{484392003,65} = \underbrace{73.3}_{DV_{484392003,baseline}} + \overbrace{\left(\underbrace{-3.3 \times 10^{-5}}_{R_{484392003,E.Texas}} \times \underbrace{123{,}000}_{\Delta E} \right)}^{NOx}$$
$$+ \overbrace{\left(\underbrace{-4.1 \times 10^{-8}}_{R_{484392003,Northeast}} \times \underbrace{111{,}000}_{\Delta E} \right)}^{NOx} + \overbrace{\left(\underbrace{-5.1 \times 10^{-7}}_{R_{484392003,Colorado}} \times \underbrace{7000}_{\Delta E} \right)}^{NOx}$$
$$+ \overbrace{\left(\underbrace{-2.6 \times 10^{-7}}_{R_{484392003,OhioRiverValley}} \times \underbrace{27{,}000}_{\Delta E} \right)}^{NOx} + \overbrace{\left(\underbrace{-1.9 \times 10^{-7}}_{R_{484392003,GreatLakes}} \times \underbrace{18{,}000}_{\Delta E} \right)}^{NOx}$$
$$= 69.3\ ppb$$

3A.3 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:

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 25 tons/yr of NO_x (see description of controls on smaller sources below);
- any control for nonEGU 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 and SNCRs – applied to coal-fired EGUs where they are in place but are idle.

VOC Reductions – VOC control measures for nonEGU and nonpoint sources that:

- emit at least 10 tons/yr of VOC;
- any control must result in a reduction of VOC of at least 1 ton/yr; and
- any replacement control must achieve 10% more reduction than the existing control.

3A.4 Application of Control Measures in Geographic Areas

Control measures were applied, to obtain the emissions reductions described in Section 3A.1 of this Appendix, to geographic areas including or adjacent to areas that were projected to exceed the baseline and alternative standards. If all non-EGU NO_x reductions were needed, then the maximum emissions reductions algorithm in CoST was used. Where less non-EGU NO_x reductions were needed than were available, these were obtained using the least cost algorithm. Where VOC reductions were needed, all potentially available VOC reductions were needed so these were identified using the maximum emissions reduction algorithm. No unidentified controls were needed for VOC emissions reductions. Tables 3A-4 and 3A-5 show where controls were applied and where unidentified controls were needed in the U.S. except California. Tables 3A-6 and 3A-7 show where controls were applied and where unidentified controls were needed in California.

Table 3A-4. Geographic Areas for Application of NOx Controls in the Baseline and Alternative Standard Analyses - U.S., except California[a]

Geographic Areas and Controls	Baseline	70 ppb	65 ppb
EAST			
North East			
Inside buffer			
Non-EGU		x	x
EGU		x	x
Outside buffer		x	x
Unidentified			U
OK + AR + LA			
Inside buffer			
Non-EGU			x
EGU			
Outside buffer			
Unidentified			
Ohio River Valley			
Inside buffer			
Non-EGU		x	x
EGU		x	x
Outside buffer		x	x
Unidentified			U
TX East			
Inside buffer			
Non-EGU		x	x
EGU			
Outside buffer		x	x
Unidentified		U	U
WEST			
AZ + NM			
Inside buffer			
Non-EGU			x
EGU			
Outside buffer			x
Unidentified			
Colorado			
Inside buffer			
Non-EGU		x	x
EGU			
Outside buffer			x
Unidentified			U

Geographic Areas and Controls	Baseline	70 ppb	65 ppb
Great Lakes			
Inside buffer			
Non-EGU		x	x
EGU			x
Outside buffer		x	x
Unidentified			U
Nevada			
Inside buffer			
Non-EGU			x
EGU			
Outside buffer			x
Unidentified			U

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

Table 3A-5. Geographic Areas for Application of VOC[a] Controls in the Baseline and Alternative Standard Analyses - U.S., except California[b]

Geographic Area	Baseline	70 ppb	65 ppb
EAST			x
North East			
New York, New Jersey, Long Island, NY-NJ-CT			
Ohio River Valley			
Louisville, KY			x
TX East			
Houston-Galveston-Brazoria, TX		x	x
WEST			
Colorado			
Denver-Boulder-Greeley-Ft.Collins-Loveland, CO			x
Great Lakes			
Chicago-Lake Michigan, WI-IL-IN-MI			x

[a] No unidentified VOC controls were needed to attain any of the standards; [b] "x" indicates known controls were applied

Table 3A-6. Geographic Areas for Application of NO_x[a] Controls in the Baseline and Alternative Standard Analyses – California[b]

Geographic Areas and Control Groups	Baseline	70 ppb	65 ppb
California			
California North Identified	x		
California North Unidentified	U	U	U
California South Identified	x		
California South Unidentified	U	U	U

[a] All reductions were calculated using the maximum reductions algorithm [b] "x" indicates known controls were applied; "U" indicates unknown control reductions.

Table 3A-7. Geographic Areas for Application of VOC[a] Controls in the Baseline and Alternative Standard Analyses – California[b]

Geographic Areas and Control Groups	Baseline	70 ppb	65 ppb
California			
California North – San Joaquin	x		
Identified	U	U	U
Unidentified			
California South – Los Angeles			
Identified	x		
Unidentified	U	U	U

[a] All reductions were calculated using the maximum reductions algorithm [b] "x" indicates known controls were applied; "U" indicates unknown control reductions.

3A.5 NO_x Control Measures for Non-EGU 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 3A-8 through 3A-11 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 3A-8. NO_x Control Measures Applied in the 70 ppb Analysis

NOx Control Measure	Reductions (tons/year)
Adjust Air to Fuel Ratio and Ignition Retard - Gas Fired IC Engines	8,723
Biosolid Injection Technology - Cement Kilns	5,383
EGU SCR & SNCR	44,951
Episodic Burn Ban	2,797
Excess O3 Control	229
Ignition Retard - IC Engines	618
Low Emission Combustion - Gas Fired Lean Burn IC Engines	17,676
Low NOx Burner - Coal Cleaning	270
Low NOx Burner - Commercial/Institutional Boilers & IC Engines	21,417
Low NOx Burner - Gas-Fired Combustion	9,237
Low NOx Burner - Glass Manufacturing	247
Low NOx Burner - Industr/Commercial/Institutional (ICI) Boilers	5,580
Low NOx Burner - Industrial Combustion	25
Low NOx Burner - Lime Kilns	2,433
Low NOx Burner - Natural Gas-Fired Turbines	6,276
Low NOx Burner - Residential Water Heaters & Space Heaters	19,900
Low NOx Burner and Flue Gas Recirculation - Coke Oven/Blast Furnace	359
Low NOx Burner and Flue Gas Recirculation - Fluid Catalytic Cracking Units	84
Low NOx Burner and Flue Gas Recirculation - Iron & Steel	399
Low NOx Burner and SCR - Industr/Commercial/Institutional Boilers	8,408
Mid-Kiln Firing - Cement Manufacturing	1,241

NOx Control Measure	Reductions (tons/year)
Non-Selective Catalytic Reduction (NSCR) - 4 Cycle Rich Burn IC Engines	39,258
Nonroad Diesel Retrofits & Engine Rebuilds - e.g., Construction Equipment	2,832
OXY-Firing - Glass Manufacturing	11,984
Replacement of Residential & Commercial/Institutional Water Heaters	8,641
Selective Catalytic Reduction (SCR) - Cement Kilns	10,176
Selective Catalytic Reduction (SCR) - Fluid Catalytic Cracking Units	1,709
Selective Catalytic Reduction (SCR) - Glass Manufacturing	3,481
Selective Catalytic Reduction (SCR) - IC Engines, Diesel	863
Selective Catalytic Reduction (SCR) - ICI Boilers	4,618
Selective Catalytic Reduction (SCR) - Industrial Incinerators	1,384
Selective Catalytic Reduction (SCR) - Iron & Steel	155
Selective Catalytic Reduction (SCR) - Process Heaters	784
Selective Catalytic Reduction (SCR) - Sludge Incinerators	100
Selective Catalytic Reduction (SCR) - Space Heaters	24
Selective Catalytic Reduction (SCR) - Utility Boilers	1,391
Selective Non-Catalytic Reduction (SNCR) - Cement Manufacturing	2,405
Selective Non-Catalytic Reduction (SNCR) - Coke Manufacturing	1,589
Selective Non-Catalytic Reduction (SNCR) - Comm./Inst. Incinerators	58
Selective Non-Catalytic Reduction (SNCR) - Industrial Incinerators	365
Selective Non-Catalytic Reduction (SNCR) - Sludge Incinerators	33
Ultra-Low NOx Burner - Process Heaters	329

Table 3A-9. VOC Control Measures Applied in the 70 ppb Analysis

VOC Control Measure	Reductions (tons/year)
Control Technology Guidelines - Wood Furniture Surface Coating	272
Control of Fugitive Releases - Oil & Natural Gas Production	9
Flare - Petroleum Flare	94
Incineration - Other	10,717
LPV Relief Valve - Underground Tanks	1,299
MACT - Motor Vehicle Coating	10
Permanent Total Enclosure (PTE) - Surface Coating	369
RACT - Graphic Arts	260
Reduced Solvent Utilization - Surface Coating	27
Reformulation - Architectural Coatings	5,246
Reformulation - Pesticides Application	171
Reformulation-Process Modification - Automobile Refinishing	220
Reformulation-Process Modification - Cutback Asphalt	655
Reformulation-Process Modification - Other	113
Reformulation-Process Modification - Surface Coating	178
Solvent Recovery System - Printing/Publishing	13
Wastewater Treatment Controls- POTWs	207

Table 3A-10. NOx Control Measures Applied in the 65 ppb Alternative Standard Analysis

NOx Control Measure	Reductions (tons/year)
Adjust Air to Fuel Ratio and Ignition Retard - Gas Fired IC Engines	16,423
Biosolid Injection Technology - Cement Kilns	5,907
EGU SCR & SNCR	109,503
Episodic Burn Ban	3,283
Ignition Retard - IC Engines	575
Low Emission Combustion - Gas Fired Lean Burn IC Engines	75,724
Low NOx Burner - Coal Cleaning	475
Low NOx Burner - Commercial/Institutional Boilers & IC Engines	36,210
Low NOx Burner - Fiberglass Manufacturing	65
Low NOx Burner - Gas-Fired Combustion	11,889
Low NOx Burner - Industr/Commercial/Institutional (ICI) Boilers	21,918
Low NOx Burner - Industrial Combustion	25
Low NOx Burner - Lime Kilns	4,616
Low NOx Burner - Natural Gas-Fired Turbines	12,109
Low NOx Burner - Residential Water Heaters & Space Heaters	51,703
Low NOx Burner - Steel Foundry Furnaces	294
Low NOx Burner - Surface Coating Ovens	26
Low NOx Burner and Flue Gas Recirculation - (ICI) Boilers	477
Low NOx Burner and Flue Gas Recirculation - Coke Oven/Blast Furnace	429
Low NOx Burner and Flue Gas Recirculation - Fluid Catalytic Cracking Units	59
Low NOx Burner and Flue Gas Recirculation - Iron & Steel	781
Low NOx Burner and Flue Gas Recirculation - Process Heaters	548
Low NOx Burner and Flue Gas Recirculation - Starch Manufacturing	67
Low NOx Burner and SCR - Industr/Commercial/Institutional Boilers	24,281
Low NOx Burner and SNCR - Industr/Commercial/Institutional Boilers	482
Natural Gas Reburn - Natural Gas-Fired EGU Boilers	590
Non-Selective Catalytic Reduction (NSCR) - 4 Cycle Rich Burn IC Engines	70,008
Non-Selective Catalytic Reduction - Nitric Acid Manufacturing	491
Nonroad Diesel Retrofits & Engine Rebuilds - e.g., Construction Equipment	8,791
OXY-Firing - Glass Manufacturing	27,100
Replacement of Residential Water Heaters	133
Selective Catalytic Reduction (SCR) - Ammonia Mfg	2,336
Selective Catalytic Reduction (SCR) - Cement Kilns	26,144
Selective Catalytic Reduction (SCR) - Coke Ovens	1,243
Selective Catalytic Reduction (SCR) - Fluid Catalytic Cracking Units	4,078
Selective Catalytic Reduction (SCR) - IC Engines, Diesel	3,574
Selective Catalytic Reduction (SCR) - ICI Boilers	9,963
Selective Catalytic Reduction (SCR) - Industrial Incinerators	1,723
Selective Catalytic Reduction (SCR) - Iron & Steel	1,777
Selective Catalytic Reduction (SCR) - Process Heaters	2,744

NOx Control Measure	Reductions (tons/year)
Selective Catalytic Reduction (SCR) - Sludge Incinerators	1,771
Selective Catalytic Reduction (SCR) - Space Heaters	286
Selective Catalytic Reduction (SCR) - Taconite	4,248
Selective Catalytic Reduction (SCR) - Utility Boilers	1,391
Selective Non-Catalytic Reduction (SNCR) - Coke Mfg	2,880
Selective Non-Catalytic Reduction (SNCR) - Comm./Inst. Incinerators	159
Selective Non-Catalytic Reduction (SNCR) - Industrial Incinerators	1,057
Selective Non-Catalytic Reduction (SNCR) - Municipal Waste Combustors	67
Selective Non-Catalytic Reduction (SNCR) - Sludge Incinerators	113
Selective Non-Catalytic Reduction (SNCR) - Utility Boilers	235
Ultra-Low NOx Burner - Process Heaters	854

Table 3A-11. VOC Control Measures Applied in the 65 ppb Alternative Standard Analysis

VOC Control Measure	Reductions (tons/year)
Control Technology Guidelines - Wood Furniture Surface Coating	2,988
Control of Fugitive Releases - Oil & Natural Gas Production	30
Flare - Petroleum Flare	108
Gas Recovery - Municipal Solid Waste Landfill	290
Improved Work Practices, Material Substitution, Add-On Controls - Printing	8
Improved Work Practices, Material Substitution, Add-On Controls -Industrial Cleaning Solvents	248
Incineration - Other	16,710
LPV Relief Valve - Underground Tanks	4,871
Low VOC Adhesives and Improved Application Methods - Industrial Adhesives	237
Low-VOC Coatings and Add-On Controls - Surface Coating	274
MACT - Motor Vehicle Coating	1,934
Permanent Total Enclosure (PTE) - Surface Coating	3,286
Petroleum and Solvent Evaporation - Surface Coating Operations	250
RACT - Graphic Arts	5,586
Reduced Solvent Utilization - Surface Coating	3,047
Reformulation - Architectural Coatings	52,378
Reformulation - Industrial Adhesives	1,110
Reformulation - Pesticides Application	3,957
Reformulation-Process Modification - Automobile Refinishing	4,879
Reformulation-Process Modification - Cutback Asphalt	2,555
Reformulation-Process Modification - Oil & Natural Gas Production	291
Reformulation-Process Modification - Other	546
Reformulation-Process Modification - Surface Coating	5,622
Solvent Recovery System - Printing/Publishing	854
Solvent Substitution and Improved Application Methods - Fiberglass Boat Mfg	14
Wastewater Treatment Controls- POTWs	234

3A.6 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.

3A.7 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.

3A.8 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 4: ENGINEERING COST ANALYSIS AND ECONOMIC IMPACTS

Overview

This chapter provides estimates of the engineering costs of the control strategies presented in Chapter 3 for the revised primary standard of 70 ppb and an alternative standard level of 65 ppb and summarizes the data sources and methodologies used to estimate the engineering costs presented in this regulatory impact analysis (RIA). As discussed in Chapter 3, identified control measures were applied to EGU, non-EGU point, nonpoint (area), and nonroad mobile sources to demonstrate attainment with the revised and alternative standards analyzed.[52] In several areas identified controls did not achieve the emissions reductions needed to attain the revised and alternative standards analyzed. In these areas, the EPA assumed that further controls would be applied to reach attainment. These additional controls are referred to as unidentified controls.

The total cost estimates include the costs of both identified and unidentified control technologies and measures. The estimated total costs of attaining the revised and alternative standards are partly a function of (1) assumptions used in the analysis, including assumptions about which areas will require emissions controls and the sources and controls available in those areas; (2) the level of sufficient, detailed information on identified control measures needed to estimate engineering costs; and (3) the future year baseline emissions from which the emissions reductions needed to attain are measured.

The remainder of the chapter is organized as follows. Section 4.1 presents the engineering costs associated with the application of identified controls. Section 4.2 discusses the challenges associated with estimating costs for unidentified controls, including a brief discussion

[52] In Chapter 3, Table 3-7 lists the specific control technologies applied in the identified control measures analysis. In addition, in the proposal RIA we discuss emissions reductions resulting from the application of known controls, as well as emissions reductions beyond known controls, or in short, *known controls* and *unknown controls*. In the final RIA we refer to those sets of emissions reductions and controls as *identified controls* or measures and *unidentified controls* or measures. This terminology has been used in prior NAAQS RIAs and reflects that we have illustrated control strategies primarily using end-of-pipe controls and many additional controls that are not end-of-pipe (e.g., energy efficiency) that we have not identified here could also be part of a states' strategies to reduce emissions.

of some of the limitations of EPA's control strategy tools and available data on NOx control technologies, and a brief discussion of the challenges in estimating baseline emissions over time. Section 4.3 presents the estimated costs associated with unidentified controls. Section 4.4 provides the total compliance cost estimates. Section 4.5 includes a discussion of potential economic impacts. Section 4.6 concludes with a discussion of the uncertainties and limitations associated with these components of the RIA.

4.1 Estimating Engineering Costs

The engineering costs described in this chapter generally include the costs of purchasing, installing, operating, and maintaining the technologies applied. The costs associated with monitoring, testing, reporting, and recordkeeping for affected sources are not included in the annualized cost estimates as this data is not generally available and can vary substantially from one facility to another. For a variety of reasons, actual control costs may vary from the estimates the EPA presents. As discussed throughout this analysis, the technologies and control strategies selected for analysis illustrate 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 a revised standard, and the EPA anticipates that state and local governments will consider programs best suited for local conditions. In addition, the EPA recognizes that there is substantial uncertainty in the portion of the engineering cost estimates associated with unidentified controls. The estimates presented herein are based on assumptions about the sectors and technologies that might become available for cost-effective control application in the future.

The engineering cost estimates are limited in their scope. This analysis focuses on the emissions reductions needed for attainment of the revised standard and an alternative standard analyzed. The EPA understands that some states will incur costs both designing State Implementation Plans (SIPs) 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.

4.1.1 Methods and Data

The EPA uses the Control Strategy Tool (CoST) (U.S. EPA, 2014a) to estimate engineering control costs. CoST was used in two parts of the analysis. First, CoST was applied to help determine potential NOx and VOC emissions reductions for each of the emissions sensitivity regions (see Chapter 2 Figure 2.2 for a map of these regions). Secondly, CoST was used to estimate the identified controls costs for the measures identified in Chapter 3. We estimated costs for non-electric generating unit point (non-EGU 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 much more detailed data, and parameters used in these equations are not 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, including cost-per-ton factors and cost equations, can be found in the tool documentation.[53] Costs for selective reduction catalysts (SCR) applied as part of the analysis for reducing NO_x emissions at coal-fired electric generating units (EGUs) were estimated using documentation for the Integrated Planning Model (IPM) (Sargent & Lundy, 2013).

When sufficient information is available to estimate a control cost using equations, the capital costs of the control equipment must be annualized. Capital costs are converted to annual costs using the capital recovery factor (CRF).[54] The engineering cost analysis uses the

[53] CoST documentation is available at: http://www.epa.gov/ttnecas1/cost.htm
[54] The capital recovery factor incorporates the interest rate and equipment life (in years) of the control equipment. 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. Using engineering convention, the annualized costs assume a 7 percent interest rate

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. Where possible, calculations are used to calculate total annual control cost (TACC), which is a function of capital costs (CC) and O&M costs. 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 the EUAC method and the TACC, 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, and 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$).[55] 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 (i) we do not know what all firms making capital investments for control measures will do and when they will do it and (ii) we do not have nor know of a better data source with possible capital investment schedules. The estimates of annualized costs include annualized capital and annual O&M costs for those controls included in the identified control strategy analysis. We make no assumptions about capital investments prior to 2025 or additional capital investment in years beyond 2025. The controls applied and their respective engineering costs are described in the Chapter 4 Appendix.

CoST relies on detailed data from the National Emissions Inventory (NEI), including detailed information by source on emissions, installed control devices, and control device efficiency. Much of this underlying NEI data serves as key inputs into the control strategy analysis. The EPA receives NEI submissions from state, local, and tribal (SLT) air agencies. Information on whether a source is currently controlled, by what control device, and control device efficiency, is required under the Air Emissions Reporting Rule (AERR) used to collect

for non-EGU point sources, nonpoint sources, and nonroad mobile sources. For EGU sources the annualized costs assume a rate of 4.77 percent. For additional discussion please see Section 4.1.2.

[55] 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.

the NEI data. This information is only required to be provided when controls are present for the sources. Since controls are not present on every source, it is not possible for the EPA to enforce systematically (i.e., through electronic reporting) the requirement to report control devices. As a result, control information may not be fully reported by SLT agencies and would therefore not be available for purposes of the control strategy analysis.

As indicated earlier, EPA needed to determine the universe of potential NOx and VOC controls and emissions reductions for each of the emissions sensitivity regions. To accomplish this, the EPA reviewed the emissions inventory and universe of potential control information from CoST to identify and employ (i) size thresholds for minimum emissions reductions (e.g., applying a control device should result in a minimum of 5 tons of NOx emissions reductions), (ii) size thresholds for application of control devices (e.g., apply a control device to sources of 25 tons of NOx emissions or more), and (iii) cost-per-ton thresholds for applying controls from the CoST database (e.g., do not apply controls that cost more than $19,000/ton to reduce NOx emissions). The above steps are taken to mitigate potential double counting of controls due to possible missing control measure information in the NEI and to reduce the number of cases where additional control measures are applied in impractical circumstances.

The highest cost-per-ton estimates are often associated with controls that reduce very small increments of NOx emissions or are unique applications of a particular control. For example, in some cases, controls that were developed primarily to address other pollutant emissions, such as SO_2, also achieve NO_X reductions and could be applied for this purpose. These controls are well characterized in the CoST database because they have been used for SO_2 control, but the degree to which sources would adopt these controls specifically to obtain NO_X reductions is uncertain. To reduce the number of cases where additional control measures are applied in impractical circumstances, we selected cost-per-ton thresholds for applying both NOx and VOC controls from the CoST database. We aggregated the raw data on all identified controls for NOx in the control measures database by cost per ton and plotted an identified control cost curve. It is important to note that this identified control cost curve is not a complete representation of the marginal abatement cost curve. A marginal abatement cost curve presents the least-cost approach to achieving any specific level of emissions reduction. In contrast, the identified control cost curve is a series of cost-per-ton estimates based on a specific emissions inventory combined with details from CoST about possible control measures that could be applied. The identified control cost curve defines how many tons of emissions reductions can be achieved at various cost levels from identified control technologies. While emissions reductions and their associated costs may be available for many different control measures, not all of these measures will be the most cost-effective way of achieving a given level of abatement, and therefore should not be used to construct the marginal abatement cost curve. In addition, we lack information on the control measures and costs for the remaining uncontrolled NOx emissions (see more detailed discussion on incomplete representation of marginal abatement cost curve in section 4.2).

Because the identified control cost curve reflects incomplete information, it is necessary to take steps to identify likely impractical control applications and to remove them from the analysis.[56] We determined that applying an exponential trend line would produce a reasonable cost threshold for identified controls, and we used the assumption in this analysis. To determine a cost threshold for identified NOx controls, we used the full dataset on NOx control measures and plotted an exponential trend line through the identified control cost curve.[57] Figure 4-1 shows the identified control cost curve for all the NOx control measures contained in the CoST database, aggregated by cost per ton, and the exponential trend line. As the figure indicates, the curves intersect at $19,000 per ton, meaning control costs above $19,000 per ton begin increasing at more than an exponential rate. We selected $19,000 per ton as the control cost value above which we would not apply additional identified NOx controls because controls above this value are not likely to be cost-effective. In the control strategy analysis for 70 ppb, there are a total of only eight control applications in three geographic areas where identified NOx controls are applied at a cost of $19,000/ton. In addition, for a standard of 70 ppb, in east Texas, the Northeast, the Great Lakes, and the Ohio River Valley there are a total of 25 control applications between $15,000/ton and $19,000/ton, representing approximately 5 percent of the

[56] Examples of control applications that could be removed from the analysis include: (i) applying SCR to small lean burn natural gas-fired reciprocating internal combustion engines to reduce NOx emissions -- these units are often in very remote locations and the requirements for ammonia or urea storage and replenishment are not practical, and ii) retrofit controls on small ICI boilers with space limitations that make the retrofit too difficult,
[57] The full dataset on NOx control measures includes approximately 120,000 individual observations, and when aggregated by cost per ton, the dataset includes 1,500 observations.

total cost of identified NOx controls and approximately 1 percent of the total NOx emissions reductions from identified controls.

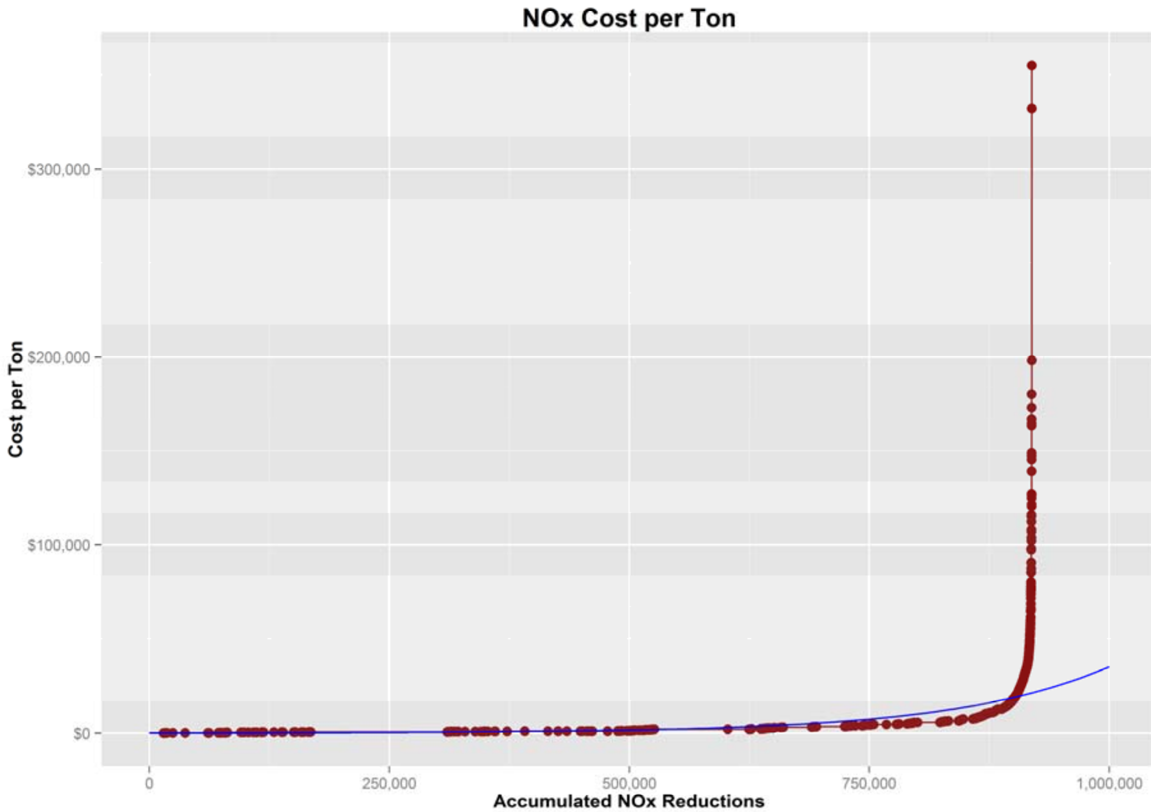

Figure 4-1. Identified Control Cost Curve for 2025 for All Identified NO_x Controls for All Source Sectors (EGU, non-EGU Point, Nonpoint, and Nonroad)

In Section 4.3 we present an average cost-per-ton approach to estimate the costs of achieving any additional NOx emission reductions that may be needed after the application of the identified controls discussed above.[58] That is, we apply a constant, average cost per ton of $15,000/ton to capture total costs associated with the NOx emissions reductions achieved through unidentified controls. The process for determining threshold values for applying identified NOx controls and the determination of a cost for valuing unidentified NOx controls are independent decisions. As discussed earlier, to determine threshold values for applying

[58] We do not apply unidentified VOC control measures in the control strategy analyses.

identified NOx controls, we review the entire data set of potential identified controls and remove likely impractical control applications. The control cost data used in Figure 4-1 reflects the entire data set of potential NOx controls from CoST prior to removing any control applications or applying any thresholds. This raw data has a median control cost of $10,400/ton and an emissions-weighted average cost of $3,000/ton; 97 percent of the emissions reductions from these controls are available at a cost less than $15,000/ton.[59] In addition, the alternative approaches for estimating costs for unidentified controls presented in Appendix 4A generated unit estimates ranging from $2,500/ton to $14,000/ton for a standard of 70 ppb and from $2,800/ton to $14,000/ton for a standard of 65 ppb. Given that both the statistics on the entire data set for identified NOx controls and the results of the alternative approaches for valuing unidentified controls provide costs below $15,000/ton, the decision to value unidentified NOx controls at $15,000/ton is both appropriate and conservative. The value of $15,000/ton captures the potential for unidentified controls to cost both above and below this value. Currently identified controls that may be applied to additional sources would likely cost less than $15,000/ton, while newly developed technologies or technologies that may be developed in the future may cost more than $15,000/ton. The assumption of an average cost of $15,000/ton does not reflect an assumption that all controls will be available at this cost. Rather, it reflects a belief that a mixture of less expensive and more expensive controls will lead to an average cost of $15,000/ton.

In the control strategy analyses, identified VOC controls are applied in the non-EGU point and nonpoint emissions sectors and in (i) fewer locations than identified NOx controls, and (ii) specific locations where the relative effectiveness of VOC controls will have a greater effect on ozone concentrations. For example, in analyzing emissions reductions needed for a standard of 70 ppb, we applied identified VOC controls only in a portion of the Houston buffer region, while we applied identified NOx controls in five larger geographic locations. Because identified VOC controls are generally more expensive than identified NOx controls and are only effective in a limited number of locations, it is reasonable to define a separate and higher cost threshold for applying VOC controls (for a detailed discussion of the contribution of VOC emissions to

[59] In the raw data, the average control cost is $17,800/ton. This average control cost is influenced by a few very high cost control applications that we do not apply in the identified control strategy analyses.

ozone formation, see Chapter 2, Section 2.1 of the November 2014 proposal RIA). We aggregated the raw data on all available identified measures for VOC in the control measures database by cost per ton and plotted an identified control cost curve for VOC controls. The dataset on VOC controls is significantly less robust with approximately 14,000 individual observations and 100 observations when aggregated by cost per ton, and the identified control cost curve revealed a clear point -- $33,000 per ton -- above which costs began increasing at more than an exponential rate. Therefore, we selected $33,000 per ton as the control cost value above which we would not apply additional identified VOC controls. In the control strategy analysis for 70 ppb, there are a total of only six applications in one geographic area (Houston) where identified VOC controls are applied at a cost of $33,000/ton. Figure 4-2 represents the identified control cost curve for all VOC control measures contained in the CoST control measures database, aggregated by cost per ton. As with the NOx identified control cost curve, it is important to note that this curve provides an incomplete representation of the marginal abatement cost curve for all VOC abatement because we do not have information on the control measures and costs for the remaining uncontrolled VOC emissions (see more detailed discussion on incomplete representation of marginal abatement cost curve in section 4.2).

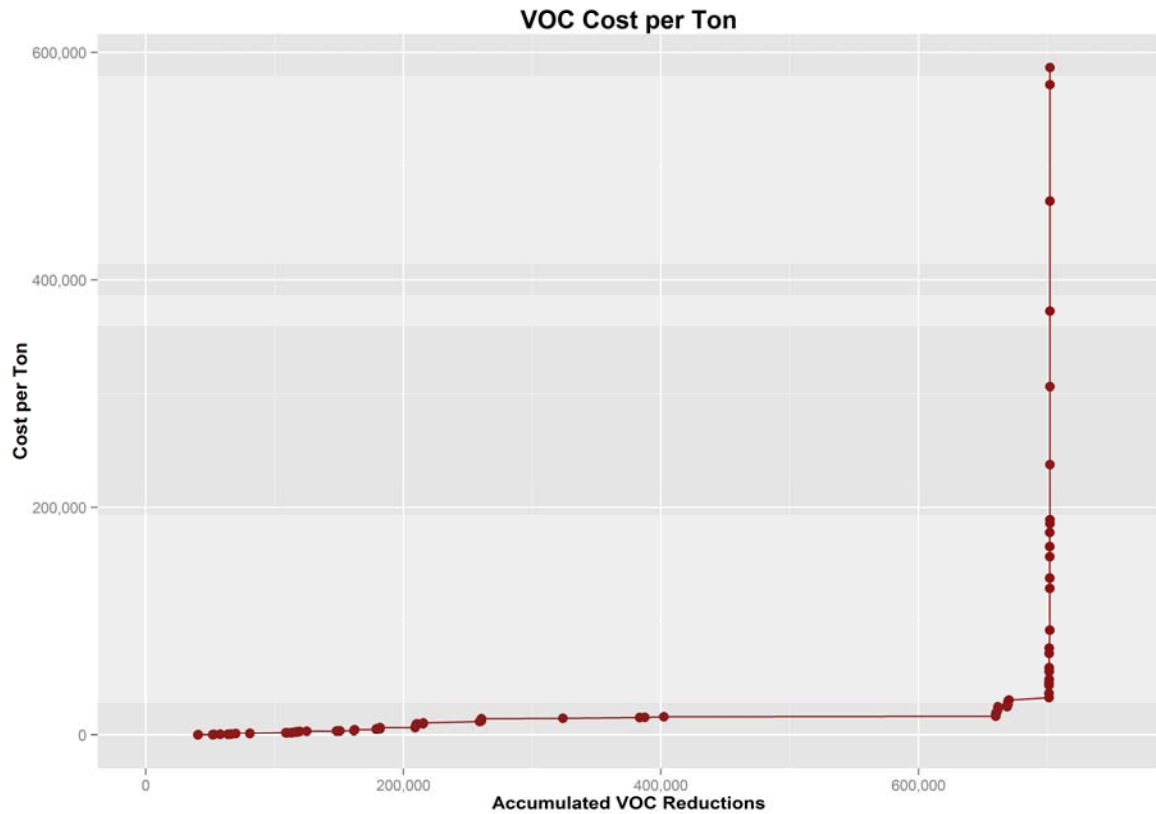

Figure 4-2. Identified Control Cost Curve for 2025 for All Identified VOC Controls for All Source Sectors (EGU, non-EGU Point, Nonpoint, and Nonroad)

4.1.2 Engineering Cost Estimates for Identified Controls

In this section, we provide engineering cost estimates for the identified controls detailed in Chapter 3 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 revised ozone standard of 70 ppb and alternative standard of 65 ppb and additional emission reductions beyond identified controls are needed (unidentified controls).

See Table 4-1 for summaries of control costs from the application of identified controls for the final standard of 70 ppb and an alternative standard of 65 ppb. Costs are listed by sector for both the eastern and western U.S., except California. Note that any incremental costs for identified controls for California (post-2025) for the revised standard of 70 ppb and an alternative standard of 65 ppb are zero because all identified controls for California were applied in the demonstration of attainment for the current standard of 75 ppb (baseline). We aggregate results by region – East and West, except California – to present cost and benefits estimates. See Figure 4.3 for a representation of these regions.

Table 4-1. Summary of Identified Annualized Control Costs by Sector for 70 ppb and 65 ppb for 2025 - U.S., except California (millions of 2011$)[a]

Geographic Area	Emissions Sector	Identified Control Costs for 70 ppb	Identified Control Costs for 65 ppb
		7 Percent Discount Rate[b]	7 Percent Discount Rate[b]
East	EGU	52[c]	130[c]
	Non-EGU Point	260[d]	750[d]
	Nonpoint	360	1,500
	Nonroad	13[e]	36[e]
	Total	**690**	**2,400**
West	EGU	-	-
	Non-EGU Point	4[d]	49[d]
	Nonpoint	<1	88
	Nonroad	-	4[e]
	Total	**4**	**140**
Total Identified Control Costs		**690**	**2,600**

[a] All values are rounded to two significant figures.
[b] The numbers presented in this table reflect the engineering costs annualized at a 7 percent discount rate, to the extent possible.
[c] EGU sector control cost data is calculated using a capital charge rate between 7 and 12 percent for retrofit controls depending on the type of equipment.
[d] A share of the non-EGU point source sector costs can be calculated using both 3 and 7 percent discount rates. When applying a 3 percent discount rate where possible, the total non-EGU point source sector costs are $250 million for 70 ppb and $740 million for 65 ppb.
[e] Nonroad sector control cost data is calculated using a 3 percent discount rate.

The total annualized engineering costs associated with the application of identified controls, using a 7 percent discount rate, are approximately $690 million for the final annual standard of 70 ppb and $2.6 billion for a 65 ppb alternative standard. Table 4-2 below provides summary statistics by emissions source category of the NOx and VOC control cost data from the identified control strategy for the revised standard of 70 ppb.[60] The costs of NO_x controls, in terms of dollars per ton of NO_x reduction for the standards analyzed were approximately $1,200/ton on average for the EGU sector.[61] The costs of NOx controls were $2,600/ton for the non-EGU point sector on average, with a range of $0/ton to $19,000/ton, a median of $960/ton, and an emissions weighted average of $2,800/ton; $760/ton for the nonpoint sector on average, with a range of $0 to $2,000/ton, a median of $970/ton, and an emissions weighted average of $1,000/ton; and $4,600/ton for the nonroad sector on average, with a range of $3,300/ton to $5,300/ton, a median of $4,600/ton, and an emissions weighted average of $4,500/ton. The costs of VOC controls in terms of dollars per ton of VOC reduction for the standards analyzed were approximately $11,000/ton for the non-EGU point sector on average, with a range of $1,200/ton to $25,000/ton, a median of $9,800/ton, and an emissions weighted average of $8,100/ton; and $11,000/ton for the nonpoint sector on average, with a range of $24 to $33,000/ton, a median of $15,000/ton, and an emissions weighted average of $14,000/ton

Table 4-2. NO_x and VOC Control Costs Applied for 70 ppb in 2025 – Average, Median, Minimum, Maximum, and Emissions Weighted Average Values ($/ton)[a]

Emissions Sector	Average Cost/Ton	Median Cost/Ton	Minimum Cost/Ton	Maximum Cost/Ton	Emissions Weighted Average Cost/Ton
NO_x Controls					
EGU	1,200	1,200	1,200	1,200	1,200
Non-EGU Point	2,600	960	0	19,000	2,800
Nonpoint	760	970	0	2,000	1,000
Nonroad	4,600	4,600	3,300	5,300	4,500
VOC Controls					
Non-EGU Point	11,000	9,800	1,200	25,000	8,100
Nonpoint	11,000	15,000	24	33,000	14,000

[a] The numbers presented in this table reflect the engineering costs annualized at a 7 percent discount rate to the extent possible. EGU control cost data is calculated using a capital charge rate between 7 and 12 percent for retrofit controls depending on the type of equipment. Nonroad control cost data is calculated using a 3 percent discount rate.

[60] Across all of the data in the control strategy analysis for a standard of 70 ppb, the average control cost is $5,000/ton and the emissions-weighted average cost is $2,000/ton.

[61] After accounting for the Clean Power Plan in the Baseline (see Chapters 2 and 3), remaining EGUs not affected by the Clean Power Plan were plants where NO_X controls existed, but had not been dispatched. This dollar per ton value represents the average operation and maintenance cost of running such controls.

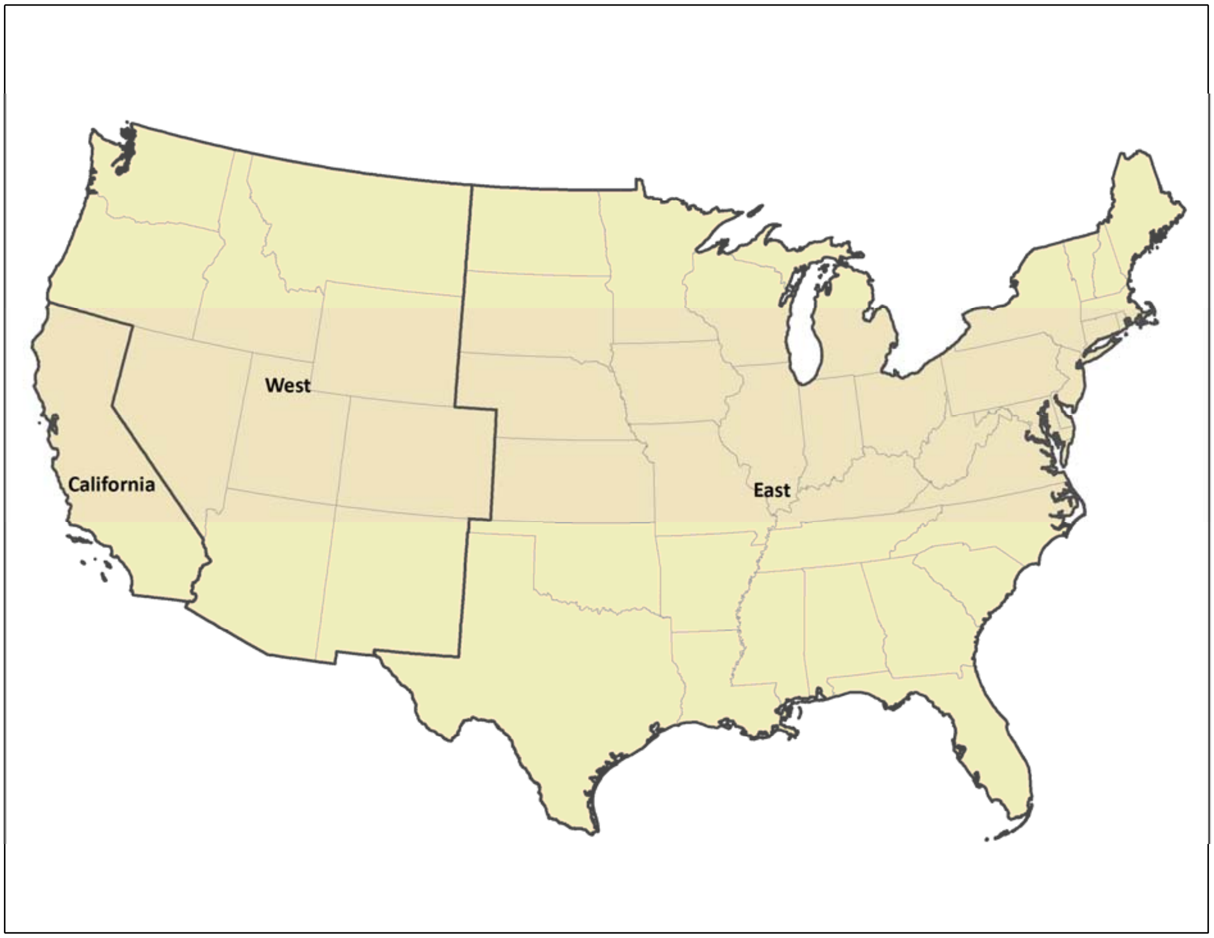

Figure 4-3. Regions Used to Present Emissions Reductions and Cost Results

The numbers presented in Tables 4-1 and 4-2 reflect the engineering costs annualized at the different discount rates discussed below and include 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.[62] A higher discount, or interest, rate results in a larger annualized cost of capital

[62] In the cost 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.
In benefits analyses, the discount rate is used to discount benefits that occur in time periods after the year in which emissions reductions take place. As a result, the way the discount rate is used in the cost analysis is different from the way it is used in the benefits analysis. For an explanation of the benefits calculations, see Chapter 6. In both cases, the values at different discount rates do not indicate that the value is the present value of a stream of annualized benefits or costs.

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.[63] If disaggregated control cost data is not available (i.e., where capital, equipment life value, and O&M costs are not separated out and where we only have a $/ton value), EPA 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 separated out) we can and do recalculate costs using a 3 percent discount rate. For the engineering costs provided in this analysis, we estimate costs for the sectors as follows:

- **For EGU controls**, the annualized EGU control costs were not estimated for either 3 or 7 percent. This is due to the complexity of investment decisions in the EGU sector. Decisions about investments in control equipment are not uniform across the sector, are made in different time frames, with different loan rates and thus, ultimately different capital recovery factors. Equipment pay off times, depreciation rates and capacities that factor into the capital charge rate vary. According to the IPM v5.13 documentation (U.S. EPA, 2013 Chapters 5 and 8), capital charge rates can vary from 7 percent to 12 percent depending on the type of equipment. See the IPM v5.13 documentation cited for a more in depth discussion. EGU control costs represent 8 percent and 5 percent of the compliance cost estimates for identified controls for the final standard of 70 ppb and an alternative standard level of 65 ppb, respectively.

- **For non-EGU point source controls**, some disaggregated data are available, and we were able to calculate costs at both 3 and 7 percent discount rates for those controls. For the final and alternative standards analyzed in this RIA, approximately 29 and 24 percent, respectively, of identified control costs for non-EGU point sources are disaggregated at a level that could be recalculated at a 3 percent discount rate. Non-EGU point source control costs represent 38 percent

[63] Data sources can include state and technical studies, which frequently do not include the original data source.

and 31 percent, respectively, of the compliance cost estimates for identified controls for the final standard of 70 ppb and alternative standard level of 65 ppb.

- **For nonpoint source controls**, because we do not have disaggregated control cost data total annualized costs for these sectors are assumed to be calculated using a 7 percent discount rate. Nonpoint source control costs represent 53 percent and 62 percent, respectively, of the compliance cost estimates for identified controls for the final standard of 70 ppb and an alternative standard level of 65 ppb.

- **For nonroad mobile source controls**, the cost estimates for control of emissions from nonroad diesel engines are prepared using a net present value (NPV) approach, which is different from the approach applied for other sources whose emissions are controlled in the illustrative control strategies applied in the RIA (U.S. EPA, 2007). To be consistent with the engineering cost estimates for other emissions sources, we would need to use the EUAC method to calculate control costs for nonroad diesel engines. To use the EUAC method we need information on the portion of annual costs that is from the annualization of the original capital expense for these nonroad controls and the portion that is from annual operation and maintenance. The cost estimates for the nonroad diesel engine retrofit controls did not include estimates for operating costs, and we do not have sufficient information to determine if the annual cost estimates reflect only capital costs. As a result, we are unable to estimate annual costs at interest rates of 3 percent and 7 percent for these controls. The nonroad diesel engine retrofit costs are estimated using a 3 percent interest rate.[64] Nonroad mobile source control costs represent 2 percent of the compliance cost estimates for identified controls for the final standard of 70 ppb and an alternative standard level of 65 ppb.

[64] The capital recovery factor, used to convert capital costs to annual costs, requires both an interest rate and an equipment life. While we do have the expected lifetime for these controls, we are not able to estimate these costs at a different interest rate using the EUAC based on the lack of annualized capital cost data.

Table 4-3 summarizes the discount rates discussed above and the percent of total identified control costs for each emissions sector for the final standard of 70 ppb and an alternative standard of 65 ppb. Because we do not have a full set of costs at the 3 percent discount rate or the 7 percent discount rate and because we believe the majority of the identified control costs is calculated at a 7 percent discount rate, Table 4-1 presents engineering cost estimates based on a 7 percent discount rate.

Table 4-3. By Sector, Discount Rates Used for Annualized Control Costs Estimates and Percent of Total Identified Control Costs

Emissions Sector	Discount Rate	Percent of Total Identified Control Costs for 70 ppb	Percent of Total Identified Control Costs for 65 ppb
EGU	7 – 12%	8	5
Non-EGU Point	3 and 7%	38	31
Nonpoint	7%	53	62
Nonroad	3%	2	2
Total		100%	100%

4.2 The Challenges of Estimating Costs for Unidentified Control Measures

Some areas are unable to attain the revised and alternative levels of the standard using only identified controls. In these areas, it is necessary to assume the application of currently unidentified control measures to estimate the full cost of attaining the standards analyzed. The EPA's application of unidentified control measures does not mean the Agency has concluded that all unidentified control measures are currently not commercially available or do not exist. Unidentified control technologies or measures can include existing controls or measures for which the EPA does not have sufficient data to accurately estimate engineering costs. Likewise, the control measures in the CoST database do not include abatement possibilities from energy efficiency measures, fuel switching, input or process changes, or other abatement strategies that are non-traditional in the sense that they are not the application of an end-of-pipe control. In addition, there will likely be some emissions reductions from currently unidentified control technologies as a result of state-specific rules that are not in the future year baseline emissions projections or are not yet finalized. See the discussion in Section 4.2.3 for examples of existing control measures for which the EPA does not have sufficient data to estimate engineering costs, as well as state-specific rules that are not in the future year baseline emissions projections.

The EPA's application of unidentified control measures does reflect the Agency's experience that some portion of controls to be applied in the future may not be currently available but will be deployed or developed over time. The EPA believes that a portion of the estimated emissions reductions needed to comply with a revised standard can be secured through future technologies, national regulatory programs, and/or state regulatory programs or measures for which information is either not currently complete or not currently available. As an example, in the 1997 ozone NAAQS RIA, NOx emissions reductions that were estimated from the mobile source Tier 2 standards were not considered as part of the "known" controls, even though the RIA acknowledged the potential for these mobile source standards to provide substantial cost-effective controls and emissions reductions. While in 1997 these emissions reductions were considered to come from "unknown", or unidentified, controls, in retrospect, they were achieved through mobile source controls. Looking forward, the EPA estimates that the Phase 2 of the Heavy Duty Greenhouse Gas Standards for New Vehicles and Engines[65] will provide additional NOx emissions reductions.[66]

The remainder of this section presents and discusses various factors that should be considered when estimating the costs of applying costs to emission reductions from unidentified control measures for the future using only information on a limited set of today's available control technologies or measures. We start with discussions about the role of technological innovation and change from the economics literature: Section 4.2.1 discusses the impact of technological innovation and diffusion on available control technologies; and Section 4.2.2 presents information on improvements in control technologies over time through learning by doing.

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[65] Greenhouse Gas Emissions Standards and Fuel Efficiency Standards for Medium- and Heavy-Duty Engines and Vehicles (Phase 1 of the Heavy Duty Greenhouse Gas Standards for New Vehicles and Engines) was included in the 2025 base case (see Chapter 1, Section 1.3.1 for a list of rules in the base case).

[66] The focus of the Phase 1 (76 FR 57106, September 15, 2011) and Phase 2 (80 FR 40138, July 13, 2015) Heavy Duty GHG rules is to reduce GHG emissions and fuel consumption, but there can also be NOx reductions that stem largely from a switch in using the on-road engine to using an auxiliary power unit (APU) during extended idling. Because the Heavy Duty GHG standards are performance-based and manufacturers can choose their own mix of technologies to meet the standards, the standards provide an incentive for APU use but do not require it. Thus, the impact on NOx emissions depends on the assumptions and projections for APU use. The EPA expects increased APU usage would result from the Phase 2 rule. After considering the revised APU projections, the EPA estimates that the two Heavy Duty GHG rules combined would reduce NOx emissions by up to 120,000 tons in 2025 and 450,000 tons in 2050.

Following the sections on the role of technological innovation and change, we include the following discussions related to limitations in the currently available information on traditional end-of-pipe technologies or measures and on projecting the future air quality problem being analyzed: Section 4.2.3 discusses the incomplete characterization of the supply of available control technologies and why the abatement supply curve from identified controls presented in the previous section provides an incomplete picture of all currently available pollution abatement opportunities; Section 4.2.4 discusses how over time as EPA reviews NAAQS standards, relevant information about future year baseline emissions and possible control technologies is revealed in the current RIA development process that was not available to analysts for previous RIAs; and Section 4.2.5 includes information on how NOx offset prices and Section 185 fees could serve as reasonable proxies for the costs associated with emissions reductions from unidentified controls. Finally, we describe how we use this information to help inform the unidentified control cost methodology applied in section 4.3.

4.2.1 Impact of Technological Innovation and Diffusion

In general, the marginal abatement cost curve (MACC) at any particular point in time for a defined set of emitting sectors will be an increasing function of the level of abatement.[67] That is, marginal costs are increasing as the amount of emissions are reduced. However, it is important to note that the MACC is not just the relationship between marginal cost and abatement, but also should be constructed as the envelope of least cost approaches for any given level of abatement. As previously noted, the identified control cost curve derived from data in CoST may include measures that may not be the most cost-effective way of achieving the emissions reduction, and as a result the cost curve derived using that data may not represent the complete MACC. The aggregated MACC is the horizontal summation of individual firms/sectors marginal abatement curves, and is generally thought to reflect the overall marginal and total abatement costs when a least cost approach is implemented. This aggregate MACC gives the efficient MAC level for each firm/sector for any aggregate emissions target for a given time period. However, the MACC represents the efficient MAC level only under some fairly restrictive conditions, including 1) all abatement opportunities across all sectors and locations have been identified and

[67] The marginal abatement cost curve (MACC) is a representation of how the marginal cost of additional emissions abatement changes with increasing levels of abatement.

included in the cost curve, and 2) information about applicability of controls is available with no uncertainty. In addition, the MACC for a current time period will only hold for a future time period if no technical change (either introduction of new technologies or reduction in cost of existing technologies) or learning by doing occurs between the present and future time periods.

In regulatory analyses of NAAQS, we typically assess costs of abatement in a future year or years selected to represent implementation of the standards. The focus has typically been on the application of existing technologies and the evolution of those technologies over time rather than on innovations that may lead to development of new pollution control technologies. As such, a MACC constructed based on currently available information on abatement opportunities will not be the best representation of a future MACC. A future MACC will likely reflect 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). Because we are unable to predict technological advances that may occur in the future, the discussion in this section focuses on the advances that might be expected in existing pollution control technologies.

Technological innovation and diffusion can affect the MACC in several ways. Some examples of the potential effects of technical change are:

1. New control technologies may be developed that cost less than existing technologies.
2. A new control technology may be developed to address an uncontrolled emissions source.
3. The efficiency of an existing control measure may increase. In some cases, the control efficiency of a measure can be improved through technological advances.
4. The cost of an existing control measure may decrease.
5. The applicability of an existing control measure to other emissions sources may increase.

Overall, these five examples describe ways that technological change can reduce both the amount of unidentified abatement needed, shift the MACC, decrease the MAC, decrease average costs, and decrease total costs relative to the case where it is assumed that the current MACC reflects all possible abatement opportunities both in the present and future. It is also possible in cases where there is a strictly binding emissions reduction target that new control 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 may not be adopted (see Section 4.2.5 for a brief discussion of Section 185 fees).

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 examples 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:

1. Selective catalytic reduction (SCR) and ultra-low NOx burners for NOx emissions;

2. Scrubbers that achieve 95 percent or greater SO_2 control on boilers;

3. Sophisticated new valve seals and leak detection equipment for refineries and chemical plants to reduce VOC and HAP emissions;

4. Low or zero VOC paints, consumer products and cleaning processes;

5. Chlorofluorocarbon (CFC) free air conditioners, refrigerators, and solvents;

6. Water and powder-based coatings to replace petroleum-based formulations to reduce VOC and HAP emissions;

7. Vehicles with lower NOx 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;

8. Idle-reduction technologies for engines to reduce NOx and $PM_{2.5}$ emissions, including truck stop electrification efforts; and

9. Improvements in gas-electric hybrid vehicles and cleaner fuels to reduce NOx emissions.

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 NOx. 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.

4.2.2 Learning by Doing

As experience is gained in the application of control technologies or pollution control practices, firms learn how to operate the controls more efficiently and learn how to apply controls to additional sources. 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. Such impacts have been identified to occur in a number of studies conducted for various production processes. These impacts 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) discuss how the cost of control technologies can decline over time using the example of post-combustion SO_2 and NOx 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. The 1997 Ozone NAAQS RIA includes information on 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. Other discrete examples include the dramatic 85 percent decline in prices of the catalyst used in operating SCR between 1980 and 2005 (Cichanowicz, 2010). In addition, analyses performed for the 2008 Ozone NAAQS RIA found controls originally developed for one source type were being applied to new source categories. For example, SCR, originally developed for use by EGUs, is now used in the cement manufacturing sector, and SNCR is now pertinent to a large number of additional boiler source categories. In some cases, these newly found controls proved to be more effective than what had been applied in the past. 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. These examples serve as evidence of a learning effect – production and implementation costs decrease as learning and repetitive use occurs.

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. The magnitude of learning curve impacts on pollution control costs was 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.[68] In the Report, learning curve adjustments were included for those sectors and technologies for which learning curve data were 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

[68] 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.

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.

Learning by doing can reduce costs in a number of ways: through the reduction of operating and maintenance costs, finding new ways to use existing technologies, etc. 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 and resources to properly generate control costs that reflect learning curve impacts.

4.2.3 Incomplete Characterization of Available NO_x Control Technologies

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, or abatement, 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.[69] Although exogenous factors, such as changes in economic conditions, may have 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.

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[69] 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.

Underlying the selection of controls described in Appendix 3A is the concept of the MACC. Adding newly developed control technologies, or changing either the abatement amount or cost of the technology, will change the shape of the overall MACC. The engineering cost estimates in section 4.1 are estimated primarily from end-of-pipe controls and only included limited process-oriented control measures, such as switching to lower-emitting fuel or energy sources and installing energy efficiency measures. As a result, the MACC derived in the previous section from identified controls represents an incomplete supply curve that only partially captures the abatement supply. An illustrative depiction of an "observed but incomplete" MACC and the complete underlying MACC is presented below in Figure 4-4. 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 increases the supply of abatement.

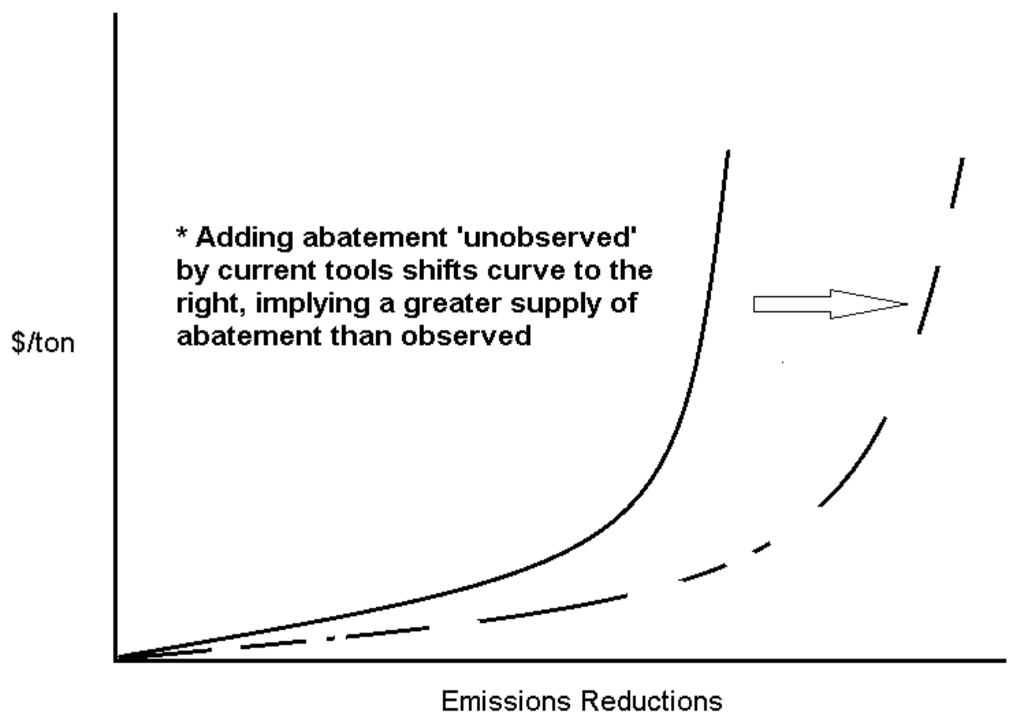

Figure 4-4. Observed but Incomplete MACC (Solid Line) Based on Identified Controls in Current Tools and Complete MACC (dashed line) where Gaps Indicate Abatement Opportunities Not Identified by Current Tools

Because of the incomplete characterization of the full range of NOx abatement possibilities, it is important to understand the composition of the cost information EPA has available and uses to construct the partial MACC. The nature of available information on the cost of NOx abatement measures is somewhat complex. EPA's control strategy tools undergo continuous improvement, and as the need for additional abatement opportunities increases, additional evaluation of uncontrolled emissions takes place. During these evaluations, additional abatement opportunities from applying identified controls typically are found. These abatement opportunities or additional controls are added to the CoST database and will be available for future analyses. In addition, in some cases we may have specific knowledge of potential additional control measures due to an impending regulation (e.g., Tier 3), but until a regulation is finalized those identified controls are not included in any concurrent analyses.

It is also important to understand that EPA's control strategy tools largely focus on end-of-pipe controls and a limited set of emissions inventory sectors, whereas opportunities for emissions reductions through non-end-of-pipe controls or measures exist. For example, we reviewed the existing control strategies indicated in the SIP for the Dallas-Fort Worth area for the 1997 ozone NAAQS, and we compared the strategies and measures in that SIP to the measures the EPA analyzed in the 1997 ozone NAAQS regulatory impact analysis. The EPA analyzed several industrial source categories and measures that were reflected in the 1997 Dallas-Fort Worth SIP, including existing control measures for stationary sources such as cement kilns, industrial boilers, iron and steel mills, as well as enhanced inspection and maintenance programs for mobile sources.[70] The Dallas-Fort Worth SIP recognized the need for additional control strategies and measures to achieve further emissions reductions – strategies and measures that were not reflected in EPA's 1997 control strategy analysis. These additional control measures included transportation control measures, additional voluntary mobile emission reduction programs, and energy efficiency/renewable energy measures. Table 4-4 below includes examples of each of these types of programs or measures.

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[70] The Dallas-Fort Worth SIP also reflected the following existing voluntary mobile emission reduction programs: alternative fuel vehicle program; employee trip reduction program; and vehicle retirement program. Information on the Dallas-Fort Worth SIP is available at http://www.nctcog.org/trans/air/sip/future/lists.asp.

Table 4-4. Control Measures in Dallas-Fort Worth SIP Not Reflected in the 1997 Ozone NAAQS RIA

Transportation Control Measures	
	Bicycle/Pedestrian Projects
	Grade Separation Projects
	HOV/Managed Lane Projects
	Intersection Improvement Projects
	Park and Ride Projects
	Rail Transit Projects
	Vanpool Projects
Voluntary Mobile Source Emission Reduction Programs	
	Clean Vehicle Program
	Employee Trip Reduction
	Locally Enforced Idling Restriction
	Diesel Freight Idling Reduction Program
Other State and Local Programs: Energy Efficiency/Renewable Energy	
	Residential Building Code
	Commercial Building Code
	Federal Facilities Projects
	Political Subdivision Projects[71]
	Electric Utility-Sponsored Programs[72]
	Wind Power Projects
Additional Measures	
	Clean School Bus Program
	Texas Low Emission Diesel
	Stationary Diesel and Dual-Fired Engine Control Measures

Further, Table 4-5 includes specific non-end-of-pipe control measures from approved SIPs in Texas and Louisiana, including measures from the Dallas-Fort Worth SIP (i.e., energy efficiency measures). The approved, non-end-of-pipe control measures include local transportation measures, local building energy efficiency requirements, and mobile source sector measures. In addition, Table 4-6 includes examples of approved non-end-of-pipe control measures in California. California is also currently developing the following additional measures:

[71] These projects are typically building system retrofits, non-building lighting projects, and other mechanical and electrical systems retrofits, such as municipal water and waste water treatment systems.
[72] These programs include air conditioner replacements, ventilation duct tightening, and commercial and industrial equipment replacement.

- Encouraging Use of Warm Mix Asphalt over Hot Mix Asphalt - European and American companies have developed several techniques, collectively known as warm-mix asphalt (WMA), to increase the workability of asphalt by lowering the viscosity at temperatures as much as 100°F below that of hot-mix asphalt (HMA). WMA was introduced in Europe in 1997 and in the United States in 2002. WMA has shown potential for reducing emissions associated with the production of asphalt for paving projects when compared to HMA. Lower temperatures required for production, storage, transport, and application translates to lower fuel consumption, which in turn reduces the criteria air pollutant emissions associated with combustion.[73]

- Replacement of gas-powered leaf blowers and mowers – The South Coast Air Quality Management District has a program that subsidizes the replacement of existing two-stroke backpack blowers currently used by commercial landscapers/gardeners with new four-stroke backpack blowers that have significantly reduced emissions and noise levels.[74]

All of these additional and non-end-of-pipe measures and associated emissions reductions are not reflected in EPA's control strategy tools and represent additional abatement opportunities not accurately captured in this analysis.

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[73] For additional information see http://www.fhwa.dot.gov/everydaycounts/technology/asphalt/intro.cfm ; slide 31 aci-na.org/static/entransit/sunday_warmmix_logan.pdf. FHWA estimates that warm mix asphalt uses 20 percent less energy than hot mix asphalt. Also see http://www.fhwa.dot.gov/everydaycounts/technology/asphalt/intro.cfm. Airports Council International reports 10-55 percent lower energy consumption and 20-55 percent reduction in emissions. (See http://aci-na.org/static/entransit/sunday_warmmix_logan.pdf, Slide 31)).

[74] Typically, ten exchange events are set up across the District, and for the convenience of the participants, the exchange events take place during consecutive weekdays. At the event site, the old leaf blowers will be tested for operation and then drained of all fluids in a responsible manner and collected for scrapping. The vendor will haul the traded-in blowers to a scrapping yard where they are crushed and recycled. The vendor will also provide training for the proper use of the equipment at each of the exchange sites. (http://www.aqmd.gov/docs/default-source/Lawn-Equipment/leafblower-brochure.pdf?sfvrsn=9)

Table 4-5. Non-End-of-Pipe Control Measures from SIPs

Measure	Projected Emissions Reductions (tons per day)	Area	Citation
Energy Efficiency[75]	0.04 (NOx)	Shreveport/Bossier City Area, Louisiana	70 FR 48880 August 22, 2005
Energy Efficiency[76]	0.72 (NOx)	Dallas/Fort Worth (DFW), Texas	73 FR 47835 August 15, 2008
Transportation Emission Reduction Measures[77]	0.72 (NOx) 0.83 (VOC)	Austin - Early Action Compact Area, Texas	70 FR 48640 August 19, 2005
Voluntary Mobile Emission Reduction Program (VMEP)	2.63 (NOx) 0.61 (VOC)	DFW area	74 FR 1906 January 14, 2009
Texas Emission Reduction Plan (TERP)[78, 79]	14.2 (NOx)	DFW area	74 FR 1906 January 14, 2009
Texas Low Emission Diesel (TxLED)[80]	Up to 6 (NOx); on-going and varies annually	East and Central Texas (includes DFW area)	79 FR 67068 November 12, 2014

[75] The measures involved installing energy conservation equipment in 33 city buildings. Measures included upgrades to the lighting, mechanical and control systems, water conservation upgrades, and other miscellaneous activities. This was an Early Action Compact (EAC) area SIP.

[76] The NOx emissions reductions in the DFW area were due to energy efficiency measures in new construction for single and multi-family residences. This measure was initially submitted within a SIP to provide emissions reductions of 5 percent in the DFW area; actual reductions fell short of the 5 percent goal, but this measure was eventually approved with other measures that, by providing additional emissions reductions, improved air quality in the area.

[77] The transportation projects were to reduce vehicle use, improve traffic flow, and/or reduce congested conditions throughout the Austin EAC area (this was an EAC SIP). The Austin EAC area included Bastrop, Caldwell, Hays, Travis, and Williamson counties and the cities of Austin, Bastrop, Elgin, Lockhart, Luling, Round Rock, and San Marcos.

[78] TERP is a discretionary economic incentive program: economic incentives to reduce emissions. The approved TERP is a grant program, unique to Texas, that provides funds through the Texas Commission on Environmental Quality in a variety of categories, including emissions reduction incentive grants, rebate grants (including grants for small businesses), and heavy and light duty motor vehicle purchase or lease programs, all with the goal of improving air quality in Texas. Examples of TERP programs include assisting small businesses in purchasing lower-emission diesel vehicles, helping school districts to reduce emissions from school buses, and providing funds to support research and development of pollution-reducing technology. TERP is available to all public and private fleet operators that operate qualifying equipment in any of the ozone nonattainment counties within Texas. TERP was also approved into the Texas EAC SIPs, providing at least 2 tons per day in NOx reductions in each of three areas (Austin, Tyler/Longview, and San Antonio).

[79] The State of Texas reports on the TERP every other year and the emissions reductions cited here were based on an estimated $6,000 per ton.

[80] The TxLED fuel program was initially approved by EPA on November 14, 2001 (66 FR 57196) and has undergone subsequent revisions, the latest on May 5, 2013 (78 FR 26255). TxLED fuel is required for use by on-highway vehicles and non-road equipment (including marine vessels) in 110 counties in eastern and central Texas. Use of this boutique fuel reduces NOx emissions.

Table 4-6. Non-End-of-Pipe Measures in California

Program or Standard	Area	Citation	Estimated Emissions Reductions
Carl Moyer Memorial Air Quality Standards Attainment Program (*selected project types*)*	San Joaquin Valley, California	79 FR 29327 (May 22, 2014)	3.78 tpd NOx credited in San Joaquin Valley
Proposition 1B: Goods Movement Emission Reduction Program (*selected project types*)*	San Joaquin Valley, California	79 FR 29327 (May 22, 2014)	1.23 tpd NOx credited in San Joaquin Valley
California Reformulated Gasoline (RFG) Program	California (statewide)	60 FR 43379 (August 21, 1995), revised 75 FR 26653 (May 12, 2010)	
California Diesel Fuel Program	California (statewide)	60 FR 43379 (August 21, 1995), revised 75 FR 26653 (May 12, 2010)	
Regional Clean Air Incentives Market (RECLAIM)	South Coast, California	63 FR 32621 (June 15, 1998), revised 71 FR 51120 (August 29, 2006) and 76 FR 50128 (August 12, 2011)	

* Program not approved into SIP but relied upon for emission reduction credit through state commitment.

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. However, other sectors may not be as well-controlled, and lower cost controls may be available.

4.2.4 Comparing Baseline Emissions and Controls across Ozone NAAQS RIAs from 1997 to 2014

Many factors affect the future year baseline emissions used in a NAAQS analysis, including the future year being analyzed, the projected air quality in that year, the emissions inventories used, emissions projections methodologies, and any federal and/or state regulatory programs or measures that are reflected in the emissions projections. The EPA believes that while these factors and changes are difficult to track individually, additional federal and/or state regulatory programs promulgated and the new data on available control technologies or measures can, over time, result in emissions reductions and control measures being reclassified from unidentified to identified measures. Each ozone NAAQS analysis since 1997 has required at least some emissions reductions from controls that were considered unidentified at the time of analysis, but evidence indicates that over time new information becomes available that changes the characterization of these emissions reductions from unidentified measures to identified measures. For example, in the 1997 ozone NAAQS RIA, NOx emissions reductions that were expected to result from the *at that time upcoming* mobile source Tier 2 standards were not characterized as resulting from identified controls, even though the RIA acknowledged the potential for these standards to provide substantial cost-effective controls and emissions reductions. As a result, in 1997 these cost-effective emissions reductions were considered to be from unidentified controls, while in retrospect they were actually from identified controls. Likewise, the 2008 ozone NAAQS RIA did not include controls on EGUs that would later be predicted to result from the Mercury and Air Toxics Standards or the Clean Power Plan. As a result, in 2008 emissions reductions needed from unidentified controls were estimated to be higher in some regions of the U.S. than those estimated in this RIA. In general, during the time between the promulgation of a NAAQS and the required date of attainment, additional rules may be developed and additional analyses performed that shed light on how emissions reductions that were once thought to be unavailable from identified control measures are obtained through tangible means. Improvements over time, both in information and engineering, lead to an increase in identified controls and as a result emission reductions obtained only through unidentified controls in one analysis may be realized through identified controls in subsequent analyses.

4.2.5 Possible Alternative Approaches to Estimate Costs of Unidentified Control Measures

In determining how to estimate the costs of achieving the emission reductions needed from unidentified control measures (see Section 4.3), we examined what information could be gleaned from existing regional NOx offset prices. 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 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.[81] 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*.[82] Lastly, we collected information on NOx offset prices in the Houston-Galveston nonattainment area for 2010 through 2013 from the *Trade Report*[83] and the New York-New Jersey-Connecticut nonattainment area from 2000 through 2013 from industry representatives.

Table 4-7 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, and represent a one-time payment or cost, not an annual payment or cost. The prices constitute average of the trades in the regions for the year given. The data series for the California regions are more complete than those for Houston and the New York region.

[81] http://www.arb.ca.gov/nsr/erco/erco.htm
[82] http://www.aqmd.gov/home/programs/business/about-reclaim/reclaim-trading-credits
[83] http://www.tceq.state.tx.us/airquality/banking/mass_ect_prog.html

Table 4-7. Average NO_x Offset Prices for Four Areas (2011$, perpetual tpy)[a]

	NOx Offset Prices ($/ perpetual tpy)			
	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.

To more directly compare offset prices to potential annual costs for unidentified emissions controls, we annualized the perpetual, 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 cost by using the capital recovery factor (CRF) discussed in Section 4.1.1. 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 4-8 presents the average and maximum annualized NO_x offset prices in 2011 dollars.

Table 4-8. Annualized NOx Offset Prices for Four Areas (2011$, tons)[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 since the purchaser of the offset chose to purchase the offset rather than curtail their business activities or purchase some other pollution control technology. It is possible that these offset prices could serve as reasonable proxies for the costs associated with emissions reductions from unidentified measures or controls. The cost information informing the identified control strategy traces out an incomplete marginal abatement cost curve in that, as discussed in Chapter 3, the controls used in the identified control analysis are primarily end-of-pipe technologies. The identified control estimates for NO_x do not account for other forms of abatement, e.g., switching to lower emitting fuels or increasing energy efficiency. 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 emissions and air quality profiles. In each region, offset supply is a function of the emissions inventory and offset demand is a function of economic growth, and neither offset supply nor demand is infinite.

An alternative proxy for estimating the costs of unidentified control measures is using the current annualized section 185 fee rate. The section 185 fee program requirement applies to any ozone nonattainment area that is classified as Severe or Extreme under the NAAQS. If a Severe or Extreme nonattainment area fails to attain the ozone NAAQS by the required date, section 185 of the Clean Air Act requires each major stationary source of VOC and NOx located in such area to pay a fee to the state for each calendar year following the attainment year for emissions above a baseline amount and until the area reaches attainment.[84,85] The fee was set in the 1990 Clean Air Act at $5,000 per ton of VOC and NOx emissions above the baseline amount and is adjusted annually for inflation based on the Consumer Price Index. The 2013 annualized section 185 fee rate was $9,398.67 per ton. Examples of states or areas that have adopted section 185 fee programs include: (a) Texas for the Houston-Galveston nonattainment area, which adopted its fee program in May 2013,[86] and (b) the South Coast Air Quality Management District, which amended Rule 317 that governs its fee program in February 2011.

4.2.6 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.[87] In addition, while we examined other approaches to estimate the costs of unidentified measures, we concluded that these approaches could potentially undervalue the costs of unidentified controls. As the EPA continuously improves its data and tools, we expect to better characterize the currently unobserved pieces of the MACC.

4.3 Compliance Cost Estimates for Unidentified Emissions Controls

This section presents the methodology and results for the costs of emissions reductions from unidentified control measures needed to demonstrate full attainment of the revised and alternative standards analyzed. We refer to the costs of emissions reductions from unidentified

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[84] For additional information on developing fee programs required by Clean Air Act Section 185, see the January 5, 2010 memorandum from the EPA's Office of Air Quality Planning and Standards, available at: http://www.epa.gov/ttn/naaqs/aqmguide/collection/cp2/bakup/20100105_page_section_185_fee_programs.pdf.

[85] In 1990, the Clean Air Act set the fee at $5,000/ton of VOC and NOx emitted by the source during the calendar year in excess of 80 percent of the baseline amount. A source's baseline amount is the lower of the amount of actual or allowable emissions under the permit for the source during the attainment year.

[86] Additional information on Texas's actions is available at http://www.tceq.texas.gov/airquality/point-source-ei/sipsection185.html

[87] See Chapter 3, Section 3.4 for additional discussion of uncertainties associated with predicting technological advancements that may occur between now and 2025.

controls as unidentified control costs.[88] As discussed in Chapter 3, the application of the identified control strategies was not sufficient in reaching full, nationwide attainment of the revised standard of 70 ppb and the alternative standard of 65 ppb analyzed. Therefore, the engineering costs detailed in Section 4.1 represent only the costs of partial attainment.

4.3.1 Methods

On the issue of estimating the costs of unidentified control measures, in 2007 the EPA's Science Advisory Board offered the following advice:

> *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 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.*[89]

While we have considered alternative methodologies to predict future abatement supply curves, we are currently unable to quantitatively predict future shifts in the abatement 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: a "fixed cost" approach, following the SAB advice, and a "hybrid" approach that has not yet

[88] In previous analyses, these costs were referred to as extrapolated costs.

[89] 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.

been reviewed by the SAB.[90] We refer to the fixed cost approach as the "average cost" approach here. The average cost approach 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 (Appendix 4A). The range of estimates reflects different assumptions about the cost of additional emissions reductions beyond those in the identified control strategies. While we use a constant, average cost per ton to estimate the costs of the emissions reductions beyond identified controls, this does not imply that the MACC is not upward sloping. The constant, average cost per ton is designed to capture total costs associated with the abatement of the emissions reductions from unidentified controls – because of the incomplete information available to inform the characterization of the MACC, a portion of those total costs is likely at a value below the average cost per ton and a portion is likely at a value above the average cost per ton.

The alternative values used in the sensitivity analysis implicitly reflect different assumptions about the amount of technological progress and innovation in emissions reduction strategies that may be expected in the future. The average cost approach reflects a view that because we have incomplete data on existing control technologies, and because no cost data exists for unidentified future technologies, measures, or strategies, it is unclear whether approaches using hypothetical cost curves will be more accurate or less accurate in forecasting total national costs of unidentified controls than an average cost approach that uses a range of national cost-per-ton values.

The hybrid approach assumed increasing marginal costs of control along an upward-sloping marginal cost curve. The hybrid approach 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.[91] Under this approach, the relative costs of unidentified controls in different geographic areas reflected the

[90] The three approaches mentioned above, outlined for SAB review, for assigning costs to unidentified measures included: (1) the fixed cost approach that assigns all unidentified measures a fixed cost per ton; (2) an approach based on an upward sloping cost curve that uses information from the identified control measure analysis on an area-specific basis; and (3) an approach that adjusts the upward sloping cost curve projections using information about cost changes over time to reflect factors such as learning by doing and induced innovation.

[91] See, for example, Section 7.2 and Appendix 7.A.2 in the December 2012 RIA for the final $PM_{2.5}$ NAAQS, available at http://www.epa.gov/ttn/ecas/regdata/RIAs/finalria.pdf.

expectation that average per-ton control costs are likely to be higher in areas needing a higher ratio of emissions reductions from unidentified and identified 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 approach 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.

For areas needing significant additional emission reductions, much pollution abatement is likely needed from sectors that historically have not been intensively regulated and thus have relatively more available potential emissions reductions. 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.

As noted in previous NAAQS analyses, the EPA continues to explore other sources of information to inform the estimates of costs associated with unidentified controls. For this RIA we examined the full set of identified controls, examined evidence that suggests that over time new information and data emerges that shifts emissions reductions from the unidentified to the identified 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 Science Advisory Board Council Advisory's advice, and the requirements of E.O. 12866 and OMB circular A-4, which provide 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 approach 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.

As discussed in Section 4.1.1, we apply a constant, average cost per ton of $15,000/ton to capture total costs associated with the NOx emissions reductions achieved through unidentified controls. To explore how sensitive total costs are to this assumption, we also use alternative assumptions of the average cost. Specifically, we conduct 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. This range is inclusive of the annualized NOx offset prices observed in recent years in the areas likely to need unidentified controls to achieve the standard (Table 4-8), and if anything, suggests the central estimate of $15,000/ton is conservative. In the RIA for the ozone NAAQS proposal, EPA requested comments on the methods presented to estimate emissions reductions needed beyond identified controls, including the parameter estimate of $15,000/ton. We received comments that the parameter estimate of $15,000/ton (2011$) was low and should be adjusted to reflect inflation. The EPA has elected to retain the parameter estimate of $15,000/ton (2011$) because inflation was low between 2006 and 2011. While the Agency received comments on its methods, alternative approaches have not been subjected to peer review and therefore could not be applied here.

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 4-7, regional factors may play a significant role in the estimation of control costs. In the RIA for proposal, EPA stated it may review alternative methodologies and sources of regional information that could result in the average cost methodology being applied more regionally. The EPA reviewed data on end-of-pipe controls more closely and worked on identifying information from measures used in SIPs. The data in CoST on end-of-pipe technologies was not sufficiently robust to generate regional data, and obtaining detailed cost information from state or local SIPs requires a longer-term effort. Despite the regional variation in offset prices, we believe the $15,000/ton estimate represents a conservative value because it is higher than the majority of the annualized offset values in Table 4-8, and because the values we use in the sensitivity analyses include the highest annualized offset value.

4.3.2 Compliance Cost Estimates from Unidentified Controls

Table 4-9 presents the control cost estimates for unidentified controls for the East and West in 2025, except for California for the final standard of 70 ppb and an alternative standard of 65 ppb using an assumed average cost of $15,000/ton, as well as values of $10,000/ton and $20,000/ton. Appendix 4A includes potential alternative methods for either assigning an average cost value or for determining values used in sensitivity analyses.

Table 4-9. Unidentified Control Costs in 2025 by Alternative Standard for 2025 – U.S., except California (7 percent discount rate, millions of 2011$)

Alternative Level	Geographic Area	Unidentified Control Cost		
		$10,000/ton	$15,000/ton	$20,000/ton
70 ppb	East	470	700	930
	West	-	-	-
	Total	**470**	**700**	**930**
65 ppb	East	8,200	12,000	16,500
	West	400	610	810
	Total	**8,600**	**13,000**	**17,000**

[a] All values are rounded to two significant figures.

Table 4-10 presents the unidentified control cost estimates for post-2025 for California for the final standard of 70 ppb and an alternative standard level of 65 ppb using an assumed average cost of $15,000/ton, as well as values of $10,000/ton and $20,000/ton.

Table 4-10. Unidentified Control Costs in 2025 by Alternative Standard for Post-2025 – California (7 percent discount rate, millions of 2011$)

Alternative Level	Geographic Area	Unidentified Control Cost		
		$10,000/ton	$15,000/ton	$20,000/ton
70 ppb	California	510	800	1,020
65 ppb	California	1,000	1,500	2,000

[a] All values are rounded to two significant figures.

4.4 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. Please see Chapter 1, Section 1.3.2 for additional detailed discussion on potential nonattainment designations and timing.

Tables 4-11 and 4-12 present summaries of the total national annual costs (identified and unidentified) of attaining the revised standard of 70 ppb and alternative standard of 65 ppb – Table 4-11 presents the total national annual costs by alternative standard for 2025 for all of the U.S., except California and Table 4-12 presents the total national annual costs by alternative standard for post-2025 for California. As discussed in Section 4.1.2, because we do not have a full set of costs at the 3 percent discount rate or the 7 percent discount rate and because we believe the majority of the identified control costs is calculated at a 7 percent discount rate, Tables 4-11 and 4-12 present engineering cost estimates based on a 7 percent discount rate.

Table 4-11. Summary of Total Control Costs (Identified and Unidentified) by Alternative Level for 2025 - U.S., except California (millions of 2011$, 7% Discount Rate)[a]

Alternative Level	Geographic Area	Total Control Costs (Identified and Unidentified)
70 ppb	East	1,400
	West	<5
	Total	**$1,400**
65 ppb	East	15,000
	West	750
	Total	**$16,000**

[a] All values are rounded to two significant figures. Unidentified control costs are based on the average cost approach.

Table 4-12. Summary of Total Control Costs (Identified and Unidentified) by Alternative Level for Post-2025 - California (millions of 2011$, 7% Discount Rate)[a]

Alternative Level	Geographic Area	Total Control Costs (Identified and Unidentified)
70 ppb	California	800
65 ppb	California	1,500

[a] All values are rounded to two significant figures. Unidentified control costs are based on the average cost approach.

4.5 Economic Impacts

4.5.1 Introduction

This section addresses the potential economic impacts of the illustrative control strategies for the 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 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 from unidentified controls 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 alternative ozone standards are anticipated to impact multiple markets in many times and places. Computable General Equilibrium (CGE) models are one possible tool for evaluating the impacts of a regulation on the broader economy because this class of models explicitly captures interactions between markets across the entire economy. 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 identified controls (excluding the unidentified control costs). EMPAX is a dynamic computable general equilibrium (CGE) model that forecasts a new equilibrium for the entire economy after a policy intervention. While a CGE model captures the effects of behavioral responses on the part of consumers or other producers to changes in price that are missed by an engineering estimate of compliance costs, most CGE models do not model the environmental externality - or the benefits that accrue to society from mitigating it. When benefits from a regulation are expected to be substantial, social cost cannot be interpreted as a complete characterization of economic welfare. To the extent that the benefits affect behavioral responses in markets, the social cost measure may also be potentially biased.

EPA included specific types of health benefits in a CGE model for the prospective analysis, *The Benefits and Costs of the Clean Air Act from 1990 to 2020* (EPA 2011), and demonstrated the importance of their inclusion when evaluating the economic welfare effects of policy. However, while the external Council on Clean Air Compliance Analysis (Council) peer review of this EPA report (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", serious technical challenges remain when attempting to evaluate the benefits and costs of potential regulatory actions using economy-wide models.

To begin to address these technical challenges, the EPA has established a new Science Advisory Board (SAB) panel on economy-wide modeling to consider the technical merits and challenges of using CGE and other economy-wide modeling tools to evaluate costs, benefits, and economic impacts of air regulations. 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 was not available in time for this analysis. Given the ongoing SAB panel on economy-wide modeling, the uncertain nature of costs, the Council's advice regarding the importance of including benefit-side effects, 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. Instead, this section proceeds with a qualitative discussion of market impacts.

4.5.2 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 to a change in price, 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.

4.6 Differences between the Proposal and Final RIAs

Several changes in the analysis for this final RIA have resulted in lower control costs compared to the proposal RIA. As discussed in Chapter 3, Section 3.3, improved emissions inventory and model inputs as well as more refined air quality modeling resulted in approximately 50 percent fewer emissions reductions needed in Texas and the Northeast to reach the revised standard of 70 ppb compared to the proposal RIA. For an alternative standard of 65 ppb, we needed approximately 20 percent fewer emissions reductions nationwide than at proposal. In addition, because of the more refined air quality modeling, control strategies were applied in smaller geographic areas closer to monitors projected to exceed 70 and 65 ppb. Also, in the proposal RIA we applied controls to reach the current standard of 75 ppb in Texas and the Northeast, and in this final RIA these controls were not needed in these areas to reach the current standard. Not applying controls to reach the current standard, as well as applying controls in smaller geographic areas had impacts on the cost estimates that are discussed below.

In the final RIA, to reach a revised standard of 70 ppb in 2025 we applied a larger number of lower cost identified controls because (i) fewer emissions reductions were needed overall, and (ii) we did not apply any identified controls to reach 75 ppb. This meant that the cost of additional reductions could be estimated from lower cost identified controls in the final RIA. As a result, total estimated costs to reach 70 ppb were lower than costs estimates in the proposal RIA by 55 percent.

In the final RIA, to reach an alternative standard of 65 ppb, while fewer reductions were needed, the area where we applied identified controls was smaller than in the proposal RIA, resulting in exhausting the supply of identified controls available in these areas. We needed additional, higher-cost unidentified controls to being these areas into attainment. For example, in the proposal RIA, in analyzing 65 ppb we applied controls across the state of Texas as well as in surrounding states inside the "Central" region (Oklahoma, Kansas, Missouri, Arkansas, Louisiana and Mississippi). Because of the size of the area in which controls were applied for proposal, there were more identified controls from which to choose. In the final RIA, we applied controls only in east Texas, which meant there were fewer identified controls available and we relied more heavily on unidentified controls. Overall, because we applied all available identified controls in the final RIA, we relied on unidentified controls for 57 percent of the emissions reductions needed, whereas in the proposal RIA we relied on unidentified controls for 40 percent of the emissions reductions needed to reach 65 ppb. Because unidentified controls are more expensive than identified controls and we relied on more unidentified controls in the final RIA, the estimated costs did not decrease in proportion to the decrease in needed emissions reductions and are about the same as in the proposal RIA.

4.7 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 identified 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 not available (i.e., where capital, equipment life value, and operation and maintenance [O&M] costs are not separated out), the EPA 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 separated out), we can and do recalculate costs using a 3 percent discount rate. In general, we have some disaggregated data available for non-EGU point source controls, but we do not have any disaggregated control cost data for nonpoint (area) source controls. In addition, while these discount rates are consistent with OMB guidance, the actual real discount rates may vary regionally or locally.

Identified 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.[92] 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 may differ from actual 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 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. A recent EPA report, the "Retrospective Study of the Costs of EPA Regulations" that examined the compliance costs of five EPA regulations in four case studies,[93] found that several of the case studies suggested that cost estimates were over-estimated *ex ante*,

[92] 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.

[93] 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.

but did not find the evidence to be conclusive. The EPA stated in the report that the small number of regulatory actions covered, as well as significant data and analytical challenges associated with the case studies limited the certainty of this conclusion.

Costs of unidentified controls: In addition to the application of identified controls, the EPA assumes the application of unidentified controls for attainment in the projection year for this analysis.

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APPENDIX 4A: ENGINEERING COST ANALYSIS

Overview

Chapter 4 describes the engineering cost analysis approach that EPA used to demonstrate attainment of the revised standard of 70 ppb and an alternative ozone standard level of 65 ppb. This Appendix contains more detailed information about the control costs of the identified control strategy analyses by control measure as well as sensitivity analyses for the average cost approach used to estimate costs for the unidentified emissions controls. Specifically, results using two alternative cost assumptions for unidentified controls are presented, and in addition the findings from six alternative approaches for estimating these costs are described and presented. These include a regression-based approach and five simulation-based variations. Table 4A-10 at the end of this Appendix provides a summary of these alternative approaches.

4A.1 Cost of Identified Controls in Alternative Standards Analyses

This section presents costs of identified 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 identified controls for California were applied as part of the baseline analysis, no identified controls were available for the alternative standards analyses in California. The costs for the standards analyzed do not include any identified control costs for California. Tables 4A-1 and 4A-2 present the costs for identified controls by measure for the 70 ppb alternative standard analysis for NOx and VOC respectively. Tables 4A-3 and 4A-4 present the costs for identified controls by measure for the 65 ppb alternative standard analysis.

Table 4A-1. Costs for Identified NO_x Controls in the 70 ppb Analysis (2011$)

NO_x Control Measure	Cost[a] (million)	Average $/ton
Adjust Air to Fuel Ratio and Ignition Retard - Gas Fired IC Engines	3.49	400
Biosolid Injection Technology - Cement Kilns	2.22	413
EGU SCR & SNCR	51.69	1,150
Episodic Burn Ban	-	- [94]

[94] An ozone season episodic burn ban is a daily ban of open burning of yard/agricultural waste on an ozone season day where ozone exceedances are predicted. There are minimal administrative costs associated with this measure, and we have not quantified those costs in this or previous analyses. For additional information on these measures go

NO_x Control Measure	Cost[a] (million)	Average $/ton
Excess O3 Control	0.01	24
Ignition Retard - IC Engines	0.73	1,185
Low Emission Combustion - Gas Fired Lean Burn IC Engines	14.66	829
Low NOx Burner - Coal Cleaning	0.30	1,125
Low NOx Burner - Commercial/Institutional Boilers & IC Engines	23.78	1,110
Low NOx Burner - Gas-Fired Combustion	8.96	970
Low NOx Burner - Glass Manufacturing	0.28	1,141
Low NOx Burner - Industr/Commercial/Institutional (ICI) Boilers	6.33	1,135
Low NOx Burner - Industrial Combustion	0.03	1,255
Low NOx Burner - Lime Kilns	2.22	913
Low NOx Burner - Natural Gas-Fired Turbines	12.27	1,955
Low NOx Burner - Residential Water Heaters & Space Heaters	22.57	1,134
Low NOx Burner and Flue Gas Recirculation - Coke Oven/Blast Furnace	1.87	5,199
Low NOx Burner and Flue Gas Recirculation - Fluid Catalytic Cracking Units	0.17	2,091
Low NOx Burner and Flue Gas Recirculation - Iron & Steel	0.25	619
Low NOx Burner and SCR - Industr/Commercial/Institutional Boilers	62.83	7,473
Mid-Kiln Firing - Cement Manufacturing	0.09	73
Non-Selective Catalytic Reduction (NSCR) - 4 Cycle Rich Burn IC Engines	41.27	1,051
Nonroad Diesel Retrofits & Engine Rebuilds - e.g., Construction Equipment	12.87	4,545
OXY-Firing - Glass Manufacturing	44.23	3,691
Replacement of Residential & Commercial/Institutional Water Heaters	--	--[95]
Selective Catalytic Reduction (SCR) - Cement Kilns	54.22	5,328
Selective Catalytic Reduction (SCR) - Fluid Catalytic Cracking Units	7.54	4,414
Selective Catalytic Reduction (SCR) - Glass Manufacturing	4.03	1,157
Selective Catalytic Reduction (SCR) - IC Engines, Diesel	3.29	3,814
Selective Catalytic Reduction (SCR) - ICI Boilers	17.35	3,756
Selective Catalytic Reduction (SCR) - Industrial Incinerators	5.27	3,805
Selective Catalytic Reduction (SCR) - Iron & Steel	0.80	5,180
Selective Catalytic Reduction (SCR) - Process Heaters	6.65	8,483
Selective Catalytic Reduction (SCR) - Sludge Incinerators	0.38	3,805
Selective Catalytic Reduction (SCR) - Space Heaters	0.11	4,661
Selective Catalytic Reduction (SCR) - Utility Boilers	0.18	128
Selective Non-Catalytic Reduction (SNCR) - Cement Manufacturing	3.13	1,303
Selective Non-Catalytic Reduction (SNCR) - Coke Manufacturing	4.25	2,673
Selective Non-Catalytic Reduction (SNCR) - Comm./Inst. Incinerators	0.11	1,842
Selective Non-Catalytic Reduction (SNCR) - Industrial Incinerators	0.60	1,645
Selective Non-Catalytic Reduction (SNCR) - Sludge Incinerators	0.06	1,842

to http://www.epa.gov/ttn/ecas/cost.htm. The EPA will continue to conduct research on possible costs for this measure, and if applicable update costs for this measure in future analyses.

[95] We have not quantified specific costs for this measure in this or previous analyses. For additional information on these measures go to http://www.epa.gov/ttn/ecas/cost.htm. The EPA will continue to conduct research on possible costs for this measure, and if applicable update costs for this measure in future analyses.

NO_x Control Measure	Cost[a] (million)	Average $/ton
Ultra-Low NOx Burner - Process Heaters	0.14	420

[a] All values are rounded to two significant figures.

Table 4A-2. Costs for Identified VOC Controls in the 70 ppb Analysis (2011$)

VOC Control Measure	Cost[a] (million)	Average $/ton
Control Technology Guidelines - Wood Furniture Surface Coating	8.87	32,595
Control of Fugitive Releases - Oil & Natural Gas Production	0.02	2,689
Flare - Petroleum Flare	0.31	3,305
Incineration - Other	155.85	14,543
LPV Relief Valve - Underground Tanks	2.29	1,763
MACT - Motor Vehicle Coating	0.00	192
Permanent Total Enclosure (PTE) - Surface Coating	4.53	12,289
RACT - Graphic Arts	1.66	6,386
Reduced Solvent Utilization - Surface Coating	0.03	1,232
Reformulation - Architectural Coatings	86.00	16,394
Reformulation - Pesticides Application	2.59	15,157
Reformulation-Process Modification - Automobile Refinishing	2.58	11,734
Reformulation-Process Modification - Cutback Asphalt	0.02	24
Reformulation-Process Modification - Other	0.35	3,102
Reformulation-Process Modification - Surface Coating	0.59	3,304
Solvent Recovery System - Printing/Publishing	0.02	1,232
Wastewater Treatment Controls- POTWs	0.70	3,366

[a] All values are rounded to two significant figures.

Table 4A-3. Costs for Identified NO_x Controls in the 65 ppb Analysis (2011$)

NO_x Control Measure	Cost[a] (million)	Average $/ton
Adjust Air to Fuel Ratio and Ignition Retard - Gas Fired IC Engines	7.25	441
Biosolid Injection Technology - Cement Kilns	2.44	413
EGU SCR & SNCR	125.93	1,150
Episodic Burn Ban	-	-[96]
Ignition Retard - IC Engines	0.73	1,262
Low Emission Combustion - Gas Fired Lean Burn IC Engines	57.85	764
Low NOx Burner - Coal Cleaning	0.69	1,451
Low NOx Burner - Commercial/Institutional Boilers & IC Engines	38.59	1,066
Low NOx Burner - Fiberglass Manufacturing	0.10	1,522
Low NOx Burner - Gas-Fired Combustion	11.53	970
Low NOx Burner - Industr/Commercial/Institutional (ICI) Boilers	56.57	2,581

[96] An ozone season episodic burn ban is a daily ban of open burning of yard/agricultural waste on an ozone season day where ozone exceedances are predicted. There are minimal administrative costs associated with this measure, and we have not quantified those costs in this or previous analyses. For additional information on these measures go to http://www.epa.gov/ttn/ecas/cost.htm. The EPA will continue to conduct research on possible costs for this measure, and if applicable update costs for this measure in future analyses.

NO_x Control Measure	Cost[a] (million)	Average $/ton
Low NOx Burner - Industrial Combustion	0.03	1,255
Low NOx Burner - Lime Kilns	4.21	913
Low NOx Burner - Natural Gas-Fired Turbines	25.63	2,117
Low NOx Burner - Residential Water Heaters & Space Heaters	103.38	1,999
Low NOx Burner - Steel Foundry Furnaces	0.27	929
Low NOx Burner - Surface Coating Ovens	0.09	3,585
Low NOx Burner and Flue Gas Recirculation - (ICI) Boilers	2.00	4,197
Low NOx Burner and Flue Gas Recirculation - Coke Oven/Blast Furnace	2.23	5,199
Low NOx Burner and Flue Gas Recirculation - Fluid Catalytic Cracking Units	0.26	4,347
Low NOx Burner and Flue Gas Recirculation - Iron & Steel	0.48	619
Low NOx Burner and Flue Gas Recirculation - Process Heaters	2.85	5,199
Low NOx Burner and Flue Gas Recirculation - Starch Manufacturing	0.35	5,199
Low NOx Burner and SCR - Industr/Commercial/Institutional Boilers	151.27	6,230
Low NOx Burner and SNCR - Industr/Commercial/Institutional Boilers	1.93	3,997
Natural Gas Reburn - Natural Gas-Fired EGU Boilers	1.41	2,388
Non-Selective Catalytic Reduction (NSCR) - 4 Cycle Rich Burn IC Engines	54.24	775
Non-Selective Catalytic Reduction - Nitric Acid Manufacturing	0.94	1,905
Nonroad Diesel Retrofits & Engine Rebuilds - e.g., Construction Equipment	39.78	4,525
OXY-Firing - Glass Manufacturing	110.92	4,093
Replacement of Residential Water Heaters	--	--[97]
Selective Catalytic Reduction (SCR) - Ammonia Mfg	6.77	2,896
Selective Catalytic Reduction (SCR) - Cement Kilns	161.93	6,194
Selective Catalytic Reduction (SCR) - Coke Ovens	9.70	7,798
Selective Catalytic Reduction (SCR) - Fluid Catalytic Cracking Units	18.56	4,551
Selective Catalytic Reduction (SCR) - IC Engines, Diesel	13.10	3,664
Selective Catalytic Reduction (SCR) - ICI Boilers	36.23	3,636
Selective Catalytic Reduction (SCR) - Industrial Incinerators	6.56	3,805
Selective Catalytic Reduction (SCR) - Iron & Steel	6.81	3,834
Selective Catalytic Reduction (SCR) - Process Heaters	20.24	7,376
Selective Catalytic Reduction (SCR) - Sludge Incinerators	10.26	5,796
Selective Catalytic Reduction (SCR) - Space Heaters	1.33	4,631
Selective Catalytic Reduction (SCR) - Taconite	27.40	6,449
Selective Catalytic Reduction (SCR) - Utility Boilers	0.18	128
Selective Non-Catalytic Reduction (SNCR) - Coke Mfg	7.70	2,673
Selective Non-Catalytic Reduction (SNCR) - Comm./Inst. Incinerators	0.29	1,842
Selective Non-Catalytic Reduction (SNCR) - Industrial Incinerators	1.97	1,861
Selective Non-Catalytic Reduction (SNCR) - Municipal Waste Combustors	0.12	1,842
Selective Non-Catalytic Reduction (SNCR) - Sludge Incinerators	0.21	1,863
Selective Non-Catalytic Reduction (SNCR) - Utility Boilers	0.33	1,390

[97] We have not quantified specific costs for this measure in this or previous analyses. For additional information on these measures go to http://www.epa.gov/ttn/ecas/cost.htm. The EPA will continue to conduct research on possible costs for this measure, and if applicable update costs for this measure in future analyses.

NO_x Control Measure	Cost[a] (million)	Average $/ton
Ultra-Low NOx Burner - Process Heaters	1.24	1,447

[a] All values are rounded to two significant figures.

Table 4A-4. Costs for Identified VOC Controls in the 65 ppb Analysis (2011$)

VOC Control Measure	Cost[a] (million)	Average ($/ton)
Control Technology Guidelines - Wood Furniture Surface Coating	97.41	32,595
Control of Fugitive Releases - Oil & Natural Gas Production	0.08	2,689
Flare - Petroleum Flare	0.36	3,305
Gas Recovery - Municipal Solid Waste Landfill	0.32	1,106
Improved Work Practices, Material Substitution, Add-On Controls - Printing	0.00	159
Improved Work Practices, Material Substitution, Add-On Controls -Industrial Cleaning Solvents	(0.34)	(1,360)
Incineration - Other	240.54	14,395
LPV Relief Valve - Underground Tanks	8.59	1,763
Low VOC Adhesives and Improved Application Methods - Industrial Adhesives	0.06	270
Low-VOC Coatings and Add-On Controls - Surface Coating	0.73	2,668
MACT - Motor Vehicle Coating	0.37	192
Permanent Total Enclosure (PTE) - Surface Coating	45.92	13,973
Petroleum and Solvent Evaporation - Surface Coating Operations	0.09	376
RACT - Graphic Arts	35.67	6,386
Reduced Solvent Utilization - Surface Coating	5.36	1,758
Reformulation - Architectural Coatings	858.67	16,394
Reformulation - Industrial Adhesives	13.34	12,017
Reformulation - Pesticides Application	59.97	15,157
Reformulation-Process Modification - Automobile Refinishing	57.25	11,734
Reformulation-Process Modification - Cutback Asphalt	0.06	24
Reformulation-Process Modification - Oil & Natural Gas Production	0.19	641
Reformulation-Process Modification - Other	1.94	3,548
Reformulation-Process Modification - Surface Coating	21.00	3,736
Solvent Recovery System - Printing/Publishing	1.05	1,232
Solvent Substitution and Improved Application Methods - Fiberglass Boat Mfg	0.06	4,310
Wastewater Treatment Controls- POTWs	0.79	3,366

[a] All values are rounded to two significant figures.

4A.2 Alternative Estimates of Costs Associated with Emissions Reductions from Unidentified Controls

This section presents alternative estimates of the unidentified control costs using alternative average cost per ton of emissions reductions from unidentified controls of $10,000 per ton and $20,000/ton.[98] Table 4A-5 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 unidentified controls. Table 4A-6 presents the estimates of the total control costs for post-2025 California when using the alternative per ton cost assumptions for emissions reductions from unidentified controls.

Table 4A-5. Summary of Total Control Costs (Identified and Unidentified) by Alternative Level for 2025 - U.S. using Alternative Cost Assumption for Unidentified Control Costs, except California (millions of 2011$)[a]

Alternative Level	Geographic Area	Total Control Costs (Identified and Unidentified)		
		Unidentified Control Cost = $10,000/ton	Unidentified Control Cost = $15,000/ton	Unidentified Control Cost = $20,000/ton
70 ppb	East	1,100	1,400	1,600
	West	<5	<5	<5
	Total	1,200	1,400	1,600
65 ppb	East	11,000	15,000	19,000
	West	550	750	950
	Total	11,000	16,000	20,000

[a] All values are rounded to two significant figures.

Table 4A-6. Summary of Total Control Costs (Identified and Unidentified) by Alternative Level for Post-2025 California - U.S. using Alternative Cost Assumption for Unidentified Control Costs (millions of 2011$)[a]

Alternative Level	Geographic Area	Total Control Cost		
		Unidentified Control Cost = $10,000/ton	Unidentified Control Cost = $15,000/ton	Unidentified Control Cost = $20,000/ton
70 ppb	California	510	800	1,020
65 ppb	California	1,000	1,500	2,000

[a] All values are rounded to two significant figures.

[98] The EPA decided to use the alternative values of $10,000 per ton and $20,000 per ton because these values were used in the following recent RIAs, and we did not identify other more appropriate alternative values: the November 2014 proposal RIA, the December 2012 *Regulatory Impact Analysis for the Final Revisions to the National Ambient Air Quality Standards for Particulate Matter* and the June 2012 *Regulatory Impact Analysis for the Proposed Revisions to the National Ambient Air Quality Standards for Particulate Matter*.

4A.3 Alternative Approaches to Estimating the Costs Associated with Emissions Reductions from Unidentified Controls

4A.3.1 Regression Approach

Using all observations under the cost per ton threshold for identified controls ($19,000/ton for NOx), a linear regression is estimated and used to predict the price of the additional unidentified controls required to attain a particular level of the standard. That is, to meet a particular level of the standard, it is assumed that all reductions that can be achieved at a cost less than the cost threshold will first be exhausted and any additional tons required can be achieved at a cost determined by the value of the regression line at those tons. Hence, the total cost of the tons of reductions for which controls are unidentified is the area under the regression line for tons between the total identified tons and the total tons of reduction required to meet the level of the standard being analyzed (see Figure 4A-1 below). Using this methodology, the light gray shaded area under the regression line represents the cost of unidentified controls needed to attain the 70 ppb level of the standard ($960 million), while the darker gray shaded area represents the additional cost of unidentified controls needed to attain the 65 ppb level of the standard (an additional $13 billion, for a total of $14 billion in unidentified control costs).

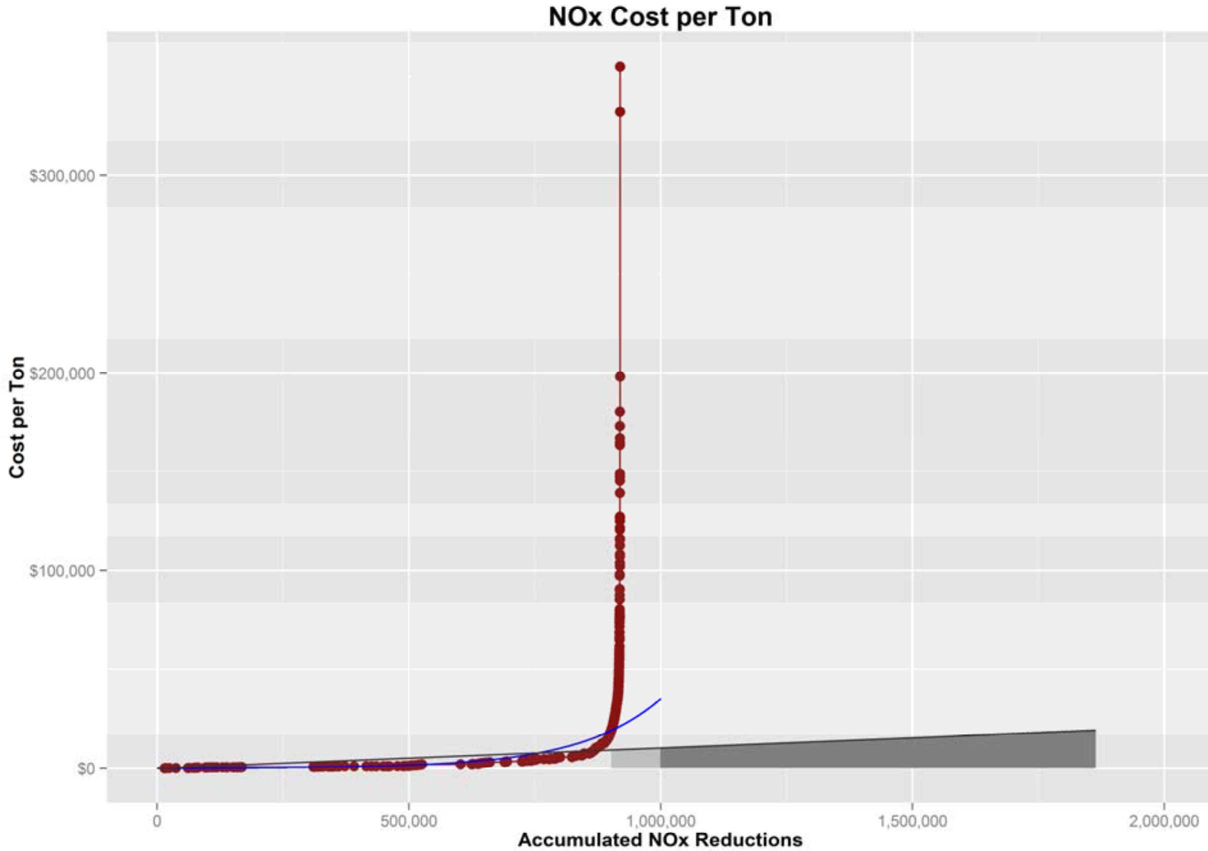

Figure 4A-1. Marginal Costs for Identified NO_x Controls for All Source Sectors with Regression Line for Unidentified Control Measures

An alternative interpretation of this approach that is in line with the discussion of the incomplete characterization of the MACC (Chapter 4, Section 4.2.3) is that the points on the regression line represent potential controls that could be applied to sectors or processes that are not characterized in the CoST tool.[99] The MACC can then be redrawn including these points, as appears below in Figure 4A-2. In this curve, the dark red line represents the expanded MACC curve including controls needed to attain the 70 ppb level of the standard. The blue portion of the line includes additional controls needed to attain the 65 ppb level of the standard. As this is simply a way of visualizing how controls estimated from the regression line could be incorporated into the MACC, the resulting unidentified control cost estimates ($960 million for

[99] Examples of controls or measures that are not in the CoST tool include local transportation measures, energy efficiency measures, or fuel switching applications.

the 70 ppb level of the standard and an additional $13 billion to attain the 65 ppb level of the standard) are unchanged.

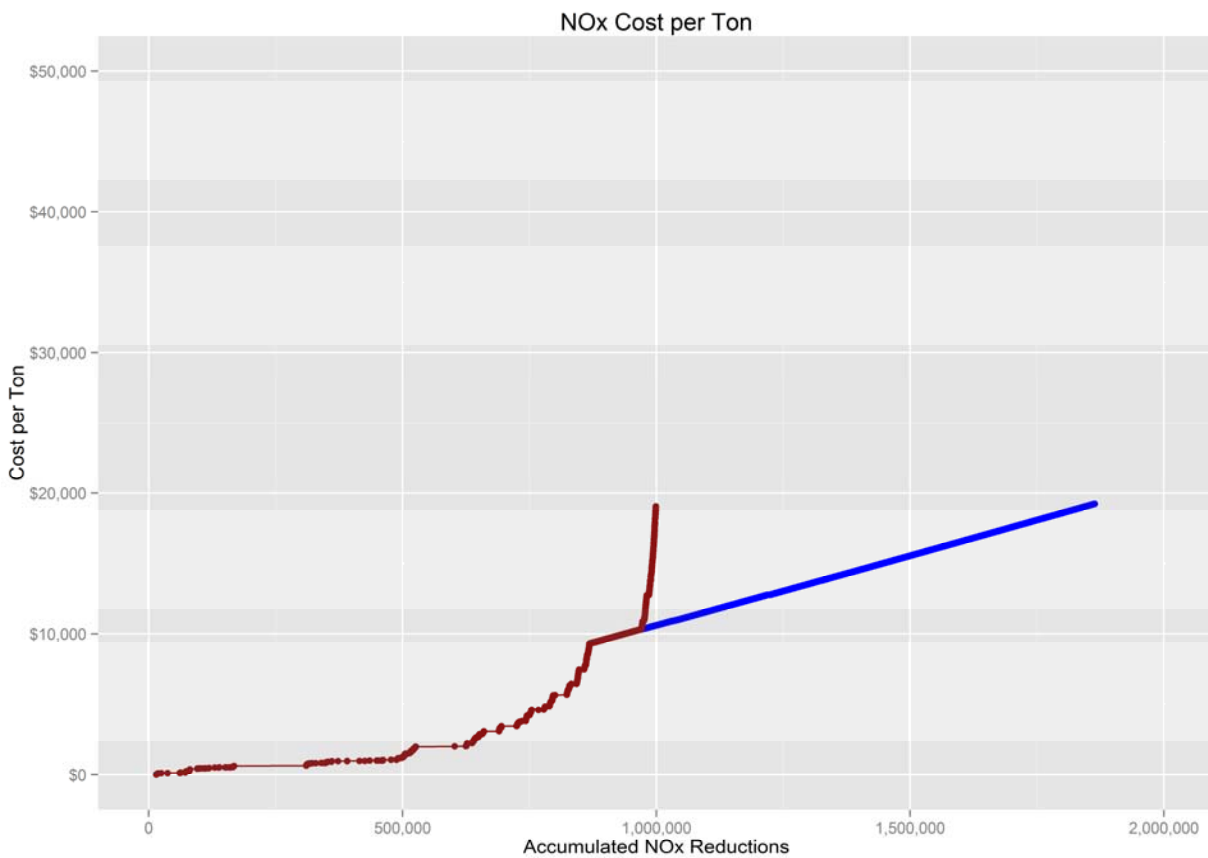

Figure 4A-2. Marginal Costs for Identified NO_x Controls for All Source Sectors with Unidentified Control Measures from Regression Line Included

4A.3.2 Simulation Approach

Another approach to estimate the cost of unidentified controls is to randomly sample from the complete MACC (without a cost threshold) to "fill in" the tons of reduction on the MACC needed for attainment. Sampling is done with replacement, and since the sampling is random, it is repeated 1000 times to limit the influence of any particular simulation. The mean of the total cost estimates is then the estimate of the cost of the unidentified controls. Alternatively, the mean cost per ton of the estimates can be interpreted as the cost per ton estimate for the unidentified controls.

The implicit assumption when applying this simulation approach is that the controls in the incomplete MACC curve are representative of the types of controls (both in cost and effectiveness) that could be applied to alternative sources not yet controlled or adequately characterized in the CoST database, and also that controls that may be developed in the future (prior to the attainment date) will be similar in cost and effectiveness to controls currently available. While we do not believe that controls with costs greater than the cost threshold are likely to be applied, they are not removed from the simulation dataset in order to provide a more complete set of identified abatement possibilities.

The CoST database used for this analysis contains approximately 120,000 individual controls applicable to five broad sectors. While there is geographic specificity in the applicability of the controls, the simulation is currently being performed on a national scale. The cost per ton and number of available controls differs considerably between sectors, as shown below in Table 4A-7, and for this reason it is important to determine how broadly to sample when selecting controls to attain a particular level of the standard.

Table 4A-7. Costs and Number of Identified NO_x Controls by Sector in the CoST Database (2011$)

Sector	Number of Controls	Cost per Ton			
		Minimum	Median	Mean	Maximum
nonpt	2,069	$413	$1,008	$1,455	$2,005
nonroad	111,943	$3,330	$4,618	$4,619	$5,300
np_oilgas	772	$78	$649	$747	$2,019
pt_oilgas	3,892	$12	$649	$1,120	$44,860
ptnonipm	2,756	$18	$3,814	$12,147	$354,974

In this simulation approach, we investigate three methods for selecting available controls to expand the MACC to simulate attainment. First, the sectors are aggregated into three broad sectors (nonroad, nonpoint, and non-EGU point sources) and controls are selected from these sectors in the same proportion as the control strategy discussed in Chapter 3 for these three sectors for each level of the standard. These percentages appear below in Table 4A-8.

Table 4A-8. Simulation Percentage of NO_x Controls from Sectors Based on Application of Identified Controls

Sector	70 ppb	65 ppb
non-EGU point	46%	56%
nonpoint	52%	42%
nonroad	2%	2%

Using this method, since only 2 percent of the controls to meet any of the standards were applied to nonroad sources, only 2 percent of the tons from random draws are allowed to come from this sector. The majority of simulated controls are then drawn from the nonpoint and non-EGU point sectors.

A second method of assigning the proportions is based upon the number of available tons of reduction remaining in each sector after the application of the controls discussed in Chapter 3. To calculate this, the tons of reduction from the control scenarios are subtracted from the projected inventories, and then the percentages used for the simulation exercise are based upon the proportion of remaining emissions in the three sectors. These percentages appear below in Table 4A-9.

Table 4A-9. Simulation Percentage of NO_x Controls from Sectors Based on Remaining Emissions in Sectors

Sector	70 ppb	65 ppb
non-EGU point	33%	30%
nonpoint	36%	36%
nonroad	31%	34%

This method leads to a larger proportion of the simulated controls being selected from the nonroad sector, because this sector has a relatively large quantity of remaining emissions in 2025. Accordingly, the proportion of simulated controls drawn from the non-EGU point and nonpoint sectors is lower using this method. A third method imposes no restrictions on the selection of controls, so controls are randomly selected from the complete set regardless of the sector.

Instead of randomly selecting from the MACC, it is also possible to simulate attainment by selecting controls from along the regression line shown in Figure 4A-1. For the purposes of this exercise, controls are first selected from the point where identified controls exceed the cost threshold up to the total tons of reduction necessary to meet a 65 ppb ozone standard. Each control is assumed to provide one ton of reduction at a cost calculated using the regression equation. This is different from using the area under the regression line to calculate the cost of unidentified controls, because in this case controls can be selected from any point along the regression line beyond the point where identified controls cross the cost threshold. As a result, the cost of the first additional ton of control as identified by the regression line is approximately $9,300 and this value rises to slightly more than $19,000 for the last ton required to attain the 65 ppb standard. A second variation on this approach selects tons of control from any point along the regression line. The results of the simulations appear in Table 4A-10.

Table 4A-10. Unidentified NO_x Control Costs by Alternative Standard using Alternative Methods for Estimation of Costs from Unidentified Controls (total costs in millions of 2011$, cost per ton in parentheses in $2011)

			Simulation Approach				
Level of Standard	**Tons of Unidentified NO_x reductions**	**Regression Approach**	**Sector Percentages from Applied Controls**	**Sector Percentages from Remaining Emissions**	**Random Draws from all Identified Controls**	**Random Draws from Regression Line Beyond Identified Controls**	**Random Draws from Entire Regression Line**
70 ppb	97,000[b]	$960 ($9,800)	$250 ($2,500)	$310 ($3,100)	$290 ($3,000)	$1,400 ($14,000)	$940 ($9,600)
65 ppb	960,000[c]	$14,000 ($14,000)	$2,700 ($2,800)	$3,000 ($3,100)	$2,900 ($3,000)	$14,000 ($14,000)	$9,300 ($9,600)

[a] All values are rounded to two significant figures.

[b] Total tons of NO_x reductions required includes 51,000 tons for Post-2025 California and 46,000 tons for the rest of the United States in 2025. Because these simulations are designed to be proof of concept, we combined the emissions reductions needed and controls applied in these analyses.

[c] Total tons of NO_x reductions required includes 100,000 tons for Post-2025 California and 860,000 tons for the rest of the United States in 2025. Because these simulations are designed to be proof of concept, we combined the emissions reductions needed and controls applied in these analyses.

Cost estimates based on the regression approach or sampling from the regression line are consistently higher than those based on sampling from the identified controls, as should be expected. While the simulations that sampled from identified controls were allowed to select controls beyond the cost per ton threshold applied in the identified control strategy described in Chapter 3, there are a limited number of controls above the cost per ton threshold in the database. As a result, the simulation is far more likely to select cheaper controls simply because of their prevalence in the data. Another observation that can be made is that requiring a certain percentage of controls from particular sectors does affect the results, but not to the extent that might be expected. While the cost of controls vary between sectors, the simulations produced similar results regardless of restrictions on the distribution of controls across sectors. Finally, while the cost per ton estimates varied considerably across the approaches, all cost per ton estimates were below the $15,000/ton estimate used as the primary estimate of the cost of unidentified controls.

The EPA continues to investigate methods to better estimate the cost of currently unidentified controls. While we have reason to believe that technological advances over the coming years will both lead to new types of controls as well as reduce the cost of currently available controls, we do not presently possess the capability to accurately predict the rate at which technological progress will occur or the potential impacts such progress will have on the cost of controls. As a result, the simulations presented herein draw upon currently identified controls and their current costs, or a linear regression of these data. The results of the simulations are sensitive to the percentage of unidentified controls required from each sector, and for this reason we plan to continue investigating methods for assigning these percentages based upon the degree to which sectors have already been controlled. Furthermore, while the simulations in this appendix were performed at the national level, we recognize that areas differ greatly in their industrial base, and for this reason the simulations should be performed at a more disaggregated level using data about the emissions sources in each area, available controls, and the degree to which sources in the area have already been controlled. Because these simulations are designed to be proof of concept, we combined the emissions reductions needed and controls applied in these analyses.

CHAPTER 5: QUALITATIVE DISCUSSION OF EMPLOYMENT IMPACTS OF AIR QUALITY

Overview

Executive Order 13563 directs federal agencies to consider regulatory impacts on job creation and employment. According to the Executive Order, "our regulatory system must protect public health, welfare, safety, and our environment while promoting economic growth, innovation, competitiveness, and job creation. It must be based on the best available science" (Executive Order 13563, 2011). Although standard benefit-cost analyses have not typically included a separate analysis of regulation-induced employment impacts,[100] we typically conduct employment analyses for economically significant rules. While the economy continues moving toward full employment, employment 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.[101]

Section 5.1 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 5.2 presents an overview of the peer-reviewed literature relevant to evaluating the effect of environmental regulation on employment. Section 5.3 discusses employment related to installation of NOx controls on coal and gas-fired electric generating units, industrial boilers, and cement kilns.

5.1 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

[100] Labor expenses do, however, contribute toward total costs in the EPA's standard benefit-cost analyses.
[101] The employment analysis in this RIA is part of EPA's ongoing effort to "conduct continuing evaluations of potential loss or shifts of employment which may result from the administration or enforcement of [the Act]" pursuant to CAA section 321(a).

declines and gains in different areas of the economy (the directly regulated sector, upstream and 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.[102] 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.[103,104]

Berman and Bui (2001) adapt this model to analyze how environmental regulations affect labor demand.[105] 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) model two components that drive changes in firm-level labor demand: output effects and substitution effects.[106] 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 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.

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[102] See Layard and Walters (1978), a standard microeconomic theory textbook, for a discussion, in Chapter 9.
[103] See Hamermesh (1993), Ch. 2, for a derivation of the firm's labor demand function from cost-minimization.
[104] In this framework, labor demand is a function of quantity of output and prices (of both outputs and inputs).
[105] Morgenstern, Pizer, and Shih (2002) develop a similar model.
[106] 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 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, for example Greenstone (2002), likely overstate employment impacts because they rely on relative comparisons between more regulated and less regulated counties, which can lead to "double counting" of impacts when production and employment shift from more regulated toward less regulated areas. Thus the empirical methods cannot be used to estimate net employment effects.[107]

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,

[107] See Greenstone (2002) p. 1212.

(3) the supply of other factors is highly elastic, or (4) labor costs are a large share of total production costs.[108] 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.[109] 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 regulatory impact analysis (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.[110] 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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[108] See Ehrenberg & Smith, p. 108.
[109] This discussion draws from Berman and Bui (2001), pp. 293.
[110] 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).[111]

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). For example, the Congressional Budget Office considered EPA's Mercury and Air Toxics Standards and regulations for industrial boilers and process heaters as potentially leading to short-run net increases in economic growth and employment, driven by capital investments for compliance with the regulations (Congressional Budget Office, 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.[112] 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

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[111] 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.
[112] E.g. Graff Zivin and Neidell (2012).

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 available. Finally, economic theory suggests that labor supply effects are also possible. In the next section, we discuss the empirical literature.

5.2 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.[113] This work focuses primarily on effects of employment policies such as labor taxes and minimum wages.[114] 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) and Ferris, Shadbegian, and Wolverton (2014), suggest that regulation-induced net employment impacts may be zero or slightly positive, but small in the regulated sector. Other research on regulated sectors suggests that employment growth may be lower in more regulated areas (Greenstone 2002, Walker 2011, 2013). However since these latter studies compare more regulated to less regulated counties, this methodological approach likely overstates employment impacts to the extent that regulation causes plants to locate in one area of the country rather than another, which would lead to “double counting” of the employment impacts. List *et al.* (2003) find some evidence that this type of geographic relocation may be occurring. 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

[113] Again, see Hamermesh (1993) for a detailed treatment.
[114] See Ehrenberg & Smith (2000), Chapter 4: “Employment Effects: Empirical Estimates” for a concise overview.

Standards (MATS).[115] 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.

5.2.1 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 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.[116] 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).[117]

A small literature examines impacts of environmental regulations on manufacturing employment. Greenstone (2002) and Walker (2011, 2013) study the impact of air quality regulations on manufacturing employment, estimating the net effects in nonattainment areas relative to attainment areas. 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

[115] U.S. EPA (2011b).

[116] 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.

[117] Including the employment effect of existing plants and plants dissuaded from opening will increase the estimated impact of regulation on employment.

an estimate of a national net effect on employment. Moreover this result is sensitive to model specification choices.

5.2.2 Economy-Wide

As noted above it is very difficult to estimate the net national employment impacts of environmental regulation. Given the difficulty with estimating national impacts of regulations, EPA has not generally estimated economy-wide employment impacts of its regulations in its benefit-cost analyses. However, in its continuing effort to advance the evaluation of costs, benefits, and economic impacts associated with environmental regulation, EPA has formed a panel of experts as part of EPA's Science Advisory Board (SAB) to advise EPA on the technical merits and challenges of using economy-wide economic models to evaluate the impacts of its regulations, including the impact on net national employment.[118] Once EPA receives guidance from this panel it will carefully consider this input and then decide if and how to proceed on economy-wide modeling of employment impacts of its regulations.

5.2.3 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.[119] 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. 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

[118] For further information see: http://yosemite.epa.gov/sab/sabproduct.nsf/0/07E67CF77B54734285257BB0004F87ED?OpenDocument
[119] For a recent review see Graff-Zivin and Neidell (2013).

billion (ppb) decreases in ozone concentrations increases worker productivity by 5.5 percent." (Graff Zivin and Neidell, 2012, p. 3654).[120]

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 macro-economy.

5.3 Employment Related to Installation and Maintenance of NO_x Control Equipment

This section discusses employment related to installation of NOx controls on coal and gas-fired electric generating units (EGUs), industrial boilers, and cement kilns, which are among the highest NOx-emitting source categories in EPA's emissions inventory (see Chapter 2 and Appendix 2A for more detail on emissions). The employment analysis in this section is an illustrative analysis, estimating the amount of labor involved with installing advanced NOx emission control systems at each of these three different types of NOx emission sources. The analysis also estimates the labor needed to operate existing advanced NOx systems more frequently (e.g., year round). Sections 5.3.1 and 5.3.2 below contain estimates of the number of direct short-term and long-term jobs that would be created by addition of NOx controls at a single facility (or unit) in each of these three categories of emissions sources, for various size units. Because the apportionment of emissions control across emissions sources in this RIA analysis is an illustrative model plant analysis, and is not necessarily representative of the controls that will be required in individual state SIPs, the EPA did not estimate short-term or long-term employment that would result from addition of NOx controls at these three source categories either everywhere throughout the country, or at facilities located in areas anticipated to need additional NOx reductions for the 65 ppb and 70 ppb standard alternatives.

5.3.1 Employment Resulting from Addition of NO_x Controls at EGUs

This section presents an illustrative analysis of the direct labor needs to install and operate SCRs at three common sizes of coal-fired EGUs: 300 MW, 500 MW and 1000 MW. As

[120] 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.

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 a 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 at 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.

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 further decrease NOx emissions by either (a) upgrading or replacing their existing NOx emissions reducing systems, or (b) operating their existing NOx systems for more hours in the year than the IPM model predicts they will in the IPM v. 5.14 base case[121] for 2025. While all existing coal-fired EGUs already have low NOx burners, there are EGUs that currently have a selective non-catalytic reduction (SNCR) post-combustion NOx control system that could be replaced with a selective catalytic reduction (SCR) system installed that would reduce their NOx emissions rate (and hence quantity of NOx) at those units.

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 in 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, such as steel, concrete, or chemicals used to manufacture NOx controls).

The analysis draws on information from seven primary sources:

- Documentation for EPA Base Case v.5.13 Using the Integrated Planning Model. November, 2013

[121] Note that the IPM v. 5.14 base case is not the base case used in the final Clean Power Plan analysis (using IPM v. 5.15), which is used for other analyses in this RIA. Furthermore neither the v. 5.14 nor the v 5.15 base cases used in this RIA include the illustrative estimated EGU responses to the Clean Power Plan. The base case, however, is only used in the labor analysis to select the three illustrative sizes of EGUs in the model plant estimates

- IPM updates included in V. 5.14 EPA Base Case v.5.14 Using IPM: Incremental Documentation. March, 2015.
- "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 Clean Power Plan Final Rule". August, 2015.
- The National Electric Energy Data System (NEEDS) Version 5.14. March, 2015.
- EPA Base Case for IPM v. 5.14 estimates for 2025. March, 2015

5.3.1.1 Existing EGUs Without SCR Systems (or SCR Systems Operating less than Full Time)

Using EPA's National Electric Energy Data System (NEEDS) v. 5.14 and the IPM 2025 estimates from the v 5.14 base case, the EPA identified all existing coal-fired EGUs in the contiguous United States that:

(1) The IPM 5.14 base case estimates will be operating in 2025.

(2) Either do not already have an SCR NOx emission control system installed, or have an SCR NOx system that could be utilized more to further reduce NOx emissions.

Coal-fired units that currently have an SNCR system but not an SCR system were identified directly from NEEDS 5.14, which includes detailed information on the type of emissions control systems that are installed. NEEDS 5.14 also includes the NOx emission rate (lbs/MMBtu) that the unit could achieve if the required state of the art (SOA) NOx emission controls were operated.[122] The 2025 estimates from IPM v. 5.14 base case, in combination with the SOA NOx rates from NEEDS, were used to determine which units with an SCR installed were capable of lowering their annual NOx emissions by operating the SCR as often as possible. If IPM estimated that the predicted NOx annual emissions in 2025 exceeded the amount of NOx that would be emitted if the 2025 quantity of coal was consumed and the unit achieved the SOA NOx emission rate from NEEDS, then the unit is assumed to be operating in 2025 running the SCR to its maximum potential all the time.

Furthermore, the EPA identified the subset of these EGUs that are in areas anticipated to need additional NOx reductions under an alternative ozone standard level of 65 ppb, as well as

[122] Mode 4 NOx emission rate. For more details see the NEEDS 5.14 documentation page 6, and IPM v. 5.13 User Guide Section 3.9.2.

the smaller subset of the EGUs that are in areas anticipated to need additional NOx reductions under a 70 ppb ozone standard.

The EPA identified 319 existing coal-fired EGUs nationwide (total capacity 122.4 GW) that are estimated to continue to be in operation in the 2025 base case that either do not already have an SCR system (30 units, 5.4 GW) or have an SCR system that is not working full time (289 units, 117.1 GW) . These 319 EGUs include units both in areas anticipated to need to reduce NOx emissions (37 units, 8.5 GW) with a 65 ppb ozone NAAQS, as well as the EGUs in all other areas (282 units, 114.0 MW). All of the 30 EGUs that do not have an SCR are in areas expected to need to reduce NOx emissions with a 65 ppb NAAQS (30 units, 5.4 GW). The areas that are expected to need to reduce NOx emissions with a 65 ppb NAAQS also have 7 units (3.1 GW) have an SCR system not working full time.

Upgrading the 30 EGUs to an SCR emission control system will reduce NOx emissions in areas expected to need additional NOx controls with a 65 ppb NAAQS by a total of 41,400 tons. Operating the 7 EGUs with SCRs system estimated to not be operated as much as possible in the 65 ppb ozone NAAQS areas an additional 18,300 tons, for a combined total of 59,700 tons of NOx reduced annually.

Of the 37 EGUs estimated to be able to reduce NOx emissions in the 65 ppb NAAQS areas, 5 EGUs (1.7 GW) are also in the areas needing additional NOx reductions under a revised 70 ppb ozone NAAQS standard. Installing SCRs on 2 of these units, and running the existing SCRs as much as possible on the other 3 units, a total of 11,200 tons of NOx in the 70 ppb ozone NAAQS areas.

Given the nationwide size distribution of the existing EGUs that do not already have an SCR, or do not operate the existing SCR system full time, we present the illustrative labor analysis for three different sized “model plants”: 300 MW capacity, 500 MW, and 800 MW. These three capacity sizes of model plants were selected by examining the distribution of existing coal-fired EGUs that can either be upgraded to an SCR, or have the existing SCR operated to as much as possible. Because of the relatively small number (37) of the identified EGUs in the 65 ppb areas, the distribution of 319 EGUs nationwide better reflects the distribution of EGUs identified. Figure 5-1 shows the nationwide capacity size distribution of the 319 units. For comparison Figure 5-2 shows the size distribution of the 37 units in the 65 ppb ozone NAAQS areas. As can be seen in Figure 5-1, the 300 MW capacity “model plant” is the most common candidate for installing an SCR system in the 65 ppb areas, but 500 MW and 800 MW units (which are common in the national set of 319 identified units) also exist in the 65 ppb areas.

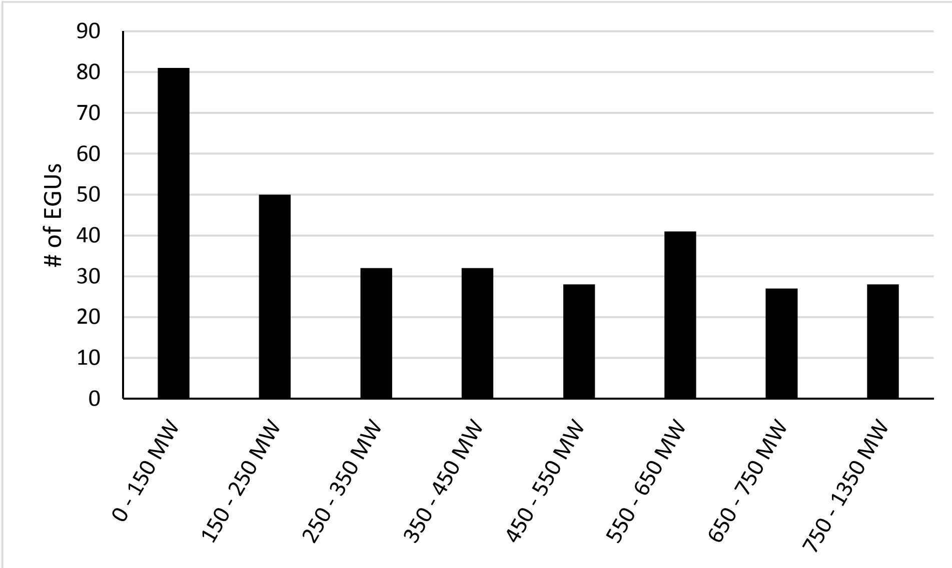

Figure 5-1. Size Distribution of Identified 319 Existing Coal-Fired EGU Units Nationwide without SCR NOx Controls (or with SCRs Operated Less Than the Maximum Possible Amount of Time)

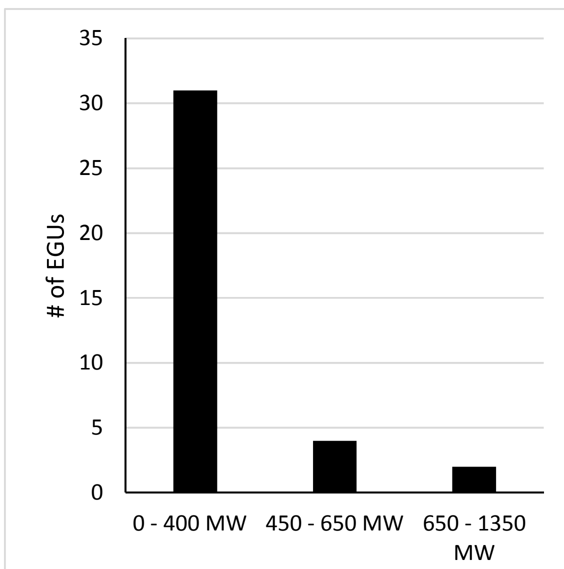

Figure 5-2. Size Distribution of 37 Existing Coal-Fired EGU Units without SCR NOx Controls (or with SCRs Operated Less Than the Maximum Possible Amount of Time) in Areas Anticipated to Need Additional NOx Controls With the Alternative 65 ppb Ozone Standard Level

5.3.1.2 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) and labor to manufacture the SCR .

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 5-1.

Table 5-1. Summary of Direct Labor Impacts for SCR Installation at EGUs (FTEs)

	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)			
Operation and Maintenance	1.9	2.8	4.6

The key assumptions used in the labor analysis are presented in Table 5-2.

Table 5-2. 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 labor hours/MW**	**Staudt, 2011**	**158.7 FTEs**	**264.4 FTEs**	**528.9 FTEs**

Assumptions	Key Factor	Source	300 MW	500 MW	1000 MW
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 FTEs**	**2.76 FTEs**	**4.57 FTEs**

5.3.2 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, and also in an illustrative analysis, the EPA estimated the amount and types of direct labor that might be used to apply and operate NOX controls for representative categories of ICI boilers and 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. 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).

5.3.2.1 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.

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 (SC&A, 2014).

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 (http://nepis.epa.gov/Adobe/PDF/P1005ODM.pdf). 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, however.. Table 5-3 below provides summary labor estimates for SCR at varying sized ICI boilers.

Table 5-3. 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

Similar to the calculations for SCR applied to EGUs, an illustrative analysis based on engineering costs estimates labor requirements in upstream sectors for SCR applied to representative ICI boilers. This includes labor requirements in selected major upstream sectors directly involved in manufacturing the materials used in SCR systems (steel), as well as the chemicals used to operate an SCR system (ammonia and the catalyst used in the construction and operation of SCR systems, such as steel, concrete, or chemicals used to manufacture NOx controls). For an SCR installed at a representative 750 MMBtu/hr coal boiler, we estimate one-time employment impacts in these related sectors as:of 8.6 FTE, annual recurring impacts of 2.17 FTE for an SCR operated year-round, and annual recurring impacts of 0.90 FTE for an SCR operated during the five-month ozone season (SC&A, 2014).

5.3.2.2 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.

The capital costs for equipment supply fabrication were estimated for an SNCR system for a mid-sized preheater and precalciner kiln (125 to 208 tons of clinker per hour). The percent of equipment supply fabrication costs of these systems attributable to labor is 44%. (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 were estimated by the vendor to be 17% of the total cost. That was converted to FTE using BLS data. This information is summarized in Table 5-4.

Table 5-4. Estimated Direct Labor Impacts for Individual SNCR Applied to a Mid-Sized Cement Kiln (125-208 tons clinker/hr)

Kiln Type	Preheater / Precalciner
Equipment Supply Fabrication FTE	1.5
Installation FTE	0.9
O&M Annual Recurring FTE	0.1

In addition, an illustrative analysis of labor requirements in upstream sectors for SNCR applied to cement kilns include the labor requirements in selected major upstream sectors directly involved in manufacturing the materials used in SNCR systems (steel), as well as the reagent used to operate an SNCR system. For an SNCR installed at a 3,000 tons clinker per day capacity cement kiln, we estimate employment impacts in these related sectors as one-time employment impacts of 0.22-0.34 FTE, annual recurring impacts of 0.41 FTE for an SNCR operated year-round, and annual recurring impacts of 0.28 FTE for an SNCR operated during the five-month ozone season (SC&A, 2014).

5.4 Conclusion

This chapter presents qualitative and quantitative discussions of potential employment impacts of the Ozone NAAQS. The qualitative discussion identifies challenges associated with estimating net employment effects and discusses anticipated impacts related to the rule. It includes an in-depth discussion of economic theory underlying analysis of employment impacts. The employment impacts for regulated firms can be decomposed into output and substitution effects, both of which may be positive or negative. Consequently, economic theory alone cannot predict the direction or magnitude of a regulation’s employment impact. It is possible to combine theory with empirical studies specific to the regulated firms and other relevant sectors if data and methods of sufficient detail and quality are available. Finally, economic theory suggests that environmental regulations may have positive impacts on labor supply and productivity as well.

We examine the peer-reviewed economics literature analyzing various aspects of labor demand, relying on the above theoretical framework. Determining the direction of employment effects in regulated industries is challenging because of the complexity of the output and substitution effects. Complying with a new or more stringent regulation may require additional inputs, including labor, and may alter the relative proportions of labor and capital used by regulated firms (and firms in other relevant industries) in their production processes. The available literature illustrates some of the difficulties for empirical estimation. Econometric studies of environmental rules converge on the finding that employment effects, whether positive or negative, have been small in regulated sectors.

The illustrative quantitative analysis in this chapter projects a subset of potential employment impacts in the electricity generation sector and for ICI boilers and cement kilns. States have the responsibility and flexibility to implement plans to meet the Ozone NAAQS. As such, given the wide range of approaches that may be used, quantifying the associated employment impacts is difficult. This analysis presents employment impact estimates based on 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 for EGUs, ICI boilers and cement kilns.

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CHAPTER 6: HUMAN HEALTH BENEFITS ANALYSIS APPROACH AND RESULTS

6.1 Summary

This chapter of the Regulatory Impact Analysis (RIA) presents the estimated human health benefits for the revised 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 emissions control strategies that reduce emissions of the ozone precursor pollutants (i.e., nitrogen oxides (NO_x) and volatile organic compounds (VOCs)) to reach the revised and alternative ozone NAAQS standard levels. We also estimate the health co-benefits of the fine particulate matter ($PM_{2.5}$)-related air quality improvements that would occur as a result of reducing NO_x emissions.[123]

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. The benefits of each standard alternative are estimated as being incremental to attaining the existing standard of 75 ppb.[124] These estimated benefits are incremental to the benefits estimated for several recent rules (e.g., U.S. EPA, 2011c and U.S. EPA, 2014a). 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 later than the rest of the U.S. In this chapter, 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.

Table 6-1 summarizes the estimated monetized benefits (total and ozone only) of attaining the revised and alternative ozone standards of 70 ppb and 65 ppb, respectively, in 2025. Table 6-2 presents the same types of benefit estimates for the scenario. These estimates reflect the sum of the economic value of estimated morbidity and mortality effects related to changes in exposure to ozone and $PM_{2.5}$. Although these tables present ozone and $PM_{2.5}$-related benefits

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[123] VOC reductions associated with simulated attainment of the revised and alternative ozone standards also have the potential to impact $PM_{2.5}$ concentrations, but we were not able to estimate those effects.

[124] The current standard is the 4th highest daily maximum 8-hour ozone concentration of 75 ppb.

separately, it is not appropriate to compare the ozone-only benefits to total costs. Reduced levels of NOx emissions needed to attain a more stringent ozone standard will affect levels of both ozone and $PM_{2.5}$. Following the standard practice for assessing the benefits of air quality rules (OMB, 2003; U.S. EPA, 2010e), this RIA quantifies the benefits of reducing both pollutants. For this reason, the costs of attaining a tighter standard should be compared against the sum of the ozone and $PM_{2.5}$ benefits.

Compared with benefit estimates generated in the proposal RIA, the total benefit estimates generated for the 2025 scenario are ~55% lower for the revised standard (70 ppb) and ~22% lower for the alternative standard (65 ppb). Benefit estimates for the post-2025 scenarios are slightly higher than those generated at proposal (~6% for the revised standard and ~2% for the alternative standard). The proposal and final RIA estimates differ principally because as discussed in Chapter 2, Section 2.4.2 and Chapter 4, Section 4.6, the additional emissions sensitivity simulations and more refined ozone response factors allowed us to more accurately represent the increased effectiveness of emissions reductions closer to some monitor locations. The more refined air quality modeling resulted in approximately 50 percent fewer emissions reductions needed to reach a revised standard of 70 ppb and approximately 20 percent fewer emissions reductions needed to reach an alternative standard of 65 ppb. We have also slightly modified our approach to estimating morbidity benefits, which had a negligible (~1%) influence on the total monetized benefits in this RIA (see sections 6.3 and 6.6.3).

Table 6-1. Estimated Monetized Benefits of Attainment of the Revised and Alternative Ozone Standards for *2025* (nationwide benefits of attaining the standards everywhere in the U.S. except California) (billions of 2011$)[a]

	Discount Rate	70 ppb	65 ppb
Ozone-only Benefits [c]	[b]	\$1.0 to \$1.7	\$5.3 to \$8.7
$PM_{2.5}$ Co-benefits of NOx Reductions [d]	3%	\$2.1 to \$4.7	\$10 to \$23
	7%	\$1.9 to \$4.2	\$9.3 to \$21
Total Benefits	3%	\$3.1 to \$6.4[e]	\$16 to \$32[e]
	7%	\$2.9 to \$5.9[e]	\$15 to \$30[e]

[a] Rounded to two significant figures. It was not possible to quantify all benefits in this analysis due to data limitations. 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.
[c] Range reflects application of effect estimates from Smith et al. (2009) and Zanobetti and Schwartz (2008).
[d] Range reflects application of effect estimates from Krewski et al. (2009) and Lepeule et al. (2012).
[e] Excludes additional health and welfare benefits which could not be quantified (see section 6.6.3.8).

Table 6-2. Estimated Monetized Benefits of Attainment of the Revised and Alternative Ozone Standards for *post-2025* (nationwide benefits of attaining the standards just in California) (billions of 2011$)[a]

	Discount Rate	70 ppb	65 ppb
Ozone-only Benefits [c]	[b]	$0.79 to $1.3	$1.6 to $2.6
$PM_{2.5}$ Co-benefits of NOx reductions [d]	3%	$0.40 to $0.91	$0.79 to $1.8
	7%	$0.37 to $0.82	$0.71 to $1.6
Total Benefits	3%	$1.2 to $2.2 [e]	$2.4 to $4.4 [e]
	7%	$1.2 to $2.1 [e]	$2.3 to $4.2 [e]

[a] Rounded to two significant figures. It was not possible to quantify all benefits in this analysis due to data limitations. These estimates reflect the economic value of avoided morbidities and premature mortalities 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.
[c] Range reflects application of effect estimates from Smith et al. (2009) and Zanobetti and Schwartz (2008).
[d] Range reflects application of effect estimates from Krewski et al. (2009) and Lepeule et al. (2012).
[e] Excludes additional health and welfare benefits which could not be quantified (see section 6.6.3.8).

The control measures (identified and unidentified) applied to reach the revised and alternative ozone standards would reduce other ambient pollutants, including 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 unable to quantify the co-benefits of reduced exposure to these pollutants. Due to limited data and methods, we were unable to estimate some anticipated health benefits associated with exposure to ozone and $PM_{2.5}$.

6.2 Overview

This chapter presents estimated health benefits for the revised and alternative ozone standards (70 ppb and 65 ppb, respectively) that the EPA could quantify, given the available resources, data and methods. This chapter characterizes the benefits of the application of the identified and unidentified control strategies identified in Chapter 3 for the revised and alternative ozone standards by answering three key questions:

1. What health effects are estimated to be avoided by reducing ambient ozone levels to attain the revised and alternative ozone standards?

2. What is the estimated 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 associated with to ozone and $PM_{2.5}$ that would be avoided by attaining a revised ozone standard. 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 mortality 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 mortality and a variety of illnesses associated with acute and chronic exposures. Air pollution can affect human health in a variety of ways. In Table 6-3 we summarize the "categories" of effects and describe those that we quantified for this analysis and those we were unable to quantify due to lack of resources, data, or methods.

This list of benefit categories is not exhaustive, and we are not always able to quantify each effect completely. In this RIA, we only quantify endpoints that are classified in the ozone and PM ISAs as being causally related, or likely to be causally related, to each pollutant. 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. This way of selecting endpoints for quantification should not be interpreted as a change in the level of evidence regarding the association between these endpoints and ozone (or $PM_{2.5}$) exposure.

This benefits analysis relies on an array of data inputs—including emissions estimates, modeled ozone concentrations, 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 6.5 and 6.7.3.

Table 6-3. 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	Source of More Information
Improved Human Health				
Reduced incidence of premature mortality from exposure to ozone	Premature mortality based on short-term exposure (all ages)	✓	✓	Section 6.6
	Premature respiratory mortality based on long-term exposure (age 30–99)	[b]	[b]	
Reduced incidence of morbidity from exposure to ozone	Hospital admissions—respiratory (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)	✓	✓	Section 6.6
	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)	—	—	
	Emergency department visits for cardiovascular effects (all ages)	—	—	
	Strokes and cerebrovascular disease (age 50–79)	—	—	

Benefits Category	Specific Effect	Effect Has Been Quantified	Effect Has Been Monetized	Source of More Information
	Other cardiovascular effects (e.g., other ages)	—	—	PM ISA[c]
	Other respiratory effects (e.g., pulmonary function, non-asthma ER visits, non-bronchitis chronic diseases, other ages and populations)	—	—	
	Reproductive and developmental effects (e.g., low birth weight, pre-term births, etc.)	—	—	PM ISA [c,d]
	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)	—	—	

[b] We quantified these benefits, but they are not part of the core monetized benefits.
[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.

The remainder of this chapter is organized as follows: Section 6.3 includes a discussion of the methodological updates represented in this analysis; Section 6.4 includes a discussion of the methodologies used in the human health benefits analyses; Section 6.5 includes a characterization of uncertainty; Section 6.6 details the data inputs used in the analysis, including demographic data, baseline incidence and prevalence estimates, effect coefficients, and economic valuation estimates; Section 6.7 presents the results; and Section 6.8 includes a brief discussion of the results. In addition, the chapter has several appendices that provide more details on the following: Appendix 6A includes a detailed characterization of uncertainty in the analysis; Appendix 6B includes additional quantitative analyses supporting uncertainty characterization.

6.3 Updated Methodology Presented in the Proposal and Final RIAs

Both the proposed and final RIAs for this ozone standard incorporate an array of policy and technical updates to the benefits analysis methods since the previous review the ozone standards in 2008 and the proposed reconsideration in 2010.

1. To be consistent with Agency guidance, EPA revised the Value of Statistical Life (VSL) it used to quantify the value of reduced mortality risk (see the NO_2 NAAQS final RIA (U.S. EPA, 2010a) for further discussion).

2. The population demographic data in BenMAP-CE (U.S. EPA, 2015a) 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 older demographic projection data from Woods and Poole (2007). This update was introduced in the final PM NAAQS RIA (U.S. EPA, 2012a).

3. 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, 2011c).

4. The cost-of-illness estimates for hospital admissions, including median wages, have been updated to reflect 2007 data. This update was introduced in the proposal PM NAAQS RIA (U.S. EPA, 2012a).

5. Updates specific to estimating ozone-related effects:

   a. *New studies used to quantify ozone-related premature mortality*.

      i. The ozone ISA (U.S. EPA, 2013a) identifies several new epidemiological studies examining the association between ozone exposure and premature mortality. We include two new multi-city studies to estimate premature mortality attributable to short-term exposure (Smith et al., 2009 and Zanobetti and Schwartz 2008). We also estimate long-term respiratory premature mortality using Jerrett et al. (2009).[125] We introduced this update in the proposal RIA (U.S. EPA, 2014c).

      ii. Following completion of the proposal RIA (U.S. EPA, 2014c), we slightly modified the methods applied in this RIA. As described in sections 6.6.3, we modified the set of epidemiology studies and associated effect coefficients used in estimating changes in asthma exacerbation and respiratory hospital admissions associated with ozone exposure. Because these changes do not involve our approach for estimating air pollution-related premature mortality,

[125] Because we do not have information on the cessation lag for premature mortality from long-term ozone exposure, we do not include the monetized benefits in the core analysis. Instead, monetized benefits associated with long-term ozone-related respiratory mortality are included as a sensitivity analysis (see Appendix 6B, section 6B.2).

which largely drives overall monetized benefits, they have a negligible (~1%) impact on total monetized benefits (see section 6.6.3 for further discussion).

b. *New studies used to quantify ozone-related morbidity effects*. 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. We introduced this update in the proposal RIA (U.S. EPA, 2014c).

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 introduced this expanded assessment in the proposal RIA (U.S. EPA, 2014c).

6. Updates specific to estimating $PM_{2.5}$-related effects:

d. When estimating $PM_{2.5}$-related health effects, EPA no longer assumed a minimum concentration at which no effects occurred, while still reporting a range of sensitivity estimates based on the EPA's $PM_{2.5}$ mortality expert elicitation (see the Portland Cement NESHAP proposal RIA (U.S. EPA, 2009a) for further discussion).

e. *New studies used to quantify $PM_{2.5}$-related premature mortality*. 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 update for the American Cancer Society cohort was introduced in the proposal RIA for the PM NAAQS review (U.S. EPA, 2012a) and the update for the Harvard Six Cities cohort was introduced in the final RIA for the PM NAAQS review (U.S. EPA, 2012c).

f. *New studies used to quantify $PM_{2.5}$ morbidity*. Based on the PM ISA and PM Provisional Assessment, we added several new studies and morbidity endpoints to our health impact assessment, including hospital admissions and emergency department visits. These updates were introduced in the proposal (U.S. EPA, 2012a) and final RIAs for the PM NAAQS review (U.S. EPA, 2012c).

g. *More recent 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 premature mortality 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, 2012c).

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

6.4 Human Health Benefits Analysis Methods

We follow a "damage-function" approach in calculating total benefits of the modeled changes in environmental quality. This approach estimates changes in individual health endpoints (i.e., 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 that EPA uses for assessing costs and benefits of environmental quality policies and has been used in several recent analyses published in the peer reviewed scientific literature as well (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 valued directly, as is the case for changes in visibility. In other cases, such as for changes in health outcomes associated with reductions in ozone and PM concentrations, 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 (HIA) is limited to those health effects that the ISA identified as causally or likely causally linked to ambient levels of 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 being analyzed. 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 the revised and alternative standard levels) were explicitly modeled and then translated into reductions in the incidence of specific health endpoints (see Chapter 2 for more information). In contrast, in estimating $PM_{2.5}$ co-benefits we utilized a reduced form approach. The details of both approaches are described in additional detail below.

6.4.1 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, 2015a). 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, 2014a). 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 applying emissions controls, such as occupational health exposures.

The HIA approach used in this analysis involves three basic steps: (1) using projections of ozone air quality[126] and estimating the change in the spatial distribution of the ambient air quality; (2) determining the subsequent change in population-level exposure; and (3) calculating health impacts by applying concentration-response (C-R) 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:

$$\Delta y = 1 - \left(e^{\beta \cdot \Delta x}\right) y_o \cdot Pop \qquad (6.1)$$

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 6-1 provides a simplified overview of this approach. Additional detail on the specific types of C-R functions (utilizing epidemiology-based effect estimates) involved in benefits modeling can be found in the Appendix C of the *User's Manual Appendices* supporting BenMAP-CE (U.S. EPA, 2015b). Specific effect estimates used in this RIA are documented in section 6.6.3.

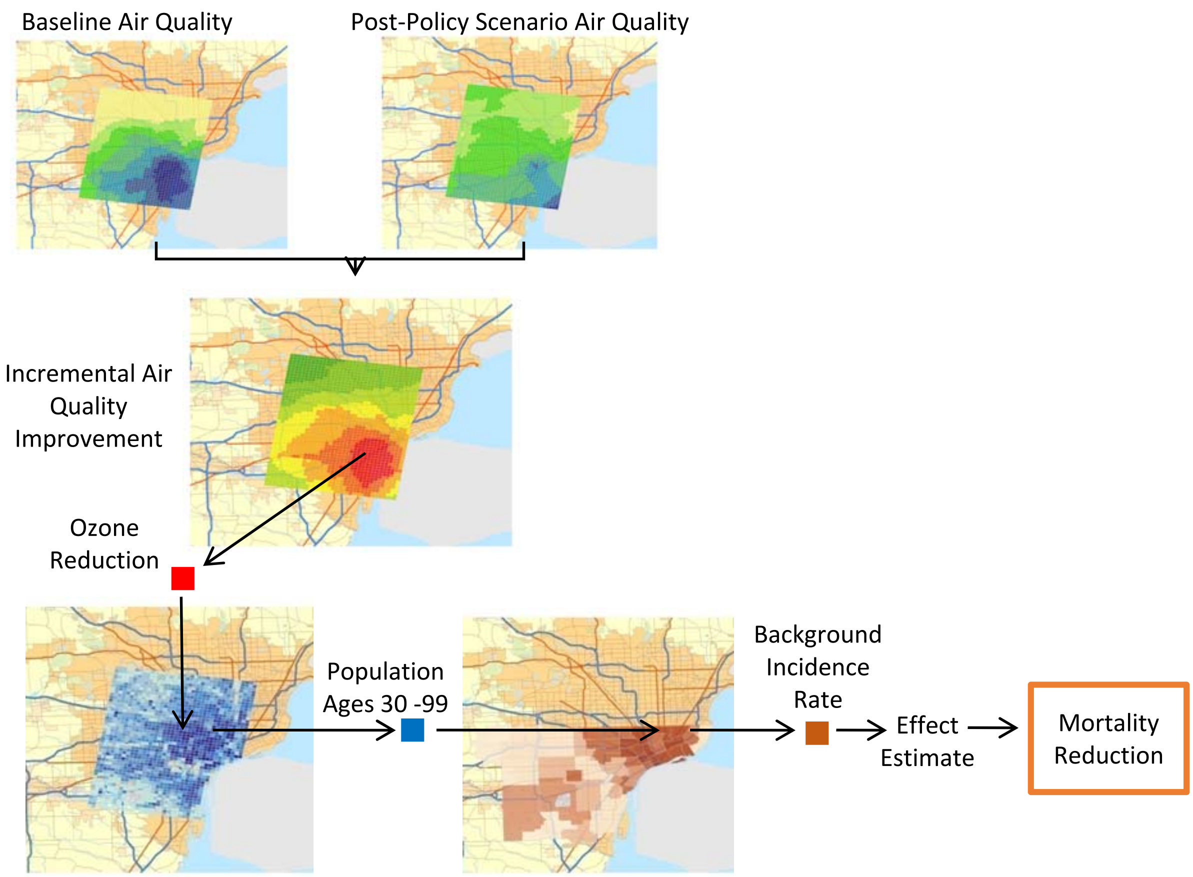

Figure 6-1. Illustration of BenMAP-CE Approach

6.4.2 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 changes in risk for 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 1.1 (U.S. EPA, 2015a, 2015b) to estimate the health impacts and monetized health benefits for the standards evaluated here. 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.[127] Figure 6-2 shows the data inputs and outputs for the BenMAP-CE program.

[127] As compared to the version that it replaces (BenMAP v4), BenMAP-CE uses the same computational algorithms and input data to calculate benefits 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; and (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 *Health Risk and Exposure Assessment for Ozone* (ozone HREA) (U.S. EPA, 2014b).

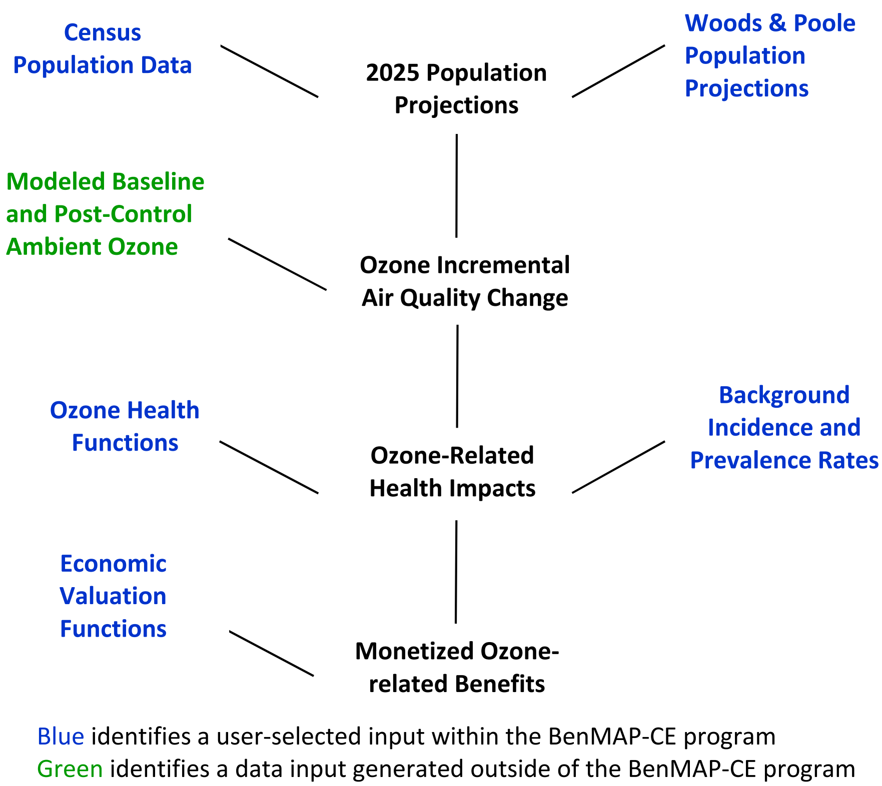

Figure 6-2. Data Inputs and Outputs for the BenMAP-CE Program

6.4.3 Estimating Benefits for 2025 and Post-2025 Analysis Years

Portions of the country with more significant air quality problems (particularly several areas in California) may not be required to meet the revised standard until as late as December 31, 2038. Consequently, for the revised and alternative standards we evaluate a 2025 scenario reflecting the nationwide benefits of attaining the standards everywhere in the U.S. except California. We then evaluate a post-2025 scenario, which represents nationwide benefits from attaining the standards in California. We report the 2025 and 2038 estimates separately because deriving a summed estimate would require us to calculate the Present Value (PV) of the stream of benefits occurring between those two years, which is not possible with the available data.

Our approach for estimating the benefits of attaining the revised and alternative ozone standards post-2025 is illustrated in Figure 6-3. In this figure, Simulation A represents our approach for estimating the benefits of attaining the revised and alternative ozone standards in every state except California in 2025. To estimate benefits for the post-2025 scenario, we first estimated the benefits occurring in 2038 from all areas (including California) attaining the revised and alternative standards (Simulation C). Next, we simulated the nationwide benefits of attaining the revised and alternative ozone standards in 2038 for every state except California (Simulation B). We then subtracted Simulation B from Simulation C to calculate the benefits of attaining the revised and alternative ozone standards after 2025— that is, to calculate the nationwide benefits from California alone attaining the standards in 2038. Important caveats are associated with this approach mentioned in Section 6.1 and discussed further in section 6.7.3.

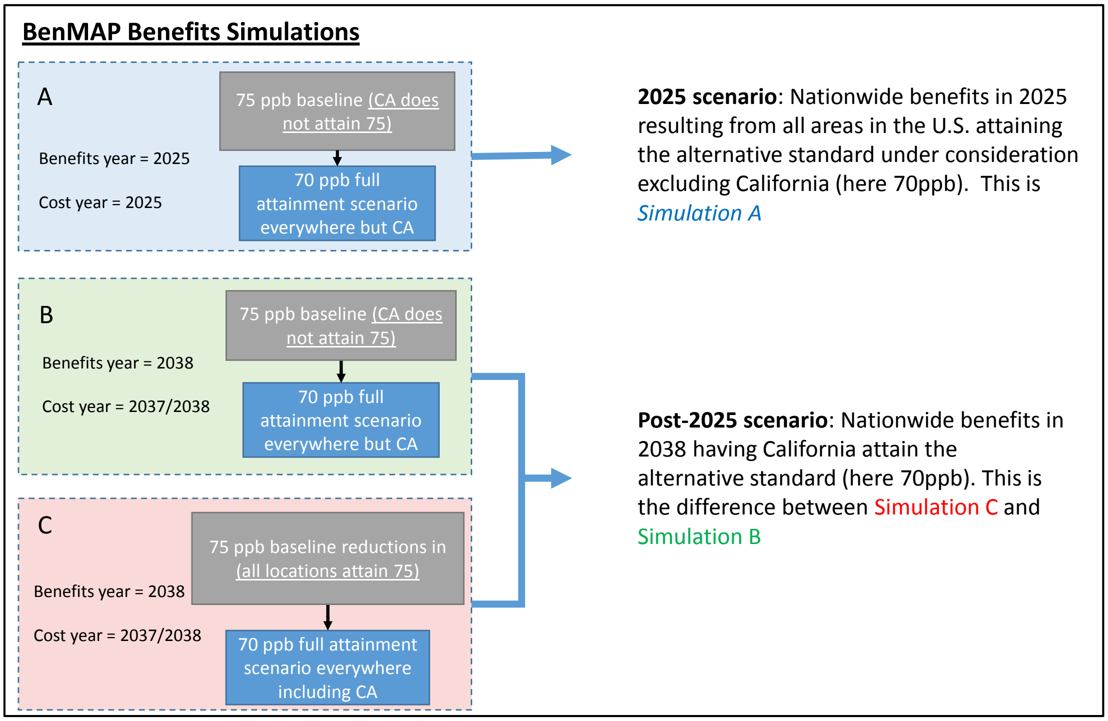

Figure 6-3. Procedure for Generating Benefits Estimates for the 2025 and Post-2025 Scenarios

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

We used a reduced form approach to estimate the $PM_{2.5}$ co-benefits in this RIA due to data and resource constraints. Specifically, we used "benefit-per-ton" estimates. 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 (reflecting the sum of premature mortality and morbidity effects) of reducing one ton of NOx (as a $PM_{2.5}$ precursor) from a specified source. In general, these estimates apply the same benefits methods (e.g., health impact assessment then economic valuation) 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.

We used 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 RIA were derived using the approach published in Fann et al. (2012b), but have since been updated to reflect the epidemiology studies and Census population data first applied in the final $PM_{2.5}$ NAAQS RIA (U.S. EPA, 2012c). The approach in Fann et al. (2012b) 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).

In Section 6.6, we describe all of the data inputs used in deriving the benefit-per-ton values for each sector, including the demographic data, baseline incidence, and valuation functions. The benefit-per-ton estimates (by sector) that resulted from this modeling are presented in Section 6.6.5. We then multiply these benefit-per-ton estimates for each sector with the NOx emission reductions from that sector to estimate the $PM_{2.5}$ co-benefits.[128] Additional information on the source apportionment modeling for each of the sectors can be found in Fann et al. (2012b) and the TSD (U.S. EPA, 2013b). 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 scenarios, respectively.[129]

Applying benefit-per-ton estimates introduces uncertainty, which we discuss in section 6.7.3 and Appendix 6A. 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 $PM_{2.5}$-related health impacts (e.g., population density,

[128] For unidentified controls, we use a weighted average of the benefit-per-ton estimates from all of the sectors.
[129] 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.

baseline health incidence rates). Therefore, using 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 addition, the use of national benefit-per-ton estimates results in a known underestimation bias in California in the post-2025 scenario due to population density.

6.5 Characterizing Uncertainty

In any complex analysis using estimated parameters and inputs from numerous models, there are likely to be many sources of uncertainty, and this analysis is no exception. This analysis includes many data sources as inputs, including emissions 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 (e.g., regulations, technology, and human behavior). Each of these inputs may be uncertain and would affect the benefits estimates. When the uncertainties from each stage of the analysis are compounded, even small uncertainties can have large effects on the total quantified benefits.

After reviewing the EPA's approach, the National Research Council (NRC) (2002, 2008), which is part of the National Academies of Science, concluded that the EPA's general methodology for calculating the benefits of reducing air pollution is reasonable and informative in spite of inherent uncertainties. The NRC also highlighted the need to conduct rigorous quantitative analyses of uncertainty and to present benefits estimates to decision makers in ways that foster an appropriate appreciation of their inherent uncertainty. Since the publication of these reports, the EPA continues to improve how it characterizes uncertainty in health incidence and benefits estimates. In response to these recommendations, we incorporated additional quantitative and qualitative characterizations of uncertainty. Although we are not yet able to perform the probabilistic uncertainty assessment the NAS envisioned, we added several quantitative and qualitative analyses. These additional analyses characterize uncertainty related to estimated premature mortality, since this endpoint is assigned the largest dollar value. For other inputs into the benefits analysis, such as the air quality data, it is too difficult to address uncertainty probabilistically for this analysis due to the complexity of the underlying air quality models and emissions inputs.

To characterize uncertainty and variability, we follow an approach that combines elements from two recent analyses by the EPA (U.S. EPA, 2010b; 2014b), and use a tiered approach developed by the World Health Organization (WHO) (WHO, 2008). We present this assessment in Appendix 6A (results of these assessments are summarized in section 6.7.3). 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: Monte Carlo assessments, and additional quantitative analyses characterizing uncertainty (see Appendix 6B).

6.5.1 Monte Carlo Assessment

Similar to other recent RIAs, we used Monte Carlo methods for characterizing random sampling error associated with the C-R 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. We report confidence intervals for ozone-related benefits, but were unable to provide confidence intervals for $PM_{2.5}$-related co-benefits because we used the benefit-per-ton estimates.

6.5.2 *Quantitative Analyses Supporting Uncertainty Characterization*

Because over 90% of the monetized benefits are from avoided premature mortality, it is particularly important to characterize the uncertainties associated with reductions in premature mortality. Each of the quantitative analyses supporting uncertainty characterization for this RIA are briefly described below and section 6.7.3 provides a discussion of the results and observations stemming from these quantitative analyses.

- **Alternative C-R functions for short-term ozone exposure-related mortality:** Alternative C-R functions are useful for assessing uncertainty beyond random statistical error, including uncertainty in the functional form of the model or alternative study designs. We used two multi-city studies (Smith et al., 2009; Zanobetti and Schwartz 2008) to estimate short-term ozone-related mortality in our core estimate. We performed a sensitivity analysis using effect coefficients from additional multi-city studies and meta-analyses utilized in prior RIAs (Bell et al., 2004; 2005, Huang, 2005; Ito et al., 2005; Levy et al., 2005), as well as alternative model specifications from the Smith et al. (2009) study (see Figure 6-4 and Appendix 6B, section 6B.1). When selecting studies for the core and uncertainty-related analyses we considered NAS (p. 80, NRC, 2008) and CASAC recommendations (U.S. EPA-SAB, 2012, 2014).

- **Potential thresholds in the long-term ozone exposure-related respiratory mortality C-R function:** Consistent with the ozone HREA, we estimate premature respiratory deaths from long-term exposure to ozone. The Jerrett et al. (2009) study explored potential thresholds in the C-R function. We use the results of the threshold analyses conducted by Jerrett et al. (2009) to conduct a quantitative uncertainty analysis evaluating models with a range of potential thresholds in addition to a non-threshold (see Appendix 6B, section 6B.3).

- **Alternative C-R functions in estimating long-term $PM_{2.5}$ exposure-related mortality:** In estimating $PM_{2.5}$ co-benefits, we use two studies (Krewski et al., 2009; Lepeule et al., 2012). To better understand the C-R relationship between $PM_{2.5}$ exposure and premature mortality, the EPA conducted an expert elicitation in 2006 (Roman et al., 2008; IEc, 2006).[130] We apply the functions from the experts as a characterization of uncertainty (see Figure 6-5 and Appendix 6B, section 6B.2).

- **Cessation lag for long-term O_3 exposure-related respiratory mortality:** We do not know how long-term O_3 exposure-related respiratory deaths are distributed over time and so we use two lag structures originally developed for $PM_{2.5}$ (the 20-year segmented lag used for $PM_{2.5}$ and an assumption of zero lag) in this quantitative uncertainty analysis (see Appendix 6B, section 6B.2).

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

- **Income elasticity in the specification of willingness-to-pay (WTP) functions used for mortality and morbidity endpoints:** The degree to which the WTP function used in valuing mortality and some morbidity endpoints changes in proportion to future changes in income is uncertain. We evaluated the potential impact of this factor on the monetized benefits in a quantitative uncertainty analysis (see Appendix 6B, section 6B.5).

Even these multiple estimates (including confidence intervals, where available) 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 identifying a reasonable upper and lower bounds for input distributions. As a result, the reported confidence intervals and range of estimates give an incomplete picture of 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.

A number of analyses provide additional perspectives on the benefits results, including:

- **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 concentrations used in estimating premature mortality associated with short-term ozone concentrations:** We characterize the distribution of premature mortality attributed to short term ozone exposure with respect to baseline ozone concentrations in the subset of 12km grid cells where the analysis predicts the premature mortalities will be avoided.

- **Analysis of baseline $PM_{2.5}$ concentrations used in estimating short-term ozone exposure-related mortality:** We also include a similar plot of the baseline annual $PM_{2.5}$ levels used in estimating $PM_{2.5}$ mortality from the earlier analysis that generated the benefit-per-ton values. This analysis is particularly important because, 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 $PM_{2.5}$ concentration in the epidemiological studies that are used to estimate the benefits.

- **Outdoor worker productivity**: In this analysis, we quantify the economic value of improved productivity among outdoor agricultural workers using Graff Zivin and Neidell (2012) in our uncertainty analysis.

6.5.3 *Qualitative Assessment of Uncertainty and Other Analysis Limitations*

To more fully address 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, 2011a, 2012a, 2012b). 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 qualitative uncertainty characterization are available in Appendix 6A.

6.6 Benefits Analysis Data Inputs

In Figure 6-2 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). As noted above, only slight modifications have been made to the epidemiology studies used to estimate two morbidity endpoints since proposal, and those modifications are described in section 6.6.3.

A brief note regarding the spatial scale associated with benefits modeling completed for this RIA: when quantifying health impacts for the ozone RIA, we apply effect coefficients from air pollution epidemiology studies among populations of various ages—either the entire population (i.e., ages 0-99) or a subset (e.g., ages 65-99). These age ranges generally correspond to those reported in the epidemiological study, though (following NRC guidance) we sometimes assign these effect coefficients to a slightly broader age range. We apply a single effect coefficient to populations throughout the United States and do not differentiate by region. The health impact functions used to quantify risk also specify population counts and baseline rates of disease or death, and most of these values are age and sex stratified, allowing us to report incidence among population subgroups.

6.6.1 Demographic Data

Quantifying the incidence and dollar value of pollution impacts requires information regarding the demographic characteristics of the exposed 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 projected population by age, sex, and race to 2040, relative to a baseline using the 2010 Census data; the 2008 ozone NAAQS 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-based" 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 by race for each year through 2040 based on historical rates of mortality, fertility, and migration.

6.6.2 Baseline Incidence and Prevalence Estimates

Epidemiological studies of the association between pollution levels and adverse health effects generally provide an 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 5 ppb decrease in 8-hour maximum daily ozone concentration 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 in that location.

Table 6-4 summarizes the sources of baseline incidence rates and provides national average (where used) 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 C-R functions to individual age groups and then summed over the relevant age range to estimate total population benefits. In many cases we used a single national incidence rate, due to a lack of more spatially disaggregated data; in these cases, whenever possible we used national average rates, 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 (U.S. EPA, 2015b). 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 (U.S. EPA, 2015b; 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, 2011c). In addition, we previously 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 symptoms and illnesses identified in the ISA as associated with ozone and $PM_{2.5}$. Also, the updated 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 the assumption that most air pollution-related hospital visits associated with ozone and $PM_{2.5}$ are likely to be unscheduled. If a portion of scheduled hospital admissions are air pollution-related, 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 6-5 lists the prevalence rates used to determine the applicable population for asthma symptoms. Note that these reflect recent asthma prevalence and assume no change in prevalence rates in future years. We last updated these rates in the CSAPR RIA (U.S. EPA, 2011c).

Table 6-4. Baseline Incidence Rates and Population Prevalence Rates for Use in Impact Functions, General Population

Endpoint	Parameter	Rates: Value	Rates: Source
Mortality	Daily or annual mortality rate projected to 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[c]	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 [d]	0.0012	Schwartz et al. (1994, Table 2)
Upper Respiratory Symptoms	Daily upper respiratory symptom incidence among asthmatic children	0.3419	Pope et al. (1991, Table 2)
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 International Classification of Diseases (ICD) codes (AHRQ, 2007).
[c] The incidence of exacerbated asthma was quantified among children of all races, using the baseline incidence rate reported in Ostro et al. (2001).
[d] Lower respiratory symptoms are defined as two or more of the following: cough, chest pain, phlegm, and wheeze.

Table 6-5. 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).

6.6.3 Effect Coefficients

In this section, we describe our general process for selecting effect coefficients from epidemiology studies. The first step in selecting effect coefficients is to identify the health endpoints to be quantified. We based our selection of health endpoints on consistency with the EPA's ISAs, with input and advice from the SAB-HES.[131] In addition, we 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, 2012d).[132] In selecting health endpoints for ozone, we also considered the suite of endpoints included in core modeling for the ozone HREA, which was supported by CASAC (U.S. EPA-SAB, 2012, 2014). In general, we follow a weight-of-evidence approach, based on the biological plausibility of effects, availability of C-R 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

[131] The SAB-HES is 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).

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

studies provide direct C-R 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 model the relationship between ozone and $PM_{2.5}$ and adverse human health effects. We evaluated epidemiological studies using the selection criteria summarized in Table 6-6. 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, using C-R 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)[133] 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 (U.S. EPA, 2015b). 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

[133] In the case of mortality, we do not pool results. Instead, we provide the results from each study separately.

the random effects model.[134] 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.

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 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.

Table 6-6. Criteria Used When Selecting C-R Functions

Consideration	Comments
Peer-Reviewed Research	Peer-reviewed research is exclusively used to select C-R functions.
Study Type	*Prospective cohort vs. ecological*: 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:* Multi-city time series studies have advantages to meta-analyses. Multi-city studies use a consistent model structure and can include factors that explain differences between effect estimates among the cities. By contrast, meta-analyses can become imprecise and the results difficult to interpret due to the aggregation of large sets of studies. In addition, meta-analyses can suffer from publication bias, which can result in high-biased effect estimates. Although we generally prefer multi-city studies, we may consider meta-analyses if multi-city studies are not available.
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 would 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. Consequently, studies matching the ozone seasons in the air quality modeling are preferred.

[134] EPA recently changed the algorithm BenMAP used 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.

Consideration	Comments
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.
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 selection of a large population or through repeated observations on a smaller population (e.g., through a symptom diary recorded for a panel of asthmatic children).
Study Location	U.S. studies are more desirable than non-U.S. studies because of potential differences in pollution characteristics, exposure patterns, medical care system, population behavior, and lifestyle. Depending on the endpoint and the study, we may consider using Canadian studies. 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, co-pollutant models are preferred for estimating PM effects because this approach controls for the potential ozone effect while not diminishing the effective sample size available for specifying the PM effect. However, when estimating the ozone effect, the use of co-pollutant models (with PM) can substantially reduce sample size since only days with both ozone and PM can be used. While these co-pollutant 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 generally favor co-pollutant models in modeling PM benefits, for ozone we generally favor single pollutant models.
Measure of PM	In general, 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. Therefore, we generally do not include these effects in benefits analyses.
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 are drawn are shown in Tables 6-7 and 6-8. 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 2008 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 6-9 summarizes those health endpoints and studies we have included in quantitative analyses supporting uncertainty characterization.

Table 6-7. Health Endpoints and Epidemiological Studies Used to Quantify Ozone-Related Health Impacts [a]

Endpoint	Study	Study Population	Relative Risk or Effect Estimate (β) (with 95th Percentile Confidence Interval or SE)
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	*Jerrett et al. (2009)*	>29 years	β = 0.003971 (0.00133)
Hospital Admissions			
Respiratory	Pooled estimate: *Katsouyanni et al. (2009)*	> 65 years	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 exacerbation	Pooled estimate: [b]	6–18 years	
	Mortimer et al. (2002)		β = 0.00929 (0.00387)
	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] The original study populations were 5 to 12 years for Schildcrout et al. (2006) and 5-9 years for the Mortimer et al. (2002) study. Based on advice from the SAB-HES, we extended the applied population to 6-18 years 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 6-8. Health Endpoints and Epidemiological Studies Used to Quantify $PM_{2.5}$-Related Health Impacts [a]

Endpoint	Study	Study Population	Relative Risk or Effect Estimate (β) (with 95th Percentile Confidence Interval or SE)
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, 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/m³ RR = 1.21 (1–1.47) per RR = 1.13 (0.86–1.48) per 10 μg/m³
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/m³
Lower respiratory symptoms	Schwartz and Neas (2000)	7–14 years	OR = 1.33 (1.11–1.58) per 15 μg/m³

[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, 2012c).
[b] The original study populations were 8 to 13 years for the Ostro et al. (2001) study and 7 to 12 years for the Mar et al. (2004) study. Based on advice from the SAB-HES, we extended the applied population to 6-18 years, 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 6-9. Health Endpoints and Epidemiological Studies Used to Quantify Ozone-Related Health Impacts in Quantitative Analyses Supporting Uncertainty Characterization [a]

Endpoint	Study	Study Population	Effect Estimate (β) (with 95th Percentile Confidence Interval)
Premature Mortality			
Premature respiratory mortality - long-term	*Jerrett et al. (2009)-based models:*		
	- non-threshold ozone only (86 cities)		β=0.00266 (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	*> 29 years*	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) (co-pollutant model with PM_{10})*		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)	*All ages*	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)		β=0.00026 (0.00009)

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

6.6.3.1 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. A more biologically relevant metric, and the one used in the ozone NAAQS since 1997, is the maximum daily 8-hour average ozone. Thus, we 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 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.[135] The conversion ratios are based on observed relationships between the 24-hour average and 8-hour maximum 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).

As part of the quantitative uncertainty 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

[135] However, 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 6A).

al. (2005) all provide national conversion ratios between daily average and 8-hour and 1-hour maxima, based on national data.

6.6.3.2 Ozone Premature Mortality Effect Coefficients

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 mortality, even as we are mindful of the uncertainty associated with the shape of the concentration response curve at lower ozone concentrations. (U.S. EPA, 2013a, page 1-7). The 2013 ozone ISA concludes that the evidence suggests that ozone effects are independent of the relationship between PM and mortality. (U.S. EPA, 2013a). However, the ISA notes that the interpretation of the potential confounding effects of PM on ozone-mortality risk estimates requires caution due to the PM sampling schedule (in most cities) which limits the overall sample size available for evaluating potential confounding of the ozone effect by PM (U.S. EPA 2013a).[136] Below we describe the evolution of EPA's understanding of the evidence supporting causality related to short-term ozone exposure and mortality (including recommendations provided to EPA by the NAS).

These observations are consistent with prior recommendations to the EPA by the NAS regarding the quantification and valuation of ozone-related short-term mortality (NRC, 2008). Chief among the NAS recommendations 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*

[136] Consequently, as noted later in this section, while we have used a single-pollutant model in generating short-term ozone-related mortality benefits for this RIA, we have included a co-pollutants model as a sensitivity analysis to address potential uncertainty associated with this assumption of an independent ozone mortality effect.

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 SAB-HES panel, and the CASAC panel, we estimate 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 with several additional studies as part of the quantitative uncertainty analysis. CASAC supported using the Smith et al. (2009) and Zanobetti and Schwartz (2008) studies for the ozone HREA (U.S. EPA-SAB, 2012, 2014), and these are multi-city studies published more recently (as compared with other multi-city studies or meta-analyses included in the quantitative uncertainty analyses – see discussion below).

Smith et al. (2009) reanalyzed the NMMAPS dataset, evaluating the relationship between short-term ozone exposure and mortality.[137] While this study reproduces the core national-scale estimates presented in Bell et al. (2004), it also explored the sensitivity of the mortality effect to different model specifications including (a) regional versus national Bayes-based adjustment,[138] (b) co-pollutant models considering PM_{10}, (c) all-year versus ozone-season based estimates, and (d) consideration of a range of ozone metrics, including the daily 8-hour max. In addition, the Smith et al. (2009) study did not use the trimmed mean approach employed in the Bell et al. (2004) study in preparing ozone monitor data.[139] In selecting among the effect estimates from Smith et al. (2009), we focused on an ozone-only estimate for non-accidental mortality using the

[137] In previous RIAs involving ozone, we have used Bell et al. (2004) as the basis for modeling short-term exposure-related mortality. However, for reasons presented here and outlined in section 7.3.2 of the final REA completed in support of the ozone review (U.S. EPA 2014b), we have substituted Smith et al. (2009) as the basis for effect estimates used in modeling this endpoint for both the REA and the benefit analysis described here. The Smith et al. (2009) effect estimate used in this RIA (0.00032, see Table 6-7) is about 22% larger than the Bell et al. (2004) effect estimate (0.000261, BenMAP-CE standard health functions, U.S. EPA 2015a).

[138] 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 if 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.

[139] 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.

8-hour max metric for the warmer ozone season.[140] For the quantitative uncertainty analysis, we included a co-pollutant model (ozone and PM_{10}) from Smith et al. (2009) for all-cause mortality, using the 8-hour max ozone metric for the ozone season. Using a single pollutant model for the core analysis and the co-pollutant model in the quantitative uncertainty analysis reflects our concern that the reduced sampling frequency for days with co-pollutant measurements (1/3 and 1/6) could affect the ability of the study to characterize the ozone effect. This choice is consistent with the ozone ISA, which concludes that ozone effects are likely to be independent of the relationship between PM and mortality (U.S. EPA, 2013a).

The Zanobetti and Smith (2008) study evaluated the relationship between ozone exposure (using an 8-hour 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 C-R 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. We used the shorter day lag based C-R function since this had the strongest effect and tighter confidence interval. We converted the effect estimate from an 8-hour mean metric to an equivalent effect estimate based on an 8-hour 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.[141]

Mortality Effect Coefficient for Long-term Ozone Exposure. Although previous advice provided by the SAB-HES was to include long-term ozone exposure-related mortality only as part of a quantitative uncertainty analysis, based on the more recent ISA, CASAC advice,

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[140] The effect estimates used for both the core and uncertainty were obtained from Smith et al., 2009. Specifically, for the core analysis, we used the national-scale ozone-only summer 8-hour max based effect estimate and standard error (Smith et al., 2009 Table 1, row seven, columns seven and eight) and for the uncertainty analysis, we included the two-pollutant model summer 8-hour max effect estimate and standard error (Smith et al., 2009 Table 1, row ten, columns seven and eight). The model results presented in Table 1 of Smith et al. (2009) represent percentage rise in mortality per 10 ppb rise in the relevant metric of ozone and consequently these had to be adjusted to represent a factor increase per unit ozone.

[141] 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 8-hour 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., 8-hour max and 8-hour 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., 24-hour or 1-hour max ratios to 8-hour max equivalents).

and HREA, the current RIA estimated long-term ozone exposure-related respiratory mortality incidence in the core analysis.[142] Support for modeling long-term exposure-related mortality incidence comes from the ozone ISA as well as recommendations provided by CASAC in their review of the ozone HREA (U.S. EPA-SAB, 2014, p. 3 and 9), despite the lower confidence in quantifying this endpoint because the ISA's consideration of this endpoint is primarily based on one study (Jerrett et al, 2009), though that study is well designed, and because of the uncertainty in that study about the existence and identification of a potential threshold in the concentration-response function. Whereas the ozone ISA concludes that evidence is *suggestive of a causal association* between total mortality and long-term ozone exposure, specifically with regard to respiratory health effects (including mortality), the ISA concludes that there is *likely to be a causal association* (U.S. EPA, 2013a). Consistent with the ozone HREA, we use Jerrett et al. (2009) to estimate premature respiratory mortality from long-term ozone exposure. In review of the ozone HREA, CASAC concluded 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." (U.S. EPA-SAB, 2014).

The Jerrett et al. (2009) study was the first to explore the relationship between long-term ozone exposure and respiratory mortality (rather than other causes of mortality). Jerrett et al. (2009) exhibits a number of strengths including (a) the study was 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 included co-pollutant 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, attributes of this study affect how we interpret the long-term exposure-related respiratory mortality estimates. First, while CASAC notes that Jerrett et al. (2009) was well designed, it is a single study and provides the only quantitative basis for estimating this endpoint. By comparison, we estimate short-term exposure-related mortality risk using several studies.

[142] As explained in section 6.3, because we do not have information on the cessation lag for premature mortality from long-term ozone exposure, we do not include the monetized benefits in the core analysis. Instead, monetized benefits associated with long-term ozone-related respiratory mortality are included as a sensitivity analysis (see Appendix 5B, section 5B.2).

The quantitative uncertainty analysis take into consideration the potential existence and location of a threshold in the C-R function relating mortality and long-term ozone concentrations, which can greatly affect the results. CASAC concluded, "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" (U.S. EPA-SAB, 2014, p. 13-14)

Reflecting this CASAC advice in the context of the HREA, we estimate long-term exposure-related respiratory mortality using a non-threshold co-pollutant model (with $PM_{2.5}$) from Jerrett et al. (2009). Because the Jerrett et al. (2009) study uses seasonal average metrics (rather than shorter single day or multi-day lagged models as with time series studies studies), co-pollutants models obtained from Jerrett et al., (2009) are not affected by the lower $PM_{2.5}$ sampling rates. Using a co-pollutant model is consistent with this study applying seasonal average metrics that are insensitive to co-pollutant monitoring for $PM_{2.5}$. The effect estimates used to model long-term ozone-attributable mortality are calculated using a seasonal average of peak (1-hour 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 PM benefits. Therefore, combining long-term and short-term ozone-attributable mortality estimates could lead to double counting. Estimates of short-term ozone mortality are for all-causes, while estimates of long-term ozone mortality are for respiratory-related mortality only.

Quantitative uncertainty Analysis: Alternate Mortality Effect Coefficients for Short-term Ozone Exposure. Although we believe the evidence supports an ozone-only effect on short-term exposure-related mortality (as supported by the ozone ISA), we recognize limitations in the ability of studies to explore copollutants effects due to lower sampling frequency for PM relative to ozone. For that reason, we conduct a quantitative uncertainty analysis using the co-pollutants model (with PM_{10}) from Smith et al. (2009).

Quantitative uncertainty Analysis: Threshold-Based Effect Coefficients for Long-term Ozone Exposure. Consistent with the ozone HREA (U.S. EPA, 2014b), we explore the sensitivity of estimated ozone-related premature mortality to a concentration threshold using the

Jerrett et al. (2009) study.[143] 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. This supports the authors' assertion that improved predictions of respiratory mortality are generated when ozone is included in the model. 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 quantitative uncertainty analyses. Specifically, we generate additional long-term risk results using unique risk models 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.[144] In addition to exploring the impact of potential thresholds, as part of the quantitative uncertainty analysis we explore the impact of

[143] The approach in the ozone HREA to explore the potential for thresholds related to long-term exposure-related mortality is described in Sasser (2014). That memorandum also describes additional data obtained from the authors of Jerrett et al. (2009) to support modeling potential thresholds.

[144] 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.

using ozone-only (non-threshold) models in estimating long-term exposure-related respiratory mortality.[145]

6.6.3.3 $PM_{2.5}$ Premature Mortality Coefficients

The co-benefits associated with NOx reductions made in the models to achieve ozone reductions are estimated in this RIA using methods consistent with those develop for the RIA for the final 2012 $PM_{2.5}$ NAAQS. Below we provide additional background for readers who are not familiar with the $PM_{2.5}$ literature that drives the approach developed for the 2012 $PM_{2.5}$ NAAQS and employed by EPA since then.

$PM_{2.5}$ 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 NAAQS, which was twice reviewed by the SAB-CASAC (U.S. EPA-SAB, 2009a, 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 exposure to 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

[145] The set of ozone-only non-threshold effect estimates include (a) a value based on the 86 cities for which there are co-pollutant monitoring data for both ozone and $PM_{2.5}$ (this is best compared with the core estimate based on the co-pollutant 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).

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 National Research Council (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 noted 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 data 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 [is] 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, 2010c, 2011a, 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) refined the earlier ACS studies 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, 2010b) used risk coefficients drawn from the Krewski et al. (2009) study, the most recent reanalysis of the ACS cohort data. The PM risk assessment cited a number of advantages that informed the selection of the Krewski et al. (2009) study as the source of the core effect estimates, including the extended period of observation, the rigorous examination of model forms and effect estimates, the coverage for ecological variables, and the large dataset with over 1.2 million individuals and 156 MSAs (U.S. EPA, 2010b). The CASAC also provided extensive peer review of the PM risk assessment and supported the use of effect estimates from this study (U.S. EPA-SAB, 2009a, b, 2010b).

Consistent with the PM risk assessment (U.S. EPA, 2010b) which was reviewed by the CASAC (U.S. EPA-SAB, 2009a, b), 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 10 μg/m^3 increase in $PM_{2.5}$). The relative risk estimate (1.06 per 10 μg/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 C-R 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.

Given that monetized benefits associated with $PM_{2.5}$ are driven largely by reductions in premature mortality, it is important to characterize the uncertainty in this endpoint. In order to do so, we utilize the results of an expert elicitation sponsored by the EPA and completed in 2006 (Roman et al., 2008; IEc, 2006). The results of that expert elicitation can be used as a characterization of uncertainty in the C-R functions. The co-benefits results derived from expert elicitation is discussed in Appendix 6B (section 6B.4).

$PM_{2.5}$ 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.[146] The PM ISA states that less evidence is

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

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 mortality estimated for adults, avoided premature mortality 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). With regard to the cohort study conducted by Woodruff et al. (1997), the SAB-HES noted 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) but also 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 Benefits and Costs of the Clean Air Act 1990 to 2020* (U.S. EPA, 2011a), we continue to rely on the earlier 1997 study in part due to the national–scale of the earlier study.

6.6.3.4 Hospital Admissions and Emergency Department Visits

We pool together the incidence estimates using several different studies for many of the hospital admission endpoints. 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.

The ozone ISA states that 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 (U.S. EPA, 2013a). The ISA found no evidence of a threshold between short term ozone exposure and respiratory hospital admissions and ED visits, although there is increasing uncertainty at lower ozone concentrations particularly at and below 20 ppb (U.S. EPA, section 2.5.4.4). The ISA also observes that effect estimates remained robust to copollutants (U.S. EPA 2013a).

Considering these observation from the ISA and a thoroughly reviewing available epidemiological studies, we estimate respiratory hospital admissions (for 65-99 year olds) using an effect estimate obtained from Katsouyanni et al. (2009) and asthma-related emergency room visits (for all ages) using several single-city studies. Although Katsouyanni et al. (2009) provides effect estimates specific to the summer season, we adjusted the 1-hour max metric to the equivalent 8-hour max effect estimates.[147] The study provides summer season single pollutant effect estimates based both on natural and penalized splines. We have re-evaluated the choice of these models in this final RIA. In contrast to the proposal RIA, where we averaged both the penalized and natural spline estimates, we decided to focus on the penalized spline model

[147] Given that Katsouyanni et al. (2009) 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 8-hour metric.

because it displays a higher degree of precision, thus less potential for random error.[148] While Katsouyanni et al. (2009) included a set of effect estimates based on co-pollutant modeling (with PM_{10}), but we could not use them because they were based on the full year rather than the summer season.

A number of studies are available to model respiratory ED visits. Because we do not yet have the information needed to value this endpoint, we focused on the narrower category of asthma-related ED visits We used 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, Maine,, 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 $PM_{2.5}$).[149] In addition, two of the studies required adjustments to reflect the 8-hour max air metric. Specifically, Glad et al. (2012) uses the 1-hour max air metric, while Sarnat et al. (2013) used the 24-hour 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.

The two main groups of hospital admissions estimated in this analysis for $PM_{2.5}$ are: 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 exposure 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 estimates from studies 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, while Glad et al. examined populations 0-99 in Pittsburgh, Pennsylvania. Mar and colleagues perform their study in

[148] We evaluated the impact from this change (i.e., use of the penalized spline model alone as contrasted with an average of estimates generated using both the natural and penalized splines), and this change made a negligible (<<1%) difference in monetized ozone benefits due to the similarity in effect coefficients between the two models.

[149] While we have included co-pollutant models as sensitivity analyses for mortality, we did not include separate co-pollutant models for any of the morbidity endpoints as sensitivity analyses because morbidity endpoint represent a small fraction of the total monetized benefits.

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 exposure to $PM_{2.5}$, 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. Total cardiovascular hospital admissions are the sum of the pooled estimate for adults over 64 and the single study estimate for adults 20 to 64. 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, the Bell et al. (2008), and Peng et al. (2008, 2009) studies estimate a subset of cardiovascular hospitalizations as well as certain cerebro- and peripheral-vascular diseases. To address the potential for the pooling of these four studies to produce a biased estimate, we match the pooled risk estimate with a baseline incidence rate that excludes cerebro- and peripheral-vascular disease. An alternative approach would be to use the Zanobetti et al. (2009) study alone, though this would prevent us from drawing upon the strengths of the three multi-city studies.

To estimate avoided incidences of respiratory hospital admissions associated with $PM_{2.5}$, we used a number of studies examining total respiratory hospital admissions as well as asthma and chronic lung disease. We estimated impacts among three age groups: adults 65 and over, 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. We pool these two studies using equal weights. Moolgavkar et al. (2003) examines $PM_{2.5}$ and chronic lung disease hospital admissions (not including asthma) in Los Angeles, California 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.

6.6.3.5 Acute Health Events

A number of acute health effects not requiring hospitalization are also associated with exposure to ozone and $PM_{2.5}$. The sources for the effect estimates used to quantify these endpoints are described below.

Asthma exacerbations. For this RIA, we 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 and to avoid double counting impacts, incidence estimates derived from studies focused only on asthmatics cannot be added to estimates from studies that consider the full population. 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 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 (U.S. 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, for the proposed RIA, we selected three multi-city studies (Mortimer et al., 2002; O'Connor et al., 2008; Schildcrout et al., 2006). All three of these studies required the application of air metric ratios to adjust effect estimates to represent the 8-hour max metric.[150] In this final RIA, following a final review of our technical approach, we removed O'Connor (2008) due to concerns about potential exposure measurement error and residual confounding from meteorological variables in its 19-day lag structure. Current evidence in the ozone ISA suggests a more immediate effect with exposure to ozone for respiratory-related effects, such as hospital admissions and emergency department visits, with additional supporting evidence from studies of respiratory symptoms (U.S. EPA, 2013a). Consequently, the 19-day lag structure reflected in the O'Connor (2008) study is not as strongly supported by the evidence as the shorter lag structures associated with the other two asthma exacerbation studies.[151] In generating the asthma exacerbation estimate, we pool estimates from Mortimer et al. (2012) and Schildcrout et al. (2006) using a random/fixed effects approach applied within BenMAP-CE (U.S. EPA, 2015a).[152]

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 exposure to $PM_{2.5}$, 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 asthma exacerbation benefits.[153]

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

[150] Mortimer et al. (2002) had effect estimates based on an 8-hour mean metric, O'Connor et al. (2008) used a 24-hour metric, and Schildcrout et al. (2006) was based on a 1-hour max metric.
[151] We evaluated the impact from excluding O'Connor (2008) in estimating asthma exacerbations, and the change made a negligible (<1%) difference in monetized ozone benefits. Furthermore, the combined changes in estimating asthma exacerbations and respiratory hospital admissions resulted in only a 1% change in monetized ozone benefits compared to the approaches used in the proposal RIA (U.S. EPA, 2014c).
[152] BenMAP-CE applies a chi-squared test to determine whether a fixed or random effect pooling approach should be used (for additional detail see U.S. EPA (2015b), p. 206).
[153] The random effects pooling used in BenMAP (U.S. EPA, 2015b) to generate a single estimate for asthma exacerbations weights estimates based on variance, and consequently those estimates with less statistical significance and wider confidence intervals will be down-weighted in generating the total aggregated estimate.

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 exposure to PM for 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 one type of acute respiratory symptom related to ozone exposure. Minor Restricted Activity Days (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.

For ozone, we modeled MRADs using Ostro and Rothschild (1989). This study provides a co-pollutant model (with $PM_{2.5}$) based on a national sample of 18-64 year olds. The original study used a 24-hour 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 8-hour max equivalent.

We estimate three types of acute respiratory symptoms related to $PM_{2.5}$ exposure: lower respiratory symptoms, upper respiratory symptoms, and 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 focused on effects in asthmatics.

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 assess 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.

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.

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,[154] 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 resulting from exposure 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.

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

Nonfatal heart attacks have been linked with short-term exposures to $PM_{2.5}$ in the United States (Mustafić et al., 2012; Peters et al., 2001; Sullivan et al., 2005; Pope et al., 2006; Zanobetti and Schwartz, 2006; Zanobetti et al., 2009) and other countries (Poloniecki et al., 1997; Barnett et al., 2006; Peters et al., 2005). In previous health impact assessments, we have relied upon a study by Peters et al. (2001) as the basis for the impact function estimating the relationship between $PM_{2.5}$ and nonfatal heart attacks. The Peters et al. (2001) study exhibits a number of strengths. In particular, it includes a robust characterization of populations experiencing 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

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

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, including: 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 C-R 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 replace the 7% survival rate previously applied across all age groups from Rosamond et al. (1999).

6.6.3.7 Worker Productivity

The EPA last quantified the effect of ozone on outdoor agricultural worker productivity in the final Regulatory Impact Analysis accompanying the Transport Rule (U.S. EPA, 2011c); that analysis reported the value of worker productivity in the core benefits analysis. That RIA applied information reported in Crocker and Horst (1981), which observed that reducing ozone by 10 percent translated to a 1.4 increase in income among outdoor citrus workers. The RIA accompanying the proposed Ozone NAAQS (US EPA, 2014c) noted that, due in part to the vintage of the data used in this study, the Agency omitted this analysis in subsequent RIA's.

A recent study by Graff Zivin, and Neidell (2012) provides new evidence of the effect of ozone exposure on productivity among outdoor agricultural workers which we use in a quantitative uncertainty analysis examining this endpoint. Specifically, the study combined individual-level daily harvest rates for outdoor agricultural workers on a 500-acre farm in the Central Valley of California, with ground-level ozone data. The authors observed that a 10 ppb increase in work-day ozone concentrations (from 6:00 am to 3:00 pm) was associated with a 5.5% decrease in productivity among outdoor agricultural workers on a given day. Additional detail on the worker productivity uncertainty analysis (including results) is presented in Appendix 6B (section 6B.8).

6.6.3.8 Unquantified Human Health Effects

Attaining a revised ozone NAAQS would reduce emissions of NOx and VOC. Although we have quantified many of the health benefits associated with reducing exposure to ozone and $PM_{2.5}$, as shown in Table 6-3, we are unable to quantify the health benefits associated with reducing direct exposure to NO_2 or VOC because of the absence of air quality modeling data for these pollutants. In addition, we are unable to quantify the effects of VOC emission reductions on ambient $PM_{2.5}$ and associated health effects. 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 (NO_2 ISA) (U.S. EPA, 2008b). The NO_2 ISA provides a comprehensive review of the current evidence of health and environmental effects of NO_2. The NO_2 ISA concluded that the evidence "is sufficient to infer a likely causal relationship between short-term NO_2 exposure and adverse effects on the respiratory system." These epidemiologic and experimental studies encompass a number of endpoints including emergency department visits and hospitalizations, respiratory symptoms, airway hyperresponsiveness, airway inflammation, and lung function. Effect estimates from epidemiologic studies conducted in the United States and Canada generally indicate a 2–20 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.

6.6.4 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 6-10. 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, which is discussed in further detail below.[155] 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.

6.6.4.1 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

[155] Income growth projections are only currently available in BenMAP through 2024, so both the 2025 and 2038 estimates use income growth 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.

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 EPA neither changed its official guidance on the use of VSL in rule-makings nor subjected the interim estimate to a scientific peer-review process through SAB or other peer-review group.

Table 6-10. Unit Values for Economic Valuation of Health Endpoints (2011$) [a]

Health Endpoint	Central Estimate of Value Per Statistical Incidence		Derivation of Distributions of Estimates
	1990 Income Level	2024 Income Level	
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
3% discount rate			period following a nonfatal MI. Lost earnings estimates are based on
Age 0–24	$100,000	$100,000	Cropper and Krupnick (1990). Direct medical costs are based on
Age 25–44	$110,000	$110,000	simple average of estimates from Russell et al. (1998) and Wittels
Age 45–54	$120,000	$120,000	et al. (1990).
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 6-10. 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 6-10. 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 6-10. 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 (U.S. EPA, 2015). Income growth projections are only currently available in BenMAP through 2024, so both the 2025 and 2038 estimates use income growth 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)[156] 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\$).[157] 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 the risk of premature mortality is the subject of continuing discussion within the economics and

[156] 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.

[157] 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.

public policy analysis communities. 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 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 6-11 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 6-11. Influence of Applied VSL Attributes on the Size of the Economic Benefits of Reductions in the Risk of Premature Mortality (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 quantitative uncertainty 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 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.[158] Changes in the cessation lag assumptions do not change the total estimated premature mortality 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 that prevented us monetizing these benefits. In the quantitative uncertainty analysis, we include both an assumption of zero lag and the PM lag structure (i.e., the SAB 20-year segmented lag). Inclusion of the zero lag reflects consideration of the possibility that the long-term respiratory mortality estimate primarily captures an accumulation of short-term mortality effects across the ozone season.[159] The use of the 20-year segmented lag reflects consideration of advice provided by the SAB-HES (U.S. EPA-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." Monetized benefit estimates

[158] 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.

[159] The ozone HREA noted,: "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." (U.S. EPA, 2014b).

generated using both lag assumptions are presented as quantitative uncertainty analyses (see section 6.7.3.1).

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).[160] 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 along with WTP to improve other individuals' survival rates. The ideal measure would also take into account the specific nature of the risk reduction commodity that is provided to individuals, as well as the context in which risk is reduced. To measure this value, it is important to assess how reductions in air pollution reduce the risk of dying from the time that reductions take effect onward and how individuals value these changes. Each individual's survival curve, or the probability of surviving beyond a given age, should shift as a result of an environmental quality improvement. For example, changing the current probability of survival for an individual also shifts future probabilities of that individual's survival. This probability shift will differ across individuals because survival curves depend on

[160] 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.

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 analysis 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, traumatic 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 because 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).

6.6.4.2 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" (U.S. EPA, 2015). 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 6-12.

Table 6-12. 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 (U.S. EPA, 2015).

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 is $480 (using the CPI-U for medical care to adjust to 2011$). 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 $440 (2011$).

6.6.4.3 Nonfatal Myocardial Infarctions Valuation

We were not able to identify a suitable WTP value for reductions in the risk of nonfatal heart attacks. 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 6-13):

- 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 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 as a result whose length of stay was substantially shorter.

- 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 6-13. 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 (U.S. EPA, 2015).
[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 6-14.

Table 6-14. 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 (U.S. EPA, 2015).
[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.

6.6.4.4 Valuation of Acute Health Events

Asthma Exacerbation Valuation. 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 Valuation. 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).

6.6.4.5 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[161] 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

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

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). More recently, in response to questions related to the adjustment for income growth over time, the SAB-EEAC noted that "EPA should adjust for changes in income in evaluating benefits of risk reduction" (U.S. EPA-SAB, 2011). 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 quantitative uncertainty analyses for alternative income growth adjustment factors.

Based on a review of the available income elasticity literature, we adjusted the valuation of human health benefits upward to account for projected growth in real U.S. income. Faced with a dearth of estimates of income elasticities derived from time-series studies, we applied estimates derived from cross-sectional studies in our analysis. Details of the procedure can be found in Kleckner and Neumann (1999). We note that the literature has evolved since the publication of this memo and that an array of newer studies identifying potentially suitable income elasticity estimates are available (IEc, 2012). The EPA anticipates seeking a peer review in the near future of these studies and its approach to adjusting WTP estimates to account for changes in personal income. 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.

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 6-15.

Table 6-15. 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.[162] We used projections of real GDP (in chained 1996 dollars) provided by Standard and Poor's (2000) for the years 2010 to 2024.[163]

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 6-16. 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.

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

[162] 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.

[163] 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.

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 6-16. 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 6-15, U.S. Census population projections, and projections of real GDP per capita.

6.6.5 Benefit per Ton Estimates Used in Modeling $PM_{2.5}$-Related Co-benefits

This section presents the benefit-per-ton estimates (dimensioned by mortality study and simulation year) used as inputs in generating $PM_{2.5}$ co-benefits estimates including (Table 6-17). 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 section 6.6.3.3). Estimates were available for 2025 and 2030, with those being used to model co-benefits for the 2025 scenario and post-2025 scenario, respectively. For additional detail on the approach used to generate $PM_{2.5}$ co-benefits estimates and the role played by these two types of inputs, see Section 6.4.4.[164]

[164] 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 co-benefits $PM_{2.5}$ estimates are based on simulated benefits for those seven sectors.

Table 6-17. Summary of $PM_{2.5}$ Benefit-per-ton Estimates [a]

Long-term mortality study	Emissions sector: air, locamotive and marine	cement kilns	coke ovens	EGU point	electric arc furnaces	ferro 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	$7,200	$5,700	$10,000	$5,200	$9,600	$4,400	$13,000	$17,000	$6,300	$7,900	$7,000	$7,600	$3,700	$6,900	$14,000	$6,000	$2,000	$6,400
Lepeule et al., 2012	$16,000	$13,000	$24,000	$12,000	$22,000	$9,800	$30,000	$38,000	$14,000	$18,000	$16,000	$17,000	$8,500	$16,000	$31,000	$14,000	$4,600	$15,000
2025 at 3% social discount																		
Krewski et al., 2009	$8,000	$6,300	$12,000	$5,800	$11,000	$4,800	$15,000	$19,000	$7,000	$8,700	$7,700	$8,400	$4,200	$7,700	$15,000	$6,600	$2,300	$7,100
Lepeule et al., 2012	$18,000	$14,000	$26,000	$13,000	$24,000	$11,000	$34,000	$42,000	$16,000	$20,000	$17,000	$19,000	$9,400	$17,000	$34,000	$15,000	$5,100	$16,000
2030 at 7% social discount																		
Krewski et al., 2009	$7,800	$6,100	$11,000	$5,600	$10,000	$4,600	$14,000	$18,000	$6,800	$8,500	$7,600	$8,200	$4,000	$7,500	$15,000	$6,400	$2,300	$6,900
Lepeule et al., 2012	$18,000	$14,000	$25,000	$13,000	$23,000	$10,000	$32,000	$41,000	$15,000	$19,000	$17,000	$19,000	$9,100	$17,000	$33,000	$14,000	$5,100	$16,000
2030 at 3% social discount																		
Krewski et al., 2009	$8,700	$6,800	$12,000	$6,200	$11,000	$5,100	$16,000	$20,000	$7,600	$9,400	$8,400	$9,100	$4,500	$8,300	$16,000	$7,100	$2,500	$7,700
Lepeule et al., 2012	$20,000	$15,000	$28,000	$14,000	$26,000	$12,000	$36,000	$46,000	$17,000	$21,000	$19,000	$21,000	$10,000	$19,000	$37,000	$16,000	$5,600	$17,000

[a] Benefit-per-ton estimates reflect application of the 20-year segmented lag at either a 3% or 7% discount rate. 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.
[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. (2012b).

6.7 Benefits Results

We estimated the benefits of attaining the revised and alternative ozone standard levels across the U.S. in 2025 except California. We estimated the benefits of attaining these standard levels in California in 2038. We report the 2025 and 2038 estimates separately because deriving a summed estimate would require us to calculate the Present Value (PV) of the stream of benefits occurring between those two years, which is not possible with the available data. Additional analyses (section 6.7.3) inform the interpretation of these core analyses.

Applying the impact and valuation functions described above to the estimated changes in ozone concentrations yields estimates of the changes in physical damages (e.g., premature deaths, cases of hospital admissions) and the associated monetary values for those changes. Not all known ozone and PM health effects could be quantified or monetized, and the monetized value of these unquantified effects is represented by adding an unknown "B" to the aggregate total. Values are rounded to two significant figures and so totals may not sum across columns or rows.

6.7.1 Benefits of Attaining a Revised Ozone Standard in 2025

This section presents the avoided health impacts and monetized benefits of attaining a more stringent ozone standard in 2025. Table 6-18 shows the population-weighted air quality change for the revised and alternative standard levels averaged across the continental U.S. Table 6-19 summarizes the tons of NOx emissions required to simulate attainment of the revised and alternative standard levels (further differentiated by geographic region including east, west and California). Tables 6-20 through 6-25 present the benefits results for the standard levels analyzed. Table 6-25 summarizes total benefits by geographic region (including east, west minus California, and California).

In addition, Figure 6-4 presents a quantitative uncertainty analysis for short-term ozone-related benefits using additional C-R functions for premature mortality, and Figure 6-5 presents a quantitative uncertainty analysis for $PM_{2.5}$-related co-benefits using additional C-R functions for premature mortality. See sections 6.6.3.2 and 6.6.3.3, respectively for additional discussion of the alternative effect estimates used in each quantitative uncertainty analysis.

Table 6-18. Population-Weighted Air Quality Change for the Revised and Alternative Annual Primary Ozone Standards Relative to the Analytical Baseline in 2025[a]

Standard	Population-Weighted Summer Season Ozone Concentration Change (8-hour max)[b]
70 ppb	0.2574
65 ppb	1.278

[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 6-19. Sector-Specific NO_x Emissions Reductions for the Revised and Alternative Standard Levels[a]

	Revised and Alternative Standard Levels			
	70ppb		65ppb	
Emissions Sector	CA NOX Emis Rdxn	nonCA NOX Emis Rdxn	CA NOX Emis Rdxn	nonCA NOX Emis Rdxn
Aircraft, locomotives and marine vessels	-	-	-	-
Area sources	-	32,224	-	69,576
Cement kilns	-	19,285	-	31,963
Electricity Generating Units	-	47,507	-	113,678
Industrial point sources	-	134,763	-	319,484
Non-road mobile sources	-	2,832	-	8,791
On-road mobile sources	-			
Pulp and paper facilities	-	42	-	265
Refineries		3,134	-	7,735
Residential wood combustion				
Unknown sector	51,000	46,542	99,500	862,803
TOTAL	51,000	286,330	99,500	1,414,296

[a]All values are tons of NOx reductions (75 ppb vs revised and alternative standard levels). 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).

Table 6-20. Estimated Number of Avoided Ozone-Related Health Impacts for the Revised and Alternative Standard Levels (Incremental to the Baseline) for the 2025 Scenario (nationwide benefits of attaining the standards in the U.S. except California) [a, b]

	Health Effectb	Revised and Alterative Standard Levels (95th percentile confidence intervals) 70 ppb	65 ppb
Avoided Short-Term Mortality			
multi-city studies	Smith et al. (2009) (all ages)	96 (47 to 140)	490 (240 to 740)
	Zanobetti and Schwartz (2008) (all ages)	160 (86 to 240)	820 (440 to 1,200)
Avoided Long-term Respiratory Mortality			
multi-city study	Jerrett et al. (2009) (30-99yrs) copollutants model ($PM_{2.5}$)	340 (110 to 560)	1,700 (580 to 2,800)
Avoided Morbidity			
	Hospital admissions - respiratory (age 65+)[d]	180 (-42 to 400)	920 (-220 to 2,000)
	Emergency department visits for asthma (all ages)	510 (47 to 1,600)	2,700 (250 to 8,300)
	Asthma exacerbation (age 6-18)[d]	220,000 (-67,000 to 440,000)	1,100,000 (-330,000 to 2,100,000)
	Minor restricted-activity days (age 18-65)	450,000 (190,000 to 720,000)	2,200,000 (920,000 to 3,500,000)
	School Loss Days (age 5-17)	160,000 (57,000 to 360,000)	790,000 (280,000 to 1,700,000)

[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.
[d] The negative estimates at the 5[th] percentile confidence estimates for these morbidity endpoints reflect the statistical power of the studies used to calculate these health impacts. These results do not suggest that reducing air pollution results will adversely affect health, but rather, that we are less confident in the magnitude of the expected benefits for this endpoint.

Table 6-21. Total Monetized Ozone-Related Benefits for the Revised and Alternative Annual Ozone Standards (Incremental to the Baseline) for the 2025 Scenario (nationwide benefits of attaining the standards everywhere in the U.S. except California) (millions of 2011$) [a]

		Revised and Alterative Standard Levels (95th percentile confidence intervals)	
	Health Effectb	70 ppb	65 ppb
Avoided Short-Term Mortality - Core Analysis			
multi-city studies	Smith et al. (2009) (all ages)	\$1,000 (\$99 to \$2,900)	\$5,300 (\$500 to \$15,000)
	Zanobetti and Schwartz (2008) (all ages)	1,700 (\$160 to \$4,800)	8,700 (\$800 to \$24,000)

[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.

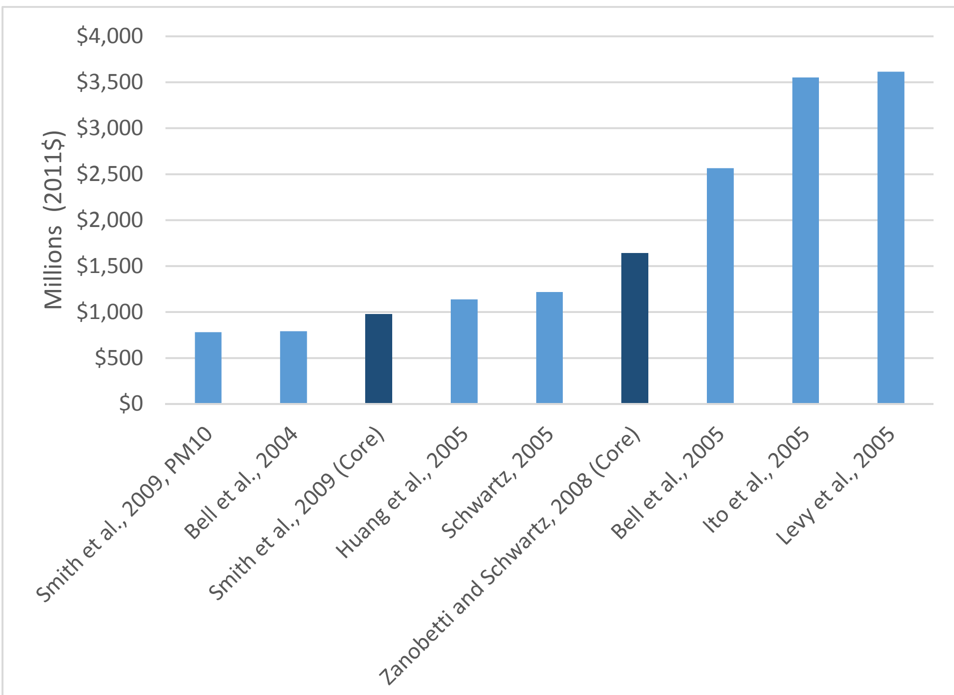

Figure 6-4. Quantitative Uncertainty Analysis for Short-Term Ozone-Related Mortality Benefits

Table 6-22. Estimated Number of Avoided $PM_{2.5}$-Related Health Impacts for the Revised and Alternative Annual Ozone Standards (Incremental to the Baseline) for the 2025 Scenario (Nationwide Benefits of Attaining the Standards in the U.S. except California) [a]

Health Effectb	Revised and Alterative Standard Levels 70ppb	65ppb
Avoided PM2.5-related Mortality		
Krewski et al. (2009) (adult mortality age 30+)	220	1,100
Lepeule et al. (2012) (adult mortality age 25+)	500	2,500
Woodruff et al. (1997) (infant mortality)	<1	2
Avoided PM2.5-related Morbidity		
Non-fatal heart attacks		
Peters et al. (2001) (age >18)	260	1,300
Pooled estimate of 4 studies (age >18)	28	140
Hospital admissions—respiratory (all ages)	66	330
Hospital admissions—cardiovascular (age > 18)	80	400
Emergency department visits for asthma (all ages)	120	600
Acute bronchitis (ages 8–12)	340	1,700
Lower respiratory symptoms (ages 7–14)	4,400	22,000
Upper respiratory symptoms (asthmatics ages 9–11)	6,300	31,000
Asthma exacerbation (asthmatics ages 6–18)	7,000	42,000
Lost work days (ages 18–65)	28,000	140,000
Minor restricted-activity days (ages 18–65)	170,000	830,000

[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 6-23. Monetized $PM_{2.5}$-Related Health Co-Benefits for the Revised and Alternative Annual Ozone Standards (Incremental to Baseline) for the 2025 Scenario (Nationwide Benefits of Attaining the Standards in the U.S. except California) (millions of 2011$) [a,b,c]

	Revised and Alternative Standard Levels	
Monetized Benefits	70 ppb	65 ppb
3% Discount Rate		
Krewski et al. (2009) (adult mortality age 30+)	$2,100	$10,000
Lepeule et al. (2012) (adult mortality age 25+)	$4,700	$23,000
7% Discount Rate		
Krewski et al. (2009) (adult mortality age 30+)	$1,900	$9,300
Lepeule et al. (2012) (adult mortality age 25+)	$4,200	$21,000

[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).

[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.

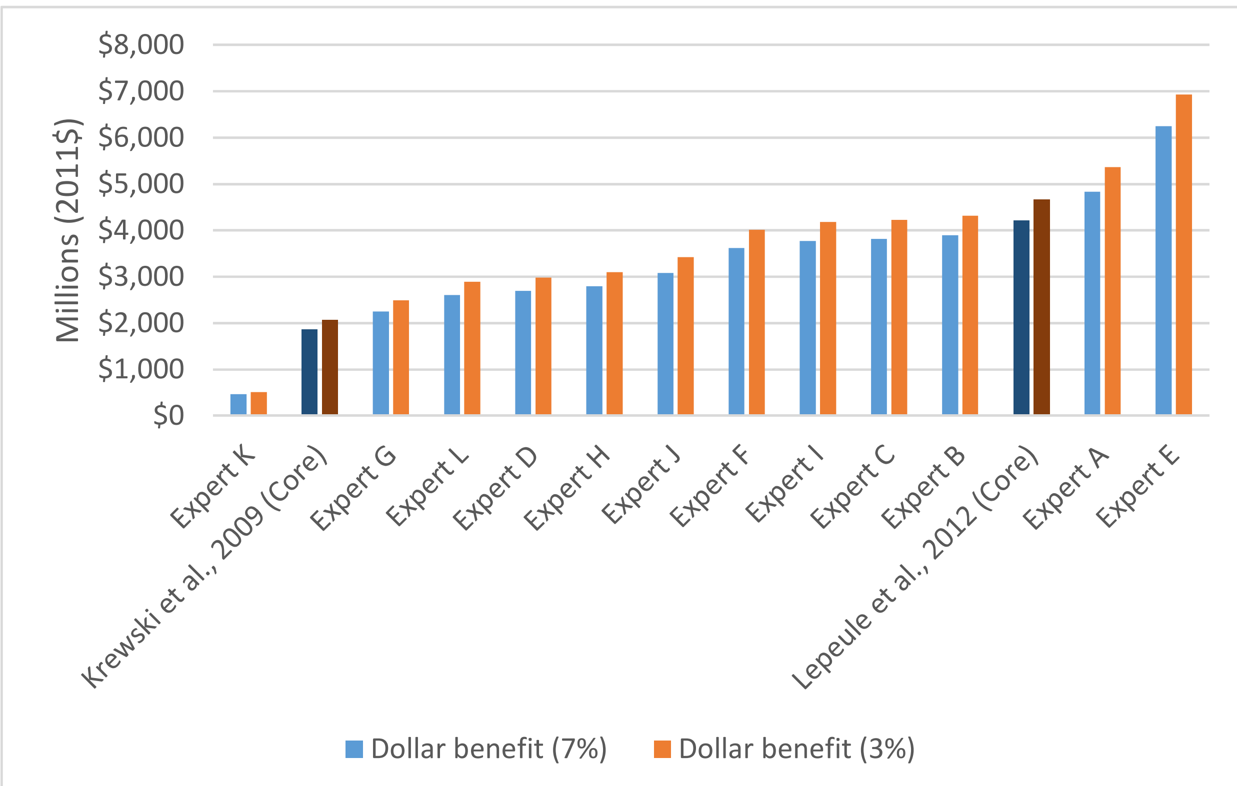

Figure 6-5. Quantitative Uncertainty Analysis Long-Term $PM_{2.5}$-Related Mortality Co-Benefits

Table 6-24. Estimated Monetized Ozone and $PM_{2.5}$ Benefits for Revised and Alternative Annual Ozone Standards Incremental to the Baseline for the 2025 Scenario (Nationwide Benefits of Attaining the Standards in the U.S. Except California) – Identified + Unidentified Control Strategies (combined) and Identified Control Strategies Only (billions of 2011$) [a]

	Discount Rate	Revised and Alternative Standard Levels	
		70 ppb	65 ppb
Identified + Unidentified Control Strategies			
Ozone-only Benefits (range reflects Smith et al. (2009) to Zanobetti and Schwartz (2008))	[b]	$1.0 to $1.7	$5.3 to $8.7
$PM_{2.5}$ Co-benefits (range reflects Krewski et al. (2009) to Lepeule et al. (2012)	3%	$2.1 to $4.7	$10 to $23
	7%	$1.9 to $4.2	$9.3 to $21
Total Benefits	3%	$3.1 to $6.4 +B	$16 to $32 +B
	7%	$2.9 to $5.9 +B	$15 to $30 +B
Identified Control Strategies Only			
Ozone-only Benefits (range reflects Smith et al. (2009) to Zanobetti and Schwartz (2008))	[b]	$0.86 to $1.4	$2.2 to $3.5
$PM_{2.5}$ Co-benefits (range reflects Krewski et al. (2009) to Lepeule et al. (2012)	3%	$1.7 to $3.9	$4.0 to $9.0
	7%	$1.6 to $3.5	$3.6 to $8.1
Total Benefits	3%	$2.6 to $5.3[c]	$6.1 to $12[c]
	7%	$2.4 to $4.9[c]	$5.7 to $12[c]

[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. 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.
[c] Excludes additional health and welfare benefits which could not be quantified (see section 6.6.3.8).

Table 6-25. Regional Breakdown of Monetized Ozone-Specific Benefits Results for the 2025 Scenario (Nationwide Benefits of Attaining the Standards in the U.S. except California) – Identified + Unidentified Control Strategies [a, b]

Region	Revised and Alterative Standard Levels	
	70 ppb	65 ppb
East [c]	98%	96%
California	~0%	~0%
Rest of West	2%	4%

[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] These regional breakdown results reflect application of identified and unidentified control strategies. Regional breakdown results are the same for benefits based on application of identified control strategies only.
[c] Includes Texas and those states to the north and east.

6.7.2 Benefits of the Post-2025 Scenario

This section presents the estimated number and economic value of avoided ozone- and $PM_{2.5}$-related effects associated with attaining a revised ozone standard after 2025 (note, sector-specific NOx emissions reductions levels used in modeling benefits for the post-2025 scenario are presented earlier in Table 6-19 - see entries under "CA NOx"). In addition, general trends and observations drawn from the quantitative uncertainty analyses presented in Figures 6-4 and 6-5 hold for the post-2025 scenario and for that reason separate plots for these quantitative uncertainty analyses (for the post-2025 scenario) are not presented (see section 6.7.3 for further discussion). While simulated attainment of standard levels for the 2025 scenario involved application of both identified and unidentified controls outside of California, simulated attainment of standard levels for the post-2025 scenario involved application exclusively of unidentified controls within California (see Table 6-18 and section 2.2.2). For that reason, results tables in this section represent exclusively, benefits based on application of unidentified controls.

Table 6-26. Population-Weighted Air Quality Change for the Revised and Alternative Annual Primary Ozone Standards Relative to Baseline for Post-2025 [a]

Standard	Population-Weighted Ozone Season Ozone Concentration Change (8-hour max)[b]
70 ppb	0.1708
65 ppb	0.3464

[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 6-27. Estimated Number of Avoided Ozone-Related Health Impacts for the Revised and Alternative Annual Ozone Standards (Incremental to the Baseline) for the Post-2025 Scenario (Nationwide Benefits of Attaining the Standards just in California) [a, b]

	Health Effectb	Revised and Alterative Standard Levels (95th percentile confidence intervals) 70 ppb	65 ppb
Avoided Short-Term Mortality			
multi-city studies	Smith et al. (2009) (all ages)	72 (35 to 110)	150 (71 to 220)
	Zanobetti and Schwartz (2008) (all ages)	120 (64 to 180)	240 (130 to 350)
Avoided Long-term Respiratory Mortality			
multi-city study	Jerrett et al. (2009) (30-99yrs) copollutants model ($PM_{2.5}$)	290 (98 to 480)	590 (200 to 970)
Avoided Morbidity			
	Hospital admissions - respiratory (age 65+)[d]	140 (-32 to 300)	270 (-65 to 610)
	Emergency department visits for asthma (all ages)	360 (33 to 1,100)	720 (67 to 2,200)
	Asthma exacerbation (age 6-18)[d]	160,000 (-49,000 to 320,000)	330,000 (-100,000 to 650,000)
	Minor restricted-activity days (age 18-65)	320,000 (130,000 to 510,000)	660,000 (270,000 to 1,000,000)
	School Loss Days (age 5-17)	120,000 (42,000 to 260,000)	240,000 (85,000 to 530,000)

[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.
[d] The negative estimates at the 5th percentile confidence estimates for these morbidity endpoints reflect the statistical power of the studies used to calculate these health impacts. These results do not suggest that reducing air pollution results in additional health impacts.

Table 6-28. Total Monetized Ozone-Only Benefits for the Revised and Alternative Annual Ozone Standards (Incremental to the Baseline) for the Post-2025 Scenario (Nationwide Benefits of Attaining the Standards just in California) (millions of 2011$) [a]

	Health Effectb	Revised and Alterative Standard Levels (95th percentile confidence intervals) 70 ppb	65 ppb
Avoided Short-Term Mortality - Core Analysis			
multi-city studies	Smith et al. (2009) (all ages)	$790 ($74 to $2,200)	$1,600 ($150 to $4,500)
	Zanobetti and Schwartz (2008) (all ages)	1,300 ($120 to $3,600)	2,600 ($240 to $7,200)

[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

Table 6-29. Estimated Number of Avoided $PM_{2.5}$-Related Health Impacts for the Revised and Alternative Annual Ozone Standards (Incremental to the Baseline) for the Post-2025 Scenario (Nationwide Benefits of Attaining the Standards just in California) [a]

Health Effectb	Revised and Alterative Standard Levels 70 ppb	65 ppb
Avoided PM2.5-related Mortality		
Krewski et al. (2009) (adult mortality age 30+)	43	84
Lepeule et al. (2012) (adult mortality age 25+)	98	190
Woodruff et al. (1997) (infant mortality)	<1	<1
Avoided PM2.5-related Morbidity		
Non-fatal heart attacks		
Peters et al. (2001) (age >18)	51	100
Pooled estimate of 4 studies (age >18)	6	11
Hospital admissions—respiratory (all ages)	13	26
Hospital admissions—cardiovascular (age > 18)	16	31
Emergency department visits for asthma (all ages)	23	44
Acute bronchitis (ages 8–12)	64	130
Lower respiratory symptoms (ages 7–14)	820	1,600
Upper respiratory symptoms (asthmatics ages 9–11)	1,200	2,300
Asthma exacerbation (asthmatics ages 6–18)	1,900	3,600
Lost work days (ages 18–65)	5,300	10,000
Minor restricted-activity days (ages 18–65)	31,000	61,000

[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 6-30. Monetized $PM_{2.5}$-Related Health Co-Benefits for the Revised and Alternative Annual Ozone Standards (Incremental to Baseline) for the Post-2025 Scenario (Nationwide Benefits of Attaining the Standards just in California) (millions of 2011$) [a,b]

Monetized Benefits	Revised and Alterative Standard Levels 70 ppb	65 ppb
3% Discount Rate		
Krewski et al. (2009) (adult mortality age 30+)	$400	$790
Lepeule et al. (2012) (adult mortality age 25+)	$910	$1,800
7% Discount Rate		
Krewski et al. (2009) (adult mortality age 30+)	$370	$710
Lepeule et al. (2012) (adult mortality age 25+)	$820	$1,600

[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).

[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 6-31. Estimate of Monetized Ozone and $PM_{2.5}$ Benefits for Revised and Alternative Annual Ozone Standards Incremental to the Baseline for the Post-2025 Scenario (Nationwide Benefits of Attaining the Standards just in California) – Identified + Unidentified Control Strategies (billions of 2011$) [a]

	Discount Rate	Revised and Alternative Standard Levels 70 ppb	65 ppb
Identified + Unidentified Control Strategies			
Ozone-only Benefits (range reflects Smith et al. (2009) to Zanobetti and Schwartz (2008))	[b]	$0.79 to $1.3	$1.6 to $2.6
$PM_{2.5}$ Co-benefits (range reflects Krewski et al. (2009) to Lepeule et al. (2012)	3%	$0.40 to $0.91	$0.79 to $1.8
	7%	$0.37 to $0.82	$0.71 to $1.6
Total Benefits	3%	$1.2 to $2.2 [c]	$2.4 to $4.4 [c]
	7%	$1.2 to $2.1 [c]	$2.3 to $4.2 [c]

[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. 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.

[c] Excludes additional health and welfare benefits which could not be quantified (see section 6.6.3.8).

Table 6-32. Regional Breakdown of Monetized Ozone-Specific Benefits Results for the Post-2025 Scenario (Nationwide Benefits of Attaining the Standards just in California) – Identified + Unidentified Control Strategies [a]

Region	Revised and Alterative Standard Levels	
	70 ppb	**65 ppb**
East [b]	3%	2%
California	90%	91%
Rest of West	7%	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.

6.7.3 Uncertainty in Benefits Results (including Results of Quantitative Uncertainty Analyses)

Avoided ozone and $PM_{2.5}$ related premature deaths account for 94% to 96% 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 monetized benefits; however, these outcomes occur among a significantly larger population. Comparing an incidence table to the monetized benefits table reveals that the number of incidences avoided and the unit value for that endpoint do not always closely correspond. For example, for ozone we estimate almost 1,000 times more cases of exacerbated ozone would be avoided than premature deaths, yet cases of exacerbated asthma account for only a very small fraction (<1%) of total monetized benefits (see Table 6-20). This is because 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 co-benefits are discussed qualitatively in Appendix 6A. Quantitative analyses completed in support of uncertainty characterization are discussed in detail in Appendix 6B.

Below we address uncertainty associated in modeling benefits for both ozone- and $PM_{2.5}$, including both key assumptions and uncertainty associated with modeling mortality along with brief summarizes of the results of the quantitative analyses completed in support of uncertainty characterization (presented in detail in Appendix 6B).

Ozone-Related Benefits

- **Key assumption and uncertainties related to modeling of ozone-related premature mortality**: Ozone-related short-term mortality represents a substantial proportion of total monetized benefits (over 94% of the ozone-related-benefits), and these estimates have the following key assumptions and uncertainties. We utilize a log-linear impact function without a threshold in modeling short-term ozone-related mortality. However, we acknowledge reduced confidence in specifying the nature of the C-R function in the range of ≤20ppb and below (ozone ISA, section 2.5.4.4). Thus, the estimates include health benefits from reducing ozone in areas with varied concentrations of ozone, including both areas that do not meet the ozone standard and those areas that are in attainment, down to the lowest modeled concentrations.

- **Avoided premature mortality according to baseline pollutant concentrations:** We recognize that, in estimating short-term ozone-related mortality, we are less confident in specifying the shape of the C-R function at lower ambient ozone concentrations (at and below 20 ppb, ozone ISA, section 2.5.4.4). As discussed in section 6.7.3.2 and (in greater detail) in Appendix 6B, section 6B.7, quantitative uncertainty analyses completed for this RIA found that the vast majority (~84%) of the reductions estimated premature mortality for simulated attainment of the revised and alternative standards (70 ppb and 65 ppb) were associated with grid cells having mean 8-hour max values between 35 and 55 ppb. Furthermore, ~100% of mortality reductions occurred above 20 ppb, where we are more confident in specifying the nature of the ozone-mortality effect (ozone ISA, section 2.5.4.4). owever, as discussed in section 6B.7, care must be taken in interpreting these results since the ambient air metric used in modeling this endpoint is the mean 8-hour max value in each grid cell (and not the full distribution of 8-hour daily max values). Had the latter been used, then the distribution would have likely been wider. The use of the mean 8-hour max metric in the RIA also means that the graphical distributions referenced here and presented in Appendix 6B.7 cannot be readily contrasted with specific ozone standard levels (since those are based on 8-hour max daily metrics and not on a seasonal-mean of those metrics).

- **Short-term ozone-exposure related premature mortality (alternative epidemiological studies and C-R functions):** We estimated the number of premature deaths using seven additional effect estimates including four multi-city studies and three meta-analysis studies. This quantitative uncertainty 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 Figure 6-4).

- **Economic value of avoided premature mortality from long-term exposure to ozone:** We estimated the economic value of long-term ozone mortality using two cessation lag structures: 20-year segment lag (as used for $PM_{2.5}$) and a zero lag. The quantitative uncertainty analysis suggests that if included in the core benefit estimate, long-term ozone exposure-related mortality could add substantially to the overall benefits. 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 premature respiratory mortality and potential thresholds:** We evaluated the impact of several assumed thresholds ranging from 40-60 ppb. This quantitative uncertainty analysis suggested that a threshold of 50 ppb or greater could have a substantial impact on estimated benefits, while thresholds below this range have a relatively minor impact.

- **Income elasticity for premature mortality and certain morbidity endpoints:** 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 quantitative uncertainty 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.

- **Value of increased productivity among outdoor agricultural workers due to reduced exposure to ozone:** Using information from the Graff Zivin and Neidell (2012) study, we estimate the economic value of improved productivity among outdoor non-livestock workers for the ozone standards in 2025 in our uncertainty analysis. We estimate the monetized worker productivity benefits of attaining a 70 ppb standard would be about $1.7 million and a 65 ppb standard would yield monetized benefits of about $8.9 million.[165]

<u>$PM_{2.5}$-Related Benefits</u>

- **Key assumption and uncertainties related to modeling of $PM_{2.5}$-related premature mortality** $PM_{2.5}$ 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 further that the health impact function for fine particles is log-linear without a threshold in this analysis. Thus, the estimates include health benefits from reducing fine particles in areas with varied concentrations of $PM_{2.5}$, including both areas that do not meet the fine particle standard and those areas that are in attainment, down to the lowest modeled concentrations. In addition, we assume that there is a "cessation" lag between

[165] We recognize that there is significant uncertainty in the generalizability of this study and the need for additional research and peer review in guiding the monetization of agriculatural productivity impacts.

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. And finally, we recognize uncertainty associated with application of the benefit-per-ton approach used in modeling $PM_{2.5}$ co-benefits. 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. (2012b). 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.

- **Avoided premature mortality according to baseline pollutant concentrations:** We recognize that, in modeling long-term $PM_{2.5}$-related mortality, we are less confident in specifying the shape of the C-R function at levels below the lowest measured level (LML) reported in the epidemiology study(s) providing the effect estimates used in modeling the mortality endpoint. As discussed in section 6.7.3.2 and (in greater detail) in Appendix 6B, section 6B.7, quantitative analyses completed in support of uncertainty characterization completed for this RIA found that, depending on the mortality study, between 67% and 93% of the long-term $PM_{2.5}$- related mortality estimate is based on modeling involving baseline $PM_{2.5}$ levels above the LML. This increases our overall confidence in the mortality estimates underlying the benefit-per-ton values used in the RIA.

- **Long-term $PM_{2.5}$ exposure-related premature mortality and alternative C-R functions (based on the Expert Elicitation):** We applied the set of expert elicitation-based functions to generate an alternative set of $PM_{2.5}$ benefit estimates (see Figure 6-5). The estimates based on Krewski et al. (2009) and Lepeule et al. (2012) fall within the range of estimates based on the functions from the 2006 expert elicitation.

6.8 Discussion

This analysis demonstrates the potential for significant health benefits of the illustrative emissions controls applied to simulate attainment with the revised and alternative primary ozone standard levels. We estimate that by 2025, the emissions reductions to reach the revised and alternative standard levels 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 rule would also yield significant welfare impacts as well (see Chapter 7). Even considering the quantified and unquantified uncertainties identified in this chapter, we believe that the revised and alternative standards would have substantial public health benefits that are likely to outweigh the costs of the control strategies for the revised and alternative standard levels analyzed (see Chapter 4).

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 C-R 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 revised and alternative standard levels.

As discussed in Chapter 1, there are important differences worth noting in the design and analytical objectives of NAAQS RIAs compared to RIAs for implementation rules, such as the Tier 3 (U.S. EPA, 2014a). Setting a NAAQS does not directly result in costs or benefits. The NAAQS RIAs illustrate the potential costs and benefits of the revised and alternative air quality standards nationwide based on an array of emissions 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 emissions reduction strategies that States may choose to enact when implementing a revised NAAQS, and as such, by contrast, the emissions reductions from implementation rules are generally for specific, well-characterized sources, such as the recent MATS rule (U.S. EPA, 2011c). In general, the EPA is more confident in the magnitude and location of the emissions reductions for implementation rules. As such, emissions reductions achieved under promulgated implementation rules, such as MATS, have been reflected in the baseline of this NAAQS analysis.[166] For this reason, the benefits estimated provided in this RIA and all other NAAQS RIAs should not be added to the

[166] The full set of rules reflected in the baseline are presented in Chapter 2, Section 2.1.3.

benefits estimated for implementation rules. Subsequent implementation rules will be reflected in the baseline for the next ozone NAAQS review.

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 emissions controls put in place to reduce concentrations at the highest monitor will simultaneously result in lower ozone concentrations throughout the entire area. In fact, the 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. 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 C-R functions for the purposes of benefits analysis.

The estimated benefits shown here are in addition to the substantial benefits estimated for several recent air quality rules (U.S. EPA, 2009a, 2011c, 2014a). Emissions reductions from rules such as Tier 3 will have substantially reduced ambient ozone concentrations by 2025 in the East, such that few additional controls would be needed to reach 70 ppb. These rules that have already been promulgated have tremendous combined benefits that explain why the number of avoided premature mortality 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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APPENDIX 6A: COMPREHENSIVE CHARACTERIZATION OF UNCERTAINTY IN OZONE BENEFITS ANALYSIS

Overview

As noted in Chapter 6, 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 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 estimates. 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, 2010a, 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 benefits, 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 $PM_{2.5}$ benefits estimates, please see the 2012 $PM_{2.5}$ NAAQS RIA (U.S. EPA, 2012).

6A.1 Description of Classifications Applied in the Uncertainty Characterization

Table 6A-1 catalogs the most significant sources of uncertainty in the ozone benefits analysis and then characterizes four dimensions of that uncertainty briefly described 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.

6A.1.1 Direction of Bias

The "direction of bias" column in Table 6A-1 is an assessment of whether, if left unaddressed, an uncertainty would likely lead to an underestimate or overestimate of 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* (Ozone ISA) (U.S. EPA, 2013). 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."

6A.1.2 Magnitude of Impact

The "magnitude of impact" column in Table 6A-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, 2010a, 2011, 2014), but we have slightly revised the category names and the cut-offs here.[167] The definitions used here are provided below.

[167] 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. 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.

- 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, 2010a, 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 (C-R) function for both categories of ozone-related mortality, and the mortality valuation, specifically for long-term exposure-related mortality.

6A.1.3 Confidence in Analytic Approach

The “confidence in analytic approach” column of Table 6A-1 is an assessment of the scientific support for the analytic approach chosen (or the inherent assumption made) to account for the relationship about which we are uncertain. In other words, based on the available evidence, how certain are we that EPA’s selected approach is the most plausible of the potential alternatives. Similar classifications have been included in previous risk and benefits analyses (U.S. EPA, 2010a, 2011).[168] 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, 2013) and by 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.

6A.1.4 Uncertainty Quantification

The column of Table 6A-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. The WHO uncertainty framework is a well-established approach to assess uncertainty in risk estimates that systematically links the characterization of uncertainty to the sophistication of the health impact assessment. The advantage of using this framework is that it clearly highlights the level of uncertainty quantification applied in this assessment and the potential sources of uncertainty that require methods development in order to assess quantitatively. Specifically, EPA applied this framework in multiple

[168] We have applied the same classification as *The Benefits and Costs of the Clean Air Act from 1990 to 2020* (U.S. EPA, 2011) 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.

risk and exposure assessments (U.S. EPA, 2010a, 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).

6A.2 Organization of the Qualitative Uncertainty Table

Table 6A-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).

Table 6A-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 Science Underlying the 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 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 Science Underlying the 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 a 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 U.S., 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 C-R 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) 8-hour maximum ozone concentrations (see Appendix 6B, section 6B.7) suggests that the vast majority of predicted reductions in mortality are associated with days having 8-hour maximum concentrations that fall within the higher confidence range.
Causal relationship between long-term ozone exposure and premature respiratory mortality	Overestimate, if long-term ozone exposure does not have a causal relationship with premature mortality.	Potentially High, if included in monetized benefits Mortality generally dominates monetized benefits, so small uncertainties could have large impacts on the total monetized benefits. However, we have not included long-term ozone mortality in the monetized benefits for this analysis due to uncertainties in the cessation lag.	Medium 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).	Tier 1 (qualitative)
Shape of the C-R functions, particularly at low concentrations for long-term ozone exposure-related respiratory mortality	Either	Potentially High, if included in monetized benefits	Medium	Tier 2 (sensitivity analysis)

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Science Underlying the Analytical Approach	Uncertainty Quantification
	The direction of bias that assuming a 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. However, we have not included long-term ozone mortality in the monetized benefits for this analysis due to uncertainties in the cessation lag.	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 model…". 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 quantitative analysis supporting uncertainty characterization.	We examined potential thresholds (from 40 to 60 ppb) in the C-R function for long-term exposure-related morality. 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 6B, Table 6B-3).
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 Science Underlying the Analytical Approach	Uncertainty Quantification
Adjustment of risk coefficients to 8-hour maximum from 24-hour average or 1-hour maximum in the epidemiology studies	We converted these metrics to maximum 8-hour average ozone concentration using standard conversion functions based on observed relationships in the underlying studies. If the relationships between air metrics reported in the studies differ systematically from the relationships seen across the modeling domain, then bias could be introduced.	This conversion does not affect the relative magnitude of the health impact function. However, the pattern of 8-hour maximum concentrations for a particular location over an ozone season could differ from the pattern of 1-hour max or 24-hour average metrics for that same location. Consequently, monetized benefits could differ for a particular location depending on the metric used in modeling benefits.	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 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.)
Confounding and effect modification by co-pollutants	Either, depending upon the pollutant. Disentangling the health responses of combustion-related pollutants (i.e., PM, SOx, NOx, ozone, and CO) is a challenge. The PM ISA states that co-pollutants may mediate the effects of PM or PM may influence the toxicity of co-pollutants (U.S. EPA, 2009, p. 1–16). Alternately, effects attributed to one pollutants may be due to another.	Medium Because this uncertainty could affect mortality and because mortality generally dominates monetized benefits, even small uncertainties could have medium impacts on total monetized benefits.	Medium 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.	Tier 1 (qualitative) (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				
	Unknown	High	Medium	Tier 3 (probabilistic)

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Science Underlying the Analytical Approach	Uncertainty Quantification
Mortality Risk Valuation/Value-of-a-Statistical-Life (VSL)	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.	Mortality generally dominates monetized benefits, so moderate uncertainties could have a large effect on total monetized benefits.	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, 2010b, U.S. EPA-SAB, 2011c).	Assessed uncertainty in mortality valuation using a Weibull distribution.
Cessation lag structure for long-term ozone mortality	Unknown We included both a zero (no) lag and 20-year segmented lag model in completing the quantitative uncertainty analysis involving dollar benefits for long-term respiratory mortality. Given that available information does not lead to the selection of a particular lag model, we are not in a position to classify the direction of potential bias associated with this source of uncertainty.	Medium, if included in the monetized benefits 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. However, we have not included long-term ozone mortality in the monetized benefits for this analysis due to uncertainties in the cessation lag.	Low As discussed in section 6.7.3.1, in presenting dollar benefit estimates as part of the quantitative analysis supporting uncertainty characterization (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 SAB-HES (USEPA-SAB, 2010, 2004c).	Tier 2 (sensitivity analysis) Using the 20-year segmented lag developed for $PM_{2.5}$–related mortality 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).
Income growth adjustments	Either Income growth increases willingness-to-pay (WTP) valuation estimates, including mortality, over time. From 1997 to 2010, personal income and GDP growth have begun to diverge. If this trend continues, the assumption that per capita GDP growth is a reasonable proxy for income growth may lead to an overstatement of benefits. (IEc, 2012).	Medium Income growth from 1990 to 2020 increases mortality valuation by 20%. Alternate estimates for this adjustment vary by 20% (IEc, 2012). Because we do not adjust for income growth over the 20-year cessation lag, this approach could also underestimate the benefits for the later years of the lag.	Medium Consistent with SAB recommendations (U.S. EPA,-SAB, 2000, pg. 16), we adjust WTP for income growth. It is difficult to forecast future income growth; however, in the absence of readily available income data projections, per capita GDP is the best available option.	Tier 2 (sensitivity analysis) As shown in Appendix 6B (section 6B.5), the use of alternate income growth adjustments would result in an increase of from 8 to 75% in the dollar benefits for short-term ozone-related mortality.
Morbidity valuation	Underestimate	Low	Low	Tier 3 (probabilistic), where available

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Science Underlying the Analytical Approach	Uncertainty Quantification
	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 valuations 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 6-10).
Uncertainties Associated with Baseline Incidence and Population Projections				
Population estimates and projections	Either The monetized benefits would change in the same direction as the over- or underestimate in population projections in areas where exposure changes.	Low–Medium Monetized benefits are substantially affected by population density. Comparisons using historical Census data show that population projections are ±5% nationally, but projection accuracy can vary by locality. Historical error for Woods & Poole's population projections has been ±8.1% for county-level projections and ±4.1% for states (Woods and Poole, 2012). The magnitude of impact on total monetized benefits depends on the specific location where ozone is reduced.	Medium We use population projections for 5-year increments for 304 race/ethnicity/gender/age groups (Woods and Poole, 2012) at Census blocks. Population forecasting is well-established but projections of future migration due to possible catastrophic events are not considered. In addition, projections at the small spatial scales used in this analysis are inherently more uncertain than projections at the county- or state-level.	Tier 1 (qualitative) (No quantitative method available)
Uncertainty in projecting baseline incidence rates for mortality	Unknown Because the mortality rate projections for future years reflect changes in mortality patterns as well as population growth, the projections are unlikely to be biased.	Low-Medium Because mortality generally dominates monetized benefits, small uncertainties could have medium impacts on total monetized benefits.	Medium The county-level baseline mortality rates reflect recent databases (i.e., 2004–2006 data) and 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	Either, depending on the health endpoint	Low	Low-Medium	Tier 1 (qualitative)

Potential Source of Uncertainty	Direction of Potential Bias	Magnitude of Impact on Monetized Benefits	Confidence in Science Underlying the Analytical Approach	Uncertainty Quantification
incidence rates and prevalence rates for morbidity	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.	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.	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).	(No quantitative method available)
Uncertainties Associated with Omitted Benefits Categories				
Unquantified ozone health benefit categories, such as worker productivity and long-term mortality	Underestimate EPA has not included monetized estimates of these benefits categories in the core benefits estimate.	High 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.	Low 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).	Tier 2 (sensitivity analysis) We include a quantitative uncertainty analysis reflecting long-term mortality, which shows that this endpoint could add substantially to the monetized benefits (see Appendix 6B, section 6B.2). We have also included a analysis reflecting application of an updated worker productivity analysis (see section 6.5.3).
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)

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APPENDIX 6B: QUANTITATIVE ANALYSES COMPLETED IN SUPPORT OF UNCERTAINTY CHARACTERIZATION

Overview

The benefits analysis presented in Chapter 6 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 uncertainty into the estimates of benefits and that alternative choices exist for some inputs to the analysis, such as the concentration-response (C-R) functions for mortality.

This appendix presents a set of quantitative analysis completed in support of uncertainty characterization including exploration of: (a) alternative studies in modeling short-term ozone exposure-related mortality (section 6B.1), (b) monetized benefits associated with mortality resulting from long-term exposure to ozone (section 6B.2), (c) the potential impact of thresholds in long-term ozone exposure-related mortality in incidence and benefits estimates (section 6B.3), (d) alternative response functions developed through expert elicitation for long-term $PM_{2.5}$ exposure-related mortality (section 6B.4), (e) alternative assumptions regarding income elasticity on benefits derived using willingness-to-pay (WTP) functions (section 6B.5), (f) 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 6B.6), (g) 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 6B.7) and (h) ozone-related impacts on outdoor worker productivity (including a detailed discussion of the methodology used in modeling and presentation of results, Section 6B.8).

For the core analyses, we estimated benefits for two scenarios: 2025 and post-2025. However, in conducting these quantitative analyses supporting uncertainty characterization, we used the 2025 scenario as the basis for making our calculations, since analytical findings for this scenario would generally hold for the post-2025 scenario.

6B.1 Alternative C-R Functions for Short-term Exposure to Ozone

Table 6B-1 presents the results of applying alternative effect estimates identified for modeling short-term exposure-related mortality for ozone, along with the set of core risk estimates (these uncertainty characterization results are also reflected in Figure 6-4 presented in the body of the document). The suite of effect estimates included in the analysis consideration for both meta-analyses and multi-city epidemiology studies. The rationale for the specific mix of effect estimates included in the quantitative analysis is presented in section 6.6.3.2.

Table 6B-1. Quantitative Analysis for Alternative C-R Functions for Short-term Exposure to Ozone [a]

Health Effect		Revised and Alternative Standards (95th percentile confidence intervals) 70 ppb	65 ppb
Avoided Short-Term Mortality - Core Analysis			
multi-city studies	Smith et al. (2009) (all ages)	96 (47 to 140)	490 (240 to 740)
	Zanobetti and Schwartz (2008) (all ages)	160 (86 to 240)	820 (440 to 1,200)
Avoided Short-Term Mortality - Uncertainty Analysis			
multi-city studies	Smith et al. (2009) (all ages) co-pollutant model with PM_{10}	77 (-21 to 170)	390 (-110 to 890)
	Schwartz (2005) (all ages)	120 (37 to 200)	610 (190 to 1,000)
	Huang et al. (2005) (cardiopulmonary)	110 (42 to 180)	580 (220 to 940)
	Bell et al. (2004) (all ages)	78 (26 to 130)	400 (130 to 660)
meta- analyses	Bell et al. (2005) (all ages)	250 (120 to 380)	1,300 (610 to 2,000)
	Ito et al. (2005) (all ages)	350 (210 to 490)	1,800 (1,100 to 2,500)
	Levy et al. (2005) (all ages)	350 (240 to 470)	1,800 (1,200 to 2,400)

[a] All estimates are rounded to two significant digits.

This quantitative analysis showed that the two core incidence estimates fall within (and towards the lower end of) the broader range resulting from application of the seven alternative effect estimates (note that these observations based on incidence would also hold for the matching set of dollar benefit estimates generated for this endpoint category).

6B.2 Monetized Benefits for Premature Mortality from Long-term Exposure to Ozone

As discussed in section 6.3, due to uncertainty in specifying the temporal lag structure associated with reductions in long-term ozone-related respiratory mortality, we have included these estimates as an uncertainty analysis and not as part of the core dollar benefit estimate. Table 6B-2 presents the dollar benefits (2011$) associated with modeled reductions in long-term ozone-related respiratory mortality. These benefit estimates are generated using non-threshold models obtained from Jerrett et al. (2009) because we present the potential threshold analysis in the section 6B.3. Additional detail on the effect estimates used in the core analysis and this uncertainty analysis is presented in section 6.6.3.2. Section 6.6.4 provides additional detail on the approach used in valuing these mortality estimates including how uncertainty related to cessation lag was addressed.

Table 6B-2. Monetized Benefits for Mortality from Long-term Exposure to Ozone (millions of 2011$) (2025 and post-2025 scenarios)[a, d]

	Health Effectb	Revised and Alterative Standard Levels (95th percentile confidence intervals) 70 ppb	65 ppb
Avoided Long-term Respiratory Mortality			
2025 scenario			
multi-city study	Jerrett et al. (2009) (age 30-99) copollutants model ($PM_{2.5}$) no lag [b]	$3,400 ($280 to $10,000)	$17,000 ($1,400 to $52,000)
	Jerrett et al. (2009) (age 30-99) copollutants model ($PM_{2.5}$) 20 yr segmented lag [c]	$2,800 to $3,100 ($250 to $8,400)	$14,000 to $16,000 ($1,200 to $47,000)
post-2025 scenario			
multi-city study	Jerrett et al. (2009) (age 30-99) copollutants model ($PM_{2.5}$) no lag [b]	$3,000 ($240 to $8,400)	$6,000 ($490 to $18,000)
	Jerrett et al. (2009) (age 30-99) copollutants model ($PM_{2.5}$) 20 yr segmented lag [c]	$2,400 to $2,700 ($200 to $8,000)	$4,900 to $5,400 ($400 to $18,000)

[a] See section 6B.3 for the quantitative analysis for potential thresholds in the C-R function for long-term exposure-related mortality. Observations from that analysis can be applied to these results.
[b] The zero-lag model is not affected by discounting. The values in parentheses reflect 95th percentile confidence intervals.
[c] The range (outside of the parentheses) results from application of 7% and 3% discount rates after applying the 20-year segmented lag (i.e., ranging from the 7% discount rate up to the 3% discount rate). The range in parentheses reflects the 95th percentile confidence interval in combination with the discount rate (i.e., ranging from the 2.5th percentile and the 7% discount rate up to the 97.5th percentile and the 3% discount rate).
[d] All estimates rounded to two significant digits.

This quantitative analysis suggests that if included in the core benefit estimate, long-term ozone exposure-related mortality could add substantially to the overall benefits. 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.

6B.3 Threshold Analysis for Premature Mortality Incidence and Benefits from Long-term Exposure to Ozone

In estimating long-term ozone mortality (including the benefit estimates presented in the last section), we employed a continuous non-threshold C-R function relating ozone exposure to premature death. However, as discussed in Section 6.6.3.2, 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 quantitative 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 the Clean Air Scientific Advisory Committee (CASAC) (U.S. EPA-SAB, 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., 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 coefficients. 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.[169] Table 6B-3 provides the results of this uncertainty analysis (based on modeling incidence) for 65 ppb and 70 ppb. The same pattern in terms of relative reductions across thresholds would also hold for the monetized benefit estimates.

Table 6B-3. Long-term Ozone Mortality Incidence at Various Assumed Thresholds [a]

Threshold Concentration	70 ppb	65 ppb
No threshold	340	1,700
40 ppb	260	1,300
45 ppb	200	910
50 ppb	59	210
55 ppb	6	66
56 ppb	5	60
60 ppb	3	19

[a] All estimates rounded to two significant digits.

The results of the uncertainty analysis based on the suite of threshold-based risk coefficients suggest that threshold models can substantially lower estimates of ozone-attributable long-term mortality. For example, estimated benefits for long-term mortality using a model that

[169] There is a separate effect coefficient (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 coefficient applied to ozone concentrations above the threshold.

includes a 55 ppb threshold are approximately 70% less than long-term mortality benefits estimated using the 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.

6B.4 Alternative C-R Functions for $PM_{2.5}$–Related Mortality

In estimating $PM_{2.5}$ co-benefits, monetized benefits are driven largely by reductions in mortality. Therefore, it is particularly important to attempt to characterize the uncertainties associated with reductions in premature mortality. In addition to the ACS and Six Cities cohort studies, several recent cohort studies conducted in North America provide evidence for the relationship between long-term exposure to $PM_{2.5}$ and the risk of premature death. Many of these additional cohort studies are described in the PM ISA (U.S. EPA, 2009) and the *Provisional Assessment* (U.S. EPA, 2012a).[170,] Table 6B-4 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[171] sponsored by the EPA (Roman et al., 2008; IEc, 2006) to demonstrate the sensitivity of the benefits estimates to 12 expert-defined C-R functions. The $PM_{2.5}$ expert elicitation and the derivation of C-R functions 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 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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[170] 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.

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

Table 6B-4. Summary of Effect Estimates from Recent Cohort Studies in North America Associated with Change in Long-Term Exposure to $PM_{2.5}$

Study	Cohort (age)	LML ($\mu g/m^3$)	Mean ($\mu g/m^3$)	Hazard Ratios per 10 $\mu g/m^3$ Change in $PM_{2.5}$ (95[th] 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. (2012)[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, role of short- and long-term exposures, potential confounding, and exposure misclassification. 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"[172] 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 nature. The experts also noted a series of limitations in these two cohort studies. In the case of

[172] These criteria are substantively similar to EPA's study selection criteria identified in Table 6-5 of Chapter 6.

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]).

It is important to 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 6B-4. 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 6B-4. However, we have completed an uncertainty analysis based on application of the full set of expert-derived effect estimates. To preserve the breadth and diversity of opinion on the expert panel, we do not combine the expert results (Roman et. al., 2008). This presentation of the expert-derived results is generally consistent with SAB advice (U.S. EPA-SAB, 2008), which suggested 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.

Table 6B-5 presents the results of this uncertainty analysis using the expert elicitation results for the 2025 scenario for 70 ppb. Overall conclusions from this analysis are also applicable to the post-2025 scenario.

Table 6B-5. $PM_{2.5}$ Co-benefit Estimates using Two Epidemiology Studies and Functions Supplied from the Expert Elicitation

C-R Function	70 ppb (2025 Scenario)
Krewski et al. (2009)	220
Lepeule et al. (2012)	500
Expert K	50
Expert G	260
Expert L	310
Expert D	320
Expert H	330
Expert J	360
Expert F	430
Expert C	450
Expert I	450
Expert B	460
Expert A	570
Expert E	740

[a] All estimates rounded to two significant digits.

The values presented in Table 6B-5 indicate that the two core incidence estimate fall within the range of alternative C-R function-based estimates obtained through expert elicitation. Figure 6-5 in the body of the document reproduces these benefit estimates, but presents them in terms of the associated dollar benefits (7% discount rate in 2011$) rather than incidence estimates (observations presented here for the uncertainty analysis results reflected modeled incidence estimates - i.e., spread in results and relative position of the two core estimates - also hold for dollar benefit estimates presented in Figure 4-6.

6B.5 Income Elasticity of Willingness-to-Pay

As discussed in Chapter 6, 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 through 2024 only and are therefore likely underestimates.

Table 6B-6 lists the ranges of elasticity values used to calculate the income adjustment factors, while Table 6B-7 lists the ranges of corresponding adjustment factors. The results of this uncertainty analysis for the two benefit categories are presented in Table 6B-8.

Table 6B-6. 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 6B-7. 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 6B-8. 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]	$31	$150	$33	$162	$38	$186
Premature Mortality [c]	$950	$4,100	$1,000	$5,200	$1,700	$8,500

[a] All estimates rounded to two significant digits. Only reflects income growth to 2024.
[b] For illustrative purposes, we evaluate minor restricted activity days (MRADS) resulting from short-term ozone exposure is the minor health effect here.
[c] Short-term mortality using 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 8% to 76% greater than the core 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 5% to 21% greater than the core estimate for minor effects. These observations (in terms of relative impact from alternative elasticities) hold for the revised and alternative standard levels analyzed under both the 2025 and post-205 scenarios.

6B.6 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, the National Research Council (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, 2004). To address these recommendations, we use simplifying assumptions to estimate the number of life years that might be gained. We also estimate the reduction in the percentage of deaths attributed to ozone resulting from the illustrative emissions reduction strategies to reach the revised 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, 2010, 2011a), the latter of which was peer reviewed by the SAB-HES (U.S. EPA-SAB, 2010a).

Changes in life years and changes in life expectancy at birth are frequently conflated, thus it is important to distinguish these two very different metrics. 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, 2011). 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 $PM_{2.5}$ NAAQs RIA for more information about the avoided life years lost from $PM_{2.5}$ exposure (U.S. EPA, 2012b). The analysis performed for the PM NAAQS RIA 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, that 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

To estimate 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. (2012). 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), we would expect average life expectancy changes associated with air pollution exposure to always be significantly smaller than the average number of life years lost by an individual 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:

$$Total\ Life\ Years = \sum_{i=1}^{n} LE_i \times M_i \qquad (6.2)$$

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 6B-9 summarizes the number of avoided deaths (by age range) attributable to ozone for the revised and alternative standards for the 2025 scenario. Table 6B-10 summarizes the modeled number of life years gained (for each age range) by reducing ozone for the revised and alternative standards analyzed for the 2025 scenario. We then calculated the average number of life years gained per avoided premature mortality.

Table 6B-9. Potential Reduction in Premature Mortality by Age Range from Attaining the Revised and Alternative Ozone Standards (2025 scenario) [a, b]

Age Range [b]	Revised and Alternative Standards	
	70 ppb	65 ppb
0-4	0.3	1.5
5-9	0.068	0.33
10-14	0.072	0.35
15-19	0.1	0.5
20-24	0.14	0.69
25–29	0.29	1.4
30–34	0.3	1.5
35–44	1.6	7.9
45–54	4.1	20
55–64	10	52
65–74	22	110
75–84	29	150
85–99	28	150
Total ozone-attributable mortality	96	490

[a] Estimates rounded to two significant digits.
[b] Effects calculated using Smith et al. (2009).

Table 6B-10. Potential Years of Life Gained by Age Range from Attaining the Revised and Alternative Ozone Standards (2025 Scenario) [a,b]

Age Range [b]	Revised and Alternative Standard	
	70 ppb	65 ppb
0-4	24	110
5-9	5	23
10-14	5	23
15-19	6	30
20-24	8	38
25–29	14	70
30–34	13	66
35–44	57	280
45–54	110	550
55–64	200	990
65–74	260	1,300
75–84	190	970
85–99	65	340
Total life years gained	960	4,800
Average life years gained per individual	10	10

[a] Estimates rounded to two significant digits.
[b] Effects calculated using Smith et al. (2009).

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 6B-9), but half of the life years would occur in populations younger than

65 (see Table 6B-10). 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 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 the revised and alternative standards by the total number of deaths in each county. Table 6B-11 shows the reduction in all-cause mortality attributed to reducing ozone exposure to the revised and alternative primary standards for the 2025 scenario.

Table 6B-11. Estimated Percent Reduction in All-Cause Mortality Attributed to the Proposed Primary Ozone Standards (2025 Scenario) [a]

Age Range [b]	Revised and Alternative Standards	
	70 ppb	**65 ppb**
0-4	0.0085%	0.0409%
5-9	0.0084%	0.0411%
10-14	0.0084%	0.0415%
15-19	0.0086%	0.0419%
20-24	0.0085%	0.0412%
25–29	0.0084%	0.0409%
30–34	0.0083%	0.0408%
35–44	0.0081%	0.0404%
45–54	0.0082%	0.0410%
55–64	0.0085%	0.0429%
65–74	0.0086%	0.0436%
75–84	0.0085%	0.0433%
85–99	0.0079%	0.0415%

[a] To illustrate the slight variations in percent reductions across age ranges, we present results rounded to three significant digits (rather than two as is typically done for other estimates in this RIA).

Results presented in Table 6B-11 highlight that when reductions in ozone-attributable mortality (in going from baseline to the revised or 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.

6B.7 Evaluation of Mortality Impacts Relative to the Baseline Pollutant Concentrations 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, 2013). 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).

Figures 6B-1 and 6B-2 compare the distribution of short-term ozone exposure-related mortality to the underlying distribution of the summer season 8-hour maximum ozone concentrations. Both figures (probability and cumulative probability) are based on Smith et al. (2009) using mortality results at each 12 km grid cell and the associated summer season 8-hour maximum concentration in the baseline. In addition, each figure includes separate plots for the revised and alternative standard levels. Figure 6B-1 shows that approximately 45% of the premature mortalities estimated for the 65 ppb alternative standard is associated with baseline ozone concentrations between 40 and 45 ppb. Because this baseline range is represents the mean across the ozone season of 8-hour max values within a given grid cell, the actual distribution of 8-hour max values on a daily basis is likely wider than the 40-45 ppb range.

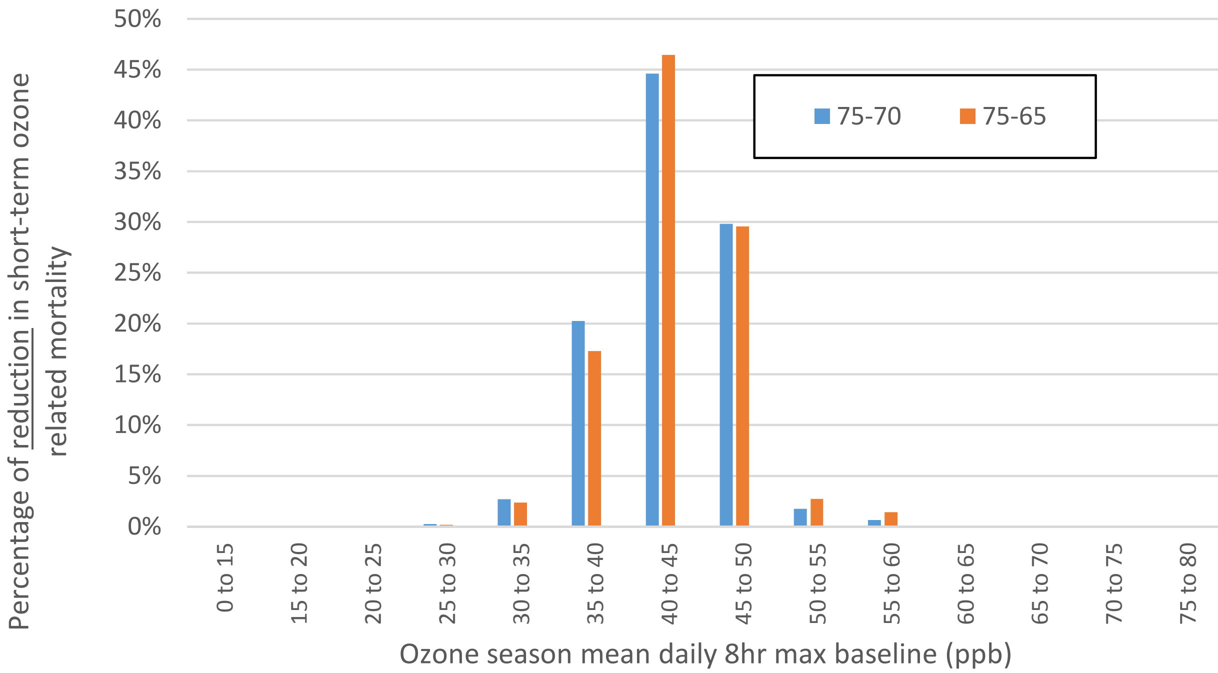


Figure 6B-1. Premature Ozone-related Deaths Avoided for the Revised and Alternative Standards (2025 scenario) According to the Baseline Ozone Concentrations

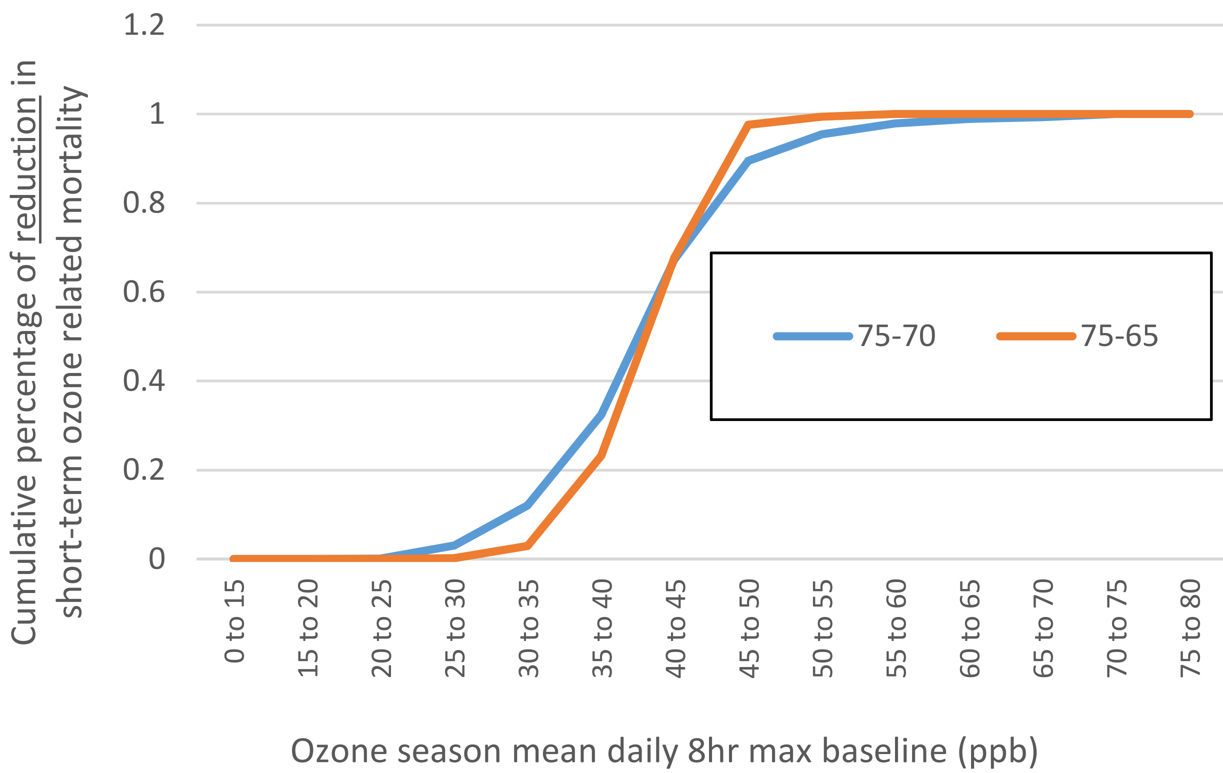


Figure 6B-2. Cumulative Probability Plot of Premature Ozone-related Deaths Avoided for the Revised and Alternative Standards (2025 scenario) According to the Baseline Ozone Concentrations

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 the revised and alternative standards, we adjust the design value at each monitor exceeding the standard 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 revised or 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 revised or alternative standards because it would omit benefits attributable to emissions reductions in non-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.

As shown in Figures 6B-1 and 6B-2, the vast majority of reductions in short-term exposure-related mortality for ozone occur in grid cells with mean 8-hour max baseline levels (across the ozone season) between 35 and 55 ppb. Comparing patterns across the revised and alternative standard levels, the upper end of the distribution shifts downwards as increasingly lower standard levels are analyzed (see Figure 6B-2).

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, 2011c). 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.[173] 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, 2009).

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, 2012b). 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 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.[174] 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

[173] 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).

[174] 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.

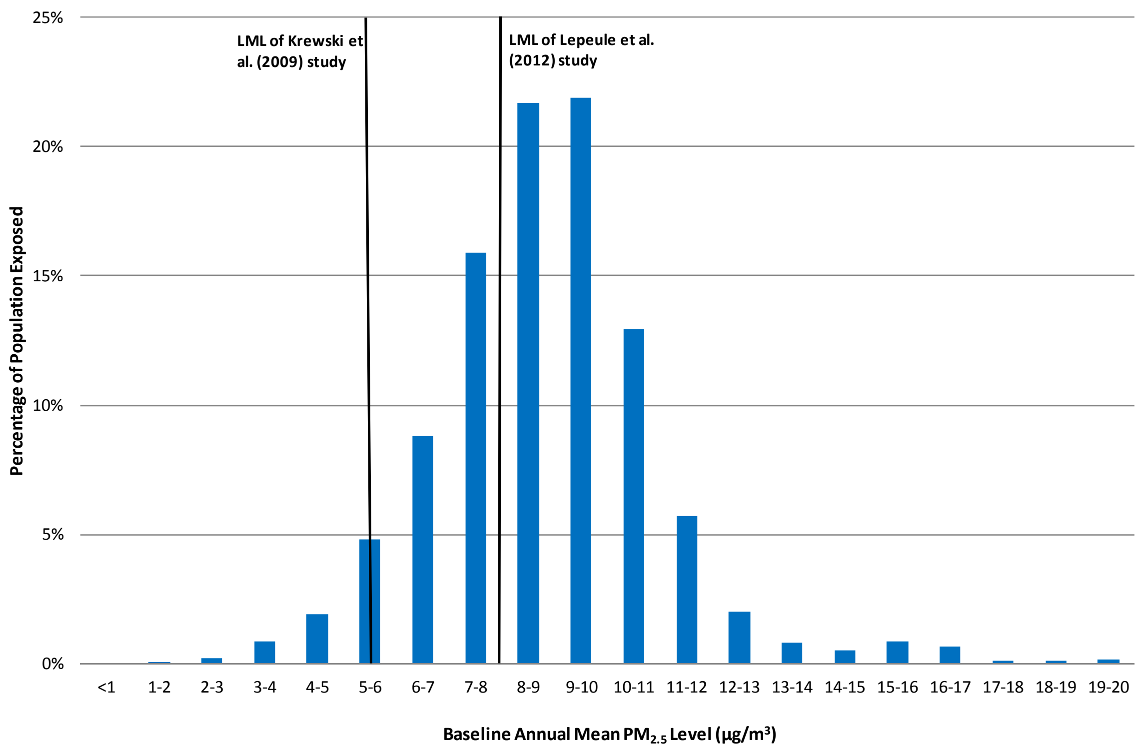
air pollution below the LML is not the same as the percentage of the population experiencing health impacts as a result of a specific emissions reduction policy. The most important aspect, which we are unable to quantify without rule-specific air quality modeling, is the shift in exposure anticipated by the revised or alternative 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 6B-12 provides the percentage of the population exposed above and below two concentration benchmarks in the modeled baseline for the sector modeling. Figure 6B-3 shows a bar chart of the percentage of the population exposed to various air quality levels in the baseline, and Figure 6B-4 shows a cumulative distribution function of the same data. Both figures identify the LML for each of the major cohort studies.

Table 6B-12. Population Exposure in the Baseline Sector Modeling (used to generate the benefit-per-ton estimates) Above and Below Various Concentration 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 et al., 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.



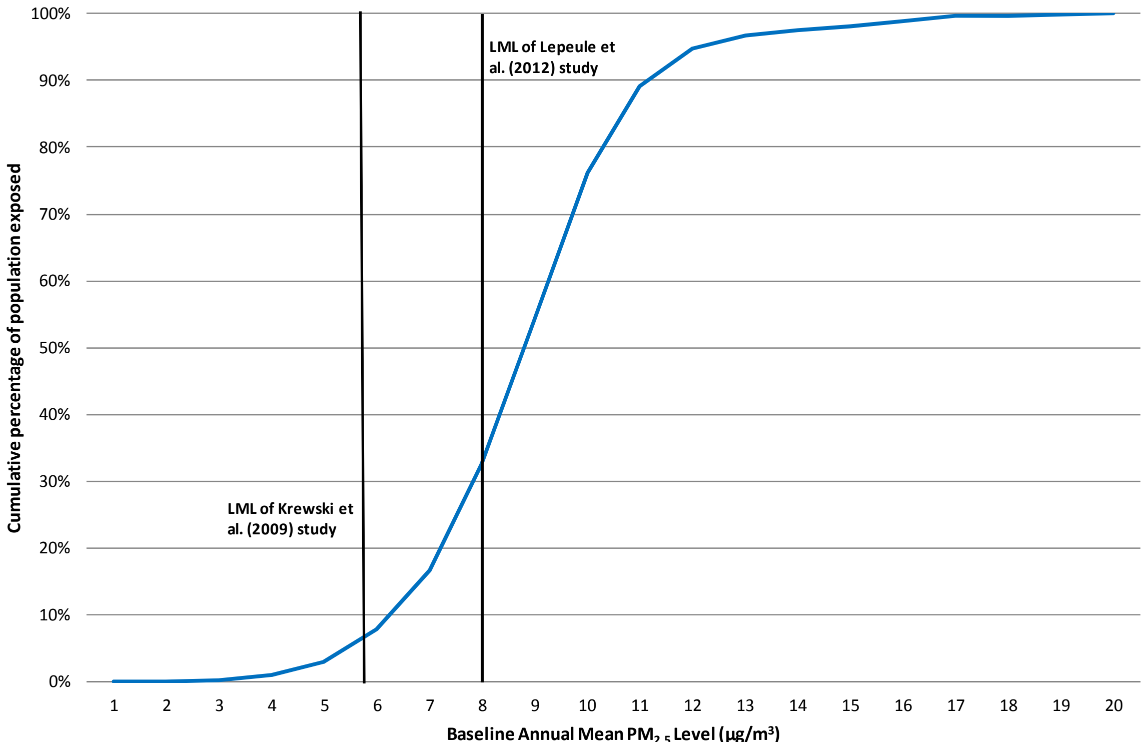
Among the populations exposed to $PM_{2.5}$ in the baseline:

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 6B-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 6B-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).

6B.8 Ozone-related Impacts on Outdoor Worker Productivity

The EPA last quantified the value of ozone-related worker productivity in the final Regulatory Impact Analysis supporting the Transport Rule (U.S. EPA, 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 a recent study, Graff Zivin and Neidell (2012) combined data on individual-level daily harvest rates for Outdoor Agricultural Workers (OWAs) with ground-level ozone concentrations to characterize changes in worker productivity as a result of ozone exposure. 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 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 recognize that there is significant uncertainty in the generalizability of this study and the need for additional research and peer review in guiding the monetization of agriculatural productivity impacts. Because we received no comments on this proposed approach, we now include the results as part of our uncertainty analysis.

Below we provide the function, input data and results for the analysis.

$$Y = \beta * DeltaAQ * DailyOutdoorWage * OutdoorAgWorkers * EmplGrow$$

Table 6B-13. Definitions of Variables Used to Calculate Changes in Worker Productivity

Variable	Definition
β	Percent change in daily outdoor worker productivity per 1ppb change in ozone (Graff Zivin and Neidell, 2012)
$DeltaAQ$	Summer season average of daily 9 hour average (6am to 3pm)
$DailyOutdoorWage$	Summer season daily wage for agricultural non-livestock workers in 17 USDA-defined regions (USDA, 2012)
$OutdoorAgWorkers$	Summer season number of workers employed in outdoor non-livestock agricultural per county (USDA, 2010; US Census, 2010)
$EmplGrow$	Growth in agricultural non-livestock workers to 2025 (Woods and Poole, 2012)

Table 6B-14. Population Estimated Economic Value of Increased Productivity among Outdoor Agricultural Workers from Attaining the Revised and Alternative Ozone Standards in 2025 (millions of 2011$)

Standard	Economic Value (95th percentile confidence interval)
70 ppb	$1.7 ($0.1 to $3.3)
65 ppb	$8.9 ($0.6 to $17)

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CHAPTER 7: 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. Although we have not conducted an analysis to represent the ozone improvements from emissions reductions estimated in the final RIA, we reference the analysis conducted in the proposal RIA (U.S. EPA, 2014b) as an indication of the potential magnitude of effects associated with changes in yields of commercial forests and agriculture, and carbon sequestration and storage. We did not update the analysis from the proposal RIA because the welfare co-benefits estimates (i) in the proposal analysis were small, and we anticipated that the estimates in the final analysis would be even smaller, and (ii) are not added to the human health benefits estimates.

The EPA has also concluded that the current secondary standard for ozone, set at a level of 75 ppb, is not requisite to protect public welfare from known or anticipated adverse effects, and is revising the standard to provide increased protection against vegetation-related effects on public welfare. Specifically, the EPA is retaining the indicator (ozone), averaging time (8-hour) and form (annual fourth-highest daily maximum, averaged over 3 years) of the existing secondary standard and is revising the level of that standard to 70 ppb. The EPA has concluded that this revision will effectively curtail cumulative seasonal ozone exposures above 17 ppm-hrs, in terms of a three-year average seasonal W126 index value, based on the three consecutive month period within the growing season with the maximum index value, with daily exposures cumulated for the 12-hour period from 8:00 am to 8:00 pm. Thus, the EPA has concluded that this revision will provide the requisite protection against known or anticipated adverse effects to the public welfare.

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. However, we are not able to quantify or monetize these benefits in this RIA.

7.1 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 7-1 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 7-1. Welfare Effects of NO_x 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	✓	✓	✓		✓		

7.2 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, 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, an essential symbiotic fungus 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 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 2014a). 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, 2014a).

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, 2014a). 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 the proposal RIA (U.S. EPA, 2014b), we were 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 analyzed 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 assessed the effects of those changes in commercial agricultural and forest yields on carbon sequestration and storage. This analysis provided 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, 2014a). Because we did not update this analysis for this RIA, we refer the reader to the results in the proposal RIA for an indication of the potential magnitude of these welfare benefits.

7.3 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 and 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.

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CHAPTER 8: COMPARISON OF COSTS AND BENEFITS

Overview

The EPA performed an illustrative analysis to estimate the costs and human health benefits of nationally attaining revised and alternative ozone standards. The EPA Administrator is revising the level of the primary ozone standard to 70 ppb. Per Executive Order 12866 and the guidelines of OMB Circular A-4, this Regulatory Impact Analysis (RIA) presents analysis of an alternative standard level of 65 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 cost and benefit 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 the revised and 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. Because of data and resource constraints, 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 will be 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 post-2025 costs and benefits estimates for California should not be added to the primary 2025 estimates for the rest of the U.S.

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 them 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 and benefits due to (1) emissions reductions 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 total costs and benefits of the revised and alternative standard levels analyzed, and show the net benefits for each of the levels across a range of modeling assumptions related to the calculation of costs and benefits.[175] Tables 8-3 and 8-4 summarize the costs and benefits resulting from identified control strategies and do not include the additional costs and benefits associated with the unidentified control strategies. Tables 8-5 and 8-6 provide information on the total costs by geographic region for the U.S., except California in 2025 and on the costs for California for post-2025. Tables 8-7 and 8-8 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 revised and alternative 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 both control strategies applied within the region and reductions in transport of ozone associated with emissions reductions in other regions.

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 at the national level, we do not estimate separately the nationwide benefits associated with the emissions reductions

[175] As discussed in Chapter 2, Section 2.2.5, of the 1,225 ozone monitors with complete ozone data, there were seven monitors, or 0.6 percent of the total, for which the DVs were influenced by wintertime ozone episodes. These seven monitors were removed from the analysis because the modeling tools are not currently sufficient to properly characterize ozone formation during wintertime ozone episodes. Because there was no technically feasible method for projecting DVs at these sites, these sites were not included in determining required NO_x and VOC emissions reductions needed to meet the revised or alternative standard levels. There could be additional emissions reductions required to lower ozone at these locations and associated additional costs and benefits not reflected in this analysis.

occurring in any specific region.[176] As a result, we are only able to provide net benefits estimates at the national level. The difference between the estimated benefits accruing to a specific region and the costs for that region is not an estimate of net benefits of the emissions reductions in that region because it ignores the benefits occurring outside of that region.

Table 8-1. Total Costs, Total Monetized Benefits, and Net Benefits of Control Strategies in 2025 for U.S., except California (billions of 2011$)[a,b]

Revised and Alternative Standard Levels	Total Costs[c]	Monetized Benefits	Net Benefits
	7% Discount Rate	7% Discount Rate	7% Discount Rate
70	$1.4	$2.9 to $5.9[d]	$1.5 to $4.5
65	$16	$15 to $30[d]	-$1.0 to $14

[a] All values are rounded to two significant figures.
[b] 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 numbers presented in this table reflect engineering costs annualized at a 7 percent discount rate to the extent possible.
[d] Excludes additional health and welfare benefits that could not be quantified (see Chapter 6, Section 6.6.3.8).

Table 8-2. Total Costs, Total Monetized Benefits, and Net Benefits of Control Strategies Applied in California, Post-2025 (billions of 2011$)[a]

Revised and Alternative Standard Levels	Total Costs	Monetized Benefits	Net Benefits
	7% Discount Rate	7% Discount Rate	7% Discount Rate
70	$0.8	$1.2 to $2.1[c]	$0.4 to $1.3
65	$1.5	$2.3 to $4.2[c]	$0.8 to $2.7

[a] 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.
[b] The numbers presented in this table reflect engineering costs annualized at a 7 percent discount rate to the extent possible.
[c] Excludes additional health and welfare benefits that could not be quantified (see Chapter 6, Section 6.6.3.8).

[176] 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.

Table 8-3. Summary of Identified Control Strategies Annualized Control Costs by Sector for 70 ppb and 65 ppb for 2025 - U.S., except California (millions of 2011$)[a]

Geographic Area	Emissions Sector	Identified Control Costs for 70 ppb	Identified Control Costs for 65 ppb
		7 Percent Discount Rate[b]	7 Percent Discount Rate[b]
East	EGU	52[c]	130[c]
	Non-EGU Point	260[d]	750[d]
	Nonpoint	360	1,500
	Nonroad	13[e]	36[e]
	Total	**690**	**2,400**
West	EGU	-	-
	Non-EGU Point	4[d]	49[d]
	Nonpoint	<1	88
	Nonroad	-	4[e]
	Total	**4**	**140**
Total Identified Control Costs		**690**	**2,600**

[a] All values are rounded to two significant figures.
[b] The numbers presented in this table reflect engineering costs annualized at a 7 percent discount rate to the extent possible.
[c] EGU control cost data is calculated using a capital charge rate between 7 and 12 percent for retrofit controls depending on the type of equipment.
d A share of the non-EGU point source costs can be calculated using both 3 and 7 percent discount rates. A share of the non-EGU point source sector costs can be calculated using both 3 and 7 percent discount rates. When applying a 3 percent discount rate where possible, the total non-EGU point source sector costs are $250 million for 70 ppb and $740 million for 65 ppb.
[e] Non-EGU point source costs at a 3 percent discount rate are $72 million for 70 ppb and $180 million for 65 ppb.

Table 8-4. Estimated Monetized Ozone and $PM_{2.5}$ Benefits for Revised and Alternative Annual Ozone Standards Incremental to the Baseline for the 2025 Scenario (Nationwide Benefits of Attaining the Standards in the U.S. except California) – Identified Control Strategies (billions of 2011$) [a]

	Discount Rate	Revised and Alternative Standard Levels	
		70 ppb	65 ppb
Identified Control Strategies			
Ozone-only Benefits (range reflects Smith et al. (2009) to Zanobetti and Schwartz (2008))	[b]	\$0.86 to \$1.4	\$2.2 to \$3.5
$PM_{2.5}$ Co-benefits (range reflects Krewski et al. (2009) to Lepeule et al. (2012)	3%	\$1.7 to \$3.9	\$4.0 to \$9.0
	7%	\$1.6 to \$3.5	\$3.6 to \$8.1
Total Benefits	3%	\$2.6 to \$5.3[c]	\$6.1 to \$12[c]
	7%	\$2.4 to \$4.9[c]	\$5.7 to \$12[c]

[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.
[c] Excludes additional health and welfare benefits that could not be quantified (see Chapter 6, Section 6.6.3.8).

Table 8-5. Summary of Total Control Costs (Identified + Unidentified) by Revised and Alternative Standard Level for 2025 - U.S., except California (millions of 2011$, 7% Discount Rate)[a]

Revised and Alternative Level	Geographic Area	Identified Control Costs	Unidentified Control Costs	Total Control Costs (Identified and Unidentified)
70 ppb	East	690	700	1,400
	West	4	-	<5
	Total	**\$690**	**\$700**	**\$1,400**
65 ppb	East	2,400	12,000	15,000
	West	140	610	750
	Total	**\$2,600**	**\$12,600**	**\$16,000**

[a] All values are rounded to two significant figures. Unidentified control costs are based on an average cost-per-ton methodology described in Chapter 4.

Table 8-6. Summary of Total Control Costs (Unidentified Control Strategies) by Revised and Alternative Level for Post-2025 - California (millions of 2011$, 7% Discount Rate)[a]

Revised and Alternative Level	Geographic Area	Total Control Costs (Unidentified)
70 ppb	California	$800
65 ppb	California	$1,500

[a] All values are rounded to two significant figures. Unidentified control costs are based on an average cost-per-ton methodology described in Chapter 4.

Table 8-7. Regional Breakdown of Monetized Ozone-Specific Benefits Results for the 2025 Scenario (nationwide benefits of attaining revised and alternative standard levels everywhere in the U.S. except California) – Identified + Unidentified Control Strategies[a]

Region	Alterative Standards	
	70 ppb	**65 ppb**
East [b]	98%	96%
California	0%	0%
Rest of West	2%	4%

[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 8-8. Regional Breakdown of Monetized Ozone-Specific Benefits Results for the Post-2025 Scenario (nationwide benefits of attaining revised and alternative standard levels just in California) – Identified + Unidentified Control Strategies[a]

Region	Alterative Standards	
	70 ppb	**65 ppb**
East	3%	2%
California	90%	91%
Rest of West	7%	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.

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 8-9 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 8-9. 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
	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

Benefits Category	Specific Effect	Effect Has Been Quantified	Effect Has Been Monetized	More Information
	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]
Reduced incidence of morbidity from exposure to NO_2	Asthma hospital admissions (all ages)	—	—	NO_2 ISA [d]
	Chronic lung disease hospital admissions (age > 65)	—	—	NO_2 ISA [d]
	Respiratory emergency department visits (all ages)	—	—	NO_2 ISA [d]
	Asthma exacerbation (asthmatics age 4–18)	—	—	NO_2 ISA [d]
	Acute respiratory symptoms (age 7–14)	—	—	NO_2 ISA [d]
	Premature mortality	—	—	NO_2 ISA [b,c]
	Other respiratory effects (e.g., airway hyperresponsiveness and inflammation, lung function, other ages and populations)	—	—	NO_2 ISA [b,c]

[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 Improvements between the Proposal and Final RIAs

In the regulatory impact analyses for both the proposed and revised ozone NAAQS, there were two geographic areas outside of California where the majority of emissions reductions were needed to meet a standard level of 70 ppb – Texas and the Northeast. In analyzing 70 ppb in this RIA for the revised NAAQS, there were approximately 50 percent fewer emissions reductions needed in these two geographic areas. For an alternative standard of 65 ppb, emissions reductions needed nationwide were approximately 20 percent lower than at proposal.

The primary reason for the difference in emissions reductions estimated for attainment is that for this RIA we conducted more geographically-refined air quality sensitivity modeling to develop improved ozone response factors (see Chapter 2, Section 2.1.4 for a more detailed discussion of this air quality modeling) and focused the emissions reduction strategies on geographic areas closer to the monitors with the highest design values (see Chapter 3, Section 3.1.2 for a more detailed discussion of the emissions reduction strategies). The improvements in air quality modeling and emissions reduction strategies account for about 80 percent of the difference in needed emissions reductions between the two RIAs.

In Texas and the Northeast, the updated response factors and more focused emissions reduction strategies resulted in larger changes in ozone concentrations in response to more geographically focused emissions reductions. In east Texas, the ppb/ton ozone response factors used in this RIA were 2 to 3 times more responsive than the factors used in the proposal RIA at controlling monitors in Houston and Dallas. In the Northeast, the ppb/ton ozone response factors used in this RIA were 2.5 times more responsive than the factors used in the proposal RIA at the controlling monitor on Long Island, NY.

A secondary reason for the difference is that in the time between developing the two RIAs we updated emissions inventories, models and model inputs for the base year of 2011. See Chapter 2, Section 2.1 and 2.2 for additional discussion of the updated emissions inventories, models and model inputs. When projected to 2025, these changes in inventories, models and inputs had compounding effects for year 2025, and in some areas resulted in lower projected base case design values for 2025. The updated emissions inventories, models, and model inputs account for about 20 percent of the difference in needed emissions reductions between the two RIAs.

These differences in the estimates of emissions reductions needed to attain the revised and alternative standard levels affect the estimates for the costs and benefits in this RIA. For a revised standard of 70 ppb, the costs were 60 percent lower than at proposal and the benefits were 55 percent lower than at proposal. The percent decrease in costs is slightly more than the percent decrease in emissions reductions because a larger number of lower cost identified controls were available to bring areas into attainment with 70 ppb.[177] The percent decrease in benefits is similar to the percent decrease in emissions reductions. For an alternative standard level of 65 ppb, the costs were less than three percent higher than those estimated at proposal and the benefits were 22 percent lower than at proposal.[178] The percent change in costs was less than the percent decrease in emissions reductions because in this analysis we applied identified controls in smaller geographic areas, resulting in fewer identified controls available within those areas and an increase in higher cost unidentified controls being applied to bring areas into attainment with 65 ppb. The percent decrease in benefits is similar to the percent decrease in emissions reductions.

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 approximately half to three-quarters of the estimated benefits, depending on the standard analyzed and on the choice of ozone and PM mortality functions used.

[177] In the final RIA, outside of California all areas were projected to meet the current standard of 75 ppb. As such, no identified controls were used to bring areas into attainment with 75 ppb. In the proposal RIA, some of these lower cost controls were used to bring areas into attainment with 75 ppb, making them unavailable for application in the analysis of 70 ppb.

[178] We have slightly modified our approach to estimating morbidity benefits since proposal, which had a negligible (~1%) influence on the total monetized benefits in this RIA.

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 identified emissions control technologies, the EPA recognized that identified 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 level. 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, end-of-pipe emissions controls, and the costs and benefits associated with those controls. The second stage took the emissions reductions beyond identified controls and used an average cost-per-ton 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 identified controls. We used the information currently available to develop reasonable approximations of the costs and benefits of the unidentified portion of the emissions reductions necessary to reach attainment. However, because of the uncertainty associated with the costs of unidentified controls, we judged it appropriate to provide separate estimates of the costs and benefits for partial attainment (based on identified controls) and full attainment (based on identified controls and unidentified controls), 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 and final 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 and revised 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 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 identified and available controls in our database. With the more stringent NAAQS, more areas will need to employ additional ways of reducing emissions. The control measures reflected in our databases include very few abatement possibilities from energy efficiency measures, fuel switching, input or process changes, or other abatement strategies that are non-traditional in the sense that they are not the application of an end-of-pipe control. So while we can speculate on what some of the future emissions reduction strategies 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 identified 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 4, Section 4.2.3 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 Net Present Value of a Stream of Costs and Benefits

The EPA believes that providing comparisons of social costs and social benefits at discount rates of 3 and 7 percent is appropriate. 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.

The EPA presents annualized costs and benefits in a single year for comparison in this RIA because there are a number of methodological complexities associated with calculating the net present value (NPV) of a stream of costs and benefits for a NAAQS. While NPV analysis allows evaluation of alternatives by summing the present value of all future costs and benefits, insights into how costs will occur over time are limited by underlying assumptions and data. Calculating a present value (PV) of the stream of future benefits also poses special challenges, which we describe below. In addition, the method requires definition of the length of that future time period, which is not straightforward for this analysis and subject to uncertainty.

To estimate engineering costs, the EPA employs the equivalent uniform annual cost (EUAC) method, which annualizes costs over varying lifetimes of control measures applied in the analysis.[179] Using the EUAC method results in a stream of annualized costs that is equal for each year over the lifetime of control measures, resulting in a value similar to the value associated with an amortized mortgage or other loan payment. Control equipment is often purchased by incurring debt rather than through a single up-front payment. Recognizing this led the EPA to estimate costs using the EUAC method instead of a method that mimics firms paying up front for the future costs of installation, maintenance, and operation of pollution control devices.

Further, because we do not know when a facility will stop using a control measure or change to another measure based on economic or other reasons, the EPA assumes the control equipment and measures applied in the illustrative control strategies remain in service for their full useful life. As a result, the annualized cost of controls in a single future year is the same throughout the lifetimes of control measures analyzed, allowing the EPA to compare the annualized control costs with the benefits in a single year for consistent comparison.

The EPA's RIAs for air quality rules generally report the estimated net benefits of improved air quality for a single year. The estimated NPV can better characterize the stream of benefits and costs over a multi-year period. However, calculating the PV of improved air quality is generally quite data-intensive and costly. Further, the results are sensitive to assumptions regarding the time period over which the stream of benefits is discounted.

The theoretically appropriate approach for characterizing the PV of benefits is the life table approach. The life table, or dynamic population, approach explicitly models the year-to-year influence of air pollution on baseline mortality risk, population growth and the birth rate—typically for each year over the course of a 50-to-100 year period (U.S. EPA SAB, 2010; Miller, 2003). In contrast to the pulse approach[180], a life table models these variables endogenously by following a population cohort over time. For example, a life table will "pass" the air pollution-modified baseline death rate and population from year to year; impacts estimated in year 50 will

[179] See Chapter 4, Section 4.1.1 for additional information on the EUAC method.
[180] The pulse approach assumes changes in air pollution in a single year and affects mortality estimates over a 20-year period.

account for the influence of air pollution on death rates and population growth in the preceding 49 years.

Calculating year-to-year changes in mortality risk in a life table requires some estimate of the annual change in air quality levels. It is both impractical to model air quality levels for each year and challenging to account for changes in federal, state and local policies that will affect the annual level and distribution of pollutants. For each of these reasons the EPA has not generally reported the PV of benefits for air rules but has instead pursued a pulse approach. While we agree that providing the NPV of a stream of costs and benefits could be informative, based on these reasons we are not able to provide the NPV of that stream in this RIA.

8.4 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.[181] These factors, summarized in Table 8.10 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 affect 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,

[181] OMB Circular A-4 indicates that qualitative discussions should be included in analyses whenever there is insufficient data to quantify uncertainty.

broader uncertainties about future trends that provide important context for the costs and benefits presented in this analysis.

Table 8-10. 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.	Recent increases 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[182] have likely led to a reduction of U.S. dependence on imports of foreign oil. Upward trends that have emerged over the last ten years in natural gas production and consumption, renewable energy installations, and energy efficiency technology installations are likely to continue.[183]
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.[184]	Recent trends in VMT illustrate some of the uncertainty around future emissions from mobile sources.[185] In 2006, projections of VMT showed a sustained increase,[186] yet VMT growth slowed in recent years and actually declined in 2008 and 2009.[187] 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	Affluence leads to increased consumption and energy use. However, this increase may not be proportional. Energy and materials use is not directly proportional to economic growth, but decrease or stabilize over time in spite of continued economic growth.[189]

[182] http://energy.gov/articles/us-domestic-oil-production-exceeds-imports-first-time-18-years

[183] 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.

[184] 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.

[185] For example, see the Transportation Research Board's National Cooperative Highway Research Program (NCHRP) 2014.

[186] https://www.fhwa.dot.gov/policy/2006cpr/chap9.htm#body

[187] https://www.fhwa.dot.gov/policyinformation/travel_monitoring/13jantvt/page2.cfm

[189] UNEP 2011, http://www.unep.org/resourcepanel/decoupling/files/pdf/decoupling_report_english.pdf

Individual Factors	Potential Implications for NAAQS Attainment	Information on Trends
	emissions making attainment potentially less costly.[188]	
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 would likely be less costly. If not, attainment would likely be more costly. A move toward investments in fuel efficiency and low emissions fuels could decrease emissions and likely lower attainment costs.	State and local policies related to energy efficiency, cleaner energy, energy security[190], 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 electric power, the use of cleaner fuels and conservation measures would likely result in decreased emissions and likely decrease attainment costs.[191] 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.[192] U.S. productivity per energy expended relative to other countries suggests that additional efficiency gains are possible.[193]
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.[194]
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.[195]

[a] Policies refer to any policies or regulations that are not environmental regulations set by U.S. EPA, states, tribes, or local authorities.

[188] For example, Bo (2011), and http://www.epa.gov/region1/airquality/nox.html for manufactures contributions to NO_X emissions.
[190] For example, http://www2.epa.gov/laws-regulations/summary-energy-independence-and-security-act.
[191] For example, see http://www.dsireusa.org/solar/solarpolicyguide/.
[192] http://bipartisanpolicy.org/sites/default/files/BPC%20SEPI%20Energy%20Report%202013_0.pdf, p. 5.
[193] http://bipartisanpolicy.org/sites/default/files/BPC%20SEPI%20Energy%20Report%202013_0.pdf, p. 69.
[194] See Jacobs (2009).
[195] http://www.eia.gov/todayinenergy/detail.cfm?id=16211

8.5 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 identified NOx and VOC controls comprise only a small part of the estimated costs of full attainment. These estimated costs for the identified set of 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.

- **An 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 fifteen (15) 2025 emissions sensitivity simulations. The 15 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 revised and alternative standard levels of 70 and 65 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.

- **NOx and VOC emissions reductions quantified from the technologies identified in this RIA may not be sufficient to attain alternative ozone NAAQS in some areas.** 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 or alternative standards. Chapter 4 contains discussion of other emissions reduction measures not quantified in this RIA, as well as discussion of technological improvement over time. After applying existing rules and the illustrative identified controls across the nation (excluding California), in order to reach 70 ppb we were able to identify controls that reduce overall NOx emissions by 240,000 tons and VOC emissions by 20,000 tons. In order to reach 65 ppb we were able to identify controls that reduce overall NOx emissions by 560,000 tons and VOC emissions by 110,000 tons. After these reductions, in order to reach 70 ppb over 47,000 tons of NOx emissions remained, and in order to reach 65 ppb over 860,000 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 unidentified 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 some 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 unidentified portion of the emissions reductions was too uncertain to be included as part of these estimates. Therefore, we did not include the economic impacts of either the identified control costs or costs of unidentified controls.
- **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.

8.6 References

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CHAPTER 9: STATUTORY AND EXECUTIVE ORDER IMPACT ANALYSES

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 Executive Order 12866: Regulatory Planning and Review

This action is an economically significant regulatory action that was submitted to the Office of Management and Budget (OMB) for review. Any changes made in response to OMB recommendations have been documented in the docket. This RIA estimates the costs and monetized human health and welfare benefits of attaining two alternative ozone NAAQS nationwide. Specifically, the RIA examines the alternatives of 65 ppb and 70 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 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 an RIA has been prepared, the results of the RIA have not been considered in issuing the revised NAAQS.

9.2 Paperwork Reduction Act

The information collection requirements for the revised NAAQS have been submitted for approval to the Office of Management and Budget (OMB) under the Paperwork Reduction Act (PRA). The information collection requirements are not enforceable until OMB approves them. The Information Collection Request (ICR) document prepared by the EPA for these revisions has been assigned EPA ICR #2313.04.

The information collected and reported under 40 CFR part 58 is needed to determine compliance with the NAAQS, to characterize air quality and associated health and ecosystems impacts, to develop emission control strategies, and to measure progress for the air pollution program. We are extending the length of the required ozone monitoring season in 32 states and the District of Columbia and the revised ozone monitoring seasons will become effective on

January 1, 2017. We are also revising the PAMS monitoring requirements to reduce the number of required PAMS sites while improving spatial coverage, and requiring states in moderate or above ozone non-attainment areas and the ozone transport region to develop an enhanced monitoring plan as part of the PAMS requirements. Monitoring agencies will need to comply with the PAMS requirements by June 1, 2019. In addition, we are revising the ozone FRM to establish a new, additional technique for measuring ozone in the ambient air. It will be incorporated into the existing ozone FRM, using the same calibration procedure in Appendix D of 40 CFR part 50. We are also making changes to the procedures for testing performance characteristics and determining comparability between candidate FEMs and reference methods.

For the purposes of ICR number 2313.04, the burden figures represent the burden estimate based on the requirements contained in this rule. The burden estimates are for the 3-year period from 2016 through 2018. The implementation of the PAMS changes will occur beyond the time frame of this ICR with implementation occurring in 2019. The cost estimates for the PAMS network (including revisions) will be captured in future routine updates to the Ambient Air Quality Surveillance ICR that are required every 3 years by OMB. The addition of a new FRM in 40 CFR part 50 and revisions to the ozone FEM procedures for testing performance characteristics in 40 CFR part 53 does not add any additional information collection requirements.

The ICR burden estimates are associated with the changes to the ozone seasons in the revised NAAQS. This information collection is estimated to involve 158 respondents for a total cost of approximately $24,597,485 (total capital, labor, and operation and maintenance) 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 $20,209,966. Also included in the total are other costs of operations and maintenance of $2,254,334 and equipment and contract costs of $2,133,185. The actual labor cost increase to expand the ozone monitoring seasons is $2,064,707. 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 $2,670,360. Burden is defined at 5 CFR 1320.3(b). State, local, and tribal entities are eligible for state assistance grants provided by the Federal government under the CAA which can be used for related activities. 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 EPA's regulations in 40 CFR are listed in 40 CFR part 9.

9.3 Regulatory Flexibility Act

This action will not have a significant economic impact on a substantial number of small entities under the RFA. This action will not impose any requirements on small entities. Rather, the 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). Similarly, the revisions to 40 CFR part 58 address the requirements for states to collect information and report compliance with the NAAQS and will not impose any requirements on small entities. Similarly, the addition of a new FRM in 40 CFR part 50 and revisions to the FEM procedures for testing in 40 CFR part 53 will not impose any requirements on small entities.

9.4 Unfunded Mandates Reform Act

This action does not contain any unfunded mandate as described in UMRA, 2 U. S. C. 1531 – 1538, and does not 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 RIA pursuant to the UMRA would not furnish any information which the court could consider in reviewing the NAAQS).

9.5 Executive Order 13132: Federalism

This action does not have federalism implications. It will not have substantial direct effects on the states, on the relationship between the national government and the states, or on the distribution of power and responsibilities among the various levels of government.

9.6 Executive Order 13175: Consultation and Coordination with Indian Tribal Governments

This action 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 as 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 action.

The EPA specifically solicited comment on this rule from tribal officials. The EPA also conducted outreach consistent with the EPA Policy on Consultation and Coordination with Indian Tribes. Outreach to tribal environmental professionals was conducted through participation in the Tribal Air call, which is sponsored by the National Tribal Air Association. Consistent with the EPA Policy on Consultation and Coordination with Indian Tribes, the EPA offered formal consultation to the tribes during the public comment period. Consultation was not requested.

9.7 Executive Order 13045: Protection of Children from Environmental Health & Safety Risks

This action is subject to Executive Order 13045 because it is an economically significant regulatory action as defined by Executive Order 12866, and the EPA believes that the environmental health risk addressed by this action may have a disproportionate effect on children. The rule will establish uniform NAAQS 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 at-risk populations196 such as all people with lung disease and people active outdoors, are at increased risk for health effects associated with exposure to ozone in ambient air. Because children are considered an at-risk lifestage, 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,

[196] As used here and similarly throughout this document, the term population refers to people having a quality or characteristic in common, including a specific pre-existing illness or a specific age or lifestage.

policy considerations, and the exposure and risk assessments pertaining to children are contained in sections II.B and II.C of the preamble.

9.8 Executive Order 13211: Actions that Significantly Affect Energy Supply, Distribution, or Use

This action is not a "significant energy action" because it is not likely to have a significant adverse effect on the supply, distribution, or use of energy. The purpose of the rule is to establish a revised NAAQS for ozone, establish an additional FRM, revise FEM procedures for testing, and revises air quality surveillance requirements. The rule does not prescribe specific pollution control strategies by which these ambient standards and monitoring revisions 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 this 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 identified controls for power plants, shown in Chapter 4, means that 4 percent of the total projected coal-fired EGU capacity nationwide in 2025 could be affected by controls for the revised standard level of 70 ppb. Similarly, 13 percent of total projected coal-fired EGU capacity in 2025 could be affected by controls for the alternative standard of 65 ppb. In addition, some fuel switching might occur that could alter these percentages, although we are not able to estimate the effect on energy impacts from fuel switching. In addition, we are not able 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 controls to sources other than EGUs for the purpose of attaining a more stringent standard.

9.9 National Technology Transfer and Advancement Act

This rulemaking involves environmental monitoring and measurement. Consistent with the Agency's Performance Based Measurement System (PBMS), the EPA is not requiring the use of specific, prescribed analytical methods. Rather, the Agency is allowing the use of any method that meets the prescribed performance criteria. Ambient air concentrations of ozone are currently measured by the Federal reference method (FRM) in 40 CFR part 50, Appendix D (Measurement Principle and Calibration Procedure for the Measurement of Ozone in the Atmosphere) or by Federal equivalent methods (FEM) that meet the requirements of 40 CFR part 53. Procedures are available in part 53 that allow for the approval of an FEM for ozone that is similar to the FRM. Any method that meets the performance criteria for a candidate equivalent method may be approved for use as an FEM. This approach is consistent with EPA's PBMS. The PBMS approach is intended to be more flexible and cost-effective for the regulated community; it is also intended to encourage innovation in analytical technology and improved data quality. The EPA is not precluding the use of any method, whether it constitutes a voluntary consensus standard or not, as long as it meets the specified performance criteria.

9.10 Executive Order 12898: Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations

The EPA believes that this action will not have disproportionately high and adverse human health or environmental effects on minority populations, low-income populations or indigenous peoples. The action described in this notice is to strengthen the NAAQS for ozone.

The primary NAAQS are established at a level that is requisite to protect public health, including the health of sensitive or at-risk groups, with an adequate margin of safety. The NAAQS decisions are based on an explicit and comprehensive assessment of the current scientific evidence and associated exposure/risk analyses. More specifically, EPA expressly considers the available information regarding health effects among at-risk populations, including that available for low-income populations and minority populations, in decisions on NAAQS. Where low-income populations or minority populations are among the at-risk populations, the decision on the standard is based on providing protection for these and other at-risk populations and lifestages. Where such populations are not identified as at-risk populations, a NAAQS that is established to provide protection to the at-risk populations would also be expected to provide protection to all other populations, including low-income populations and minority populations.

The Integrated Science Assessment, the Health Risk and Exposure Assessment, and the Policy Assessment for this review, which include identification of populations at risk from ozone health effects, are available in the docket EPA-HQ-OAR-2008-0699. The information on at-risk populations for this NAAQS review is summarized and considered in the rule preamble (see section II.A). The final rule increases the level of environmental protection for all affected populations without having any disproportionately high and adverse human health or environmental effects on any population, including any minority populations, low-income populations or indigenous peoples. The rule establishes uniform national standards for ozone in ambient air that, in the Administrator's judgment, protect public health, including the health of sensitive groups, with an adequate margin of safety.

Although it has a separate docket and is not part of the rulemaking record for this action, EPA has prepared a RIA of this decision. As part of the RIA, a demographic analysis was conducted. While, as noted in the RIA, the demographic analysis is not a full quantitative, site-specific exposure and risk assessment, that analysis examined demographic characteristics of persons living in areas with poor air quality relative to the revised standard. Specifically, Appendix 9A describes the proximity and socio-demographic analysis. The analysis found that in areas with poor air quality relative to the revised standard,197 the representation of minority populations was slightly greater than in the U.S. as a whole. Because the air quality in these areas does not currently meet the revised standard, populations in these areas would be expected to benefit from implementation of the strengthened standard, and, thus, would be more affected by strategies to attain the revised standard. The analysis, which evaluates the potential implications for minority populations and low-income populations of future air pollution control actions that state and local agencies may consider in implementing the revised ozone NAAQS described in the decision notice, is discussed in Appendix 9A. The RIA is available on the Web, through the EPA's Technology Transfer Network website at http://www.epa.gov/ttn/naaqs/standards/ozone/s_o3_index.html and in the RIA docket (EPA-HQ-OAR-2013-0169). As noted above, although an RIA has been prepared, the results of the RIA have not been considered in issuing this final rule.

[197] This refers to monitored areas with ozone design values above the revised and alternative standards.

9.11 Congressional Review Act (CRA)

This action is subject to the CRA, and the EPA will submit a rule report to each House of the Congress and to the Comptroller General of the United States. This action is a "major rule" as defined by 5 U.S.C. 804(2).

APPENDIX 9A: SOCIO-DEMOGRAPHIC CHARACTERISTICS OF POPULATIONS IN CORE BASED STATISTICAL AREAS WITH OZONE MONITORS EXCEEDING REVISED AND ALTERNATIVE OZONE STANDARDS

Overview

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 revisions to the National Ambient Air Quality Standards (NAAQS) for ozone. 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 (2012-2014) design value exceeding the revised and alternative ozone standard levels of 70 and 65 parts per billion (ppb). This analysis does not include a quantitative assessment of exposure and/or risk for 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.

The EPA Administrator is revising the NAAQS for ozone from the current level of 75 ppb to a level of 70 ppb. The revisions will establish uniform national standards for ozone in ambient air and 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 with poor ozone air quality, defined for this analysis as any Core Based Statistical Area (CBSA) with at least one county having an ozone monitor with a current (2012-2014) design value exceeding the revised and alternative ozone standard levels (70 and 65 ppb), including individual counties not included in a CBSA with a design value exceeding the revised and alternative standards. These areas are called "study areas" for the purposes of this analysis.

9A.1 Design of Analysis

To gain a better understanding of the populations within the study areas, the EPA conducted an analysis at the county level for this ozone NAAQS review. The study areas for these analyses were defined as all counties contained within any CBSA with at least one monitor with a current (2012-2014) design value above the revised and alternative standard levels (70 and 65 ppb) as well as counties not in a CBSA with a current (2012-2014) design value above the revised and alternative standard levels (70 and 65 ppb). The study areas were designed to capture population and communities with poor ozone air quality, and areas most likely to benefit from improved air quality following the implementation of the revised ozone NAAQS.

For the revised standard levels of 70 ppb, 511 counties were analyzed in 143 areas exceeding the standard levels, including 495 counties in 127 CBSAs and 16 counties outside CBSAs. For the alternative standard levels of 65 ppb, 891 counties were analyzed in 294 areas exceeding the standard level, including 847 counties in 250 CBSAs and 44 counties outside CBSAs. The population identified within these study areas made up about 52% of the U.S. population for the revised standard levels and about 69% of the U.S. population for the alternative standard levels.

Demographic data from the study areas identified for the revised and alternative standard levels were aggregated nationally for comparison with the U.S. population demographics. The demographic data used in this analysis include race, ethnicity, age, income, and education variables. Details on these demographic groups are provided in the following section (9A.1.1). The aggregated demographic values across the study areas are compared to the national data in Table 9A-2 of Section 9A-3.

This analysis identifies, on a limited basis, the populations that are most likely to experience reductions in ozone concentrations as a result of actions taken to meet the revised 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 populations are disproportionately impacted by ozone levels because they reside in a study area, that population will also experience increased environmental and health benefits from meeting the revised standard levels.

9A.1.1 Demographic Variables Included in Analysis

This analysis includes race, ethnicity, and age data derived from the 2010 Census SF1 dataset[198] and income and education data from the Census Bureau's 2006-2010 American Community Survey (ACS) 5-Year Estimates.[199] 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 American	Number of African Americans (may include Hispanics)
Native American	Number of Native Americas (may include Hispanics)
Other and multiracial	Number of other race and multiracial (may include Hispanics)
Minority	Total Population less White Population
Hispanic	Number of Hispanics
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
Education level	Number of adults age 25 years and up without a high school diploma
Low Income	Number of people living in households with income below twice the poverty line
Linguistic Isolation	Number of people linguistically isolated

*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

198 2010 Census Summary File 1 Delivered via FTP, http://www2.census.gov/census_2010/04-Summary_File_1/
199 U.S. Census Bureau 2006-2010 American Community Survey 5-Year Estimates, http://www.census.gov/acs/www/data_documentation/2010_release/

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 provides 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. 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.

9A.2 Considerations in Evaluating and Interpreting Results

This analysis characterizes the demographic attributes of populations located in areas defined by a county or a CBSA containing a county with a monitored 2012-2014 design value greater than the revised standard levels of 70 ppb, or the alternative standard levels of 65 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 revised and alternative ozone standard levels, and thus may be more affected by implementation of the revised standards. This analysis does not include a quantitative assessment of exposure 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 could result from implementation of the revised ozone standards.

The analysis simply represents a national depiction of the baseline characteristics of populations residing in areas with measured ozone air quality above the revised and alternative standard levels.

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). EPA does not have the ability to conduct such a rigorous technical analysis at this time.

9A.3 Presentation of Results

This section presents a summary of the demographics of populations in areas with 2012-2014 design values greater than the revised and alternative ozone standard levels. The results are provided in Table 9A-2. As a whole, the demographic distributions within the study areas estimated for the revised and alternative standard levels (i.e., 70 ppb and 65 ppb) are similar to the national averages. The largest difference is only 5%, between the national and study area percentages for the Minority demographic group. Overall, these qualitative results support the determination that the revised rule will tend to benefit geographic areas that have a higher proportion of minority residents than the national average.

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	Hispanic
Area Total - 70 ppb	163,900,459	109,711,305	23,448,540	1,196,130	29,544,484	54,189,154	33,651,929
% of Area Total - 70 ppb		67%	14%	1%	18%	33%	21%
Area Total - 65 ppb	216,932,408	150,701,600	29,091,602	1,699,450	35,439,756	66,230,808	39,746,762
% of Area Total - 65 ppb		69%	13%	1%	16%	31%	18%
National Total	312,861,256	226,405,205	39,475,216	2,952,087	44,028,748	86,456,051	54,181,245
% of National Total		72%	13%	1%	14%	28%	17%

Demographic Summary	Population	Age 0 to 4	Age 0 to 17	Age 65+	No High School Diploma	Low Income	Linguistically Isolated
Area Total - 70 ppb	163,900,459	11,028,428	40,546,150	19,510,553	15,960,333	49,392,881	11,150,433
% of Area Total - 70 ppb		7%	25%	12%	10%	30%	7%
Area Total - 65 ppb	216,932,408	14,387,223	52,913,376	26,596,432	20,469,632	65,336,396	13,050,626
% of Area Total - 65 ppb		7%	24%	12%	9%	30%	6%
National Total	312,861,256	20,465,065	75,217,176	40,830,262	30,952,789	101,429,436	19,196,507
% of National Total		7%	24%	13%	10%	32%	6%

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