Regulatory Impact Analysis

Final National Ambient Air Quality Standard for Ozone

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Summary of the Supplemental Regulatory Impact Analysis (RIA) for the Reconsideration of the 2008 Ozone National Ambient Air Quality Standard (NAAQS)

On September 16, 2009, EPA committed to reconsidering the ozone NAAQS standard promulgated in March 2008. Today's rule sets the ozone NAAQS at 0.070 ppm, based on this reconsideration of the evidence available at the time the last standard was set. Today's rule also includes a separate secondary NAAQS, for which this RIA provides only qualitative analysis due to the limited nature of available EPA guidance for attaining this standard

This supplement to the RIA contains an updated illustrative analysis of the potential costs and human health and welfare benefits of nationally attaining a new primary ozone standard of 0.070 ppm. The basis for this updated economic analysis is the RIA published in March 2008 with changes. These changes reflect some significant methodological improvements to air pollution benefits estimation, which EPA has adopted since the ozone standard was last promulgated. These significant changes include the following:

- We have adopted several key methodological updates to benefits assessment since the 2008 Ozone NAAQS RIA. These updates have already been incorporated into previous RIAs for the Portland cement NESHAP, NO_2 NAAQS RIA, and Category 3 Marine Diesel Engine Rule, and are therefore now incorporated in this analysis. Significant updates include:
    - We removed the assumption of no causality for ozone mortality, as recommended by the National Academy of Science (NAS).
    - We included two more ozone multi-city studies, per NAS recommendation.
    - We revised the Value of a Statistical Life (VSL) to be consistent with the value used in current EPA analyses.
    - We removed thresholds from the concentration-response functions for $PM_{2.5}$, consistent with EPA's Integrated Science Assessment for Particulate Matter.

The other elements of the illustrative analysis included in the March 2008 RIA were not changed for this supplemental analysis. The March 2008 RIA was based on the best available air quality modeling available and reflected emission reductions expected from federal rules promulgated and proposed at that time. Because of the fundamental similarities between the original and more recent air quality modeling simulations, EPA has elected not to update the original analysis of emissions reductions needed to attain the ozone NAAQS as described in

Chapter 4 of the 2008 RIA. See section S1.3 below for discussion of the air quality baseline used in this supplemental analysis.

Structure of this Updated RIA

As part of the ozone NAAQS reconsideration, this RIA supplement takes as its foundation the 2008 ozone NAAQS RIA. Detailed explanation of the majority of assumptions and methods are contained within that document and should be relied upon, except as noted in this summary.

This supplement itself consists of four parts:

- Section 1 provides an overview of the changes to the analysis and summary tables of the illustrative cost and benefits of obtaining a revised standard and alternatives of 0.065 ppm and 0.075 ppm.
- Section 2 contains a supplemental benefits analysis outlining the adopted changes in the methodology, updated results for the final NAAQS of 0.070 ppm and standard alternatives of 0.065 and 0.075 ppm using the revised methodology and assumptions.
- Section 3 contains supplemental evaluation of a separate secondary ozone NAAQS of 13 ppm-hr, as well as a less stringent alternative of 15 ppm-hr and a more stringent alternative of 11 ppm-hr. This supplemental includes an explanation of the complexities associated with quantifying the costs and benefits of a secondary standard at this time. In addition, we have incorporated an assessment of which counties would have an additional requirement to reduce ozone concentrations to meet a secondary standard beyond the reductions needed to meet the primary standard, the qualitative benefits of reducing ozone exposure on vegetation, and maps of biomass/yield loss avoided by attaining the primary and secondary ozone standards.

S1.1 Results of Benefit-Cost Analysis

This updated RIA consists of multiple analyses, including an assessment of the nature and sources of ambient ozone; estimates of current and future emissions of relevant ozone precursors; air quality analyses of baseline and alternative control strategies; illustrative control strategies to attain the standard alternatives in future years; estimates of the incremental costs and benefits of attaining the final standard and three alternative standards, together with an examination of key uncertainties and limitations; and a series of conclusions and insights gained from the analysis. It is important to recall that this RIA rests on the analysis done in 2008; no new air quality modeling or other assessments were completed except those outlined above.

The supplement includes a presentation of the benefits and costs of attaining various alternative ozone National Ambient Air Quality Standards in the year 2020. These estimates only include areas assumed to meet the current standard by 2020. They do not include the costs or benefits of attaining the alternate standards in the San Joaquin Valley and South Coast air basins in California, because we expect that nonattainment designations under the Clean Air Act for these areas would place them in categories afforded extra time beyond 2020 to attain the ozone NAAQS.

[Hold for reference to Addendum]

In Table S1.1 below, the individual row estimates reflect the different studies available to describe the relationship of ozone exposure to premature mortality. These monetized benefits include reduced health effects from reduced exposure to ozone, reduced health effects from reduced exposure to $PM_{2.5}$, and improvements in visibility. The ranges within each row reflect two PM mortality studies (i.e. Pope and Laden).

Ranges in the total costs column reflect different assumptions about the extrapolation of costs as discussed in Chapter 5 of the 2008 Ozone NAAQS RIA. The low end of the range of net benefits is constructed by subtracting the highest cost from the lowest benefit, while the high end of the range is constructed by subtracting the lowest cost from the highest benefit. The presentation of the net benefit estimates represents the widest possible range from this analysis.

Table S1.2 presents the estimate of total ozone and $PM_{2.5}$-related premature mortalities and morbidities avoided nationwide in 2020 as a result of this regulation.

Table S1. 1: Total Monetized Costs with Ozone Benefits and $PM_{2.5}$ Co-Benefits in 2020 (in Billions of 2006$) [A]

	Ozone Mortality Function	Reference	Total Benefits [B]		Total Costs [C]	Net Benefits	
			3%	7%	7%	3%	7%
0.075 ppm	Multi-city	Bell et al. 2004	$6.9 to $15	$6.4 to $13	$7.6 to $8.8	$-1.9 to $7.4	$-2.4 to $5.4
		Schwartz 2005	$7.2 to $16	$6.8 to $13	$7.6 to $8.8	$-1.6 to $8.4	$-2.1 to $5.4
		Huang 2005	$7.3 to $16	$6.9 to $13	$7.6 to $8.8	$-1.5 to $8.4	$-2.0 to $5.4
	Meta-analysis	Bell et al. 2005	$8.3 to $17	$7.9 to $14	$7.6 to $8.8	$-0.50 to $9.4	$-1.0 to $6.4
		Ito et al. 2005	$9.1 to $18	$8.7 to $15	$7.6 to $8.8	$0.30 to $10	$-0.20 to $7.4
		Levy et al. 2005	$9.2 to $18	$8.8 to $15	$7.6 to $8.8	$0.40 to $10	$-0.10 to $7.4
0.070 ppm	Multi-city	Bell et al. 2004	$13 to $29	$11 to $24	$19 to $25	$-12 to $10	$-14 to $5.0
		Schwartz 2005	$15 to $30	$12 to $25	$19 to $25	$-10 to $11	$-13 to $6.0
		Huang 2005	$15 to $30	$13 to $26	$19 to $25	$-10 to $11	$-12 to $7.0
	Meta-analysis	Bell et al. 2005	$18 to $34	$16 to $29	$19 to $25	$-7.0 to $15	$-9.0 to $10
		Ito et al. 2005	$21 to $37	$18 to $31	$19 to $25	$-4.0 to $18	$-6.0 to $12
		Levy et al. 2005	$21 to $37	$18 to $31	$19 to $25	$-4.0 to $18	$-6.0 to $12
0.065 ppm	Multi-city	Bell et al. 2004	$22 to $47	$19 to $40	$32 to $44	$-22 to $15	$-25 to $7.0
		Schwartz 2005	$24 to $49	$21 to $42	$32 to $44	$-20 to $17	$-23 to $9.0
		Huang 2005	$25 to $50	$22 to $42	$32 to $44	$-19 to $18	$-23 to $10
	Meta-analysis	Bell et al. 2005	$31 to $56	$27 to $48	$32 to $44	$-13 to $24	$-17 to $16
		Ito et al. 2005	$36 to $61	$32 to $53	$32 to $44	$-8.0 to $29	$-13 to $20
		Levy et al. 2005	$36 to $61	$32 to $53	$32 to $44	$-7.0 to $29	$-12 to $20

[A] All estimates rounded to two significant figures. As such, they may not sum across columns. Only includes areas required to meet the current standard by 2020; does not include San Joaquin and South Coast areas in California.

[B] Includes ozone benefits, and $PM_{2.5}$ co-benefits. Range was developed by adding the estimate from the ozone premature mortality function to estimates from the $PM_{2.5}$ premature mortality functions from Pope et al. and Laden et al. Tables exclude unquantified and nonmonetized benefits.

[C] Range reflects lower and upper bound cost estimates. Data for calculating costs at a 3% discount rate was not available for all sectors, and therefore total annualized costs at 3% are not presented here. Additionally, these estimates assume a particular trajectory of aggressive technological change. An alternative storyline might hypothesize a much less optimistic technological trajectory, with increased costs, or with decreased benefits in 2020 due to a later attainment date.

Table S1.2: Summary of Total Number of Ozone and $PM_{2.5}$-Related Premature Mortalities and Premature Morbidity Avoided: 2020 National Benefits [A]

Combined Estimate of Mortality		0.075 ppm	0.070 ppm	0.065 ppm
NMMAPS	Bell et al. (2004)	760 to 1,900	1,500 to 3,400	2,500 to 5,600
	Schwartz	800 to 1,900	1,600 to 3,600	2,700 to 5,800
	Huang	820 to 1,900	1,700 to 3,600	2,800 to 5,900
Meta-analysis	Bell et al. (2005)	930 to 2,000	2,000 to 4,000	3,500 to 6,600
	Ito et al.	1,000 to 2,100	2,400 to 4,300	4,000 to 7,200
	Levy et al.	1,000 to 2,100	2,400 to 4,300	4,100 to 7,200
Combined Estimate of Morbidity		**0.075 ppm**	**0.070 ppm**	**0.065 ppm**
Acute Myocardial Infarction [B]		1,300	2,200	3,500
Upper Respiratory Symptoms [B]		9,900	19,000	31,000
Lower Respiratory Symptoms [B]		13,000	25,000	41,000
Chronic Bronchitis [B]		470	880	1,400
Acute Bronchitis [B]		1,100	2,100	3,400
Asthma Exacerbation [B]		12,000	23,000	38,000
Work Loss Days [B]		88,000	170,000	270,000
School Loss Days [C]		190,000	600,000	1,100,000
Hospital and ER Visits		2,600	6,600	11,000
Minor Restricted Activity Days		1,000,000	2,600,000	4,500,000

[A] All estimates rounded to two significant figures. Only includes areas required to meet the current standard by 2020; does not include San Joaquin Valley and South Coast air basins in California. Includes ozone benefits, and $PM_{2.5}$ co-benefits. Mortality incidence range was developed by adding the estimate from the ozone premature mortality function to estimates from the $PM_{2.5}$ premature mortality functions from Pope et al. (2002) and Laden et al. (2006).
[B] Estimated reduction in premature morbidity due to $PM_{2.5}$ reductions only.
[C] Estimated reduction in premature morbidity due to ozone reductions only.

The following set of graphs is included to provide the reader with a richer presentation of the range of costs and benefits of the alternative standards. The graphs supplement the tables by displaying all possible combinations of net benefits, utilizing the six different ozone functions, the fourteen different PM functions, and the two cost methods. Each of the 168 bars in each graph represents a separate point estimate of net benefits under a certain combination of cost and benefit estimation methods. Because it is not a distribution, it is not possible to infer the likelihood of any single net benefit estimate. The blue bars indicate combinations where the net benefits are negative, whereas the green bars indicate combinations where net benefits are positive. Figures S1.1 through S1.3 shows all of these combinations for all standards analyzed. Figure S1.4 shows the comparison of total monetized benefits with costs using the two benefits anchor points based on Pope/Bell 2004 and Laden/Levy.

Figure S1.1:

Net Benefits for an Alternate Standard of 0.075 ppm (7% discount rate)

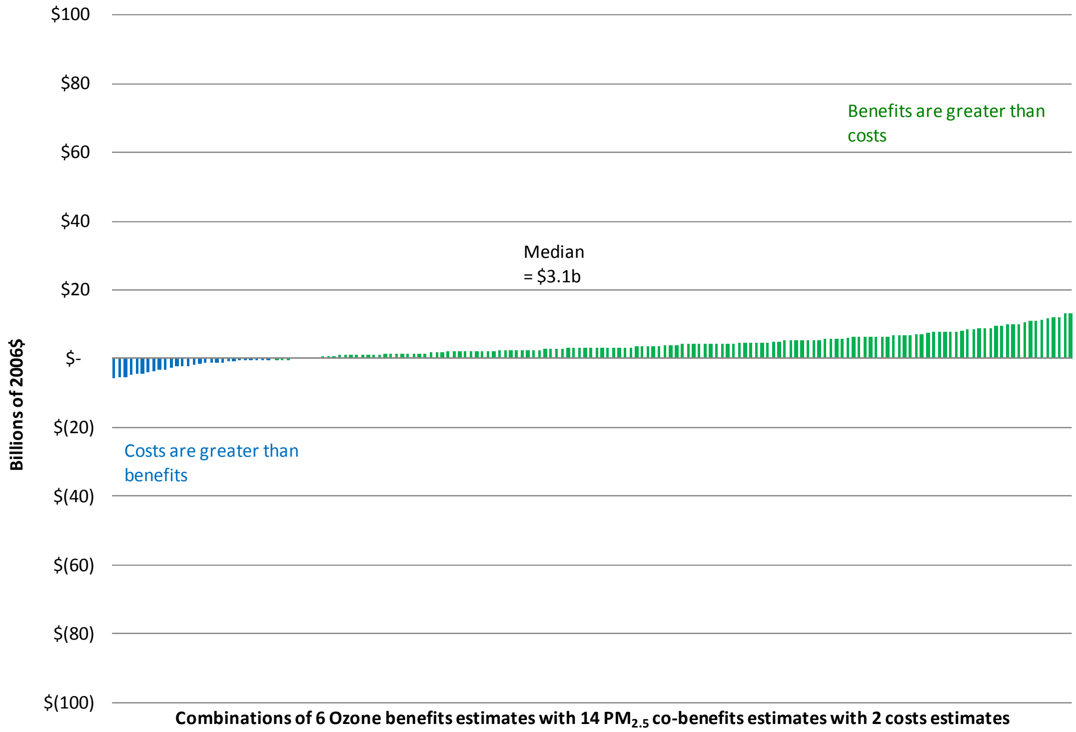
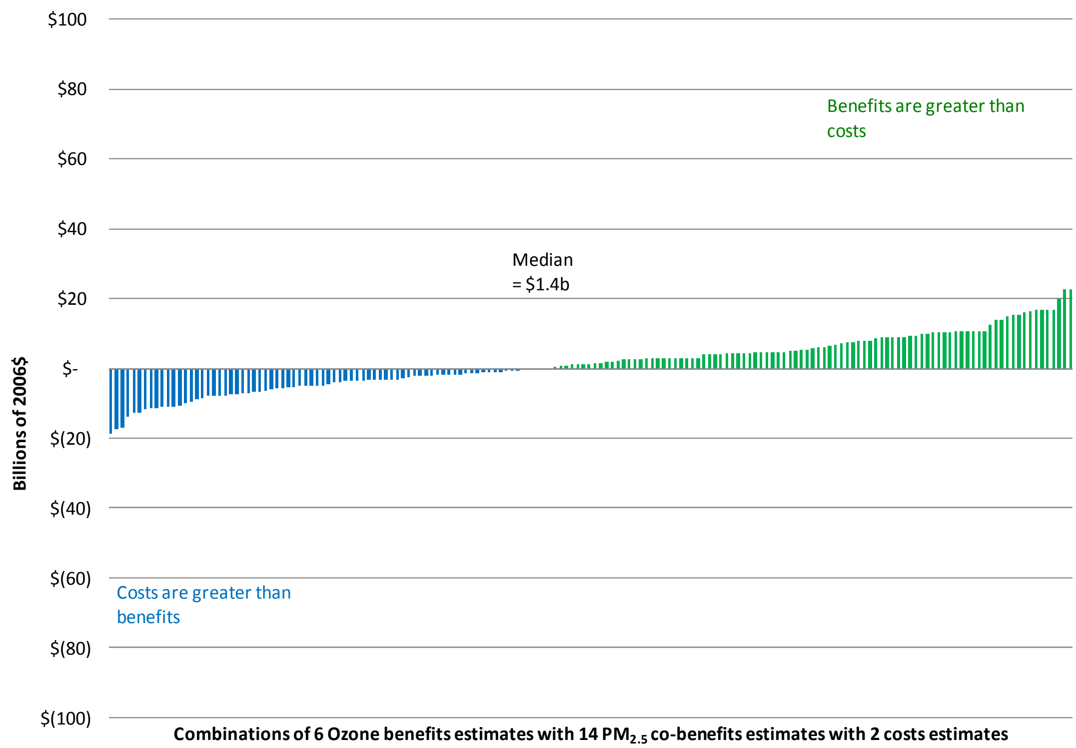


Figure S1.2:

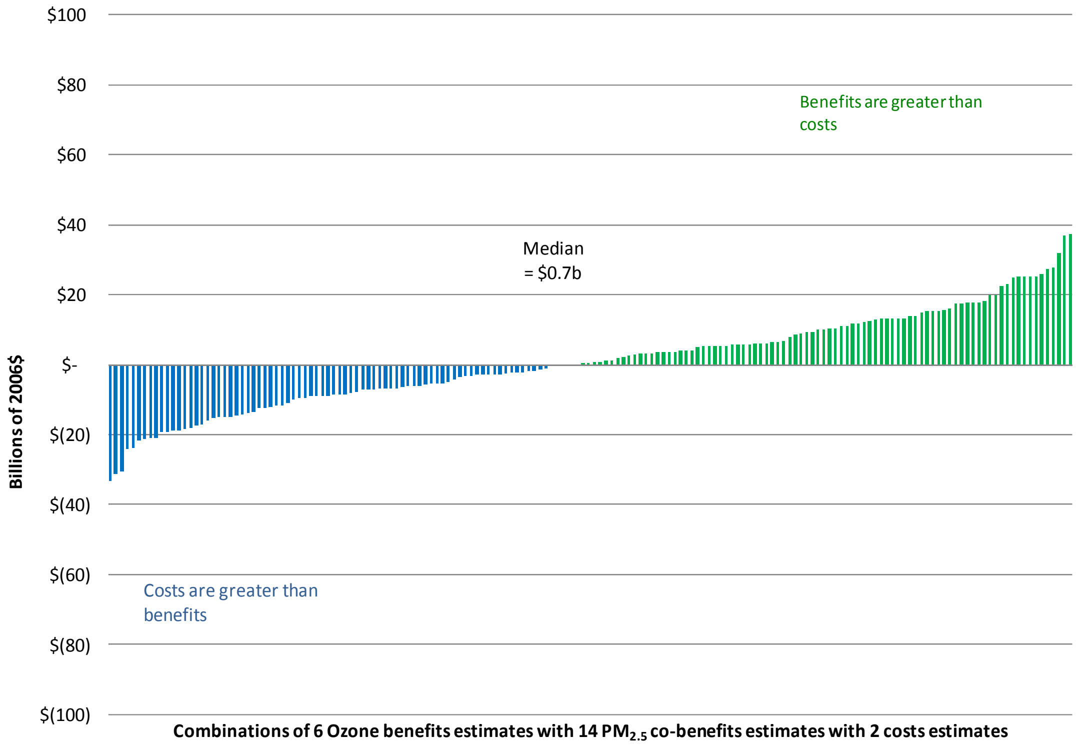
Net Benefits for an Alternate Standard of 0.070 ppm (7% discount rate)



These graphs show all 168 combinations of the 6 different ozone mortality functions and assumptions, the 14 different PM mortality functions, and the 2 cost methods. These combinations do not represent a distribution.

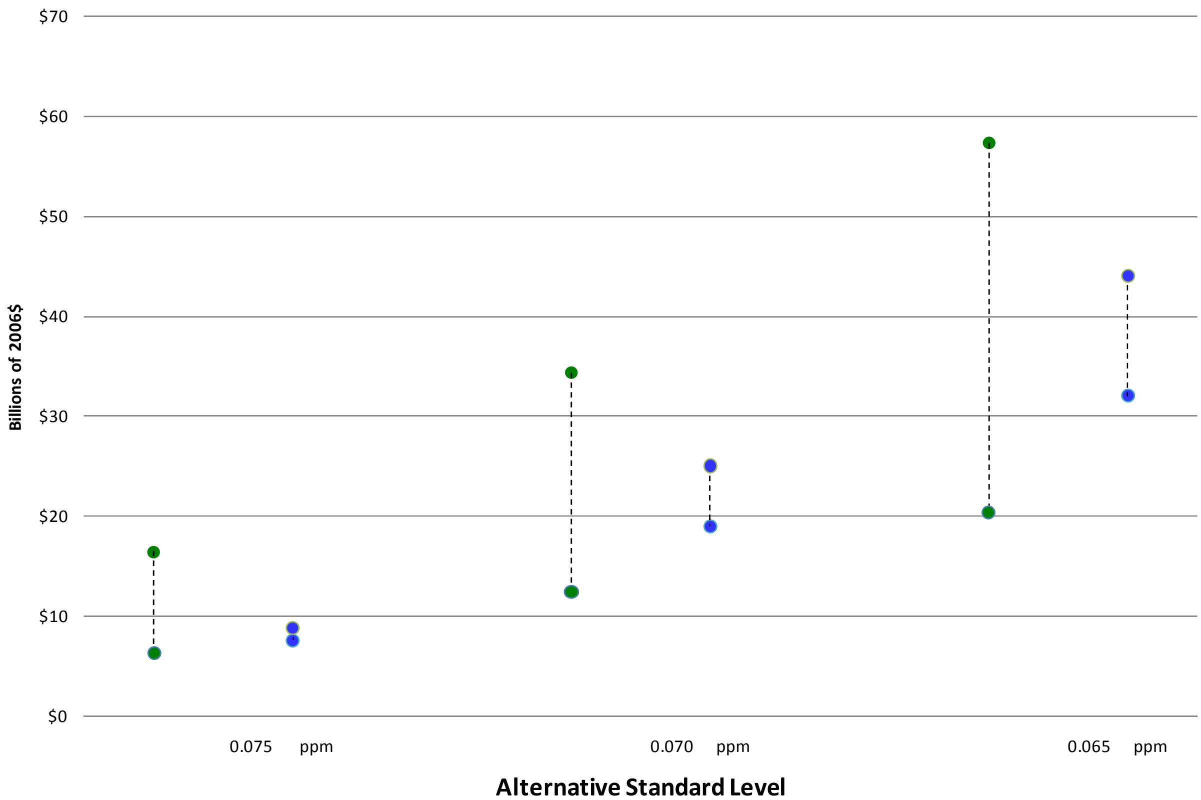
Figure S1.3:

Net Benefits for an Alternate Standard of 0.065 ppm (7% discount rate)



These graphs show all 168 combinations of the 6 different ozone mortality functions and assumptions, the 14 different PM mortality functions, and the 2 cost methods. These combinations do not represent a distribution.

Figure S1.4:
Comparison of Total Monetized Benefits to Costs for Alternative Standard Levels in 2020 (Updated results, 7% discount rate)



The low benefits estimate is based on Pope/Bell 2004 and the high benefits estimate is based on Laden/Levy. The two cost estimates are based on two different extrapolated cost methodologies. These endpoints represent separate estimates based on separate methodologies. The dotted lines are a visual cue only, and these lines do not imply a uniform range between these endpoints.

S1.2 Analysis of the Proposed Secondary NAAQS for Ozone

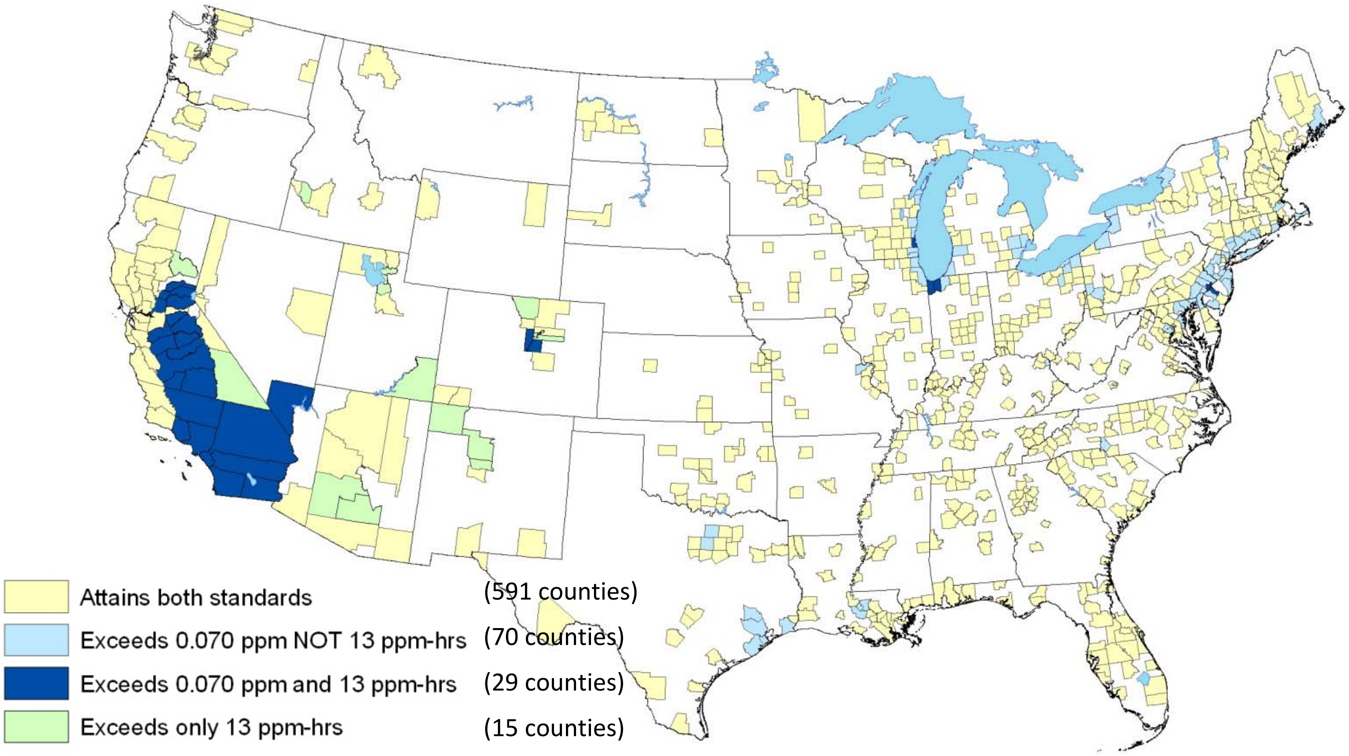
Exposure to ozone has been associated with a wide array of vegetation and ecosystem effects in the published literature. Sensitivity to ozone is highly variable across species, with over 65 plan species identified as "ozone-sensitive", many of which occur in state and national parks and forests. These effects include those that damage or impair the intended use of the plant or ecosystem. Such effects are considered adverse to the public welfare and can include reduced growth and/or biomass production in sensitive plant species, including forest trees, reduced crop yields, visible foliar injury, reduced plant vigor (e.g., increased susceptibility to harsh weather, disease, insect pest infestation, and competition), species composition shift, and changes in ecosystems and associated ecosystem services.

This secondary NAAQS standard for ozone is the first secondary standard to be promulgated with a form, averaging time, and level that is distinct from the health-based primary standard apart from the PM and SO_2 regulations originally set in the early 1970s. The index would be cumulated over the 12-hour daylight window (8:00 a.m. to 8:00 p.m.) during the consecutive 3-month period during the ozone season with the maximum index value (hereafter, referred to as W126). After reviewing the scientific evidence and public comments, the Administrator selected a secondary ozone NAAQS at a level of 13 ppm-hrs, using the W126 form, calculated as a 3-year average of the annual sums.

Quantifying the costs and benefits of attaining a secondary NAAQS is an exceptionally complex task, including unresolved issues related to the RIA analysis, air quality projection, monitoring expansion, and implementation.[1] Because of these complexities as well as limited time and resources within the expedited schedule, we are limited in our ability to quantify the costs and benefits of attaining a separate secondary NAAQS for ozone for this proposal. However, we have incorporated an assessment of which counties would have an additional requirement to reduce ozone concentrations to meet a secondary standard beyond the reductions needed to meet the primary standard, the qualitative benefits of reducing ozone exposure on vegetation, and maps of biomass/yield loss avoided by attaining the primary and secondary ozone standards. Using a cumulative seasonal secondary standard (i.e., W126), we evaluated alternate standard levels at 11, 13, and 15 ppm-hours. Figure S1.5 shows the counties projected to exceed a primary standard at 0.070 ppm and/or a secondary standard at 13 ppm-hrs in the 2020 baseline.

[1] These complexities are described in detail in Section S3.3.

Figure S1.5: Counties Projected to Exceed the Selected Primary and Secondary Standards in the Baseline in 2020*



* Many of the counties projected to exceed are in the South Coast and San Joaquin areas of California, which are not required to attain the primary standards by 2020.

S1.3 Baseline Emissions Inventory

EPA expects that the emissions reductions needed to attain the new ozone primary standard may be less than what EPA originally predicted in the March 2008 RIA. Recent updates to the emission and air quality modeling platform suggest that future baseline air quality will be better than what was projected in the 2008 RIA. If the more recent projections are better estimates of future ozone nonattainment in these areas, then the costs and benefits of attaining the ozone NAAQS incremental to the current standard will likely be less than what was projected as part of the 2008 RIA. However, there have also been a few rules promulgated since the 2008 RIA baseline was developed that significantly affect ozone precursor emissions. It is difficult to assess retroactively the net emissions impacts of these rules and how they would likely affect total costs and benefits of the ozone NAAQS if they had been included in the baseline. We discuss each of these baseline issues below.

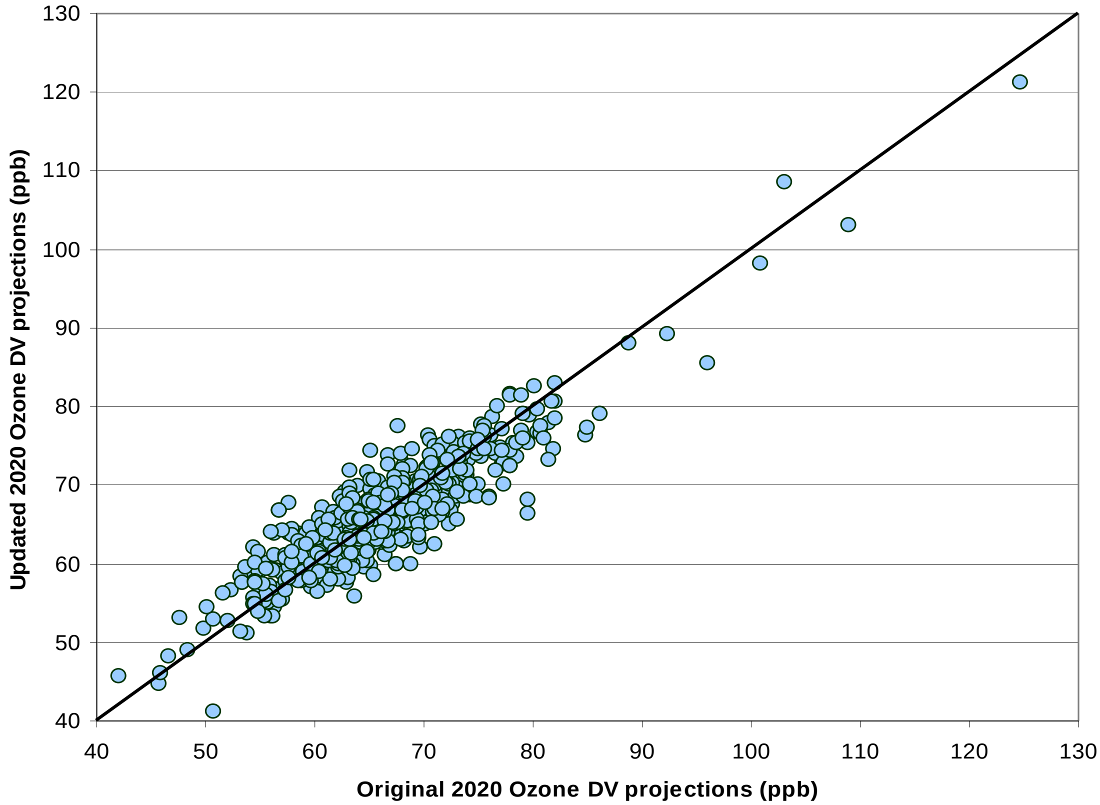
Modeling Platform

In March 2008, EPA completed a regulatory impacts analysis (RIA) that estimated the potential costs and benefits of attaining a 0.075 ppm standard as well as several alternatives. This illustrative analysis was based on the best available air quality modeling available at the time of the original analysis. As described in Chapter 2 of the 2008 RIA, EPA used the Community Multiscale Air Quality (CMAQ) model with inputs from a 2002 base year to project: a) ozone concentrations in the future (i.e., 2020) and b) the amount of emissions reductions that would be necessary to meet specified ozone targets. Since the original analysis, the CMAQ model has been updated with several new science algorithms (Foley et al., 2010) and the base year platform has been updated to include 2005 ambient data and model inputs. As part of this NAAQS reconsideration, EPA completed a quick analysis to determine if the updates to the air quality modeling system would substantially affect the original 2008 estimates of the control costs needed to attain the new ozone standard.

One of the key elements in determining the amount of controls needed to attain a particular ozone target is the estimate of how many areas will be above the chosen threshold in the future and by how much they will exceed the goal. Greater amounts of residual nonattainment will lead to greater amounts of needed emissions reductions which will lead to higher attainment costs and benefits attributable to the new standard.

EPA compared model projections of 2020 eight-hour ozone design values from the original RIA (based on the 2002 platform) against the same 2020 model projections from the latest air quality modeling simulations that use the current version of CMAQ and the more recent base year (2005). In general, the 2020 design value estimates were very similar between the two modeling exercises, as shown in Figure S1.6. For the 635 counties with eligible 2020 projections in both cases, the average difference between the most recent and the original analysis was -0.15 ppb. That is, the updated analysis estimated slightly cleaner ozone values in 2020 than what EPA previously estimated in the original RIA. However, the two sets of projections are very similar.

Figure S1.6: Comparison of projected 2020 eight-hour ozone design values over 635 counties in the U.S. from the original 2008 RIA air quality modeling (x-axis) and a more recent modeling analysis based on an updated model with updated model inputs (y-axis)



The majority of the cost in attaining a new ozone standard will come from meeting the target in the areas projected to be most polluted in the future. Limiting the analysis to only those 61 counties where the ozone design values are projected to exceed 0.070 ppm in 2020 after the implementation of the controls in the hypothetical RIA control scenario, we see that there is an even stronger tendency for the updated modeling to project cleaner conditions in the future. As discussed in the original RIA, these 61 counties are primarily located in four areas: California, Houston, the western Lake Michigan region, and the Northeast Corridor. For this subset of locations, the average 2020 projected design value difference was -3.3 ppb. In other words, the more recent EPA modeling predicts slightly cleaner ozone conditions (78.1 vs. 81.4 ppb) at the most polluted locations in the future. If the more recent projections are better estimates of future ozone nonattainment in these areas, then the costs and benefits of attaining the ozone NAAQS incremental to the current standard will likely be less than what was projected as part of the 2008 RIA.

Because of the fundamental similarities between the original and more recent air quality modeling simulations, EPA has elected not to update the original analysis of emissions reductions needed to attain the ozone NAAQS as described in Chapter 4 of the 2008 RIA. Based on the latest air quality modeling information, however, it is expected that the original RIA estimates of needed emissions reductions are greater than what is necessary to attain the new primary standard.

Federal Rulemakings Included in the Baseline

The starting point for this analysis is the "baseline", which represents what ambient air quality would be nationwide in 2020 absent the revised ozone NAAQS. (2020 is when the ozone NAAQS would be expected to be fully implemented in all areas except those with the most significant air quality problems. Our analysis recognizes that two areas in Southern California are not planning to meet the current standard by 2020.) The baseline for the revised ozone standard is calculated using emissions estimates that include emission controls that will be needed to attain the "current" standard by 2020. Since this rulemaking is a reconsideration of the 0.080 ppm NAAQS, for this analysis the "current" standard is considered to be 0.08 ppm (effectively 0.084 with rounding).

Two steps were used to develop the baseline for the March 2008 RIA. First, the reductions expected nationwide in ozone concentrations from Federal rules in effect or proposed at that time were included, as well as the controls applied as part of the PM2.5 NAAQS RIA analysis. The rules reflected in the modeling include:

- Clean Air Interstate Rule (EPA, 2005b)
- Clean Air Mercury Rule (EPA, 2005c)
- Regional Haze Regulations and Guidelines for Best Available Retrofit Technology Determinations (EPA, 2005d)
- Clean Air Nonroad Diesel Rule (EPA, 2004)
- Light-Duty Vehicle Tier 2 Rule (EPA, 1999)
- Heavy Duty Diesel Rule (EPA, 2000)
- Proposed rules for Locomotive and Marine Vessels (EPA, 2007a) and for Small Spark-Ignition Engines (EPA, 2007b)
- Proposed C3 Emission Control Area Rule (2009)
- State and local level mobile and stationary source controls identified for additional reductions in emissions for the purpose of attaining the current PM 2.5 and Ozone standards.

Second, since these reductions alone were not predicted to bring all areas into attainment with the current standard, we used a hypothetical control strategy to apply additional known controls. Additional control measures were used in four sectors to establish the baseline: Non-Electricity Generating Unit Point Sources (NonEGUs), Non-Point Area Sources (Area), Onroad Mobile Sources and Nonroad Mobile Sources.

Since the 2008 RIA was completed, a few other Federal rules significantly affecting ozone precursor emissions have been promulgated. Also since that time, the Clean Air Interstate Rule (CAIR) was remanded to EPA by the U.S. Court of Appeals for the D.C. Circuit and the Clean Air Mercury Rule (CAMR) was vacated by the Court. These new developments suggest that the baseline for this supplementary analysis does not reflect emission impacts expected from some recent rules, and that it does reflect emission impacts from some rules that are no longer in place.

Three major rules that were promulgated in 2010, and which affect large categories of NOx emissions, should be represented in the baseline but are not. These are the Renewable Fuel Standard (RFS2) and the Reciprocating Internal Combustion Engines (RICE) NESHAPs (2004 and 2010). It is difficult to assess retroactively how these rules would likely affect total costs of the ozone NAAQS if they had been included in the baseline. NOx emissions from the two RICE rules are estimated to have decreased by a total of about 165,000 tons per year in 2020. However, NOx emissions are expected to increase by 247,600 tons in 2020 as a result of RFS2.

It is difficult to quantify the emission implications of having CAIR in the baseline for the ozone analysis. In 2008, the U.S. Court of Appeals for the D.C. Circuit remanded CAIR to EPA. (See http://www.epa.gov/CAIR/ for more background on CAIR and the Court ruling.) On July 6, 2010, EPA proposed the Transport Rule as a replacement for CAIR. For NOx, the Transport Rule budget is lower in the near term and higher after 2015 relative to CAIR adjusting for differences in the spatial coverage of the two rules. On net, annual NOx emissions are higher than under CAIR once the replacement rule is in effect. Seasonal NOx emissions are lower with the replacement rule, but this is because of differences in baseline emissions and is not attributable to the replacement rule as emissions in the base case are lower than what is forecast with CAIR compliance. Table S1.3 below summarizes the modeled emissions under CAIR and the Transport Rule in various years.

Table S1.3. IPM Estimated Emissions Under CAIR and CAIR Replacement Rule (Transport Rule)

National NOx Annual Emissions (Million Tons)					
	2010	**2012**	**2015**	**2020**	**2025/2026**
CAIR baseline	3.6	NA	3.7	3.7	NA
CAIR	2.4	NA	2.1	2.1	NA

Transport Rule baseline	NA	3.0	3.0	3.1	3.1
Transport Rule main remedy	NA	2.2	2.2	2.3	2.3
CAIR-Region NOx Seasonal Emissions (Million Tons)					
	2010	**2012**	**2015**	**2020**	**2025/2026**
CAIR baseline	0.80	NA	0.80	0.80	NA
CAIR	0.70	NA	0.60	0.60	NA
Transport Rule baseline	NA	0.40	0.40	0.41	0.42
Transport Rule main remedy	NA	0.39	0.38	0.39	0.40

Source: CAIR results are taken from "EPA Base Case 2004" and "IPM Run CAIR 2004 Final" modeling output, available at: http://www.epa.gov/airmarkt/progsregs/epa-ipm/cair/index.html. Transport Rule results are taken from "TR Base Case" and "TR SB Limited Trading" modeling output, available at: http://www.epa.gov/airmarkets/progsregs/epa-ipm/transport.html

At this time we are unable to assess the relative emission reductions expected from the CAMR replacement rule relative to CAMR. In 2008, the U.S. Court of Appeals for the D.C. Circuit vacated the Clean Air Mercury Rule (CAMR). (See http://www.epa.gov/mercuryrule/ for more background CAMR and the Court ruling.) EPA intends to propose air toxics standards for power plants consistent with the D.C. Circuit's opinion regarding the CAMR by March 10, 2011 and finalize a rule by November 16, 2011.

S1.4 Caveats and Conclusions

Of critical importance to understanding these estimates of future costs and benefits is that they are not intended to be forecasts of the actual costs and benefits of implementing revised standards. There are many challenges in estimating the costs and benefits of attaining a tighter ozone standard, which are fully discussed in 2008 Ozone NAAQS RIA and the supplement to this analysis accompanying today's final rule.

There are significant uncertainties in both cost and benefit estimates for the full range of standard alternatives. Below we summarize some of the more significant sources of uncertainty common to all level analyzed in the 2008 ozone NAAQS RIA and this supplemental analysis:

- Benefits estimates are influenced by our ability to accurately model relationships between ozone and PM and their associated health effects (e.g., premature mortality).
- Benefits estimates are also heavily dependent upon the choice of the statistical model chosen for each health benefit.

- PM co-benefits are derived primarily from reductions in nitrates (associated with NOx controls). As such, these estimates are strongly influenced by the assumption that all PM components are equally toxic. Co-benefit estimates are also influenced by the extent to which a particular area chooses to use NOx controls rather than VOC controls.

- There are several nonquantified benefits (e.g., effects of reduced ozone on forest health and agricultural crop production) and disbenefits (e.g., decreases in tropospheric ozone lead to reduced screening of UV-B rays and reduced nitrogen fertilization of forests and cropland) discussed in this analysis in Chapter 6 of the 2008 Ozone NAAQS RIA.

- Changes in air quality as a result of controls are not expected to be uniform over the country. In our hypothetical control scenario some increases in ozone levels occur in areas already in attainment, though not enough to push the areas into nonattainment.

- As explained in Chapter 5 of the 2008 Ozone NAAQS RIA, there are several uncertainties in our cost estimates. For example, the states are likely to use different approaches for reducing NOx and VOCs in their state implementation plans to reach a tighter standard. In addition, since our modeling of known controls does not get all areas into attainment, we needed to make assumptions about the costs of control technologies that might be developed in the future and used to meet the tighter alternative. For example, for the 21 counties (in four geographic areas) that are not expected to attain 0.075 ppm[2] in 2020[3], assumed costs of unspecified controls represent a substantial fraction, of the costs estimated in this analysis ranging from 50% to 89% of total costs depending on the standard being analyzed.

    - As discussed in Chapter 5 of the 2008 Ozone NAAQS RIA, advice from EPA's Science Advisory Board has questioned the appropriateness of an approach similar to one of those used here for estimating extrapolated costs. For balance, EPA also applied a methodology recommended by the Science Advisory Board in an effort to best approximate the costs of control technologies that might be developed in the future.

[2] Areas that do not meet 0.075 ppm are Chicago, Houston, the Northeastern Corridor, and Sacramento. For more information see chapter 4 section 4.1.1 of the 2008 Ozone NAAQS RIA.
[3] This list of areas does not include the San Joaquin and South Coast air basins who are not expected to attain the current 0.084 ppm standard until 2024.

- Both extrapolated costs and benefits have additional uncertainty relative to modeled costs and benefits. The extrapolated costs and benefits will only be realized to the extent that unknown extrapolated controls are economically feasible and are implemented. Technological advances over time will tend to increase the economic feasibility of reducing emissions, and will tend to reduce the costs of reducing emissions. Our estimates of costs of attainment in 2020 assume a particular trajectory of aggressive technological change. This trajectory leads to a particular level of emissions reductions and costs which we have estimated based on two different approaches, the fixed cost and hybrid approaches. An alternative storyline might hypothesize a much less optimistic technological change path, such that emissions reductions technologies for industrial sources would be more expensive or would be unavailable, so that emissions reductions from many smaller sources might be required for 2020 attainment, at a potentially greater cost per ton. Under this alternative storyline, two outcomes are hypothetically possible: Under one scenario, total costs associated with full attainment might be substantially higher. Under the second scenario, states may choose to take advantage of flexibility in the Clean Air Act to adopt plan with later attainment dates to allow for additional technologies to be developed and for existing programs like EPA's Onroad Diesel, Nonroad Diesel, and Locomotive and Marine rules to be fully implemented. If states were to submit plans with attainment dates beyond our 2020 analysis year, benefits would clearly be lower than we have estimated under our analytical storyline. However, in this case, state decision makers seeking to maximize economic efficiency would not impose costs, including potential opportunity costs of not meeting their attainment date, when they exceed the expected health benefits that states would realize from meeting their modeled 2020 attainment date. In this case, upper bound costs are difficult to estimate because we do not have an estimate of the point where marginal costs are equal to marginal benefits plus the costs of nonattainment. Clearly, the second stage analysis is a highly speculative exercise, because it is based on estimating emission reductions and air quality improvements without any information about the specific controls that would be available to do so.

Appendix S1.A: Reductions of Criteria Air Pollutants from Travel Efficiency Strategies

The RIA contains only a minimal analysis of travel efficiency strategies to reduce vehicle miles traveled, and thus reduce emissions of NOx and other pollutants. A recent report titled, *Moving Cooler: An Analysis of Transportation Strategies for Reducing Greenhouse Gas Emissions*[4]*,* which EPA and US DOT helped to fund, analyzed the potential levels of emissions reductions from light-duty travel efficiency. Moving Cooler included six different bundles of strategies to reflect different potential groups of strategies that could be implemented. Using data from this report, EPA conducted an analysis of the air quality benefits of a subset of the travel efficiency strategies evaluated in the report. Below are preliminary results based on EPA's draft MOVES2009 Model.

For the purposes of EPA's analysis, we chose the "Low Cost" bundle because we believed that it represented the best combination of strategies based on cost, likelihood of success, and accuracy of the research results. This bundle included strategies like smart growth/transit, commuter strategies, system operations (e.g., eco-driving, ramp metering), pricing (e.g., parking taxes, congestion pricing, intercity tolls), speed limit restrictions, and multimodal freight strategies. Note that this bundle did not include a VMT tax or cap-and-trade assumptions.

Moving Cooler made assumptions about the geographic scope for which each strategy could be implemented, with certain strategies like transit being dependent on greater populations, while other strategies like speed limit restrictions could be implemented in both urban and rural areas. Adjustments were also made to operational and commuter strategies to account for induced demand impacts. Scenarios A and B represent aggressive and maximum deployment, respectively, of the "Low Cost" bundle of strategies in Moving Cooler.

Summary of Results

- Nationally, the modeled travel efficiency strategies would reduce exhaust $PM_{2.5}$, NOx, HC and CO from cars and light trucks by approximately 2% in 2020, to approximately 7% in 2045, under the "aggressive" Moving Cooler assumptions.
- The modeled travel efficiency strategies would reduce these emissions by approximately 5% in 2020, to approximately 11% in 2045, under the "maximum" Moving Cooler assumptions.

[4] Cambridge Systematics, Inc. (2009). Moving Cooler: An Analysis of Transportation Strategies for Reducing Greenhouse Gas Emissions. Urban Land Institute: Washington, D.C.

- Percent reductions would be larger in urban areas, where Moving Cooler VMT reductions are concentrated.

Detailed Results

U.S. Annual Ton Reductions from Moving Cooler Bundle 6

	"Maximum" Reductions							
	HC		NOx		PM2.5		CO	
	Tons	% LD	Tons	% LD	Tons	% LD	Tons	% LD
2010	3,437	0.2%	6,542	0.2%	116	0.2%	63,414	0.2%
2015	21,902	1.5%	44,447	1.4%	909	1.5%	486,532	1.4%
2020	50,756	5.2%	101,773	4.8%	2,881	5.2%	1,379,197	4.9%
2025	55,130	7.4%	109,888	6.9%	4,157	7.6%	1,848,978	7.1%
2030	55,569	8.5%	109,193	7.9%	5,039	9.0%	2,157,685	8.3%
2035	56,701	9.3%	111,183	8.7%	5,794	10.0%	2,401,437	9.2%
2040	62,517	10.1%	125,017	9.5%	6,659	10.9%	2,723,906	10.1%
2045	69,934	11.0%	142,747	10.3%	7,631	11.9%	3,088,375	11.0%

	"Aggressive" Reductions							
	HC		NOx		PM2.5		CO	
	Tons	% LD	Tons	% LD	Tons	% LD	Tons	% LD
2010	1,222	0.1%	2,325	0.1%	41	0.1%	22,536	0.1%
2015	6,345	0.4%	13,079	0.4%	265	0.4%	143,319	0.4%
2020	22,088	2.3%	44,291	2.1%	1,254	2.3%	600,223	2.1%
2025	30,592	4.1%	61,366	3.8%	2,301	4.2%	1,029,945	3.9%
2030	33,256	5.1%	65,633	4.8%	3,005	5.4%	1,292,924	5.0%
2035	35,727	5.8%	70,298	5.5%	3,638	6.3%	1,513,454	5.8%
2040	39,897	6.5%	80,020	6.1%	4,236	6.9%	1,738,288	6.5%
2045	45,181	7.1%	92,460	6.7%	4,916	7.6%	1,995,082	7.1%

1 SECTION 2: RE-ANALYSIS OF THE BENEFITS OF ATTAINING ALTERNATIVE OZONE STANDARDS TO INCORPORATE CURRENT METHODS

Synopsis

This chapter presents a benefits analysis of three alternate ozone standards updated to reflect key methodological changes that EPA implemented after publishing the 2008 Ozone NAAQS RIA. Since the completion of this analysis EPA has introduced several methodological improvements in other RIA's that are not incorporated in this analysis.[5] In this updated analysis we re-estimate the human health benefits of reduced exposure to ambient ozone and $PM_{2.5}$ co-benefits from simulated attainment with the selected daily 8hr maximum standard of 0.070 ppm and two alternate standards of 0.075 ppm and 0.065 ppm. For the selected standard of 0.070 ppm, EPA estimates the monetized benefits to be $13 to $37 billion (2006$, 3% discount rate) in 2020. For an alternative standard at 0.075 ppm, EPA estimates the monetized benefits to be $6.9 to $18 billion (2006$, 3% discount rate) in 2020.[6] For the alternative standard at 0.065 ppm, EPA estimates the monetized benefits to be $22 to $61 billion (2006$, 3% discount rate) in 2020. Higher or lower estimates of benefits are possible using other assumptions. These updated estimates reflect three key methodological changes we have implemented since the publication of the 2008 RIA that reflect EPA's most current interpretation of the scientific literature and include: (1) a no-threshold model for $PM_{2.5}$ that calculates incremental benefits down to the lowest modeled air quality levels; (2) removal of the assumption of no causality for the relationship between ozone exposure and premature mortality; (3) a different Value of Statistical Life (VSL). These benefits are incremental to an air quality baseline that reflects attainment with the 1997 ozone and 2006 $PM_{2.5}$ National Ambient Air Quality Standards (NAAQS). Methodological limitations prevented EPA from monetizing the benefits from several important benefit categories, including ecosystem effects.

S2.1 Background

In response to the recent court vacatur of the 2008 Ozone NAAQS, EPA is reconsidering this rulemaking. Consistent with EPA's decision to, in general, use the "existing record" for this reconsideration, we present a benefits analysis based on the same air quality modeling inputs as the 2008 analysis. However, we update this analysis to make the results consistent with an array of methodological updates that EPA has incorporated since the release of Regulatory Impact Analysis (RIA) for the 2008 Ozone NAAQS (U.S. EPA, 2008). Because the rulemaking period for the reconsideration is condensed, we only provide estimates associated with the promulgated standard level of 0.070 ppm and the two less stringent standard levels previously analysis (i.e.,

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[5] Such improvements include the use of more current baseline mortality and morbidity rates to calculate health impacts and the use of more recent PM health studies to calculate health impacts. The effect of these changes would be to reduce certain ozone and PM2.5-related health impacts reported in this RIA by a modest amount.

[6] Results are shown as a range from Bell et al. (2004) with Pope et al. (2002) to Levy (2005) with Laden et al. (2006). $PM_{2.5}$ co-benefits using a 7% discount rate would be approximately 9% lower.

0.065 ppm and 0.075 ppm). All benefits estimates in this analysis are incremental to the 1997 Ozone NAAQS standard at 0.08 ppm and the 2006 $PM_{2.5}$ NAAQS standard at 15/35 µg/m^3.

S2.2 Key updates to the benefits assessment

In this analysis, we update several aspects of our benefits assessment for the human health benefits of reducing exposure to ozone and $PM_{2.5}$.[7] Both ozone benefits and $PM_{2.5}$ co-benefits incorporate the updated population projections in BenMAP. In addition, both ozone benefits and $PM_{2.5}$ co-benefits reflect EPA's current interpretation of the economic literature on mortality valuation to use the value-of-a statistical life (VSL) based on meta-analysis of 26 studies.[8]

For ozone benefits, these updates are a response to recent recommendations from the National Research Council (NRC, 2008). In this analysis, we have incorporated three of NRC's recommendations:

1) We no longer include estimates of ozone benefits with an assumption of no causal relationship between ozone exposure and premature mortality.
2) We include two additional ozone mortality estimates, one based on the National Morbidity, Mortality and Air Pollution Study (NMMAPS) (Huang, 2005), and one 14-city study (Schwartz, 2005), placing the greatest emphasis on the multi-city studies, such as NMMAPS.
3) We present additional risk metrics, including the change in the percentage of baseline mortality attributable, and the number of life years lost due, to ozone-related premature mortality.

In addition to these recommendations, we modify the health functions used to estimate the number of emergency department visits for asthma avoided by reducing exposure to ozone. Specifically, we removed the Jaffe et al. (2003) function because the age range overlaps partially with Wilson et al. (2005) and Peel et al. (2005) functions. This change results in a slightly larger estimate of ozone-related emergency department visits as compared to the 2008 analysis.

For $PM_{2.5}$ co-benefits, this analysis is consistent with proposed Portland Cement NESHAP RIA (U.S. EPA, 2009a) and proposed NO_2 NAAQS RIA (U.S. EPA, 2009b). In this analysis, we incorporate four updates:

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[7] This analysis does not attempt to describe the overall methodology for estimating the benefits of reducing ozone and $PM_{2.5}$. For more information, please consult Chapter 6 of the 2008 Ozone NAAQS RIA (U.S. EPA, 2008).

[8] For more information regarding mortality valuation, please consult section 5.7 of the proposed NO_2 RIA (U.S. EPA, 2009b).

1) We removed assumed thresholds from the mortality and morbidity concentration-response functions for $PM_{2.5}$.[9] Removing the assumed 10 μg/m^3 threshold is a key difference between the method used in this analysis of $PM_{2.5}$-co benefits and the methods used in RIAs prior to Portland Cement, and we now calculate incremental benefits down to the lowest modeled $PM_{2.5}$ air quality levels. This change results in a larger estimate of PM-related premature mortality as compared to the 2008 analysis.
2) We now present the $PM_{2.5}$ co-benefits results using concentration-response functions for mortality from two cohort studies (Pope et al. (2002) and Laden et al. (2006)) instead of range between the minimum and maximum results from an expert elicitation of the relationship between exposure to $PM_{2.5}$ and premature mortality (Roman et al., 2008). This change produces a slightly narrower range of PM-related mortality estimates as compared to the 2008 analysis.
3) When adjusting the benefits of the modeled PM co-benefits for alternate standard levels, we apply $PM_{2.5}$ benefit per ton estimates calculated using a broader geographic area, which, when compared to the 2008 analysis, produces more reliable and generally larger PM-related benefits estimates.
4) We incorporated an updated methodology for quantifying the health incidences associated with the benefit-per-ton estimates. This change should produce more reliable estimates of PM-related health impacts.

In this analysis we estimate ozone-related premature mortality using risk coefficients drawn from short-term mortality studies. Two recent epidemiologic studies assessed the relationship between long-term exposure to ozone and premature mortality. Jerrett et al. (2009) utilized the ACS cohort with air quality data from 1977 through 2000 (April through September). Jarrett et al. reported a positive and statistically significant association between ambient ozone concentration and respiratory causes of death after controlling for $PM_{2.5}$ using co-pollutant models. Further examination of the association between ozone exposure and respiratory-related mortality revealed the association was increased by higher temperatures and geographic variation. In single pollutant models, long-term ozone exposure was also associated with cardiopulmonary, cardiovascular, and ischemic heart disease mortality, but the associations were not present in the co-pollutant model. Krewski et al. (2009) also utilized data from the ACS cohort with air quality data from 1980 (April through September) and observed a positive association between ozone exposure and all-cause and cardiopulmonary disease mortality. This association was robust to control for ecologic variables, but no association was

[9] For more information regarding thresholds in the $PM_{2.5}$ mortality relationship, please consult the proposed Portland Cement NESHAP RIA (U.S. EPA, 2009a).

observed with ischemic heart disease or lung cancer. In addition, Krewski et al. observed no association with year-round ozone exposure.

S2.3 Presentation of results

Tables S2.1 through S2.6 show the results of this updated analysis. Figures S2.1 and S2.2 show the breakdown of ozone benefits and $PM_{2.5}$ co-benefits by endpoint category using a single mortality study as an example. Figures S2.3 and S2.4 show the ozone benefits and $PM_{2.5}$ co-benefits by mortality study. Figures S2.5 and S2.6 show the breakdown of monetized benefits between ozone, PM, morbidity, mortality, and visibility. Figure S2.7 shows the results of this updated analysis graphically.

Table S2.1: Summary of Total Number of Ozone and $PM_{2.5}$-Related Premature Mortalities and Morbidity Incidences Avoided in 2020 [A]

Combined Estimate of Mortality		0.075 ppm			0.070 ppm			0.065 ppm		
Multi-city	Bell et al. (2004)	760	to	1,900	1,500	to	3,400	2,500	to	5,600
	Schwartz	800	to	1,900	1,600	to	3,600	2,700	to	5,800
	Huang	820	to	1,900	1,700	to	3,600	2,800	to	5,900
Meta-analysis	Bell et al. (2005)	930	to	2,000	2,000	to	4,000	3,500	to	6,600
	Ito et al.	1,000	to	2,100	2,400	to	4,300	4,000	to	7,200
	Levy et al.	1,000	to	2,100	2,400	to	4,300	4,100	to	7,200

Combined Estimate of Morbidity	0.075 ppm	0.070 ppm	0.065 ppm
Acute Myocardial Infarction [B]	1,300	2,200	3,500
Upper Respiratory Symptoms [B]	9,900	19,000	31,000
Lower Respiratory Symptoms [B]	13,000	25,000	41,000
Chronic Bronchitis [B]	470	880	1,400
Acute Bronchitis [B]	1,100	2,100	3,400
Asthma Exacerbation [B]	12,000	23,000	38,000
Work Loss Days [B]	88,000	170,000	270,000
School Loss Days [C]	190,000	600,000	1,100,000
Hospital and ER Visits	2,600	6,600	11,000
Minor Restricted Activity Days	1,000,000	2,600,000	4,500,000

[A] All estimates rounded to two significant figures. Only includes areas required to meet the current standard by 2020; does not include San Joaquin Valley and South Coast air basins in California. Includes ozone benefits, and $PM_{2.5}$ co-benefits. Mortality incidence range was developed by adding the estimate from the ozone premature mortality function to estimates from the $PM_{2.5}$ premature mortality functions from Pope et al. (2002) and Laden et al. (2006).

[B] Estimated reduction in premature morbidity due to $PM_{2.5}$ reductions only.

[C] Estimated reduction in premature morbidity due to ozone reductions only.

Table S2.2: Summary of Total Monetized Benefits in 2020 (3% discount rate, in millions of 2006$)[A, B, C]

Combined Estimate of Mortality		0.075 ppm			0.070 ppm			0.065 ppm		
NMMAPS	Bell et al. (2004)	$6,900	to	$15,000	$13,000	to	$29,000	$22,000	to	$47,000
	Schwartz	$7,200	to	$16,000	$15,000	to	$30,000	$24,000	to	$49,000
	Huang	$7,300	to	$16,000	$15,000	to	$30,000	$25,000	to	$50,000
Meta-analysis	Bell et al. (2005)	$8,300	to	$17,000	$18,000	to	$34,000	$31,000	to	$56,000
	Ito et al.	$9,100	to	$18,000	$21,000	to	$37,000	$36,000	to	$61,000
	Levy et al.	$9,200	to	$18,000	$21,000	to	$37,000	$36,000	to	$61,000

[A] Does not reflect estimates for the San Joaquin and South Coast Air Basins

[B] All estimates rounded to two significant digits

[c] Includes Visibility benefits of $160,000

Table S2.3: Summary of Total Monetized Benefits in 2020 (7% discount rate, in millions of 2006$)[A, B, C]

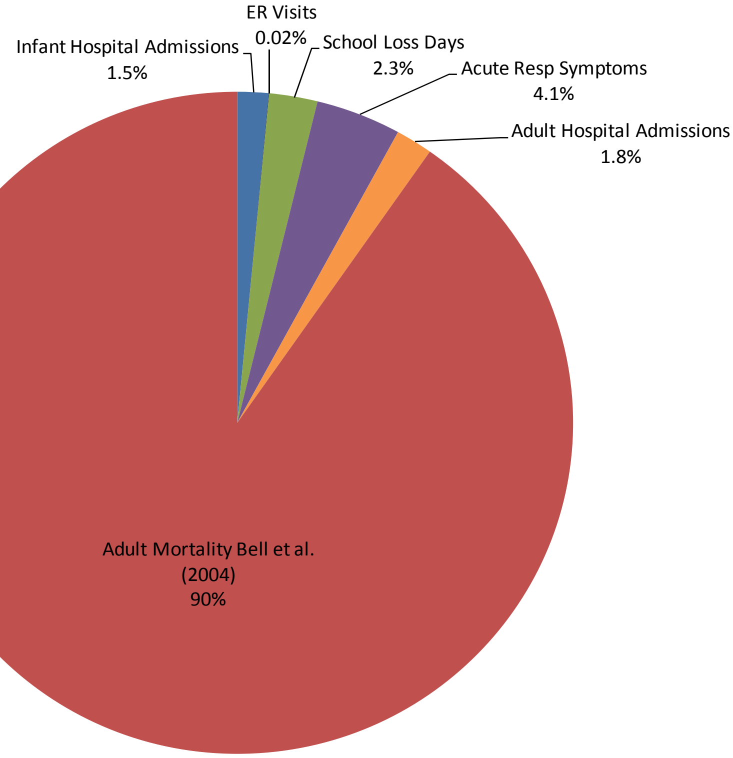
Combined Estimate of Mortality		0.075 ppm			0.070 ppm			0.065 ppm		
NMMAPS	Bell et al. (2004)	$6,400	to	$13,000	$11,000	to	$24,000	$19,000	to	$39,000
	Schwartz	$6,700	to	$13,000	$12,000	to	$25,000	$21,000	to	$41,000
	Huang	$6,800	to	$13,000	$13,000	to	$26,000	$21,000	to	$42,000
Meta-analysis	Bell et al. (2005)	$7,800	to	$14,000	$16,000	to	$29,000	$27,000	to	$48,000
	Ito et al.	$8,600	to	$15,000	$18,000	to	$31,000	$31,000	to	$52,000
	Levy et al.	$8,700	to	$15,000	$18,000	to	$31,000	$32,000	to	$52,000

[A] Does not reflect estimates for the San Joaquin and South Coast Air Basins

[B] All estimates rounded to two significant digits

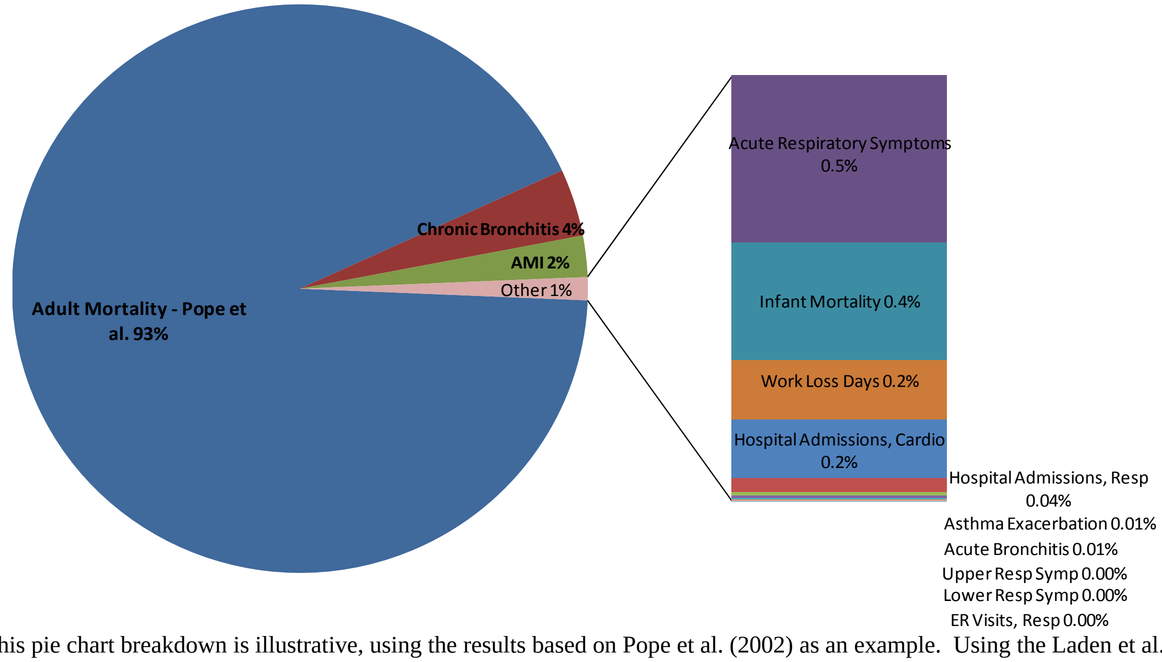
[c] Includes Visibility benefits of $160,000

Figure S2-1: Breakdown of Ozone Health Benefits (using Bell 2004)*



*This pie chart breakdown is illustrative, using the results based on Bell et al. (2004) as an example. Using the Levy et al. (2006) function for premature mortality, the percentage of total monetized benefits due to adult mortality would be 97%.

Figure S2-2: Breakdown of $PM_{2.5}$ Health Benefits (using Pope)*



*This pie chart breakdown is illustrative, using the results based on Pope et al. (2002) as an example. Using the Laden et al. (2006) function for premature mortality, the percentage of total monetized benefits due to adult mortality would be 97%. This chart shows the breakdown using a 3% discount rate, and the results would be similar if a 7% discount rate was used.

Table S2.4: Summary of National Ozone Benefits by Standard Level with 95th percentile confidence intervals (in millions of 2006$)[A, B, C]

Endpoint Group	Author		0.075 ppm Valuation	0.075 ppm Incidence	0.070 ppm Valuation	0.070 ppm Incidence	0.065 ppm Valuation	0.065 ppm Incidence
Infant Hospital Admissions, Respiratory			$11 ($5.7 -- $16)	550 (310 -- 830)	$17 ($8.5 -- $25)	1,700 (960 -- 2,600)	$30 ($15 -- $43)	3,000 (1,700 -- 4,500)
Emergency Room Visits, Respiratory			$0.11 (-$.21 -- $.35)	290 (-310 -- 930)	$0.36 (-$.71 -- $1.2)	990 (-890 -- 3,200)	$0.66 (-$1.3 -- $2.2)	1,800 (-1,600 -- 5,800)
School Loss Days			$17 ($7.5 -- $24)	190,000 (93,000 -- 280,000)	$53 ($23 -- $76)	600,000 (300,000 -- 880,000)	$96 ($42 -- $140)	1,100,000 (550,000 -- 1,600,000)
Acute Respiratory Symptoms			$30 ($12 -- $56)	510,000 (280,000 -- 790,000)	$96 ($37 -- $180)	1,600,000 (910,000 -- 2,500,000)	$170 ($68 -- $320)	2,900,000 (1,700,000 -- 4,500,000)
Hospital Admissions, Respiratory			$13 ($1.7 -- $22)	550 (130 -- 980)	$45 ($5.6 -- $77)	1,900 (550 -- 3,400)	$81 ($11 -- $140)	3,400 (1,000 -- 6,100)
Mortality	Bell et al.	2004	$660 ($54 -- $2,000)	74 (36 -- 120)	$2,200 ($180 -- $6,600)	250 (130 -- 410)	$4,000 ($330 -- $12,000)	450 (240 -- 730)
Mortality	Schwartz		$1,000 ($82 -- $3,000)	110 (54 -- 190)	$3,400 ($270 -- $10,000)	380 (190 -- 630)	$6,200 ($500 -- $19,000)	700 (350 -- 1,100)
Mortality	Huang		$1,100 ($95 -- $3,300)	130 (66 -- 200)	$3,800 ($320 -- $11,000)	420 (230 -- 670)	$6,800 ($580 -- $20,000)	770 (420 -- 1,200)
Mortality	Bell et al.	2005	$2,000 ($190 -- $6,100)	240 (140 -- 350)	$7,000 ($630 -- $21,000)	800 (490 -- 1,200)	$10,000 ($1,100 -- $37,000)	1,500 (910 -- 2,200)
Mortality	Ito et al.		$2,900 ($280 -- $8,200)	330 (230 -- 450)	$9,900 ($930 -- $28,000)	1,100 (790 -- 1,500)	$18,000 ($1,700 -- $50,000)	2,000 (1,400 -- 2,800)
Mortality	Levy et al.		$3,000 ($280 -- $8,200)	340 (260 -- 430)	$10,000 ($930 -- $28,000)	1,100 (870 -- 1,500)	$18,000 ($1,700 -- $50,000)	2,100 (1,600 -- 2,600)

[A] Does not reflect estimates for the San Joaquin and South Coast Air Basins

[B] Confidence intervals are not available for PM co-benefits because of methodological limitations when using benefit-per-ton estimates.

[C] All estimates rounded to two significant digits

Table S2.5: Summary of National Ozone Benefits and $PM_{2.5}$ Co-Benefits by Standard Level (in millions of 2006$ at a 3% discount rate)[A, B, C]

	Endpoint Group	Author	0.075 ppm Valuation	0.075 ppm Incidence	0.070 ppm Valuation	0.070 ppm Incidence	0.065 ppm Valuation	0.065 ppm Incidence
Ozone	Infant Hospital Admissions, Respiratory		$11	550	$17	1,700	$30	3,000
	Emergency Room Visits, Respiratory		$0.11	290	$0.36	990	$0.66	1,800
	School Loss Days		$17	190,000	$53	600,000	$96	1,100,000
	Acute Respiratory Symptoms		$30	510,000	$96	1,600,000	$170	2,900,000
	Hospital Admissions, Respiratory		$13	550	$45	1,900	$81	3,400
	Mortality	Bell et al. (2004)	$660	74	$2,200	250	$4,000	450
	Mortality	Schwartz	$1,000	110	$3,400	380	$6,200	700
	Mortality	Huang	$1,100	130	$3,800	420	$6,800	770
	Mortality	Bell et al. (2005)	$2,100	240	$7,100	800	$13,000	1,500
	Mortality	Ito et al.	$2,900	330	$9,900	1,100	$18,000	2,000
	Mortality	Levy et al.	$3,000	340	$10,000	1,100	$18,000	2,100
$PM_{2.5}$	Chronic Bronchitis		$230	470	$430	880	$700	1,400
	Acute Myocardial Infarction		$140	1,300	$240	2,200	$380	3,500
	Hospital Admissions, Respiratory		$2.5	180	$4.3	310	$6.8	490
	Hospital Admissions, Cardiovascular		$11	390	$18	670	$29	1,000
	Emergency Room Visits, Respiratory		$0.22	590	$0.39	1,100	$0.63	1,700
	Acute Bronchitis		$0.08	1,100	$0.15	2,100	$0.25	3,400
	Work Loss Days		$11	88,000	$20	170,000	$34	270,000
	Asthma Exacerbation		$0.64	12,000	$1.2	23,000	$2.0	38,000
	Acute Respiratory Symptoms		$31	520,000	$58	980,000	$95	1,600,000
	Lower Respiratory Symptoms		$0.24	13,000	$0.45	25,000	$0.75	41,000
	Upper Respiratory Symptoms		$0.29	9,900	$0.54	19,000	$0.89	31,000
	Infant Mortality		$22	3	$44	5	$73	8
	Mortality	Pope et al	$5,500	690	$10,000	1,200	$16,000	2,000
	Mortality	Laden et al	$14,000	1,800	$26,000	3,200	$41,000	5,100
	Mortality	Expert K	$1,900	230	$3,500	430	$5,700	700
	Mortality	Expert E	$19,000	2,300	$34,000	4,200	$55,000	6,800

[A] Does not reflect estimates for the San Joaquin and South Coast Air Basins

[B] Does not include confidence intervals

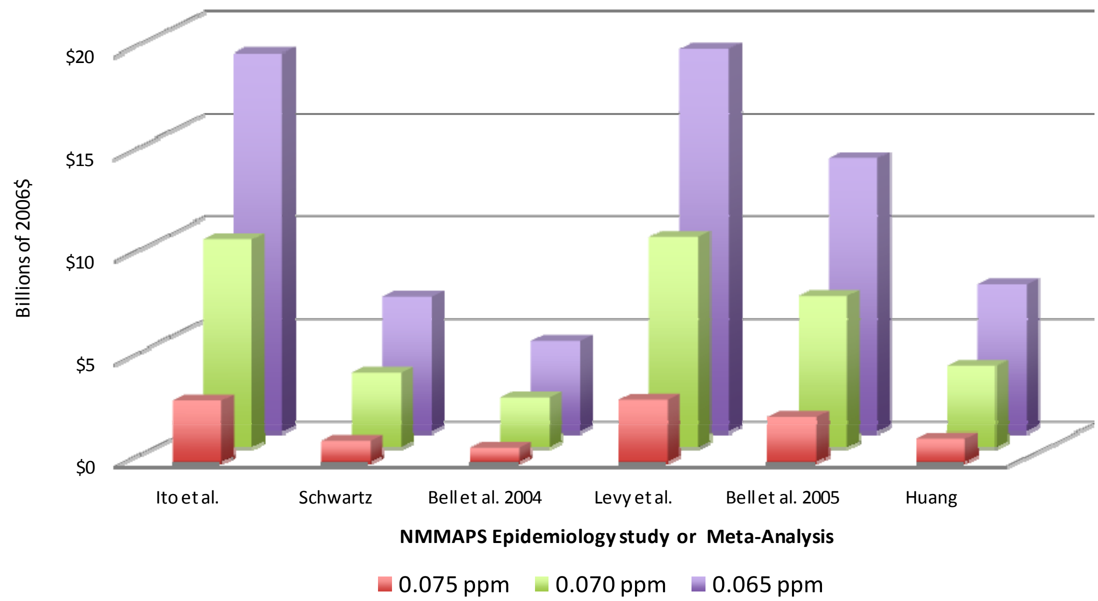
[C] All estimates rounded to two significant digits

Table S2.6: Summary of National Ozone Benefits and $PM_{2.5}$ Co-Benefits by Standard Level (in millions of 2006$ at a 7%

	Endpoint Group	Author	0.075 ppm Valuation	0.075 ppm Incidence	0.070 ppm Valuation	0.070 ppm Incidence	0.065 ppm Valuation	0.065 ppm Incidence
Ozone	Infant Hospital Admissions, Respiratory		$11	550	$17	1,700	$30	3,000
	Emergency Room Visits, Respiratory		$0.11	290	$0.36	990	$0.66	1,800
	School Loss Days		$17	190,000	$53	600,000	$96	1,100,000
	Acute Respiratory Symptoms		$30	510,000	$96	1,600,000	$170	2,900,000
	Hospital Admissions, Respiratory		$13	550	$45	1,900	$81	3,400
	Mortality	Bell et al. (2004)	$660	74	$2,200	250	$4,000	450
	Mortality	Schwartz	$1,000	110	$3,400	380	$6,200	700
	Mortality	Huang	$1,100	130	$3,800	420	$6,800	770
	Mortality	Bell et al. (2005)	$2,100	240	$7,100	800	$13,000	1,500
	Mortality	Ito et al.	$2,900	330	$9,900	1,100	$18,000	2,000
	Mortality	Levy et al.	$3,000	340	$10,000	1,100	$18,000	2,100
$PM_{2.5}$	Chronic Bronchitis		$230	470	$430	880	$700	1,400
	Acute Myocardial Infarction		$140	1,300	$240	2,200	$380	3,500
	Hospital Admissions, Respiratory		$2.5	180	$4.3	310	$6.8	490
	Hospital Admissions, Cardiovascular		$11	390	$18	670	$29	1,000
	Emergency Room Visits, Respiratory		$0.22	590	$0.39	1,100	$0.63	1,700
	Acute Bronchitis		$0.08	1,100	$0.15	2,100	$0.25	3,400
	Work Loss Days		$11	88,000	$20	170,000	$34	270,000
	Asthma Exacerbation		$0.64	12,000	$1.2	23,000	$2.0	38,000
	Acute Respiratory Symptoms		$31	520,000	$58	980,000	$95	1,600,000
	Lower Respiratory Symptoms		$0.24	13,000	$0.45	25,000	$0.75	41,000
	Upper Respiratory Symptoms		$0.29	9,900	$0.54	19,000	$0.89	31,000
	Infant Mortality		$22	3	$44	5	$73	8
	Mortality	Pope et al	$5,000	690	$9,000	1,200	$14,000	2,000
	Mortality	Laden et al	$13,000	1,800	$23,000	3,200	$37,000	5,100
	Mortality	Expert K	$1,700	230	$3,100	430	$5,100	700
	Mortality	Expert E	$17,000	2,300	$31,000	4,200	$49,000	6,800

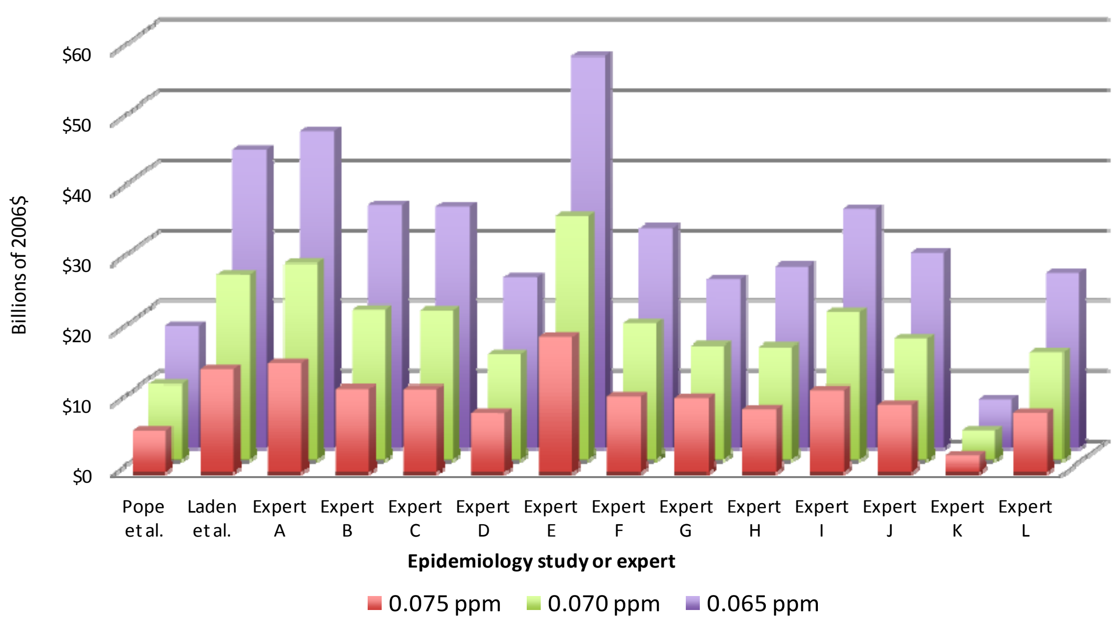
[A] Does not reflect estimates for the San Joaquin and South Coast Air Basins
[B] Does not include confidence intervals
[C] All estimates rounded to two significant digits

Figure S2.3: Ozone benefits for Alternate Standard Levels*



*This graph shows the estimated ozone benefits in 2020 using three NMMAPS-based epidemiology studies and three meta-analyses. The results shown are not the direct results from the studies; rather, the estimates are based in part on the concentration-response function provided in those studies. Because all ozone-related health effects are short-term, the discount rate does not affect the results.

Figure S2.4: $PM_{2.5}$ co-benefits for Alternate Standard Levels*



*This graph shows the estimated $PM_{2.5}$ co-benefits in 2020 using the no-threshold model at discount rates of 3% using effect coefficients using the Pope et al. study and the Laden et al study, as well as 12 effect coefficients derived from EPA's expert elicitation on PM mortality. The results shown are not the direct results from the studies or expert elicitation; rather, the estimates are based in part on the concentration-response function provided in those studies. Results using a 7% discount rate would be similar, but approximately 9% lower.

Figure S2.5: Breakdown of total monetized benefits for Alternate Standard Levels (Low)

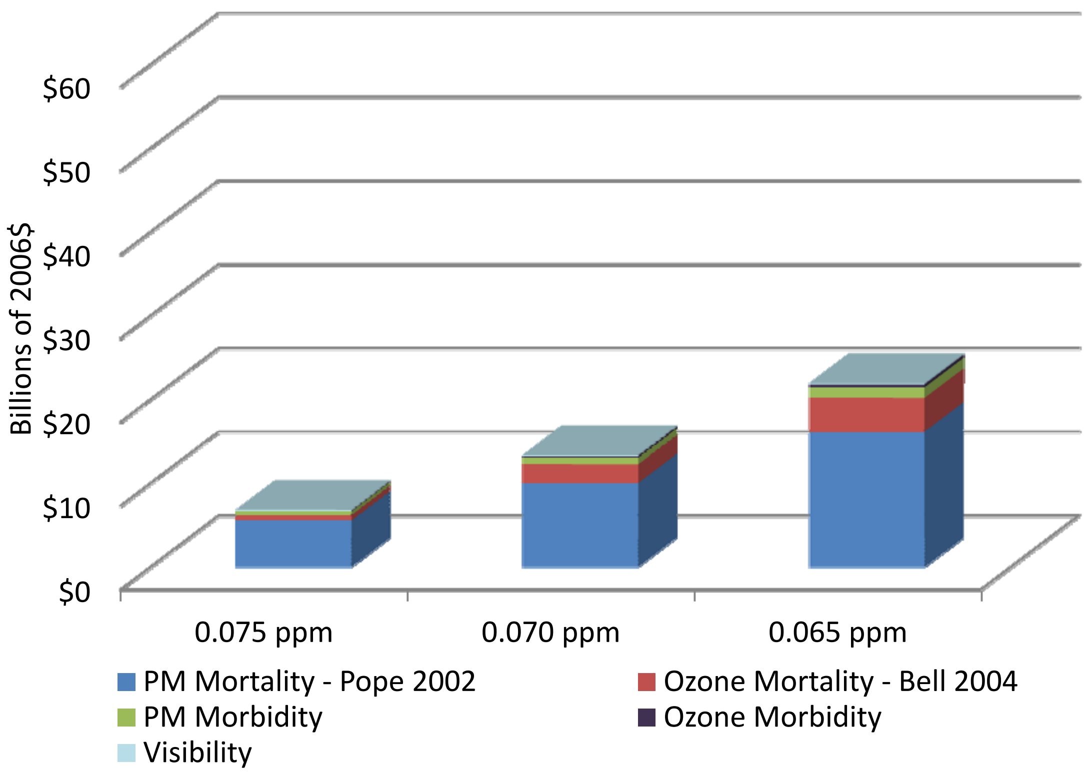


Figure S2.6: Breakdown of total monetized benefits for Alternate Standard Levels (High)

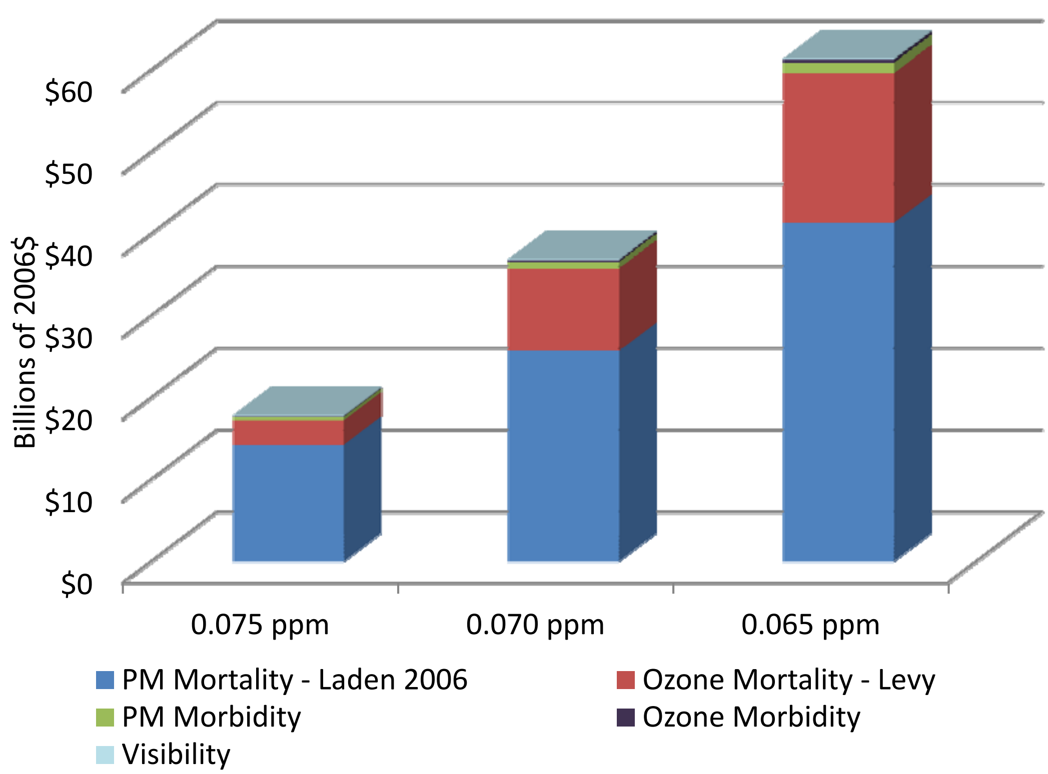
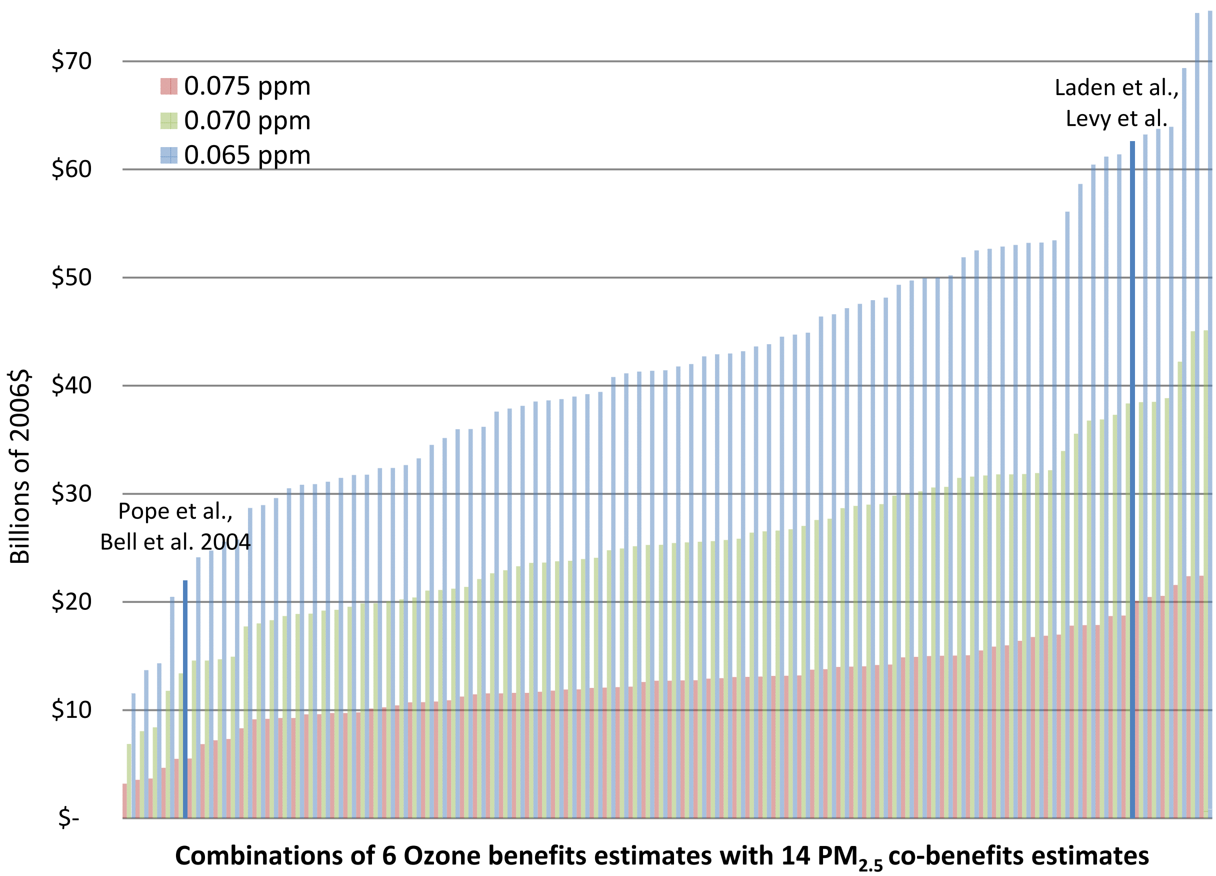


Figure S2.7: Total Monetized Benefits for Alternate Standard Levels*



*This graph shows the estimated total monetized benefits in 2020 using the no-threshold model at discount rates of 3% using effect coefficients derived from the 6 ozone mortality studies and PM co-benefits estimates using the Pope et al. study and the Laden et al study, as well as 12 effect coefficients derived from EPA's expert elicitation on PM mortality. The highlighted results represent the combined estimates from Bell et al. (2004) with Pope et al. (2002) and Levy (2005) with Laden et al. (2006). The results shown are not the direct results from the studies or expert elicitation; rather, the estimates are based in part on the concentration-response function provided in those studies. PM co-benefit results using a 7% discount rate would be similar, but approximately 9% lower.

In 2008, the National Research Council (NRC) evaluated the EPA's approach to estimating ozone-related mortality benefits. Among other recommendation, in its report the NRC indicated that "EPA should consider placing greater emphasis on reporting decrease in age-specific death rates and increases in life expectancy…" (NRC, 2008). As a first step in implementing this recommendation, below for two of the three scenarios, we present changes in the percentage of total cause-specific mortality attributable to ozone and the change in the number of life years.[10] Table 7 summarizes the estimated number of life years gained resulting from simulated attainment with the 0.065 ppm and 0.070 ppm standard alternatives. To simplify this presentation

[10] Here we omit the results for the 0.075 ppm alternative. We estimated the benefits of attaining this alternative through an interpolation approach that made subsequent estimation of life years and changes in death rates technically challenging.

we include results based on the estimates of ozone mortality reported in Levy et al. (2005) and Bell et al. (2004), which provide upper and lower-bound estimates, respectively.

Table S2.7: Estimated Reduction in Ozone-Related Premature Mortality in Terms of Life Years Gained from Increases in Life Expectancy

	Bell et al. (2004) mortality estimate		*Levy et al. (2005) mortality estimate*	
Age Range	0.070 ppm	0.065 ppm	0.070 ppm	0.065 ppm
25-29	75 (32—120)	130 (58—210)	660 (780—830)	1,200 (850—1,500)
30-34	66 (28—100)	120 (51—180)	580 (420—740)	1,000 (750—1,300)
35-44	260 (110—410)	460 (200—730)	1,600 (1,200—2,000)	2,800 (2,000—3,500)
45-54	520 (220—830)	930 (400—1,500)	2,600 (1,900—3,300)	4,500 (3,300—5,700)
55-64	1,000 (440—1,600)	1,800 (780—2,800)	4,600 (3,400—5,900)	8,100 (5,900—10,000)
65-74	1,200 (500—1,900)	2,100 (900—3,300)	5,200 (3,800—6,600)	9,100 (6,700—12,000)
75-84	810 (340—1,300)	1,400 (620—2,200)	3,500 (2,600—4,500)	6,200 (4,600—7,900)
85-99	400 (170—630)	720 (310—1,100)	1,800 (1,300—2,200)	3,100 (2,300—4,000)

Table S2.8 summarizes the percentage of total mortality attributable to ozone. As above, we include estimates based on the Bell et al. (2004) and Levy et al. (2005) risk coefficients.

Table S2.8: Percentage of Total Mortality Attributable to Ozone

	Bell et al. (2004) mortality estimate		*Levy et al. (2005) mortality estimate*	
Age Range	0.070 ppm	0.065 ppm	0.070 ppm	0.065 ppm
25-29	0.030%	0.054%	0.126%	0.224%
30-34	0.029%	0.052%	0.123%	0.217%
35-44	0.029%	0.051%	0.123%	0.217%
45-54	0.030%	0.052%	0.127%	0.224%
55-64	0.028%	0.050%	0.122%	0.212%
65-74	0.027%	0.047%	0.114%	0.200%
75-84	0.026%	0.046%	0.112%	0.197%
85-99	0.027%	0.048%	0.115%	0.206%

Based on our review of the current body of scientific literature, EPA estimated PM-related mortality without applying an assumed concentration threshold. EPA's Integrated Science Assessment for Particulate Matter (U.S. EPA, 2009c), which was recently reviewed by EPA's Clean Air Scientific Advisory Committee, concluded that the scientific literature consistently finds that a no-threshold log-linear model most adequately portrays the PM-mortality concentration-response relationship while recognizing potential uncertainty about the exact shape of the concentration-response function. Consistent with this finding, we have conformed the threshold sensitivity analysis to the current state of the PM science improved upon our previous approach for estimating the sensitivity of the benefits estimates to the presence of an assumed threshold by incorporating a new "Lowest Measured Level" (LML) assessment.

This approach summarizes the distribution of avoided PM mortality impacts according to the baseline $PM_{2.5}$ levels (i.e. those levels that exist prior to the implementation of the ozone attainment scenario) experienced by the population receiving the $PM_{2.5}$ mortality benefit (Figure S2.8 and S2.9). We identify on this figure the lowest air quality levels measured in each of the two primary epidemiological studies EPA uses to quantify PM-related mortality. This information allows readers to determine the portion of PM-related mortality benefits occurring above or below the LML of each study; in general, our confidence in the estimated PM mortality decreases as we consider air quality levels further below the LML in the two epidemiological studies. While the LML analysis provides some insight into the level of uncertainty in the estimated PM mortality benefits, EPA does not view the LML as a threshold and continues to quantify PM-related mortality impacts using a full range of modeled air quality concentrations.

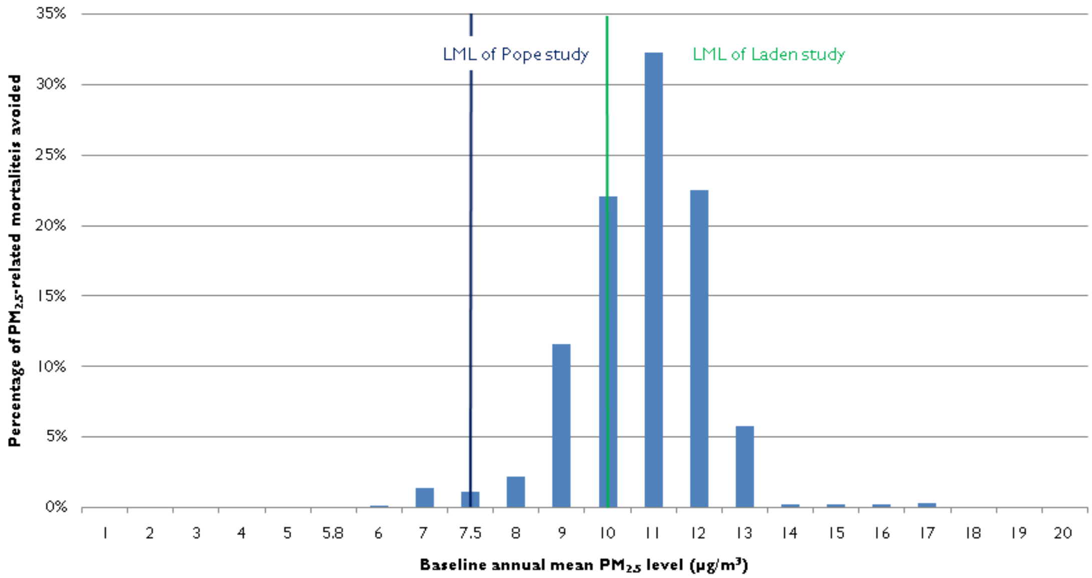
The very large proportion of the avoided PM-related impacts we estimate in this illustrative analysis occur among populations exposed at or above the LML of each study (Figures S2.8 and S2.9), increasing our confidence in the PM mortality analysis. Approximately 62% of the avoided impacts occur at or above an annual mean $PM_{2.5}$ level of 10 $\mu g/m^3$ (the LML of the Laden et al. 2006 study); about 97% occur at or above an annual mean $PM_{2.5}$ level of 7.5 $\mu g/m^3$ (the LML of the Pope et al. 2002 study). As we model mortality impacts among populations exposed to levels of $PM_{2.5}$ that are successively lower than the LML of each study our confidence in the results diminishes. However, the analysis above confirms that the great majority of the impacts occur at or above each study's LML.

Because time and resource limitations prevented EPA from performing air quality modeling of the $PM_{2.5}$-related co-benefits of the illustrative ozone attainment strategies, this LML analysis considers only a single air quality modeling scenario. This single scenario represents only a portion of $PM_{2.5}$ reductions we anticipate to occur as a result of the NOx emission reductions needed to attain a new standard of 0.065 ppm. As such, this LML analysis provides an incomplete representation of the distribution of avoided mortality impacts and reductions in $PM_{2.5}$ exposure that might occur under a air quality modeling scenario that simulated full attainment with the 0.065 ppm standard.

Finally, Figure S2.10 illustrates the percentage of population exposed to different levels of annual mean $PM_{2.5}$ levels in the baseline and after the implementation of the illustrative ozone attainment strategy in 2020. This strategy achieves fairly modest reductions of $PM_{2.5}$ as a co-benefit of the ozone attainment strategy. Much of this small benefit occurs among highly exposed populations and we find that prior to the implementation of this illustrative scenario, 83% of the population live in areas where $PM_{2.5}$ levels are projected to be above the lowest measured levels of the Pope study. Taken together, this information increases our confidence in the estimated mortality reductions for this rule.

While the LML of each study is important to consider when characterizing and interpreting the overall level PM-related benefits, as discussed earlier in this chapter, EPA believes that both cohort-based mortality estimates are suitable for use in air pollution health impact analyses. When estimating PM mortality impacts using risk coefficients drawn from the Laden et al. analysis of the Harvard Six Cities and the Pope et al. analysis of the American Cancer Society cohorts there are innumerable other attributes that may affect the size of the reported risk estimates—including differences in population demographics, the size of the cohort, activity patterns and particle composition among others. The LML assessment presented here provides a limited representation of one key difference between the two studies.

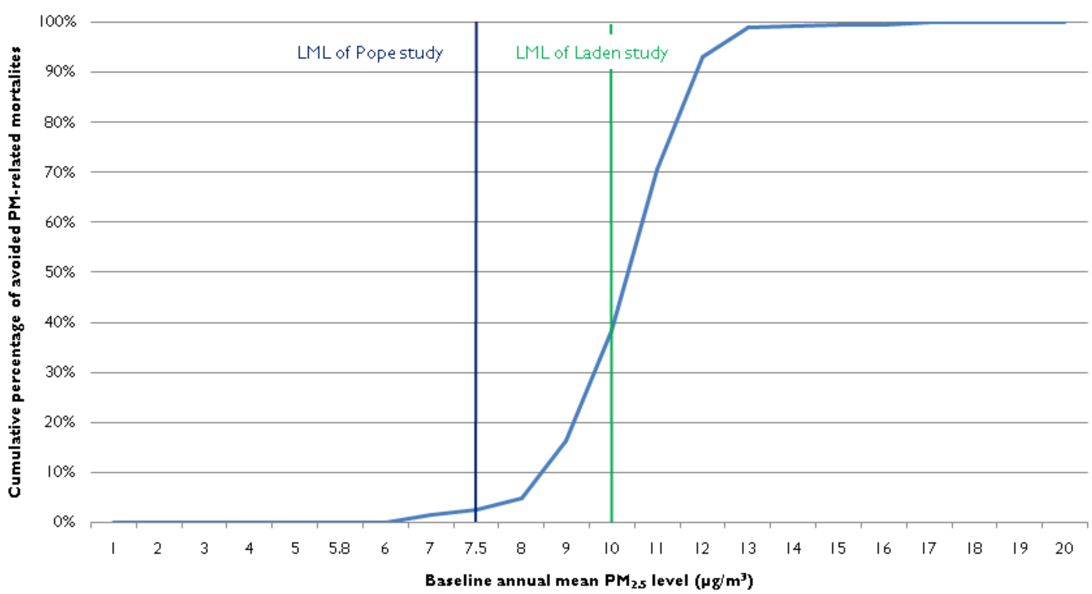
Figure S2.8: Percentage of PM-related mortalities avoided by baseline $PM_{2.5}$ air quality level



Of the total mortalities avoided:

97% occur among populations exposed to PM levels at or above the LML of the **Pope et al.** study.
62% occur among populations exposed to PM levels at or above the LML of the **Laden et al.** study.

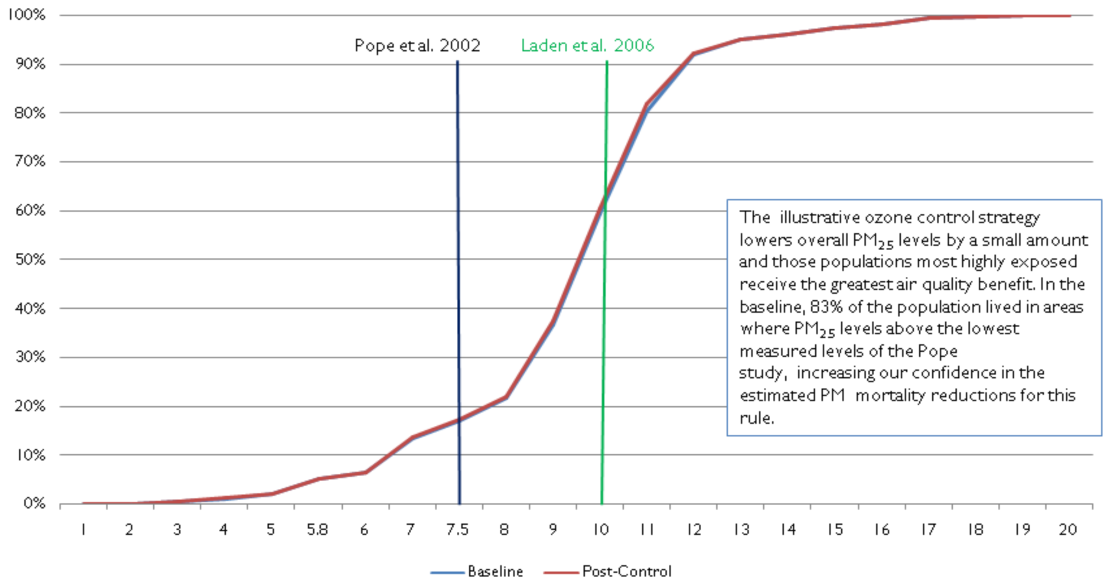
Figure S2.9: Cumulative percentage of total PM-related mortalities avoided by baseline $PM_{2.5}$ air quality level



Of the total mortalities avoided:

97% occur among populations exposed to PM levels at or above the LML of the **Pope et al.** study.
62% occur among populations exposed to PM levels at or above the LML of the **Laden et al.** study.

S2.10: Cumulative distribution of adult population at annual mean PM2.5 levels (pre- and post- policy scenario)



S2.4 Comparison of results to previous results in 2008 Ozone NAAQS RIA

The overall effect of incorporating the array of methodological changes was to increase the estimated benefits of attaining alternate ozone standards estimates presented in the 2008 Ozone NAAQS RIA. In general, the key update that had the largest effect on the valuation and the incidence results is removing the threshold from the PM concentration-response functions. Tables S2.9 and S2.10 show the total monetized benefits, costs, and net benefits for the 2008 Ozone RIA analysis and this updated analysis, respectively. Figure 6 shows a comparison of the range of net benefits estimates in this updated analysis compared to the net benefits presented in the 2008 Ozone NAAQS RIA.[11]

[11] Net benefits are total monetized benefits minus total monetized costs. Total monetized benefits include ozone health benefits, $PM_{2.5}$ health co-benefits, visibility benefits, but not other unquantified benefit categories.

Table S2.9: Total Monetized Costs with Ozone Benefits and $PM_{2.5}$ Co-Benefits in 2020 (in Billions of 2006$) [A] 2008 RIA

	Ozone Mortality Function	Reference	Total Benefits [B]		Total Costs [C]	Net Benefits	
			3%	7%	7%	3%	7%
0.075 ppm	NMMAPS and Multi-city	Bell et al. 2004	$4.4 to $8.5	$4.1 to $7.7	$7.6 to $8.8	$-4.4 to $0.9	$-4.7 to $0.1
		Schwartz 2005	N/A	N/A	N/A	N/A	N/A
		Huang 2005	N/A	N/A	N/A	N/A	N/A
	Meta-analysis	Bell et al. 2005	$5.6 to $9.7	$5.3 to $9.0	$7.6 to $8.8	$-3.2 to $2.1	$-3.5 to $1.4
		Ito et al. 2005	$6.3 to $10	$5.9 to $9.6	$7.6 to $8.8	$-2.5 to $2.7	$-2.9 to $2.0
		Levy et al. 2005	$6.3 to $10	$6.0 to $9.7	$7.6 to $8.8	$-2.5 to $2.8	$-2.8 to $2.1
0.070 ppm	NMMAPS and multi-city	Bell et al. 2004	$8.8 to $16	$8.2 to $15	$19 to $25	$-16 to $-2.8	$-17 to $4.1
		Schwartz 2005	N/A	N/A	N/A	N/A	N/A
		Huang 2005	N/A	N/A	N/A	N/A	N/A
	Meta-analysis	Bell et al. 2005	$13 to $21	$13 to $19	$19 to $25	$-12 to $1.5	$-12 to $0.2
		Ito et al. 2005	$15 to $23	$15 to $21	$19 to $25	$-9.6 to $3.8	$-10 to $2.5
		Levy et al. 2005	$16 to $23	$15 to $22	$19 to $25	$-9.3 to 4.1	$9.9 to $2.7
0.065 ppm	NMMAPS and multi-city	Bell et al. 2004	$15 to $27	$14 to $24	$32 to $44	$-29 to $-5.4	$-30 to $-7.5
		Schwartz 2005	N/A	N/A	N/A	N/A	N/A
		Huang 2005	N/A	N/A	N/A	N/A	N/A
	Meta-analysis	Bell et al. 2005	$22 to $34	$21 to $32	$32 to $44	$-22 to $2.4	$-23 to $0.3
		Ito et al. 2005	$27 to $39	$26 to $36	$32 to $44	$-17 to $6.6	$-18 to $4.4
		Levy et al. 2005	$27 to $39	$26 to $37	$32 to $44	$-17 to $7.0	$-18 to $4.9

[A] All estimates rounded to two significant figures. As such, they may not sum across columns. Only includes areas required to meet the current standard by 2020; does not include San Joaquin and South Coast areas in California.

[B] Includes ozone benefits, and $PM_{2.5}$ co-benefits. Range was developed by adding the estimate from the ozone premature mortality function to estimates from the $PM_{2.5}$ premature mortality functions from Pope et al. and Laden et al. Tables exclude unquantified and nonmonetized benefits.

[C] Range reflects lower and upper bound cost estimates. Data for calculating costs at a 3% discount rate was not available for all sectors, and therefore total annualized costs at 3% are not presented here. Additionally, these estimates assume a particular trajectory of aggressive technological change. An alternative storyline might hypothesize a much less optimistic technological trajectory, with increased costs, or with decreased benefits in 2020 due to a later attainment date.

Table S2.10: Total Monetized Costs with Ozone Benefits and $PM_{2.5}$ Co-Benefits in 2020 (in Billions of 2006$) [A] Updated Analysis

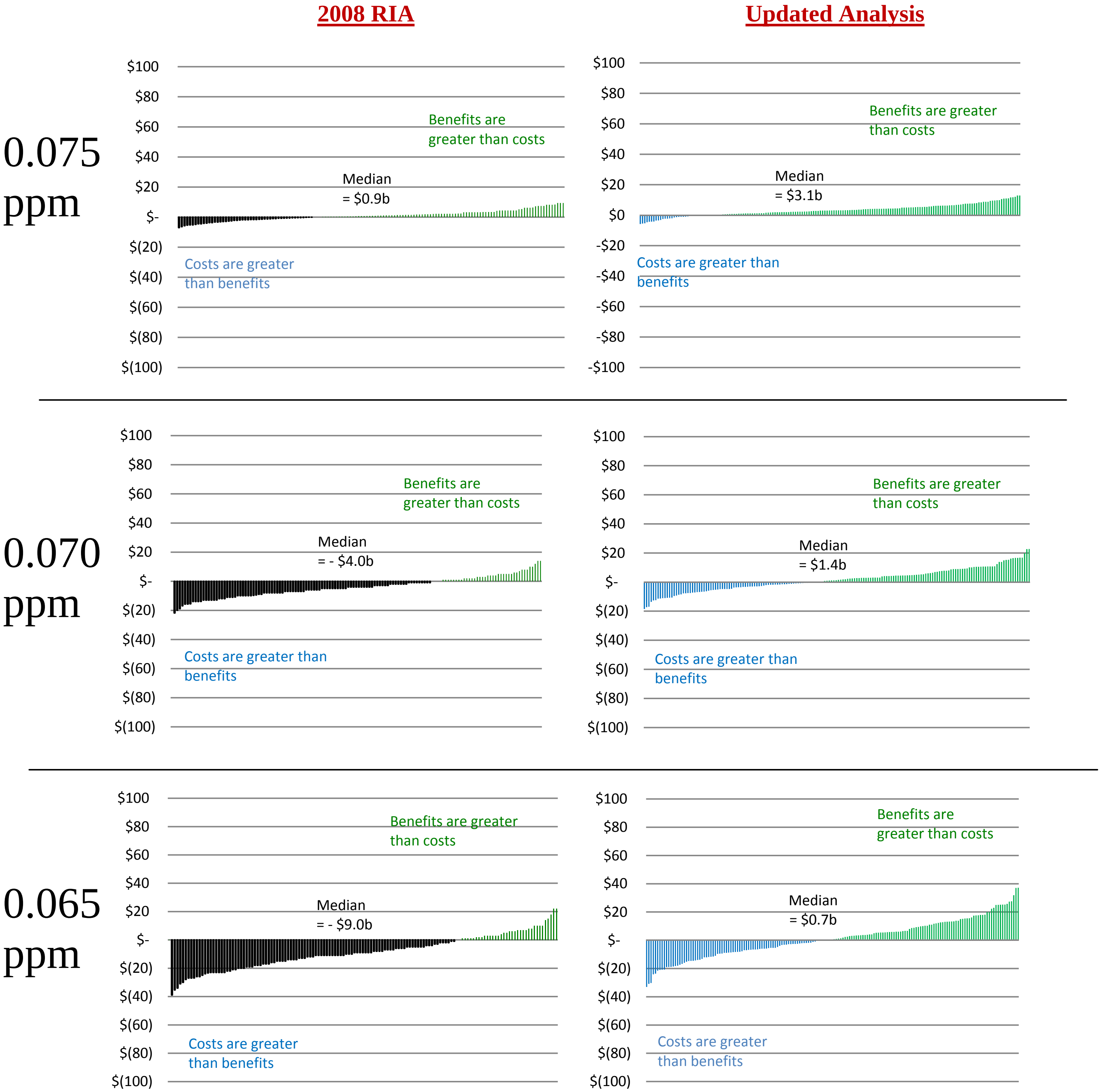
	Ozone Mortality Function	Reference	Total Benefits [B]		Total Costs [C]	Net Benefits	
			3%	7%	7%	3%	7%
0.075 ppm	NMMAPS and multi-city	Bell et al. 2004	$6.9 to $15	$6.4 to $13	$7.6 to $8.8	$-1.9 to $7.4	$-2.4 to $5.4
		Schwartz 2005	$7.2 to $16	$6.8 to $13	$7.6 to $8.8	$-1.6 to $8.4	$-2.1 to $5.4
		Huang 2005	$7.3 to $16	$6.9 to $13	$7.6 to $8.8	$-1.5 to $8.4	$-2.0 to $5.4
	Meta-analysis	Bell et al. 2005	$8.3 to $17	$7.9 to $14	$7.6 to $8.8	$-0.50 to $9.4	$-1.0 to $6.4
		Ito et al. 2005	$9.1 to $18	$8.7 to $15	$7.6 to $8.8	$0.30 to $10	$-0.20 to $7.4
		Levy et al. 2005	$9.2 to $18	$8.8 to $15	$7.6 to $8.8	$0.40 to $10	$-0.10 to $7.4
0.070 ppm	NMMAPS and multi-city	Bell et al. 2004	$13 to $29	$11 to $24	$19 to $25	$-12 to $10	$-14 to $5.0
		Schwartz 2005	$15 to $30	$12 to $25	$19 to $25	$-10 to $11	$-13 to $6.0
		Huang 2005	$15 to $30	$13 to $26	$19 to $25	$-10 to $11	$-12 to $7.0
	Meta-analysis	Bell et al. 2005	$18 to $34	$16 to $29	$19 to $25	$-7.0 to $15	$-9.0 to $10
		Ito et al. 2005	$21 to $37	$18 to $31	$19 to $25	$-4.0 to $18	$-6.0 to $12
		Levy et al. 2005	$21 to $37	$18 to $31	$19 to $25	$-4.0 to $18	$-6.0 to $12
0.065 ppm	NMMAPS and multi-city	Bell et al. 2004	$22 to $47	$19 to $40	$32 to $44	$-22 to $15	$-25 to $7.0
		Schwartz 2005	$24 to $49	$21 to $42	$32 to $44	$-20 to $17	$-23 to $9.0
		Huang 2005	$25 to $50	$22 to $42	$32 to $44	$-19 to $18	$-23 to $10
	Meta-analysis	Bell et al. 2005	$31 to $56	$27 to $48	$32 to $44	$-13 to $24	$-17 to $16
		Ito et al. 2005	$36 to $61	$32 to $53	$32 to $44	$-8.0 to $29	$-13 to $20
		Levy et al. 2005	$36 to $61	$32 to $53	$32 to $44	$-7.0 to $29	$-12 to $20

[A] All estimates rounded to two significant figures. As such, they may not sum across columns. Only includes areas required to meet the current standard by 2020; does not include San Joaquin and South Coast areas in California.

[B] Includes ozone benefits, and $PM_{2.5}$ co-benefits. Range was developed by adding the estimate from the ozone premature mortality function to estimates from the $PM_{2.5}$ premature mortality functions from Pope et al. and Laden et al. Tables exclude unquantified and nonmonetized benefits.

[C] Range reflects lower and upper bound cost estimates. Data for calculating costs at a 3% discount rate was not available for all sectors, and therefore total annualized costs at 3% are not presented here. Additionally, these estimates assume a particular trajectory of aggressive technological change. An alternative storyline might hypothesize a much less optimistic technological trajectory, with increased costs, or with decreased benefits in 2020 due to a later attainment date.

Figure S2.11: Comparison of Net Benefits in Updated Analysis to 2008 Ozone NAAQS RIA*



These graphs shows all combinations of the 6 different ozone mortality functions and assumptions, the 14 different PM mortality functions, and the 2 cost methods. These combinations do not represent a distribution.

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SECTION 3: SECONDARY OZONE NAAQS EVALUATION

1.1

1.2 Synopsis

This section contains an evaluation of the regulatory impacts associated with a distinct secondary NAAQS for ozone. The purpose of a secondary NAAQS is to protect the public welfare against the negative effects of criteria air pollutants, including decreased visibility, damage to animals, crops, vegetation, and buildings. Exposure to ozone has been associated with a wide array of vegetation and ecosystem effects, including those that damage or impair the intended use of the plant or ecosystem. Such effects are considered adverse to the public welfare. This secondary NAAQS standard for ozone is the first secondary standard to be promulgated with a form, averaging time, and level that is distinct from the health-based primary standard, apart from the PM and SO_2 regulations originally set in the early 1970s. Quantifying the costs and benefits of attaining a secondary NAAQS is an exceptionally complex task, including unresolved issues related to the RIA analysis, air quality projections, monitoring expansion, and implementation.[12] Because of these complexities as well as limited time and resources within the expedited schedule, we are limited in our ability to quantify the costs and benefits of attaining a distinct secondary NAAQS for ozone for this rule. However, we provide a semi-quantitative assessment in this analysis, including identifying which counties would have an additional requirement to reduce ozone concentrations to attain a secondary standard beyond the reductions needed to attain the primary standard, qualitative descriptions of available pollution control strategies, qualitative benefits of reducing ozone exposure on forests, crops, and ornamental plants, and maps of avoided biomass/yield loss for the currently monitor locations. The Administrator selected a secondary ozone NAAQS at a level of 13 ppm-hrs using the W126 form. Using a cumulative seasonal secondary standard (i.e., W126), we evaluated alternate standard levels at 11, 13, and 15 ppm-hours.

S2.6 Introduction

As defined by section 109(b)(2) of the Clean Air Act (CAA), the purpose of a secondary NAAQS is to protect the public welfare against any known or anticipated negative effects associated with criteria air pollutants. These welfare effects 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 wellbeing.’’

The secondary NAAQS for ozone is focused on the negative effects on vegetation associated with direct ozone exposure. Exposure to ozone has been associated with a wide array

[12] These complexities are described in detail in Section S3.3.

of vegetation and ecosystem effects in the published literature (U.S. EPA, 2006). Sensitivity to ozone is highly variable across plant species, with over 65 plant species identified as "ozone-sensitive", many of which occur in state and national parks and forests.[13] These effects include those that damage or impair the intended use of the plant or ecosystem. Such effects are considered adverse to the public welfare and can include reduced growth and/or biomass production in sensitive plant species, including forest trees, reduced crop yields, visible foliar injury, reduced plant vigor (e.g., increased susceptibility to harsh weather, disease, insect pest infestation, and competition), species composition shift, and changes in ecosystems and associated ecosystem services.

Vegetation effects research has shown that seasonal air quality indices that cumulate peak-weighted hourly ozone concentrations are the best candidates for relating exposure to plant growth effects (U.S. EPA, 2006). Based on this research, the 2007 Ozone Staff Paper (hereafter, "the Staff Paper") concluded that the cumulative, seasonal index referred to as "W126" is the most appropriate index for relating vegetation response to ambient ozone exposures (U.S. EPA, 2007b). Based on additional conclusions regarding appropriate diurnal and seasonal exposure windows, the Staff Paper recommended a cumulative seasonal secondary standard, expressed as an index of the annual sum of weighted hourly concentrations (using the W126 form), set at a level in the range of 7 to 21 ppm-hours. The index would be cumulated over the 12-hour daylight window (8:00 a.m. to 8:00 p.m.) during the consecutive 3-month period during the ozone season with the maximum index value (hereafter, referred to as W126). After reviewing the recommendations in the Staff Paper, EPA's Clean Air Scientific Advisory committee (CASAC) agreed with the form of the secondary standard, but instead recommended a range of 7 to 15 ppm-hours (U.S. EPA-SAB, 2007). In January 2010, EPA's Administrator proposed a range of secondary standards based on the W126 index between 7 and 15 ppm-hrs (U.S. EPA, 2010). After reviewing the scientific evidence and public comments, the Administrator selected a secondary ozone NAAQS at a level of 13 ppm-hrs, using the W126 form, calculated as a 3-year average of annual sums.

To comply with Circular A-4 (OMB, 2003), this analysis includes the selected standard level as well as one more stringent and one less stringent alternative. Therefore, this analysis focuses on secondary standards at 13 ppm-hrs, as well as 15 ppm-hrs and 11 ppm-hrs.

S2.7 Air Quality Analysis

Ozone is a secondary pollutant formed by atmospheric reactions involving two classes of precursor compounds: nitrogen oxides (NOx) and volatile organic compounds (VOCs) (U.S. EPA,

[13] Appendix S3A contains a list of plant species identified as "ozone-sensitive".

2007b). The W126 standard is a specific peak-weighted index that is summed over 12 hours per day during the maximum 3-month period within the ozone season and calculated as the 3-year average of the annual sums. An example of this calculation is described in more detail in Appendix S3-B of this RIA. The 3-year average provides increased stability due to large year-to-year variability. As described in the Staff Paper, using the highest PRB estimate of 0.035 ppm from Fiore et al. (2003) as a constant value would only add up to a 3-month 12-hr W126 of less than 1 ppm-hr (U.S. EPA, 2007b).

a. Ambient Monitoring Data (2007 – 2009)

The monitoring data for this analysis has been updated since the proposal. In addition to incorporating more recent monitoring data, we have also excluded monitoring data from CASTNET that cannot be used for nonattainment designations. Ozone concentrations were generally lower in 2009, and thus the 2007-2009 design values indicate fewer counties would violate the secondary standard compared to the counties shown in the proposal analysis. These monitoring data are limited to the existing monitoring network. It is important to note that nonattainment designations are likely to be based on 2008-2010 data, not 2007-2009 data. [14]

In this analysis, we considered the extent to which there is overlap between county-level air quality measured in terms of the 8-hour average form of the current standard and that measured in terms of the cumulative W126, seasonal form. Using monitoring data collected from 2007 to 2009, Table S3-1 shows the number of counties that exceed the alternate secondary standard levels in comparison to the number of counties that exceed the selected primary standard at 0.070 ppm. Figure S3-1 maps the counties that correspond with Table S3-1.

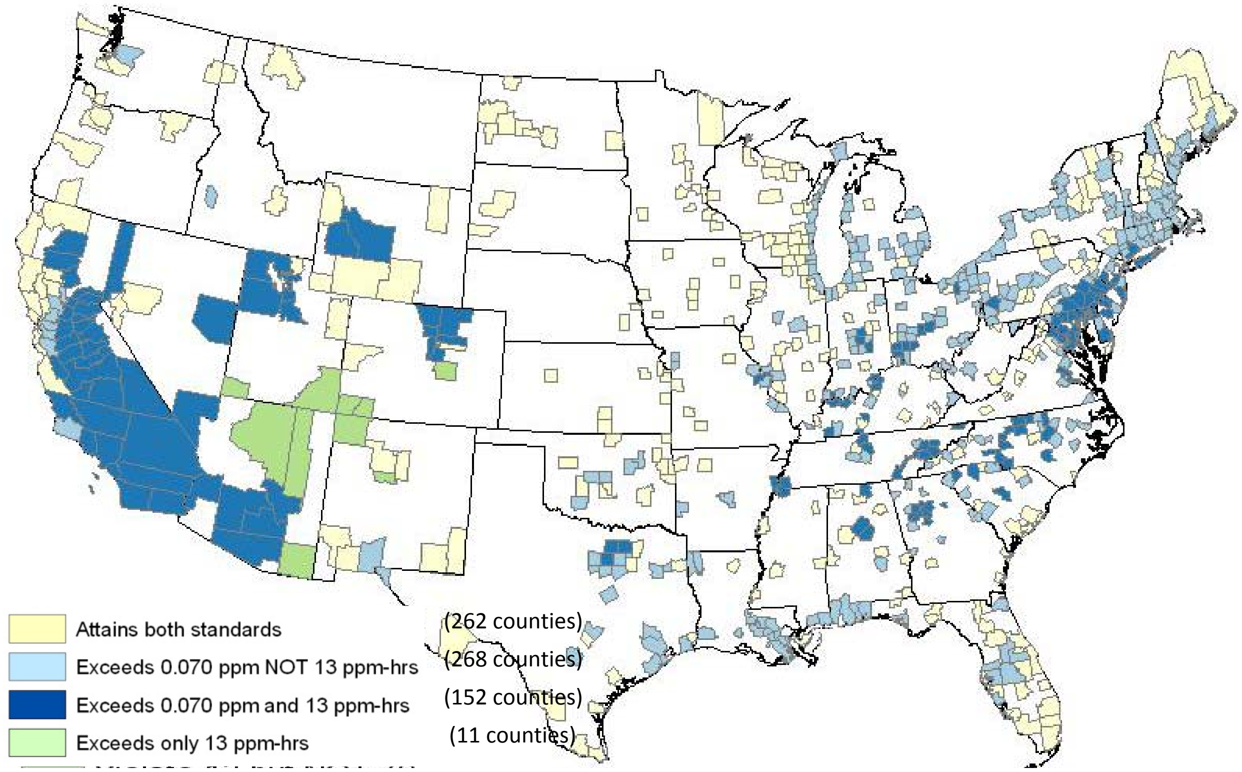
Table S3-1: Number of Counties Exceeding Alternate Secondary Standards (2007-2009 monitoring data)

Monitor Baseline	15 ppm-hrs	13 ppm-hrs	11 ppm-hrs
Attain primary (0.070 ppm) and secondary	270	262	257
Exceed only primary (0.070 ppm)	335	268	194
Exceed primary (0.070 ppm) and secondary	85	152	226
Exceed only secondary	3	11	16

* As these estimates are limited to existing ozone monitoring data, there might be other non-monitored areas after the monitoring network is expanded that would exceed the secondary standard. There are 693 currently monitored counties with sufficient data for this analysis.

[14] Monitoring data for 2010 is not yet available.

Figure S3-1: Counties exceeding Primary Standard at 0.070 ppm or Secondary Standard at 13 ppm-hours (based on 2007–2009 monitoring data)



b. Modeling Projection Data (2020)

In this analysis, we also projected W126 levels for two scenarios in 2020 developed as part of the 2008 analysis of the primary standard: the baseline scenario and the after hypothetical RIA controls scenario. The modeling methodology used to project W126 levels into the future utilizes the same approach as used to project design values of the primary standard, as described in EPA modeling guidance (U.S. EPA, 2007a). The 2020 baseline and hypothetical RIA control scenario are fully described in Chapter 3 of the 2008 Ozone NAAQS RIA (U.S. EPA, 2008a). The baseline includes current state and federal programs plus additional controls EPA estimated would be necessary to attain the previous ozone and $PM_{2.5}$ standards. For the hypothetical RIA control scenario, EPA applied additional known NOx and VOC controls in those specific geographic areas that were predicted to exceed an 0.070 ppm primary standard in 2020.[15]

Additionally, EPA estimated the counties that are projected to attain the primary standard in 2020 but would still exceed the alternate secondary standards. These data are listed in Table S3-2, and mapped in Figures S3-2 through S3-5. Because this projection approach is prefaced on

[15] It is important to note that the modeled hypothetical RIA controls did not fully attain the primary standard of 0.070 ppm, especially in Southern California, Houston, Eastern Lake Michigan, and the Northeast corridor.

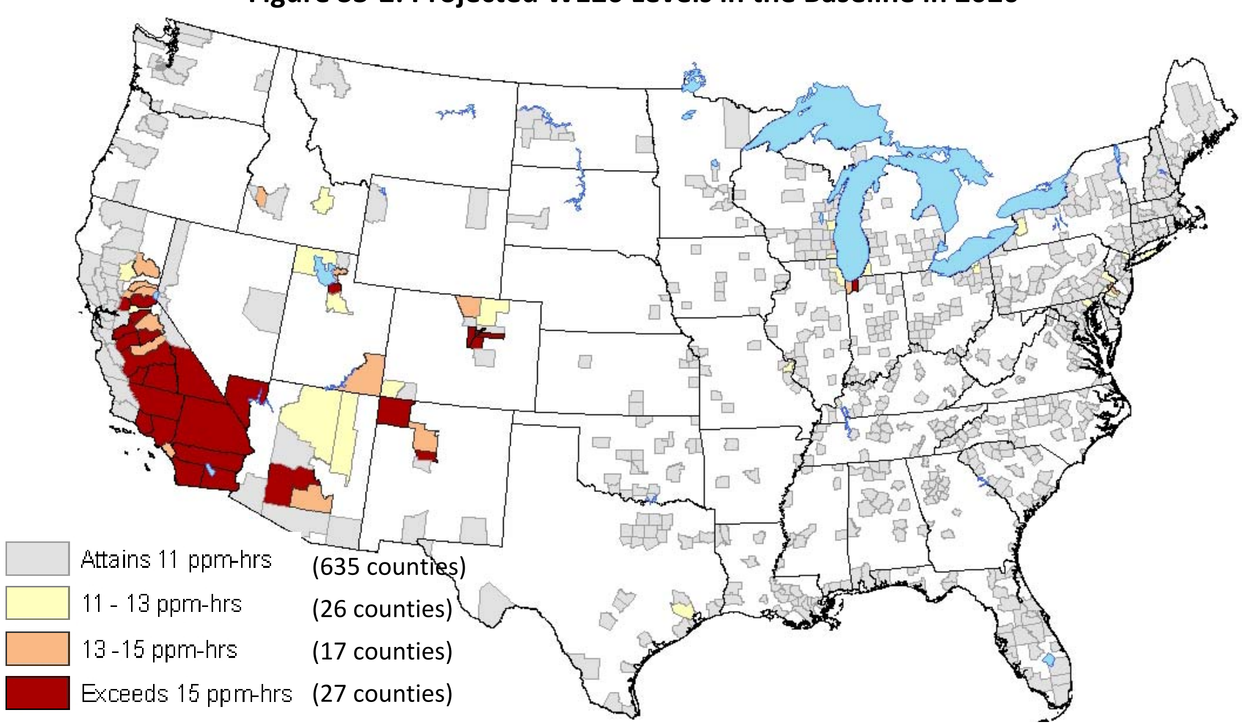
ambient data, projections can only be made for counties with ozone monitoring data for the base period. As a result, Table S3-2 and the associated figures may not capture other, currently unmonitored, locations.

Table S3-2: Number of Counties Projected to Exceed Alternate Secondary Standards in 2020*

2020 Baseline	15 ppm-hrs	13 ppm-hrs	11 ppm-hrs
Attain primary (0.070 ppm) and secondary	599	591	580
Exceed only primary (0.070 ppm)	79	70	55
Exceed primary (0.070 ppm) and secondary	20	29	44
Exceed only secondary	7	15	26
After Hypothetical RIA controls	**15 ppm-hrs**	**13 ppm-hrs**	**11 ppm-hrs**
Attain primary (0.070 ppm) and secondary	633	624	613
Exceed only primary (0.070 ppm)	48	41	36
Exceed primary (0.070 ppm) and secondary	17	24	29
Exceed only secondary	7	16	27

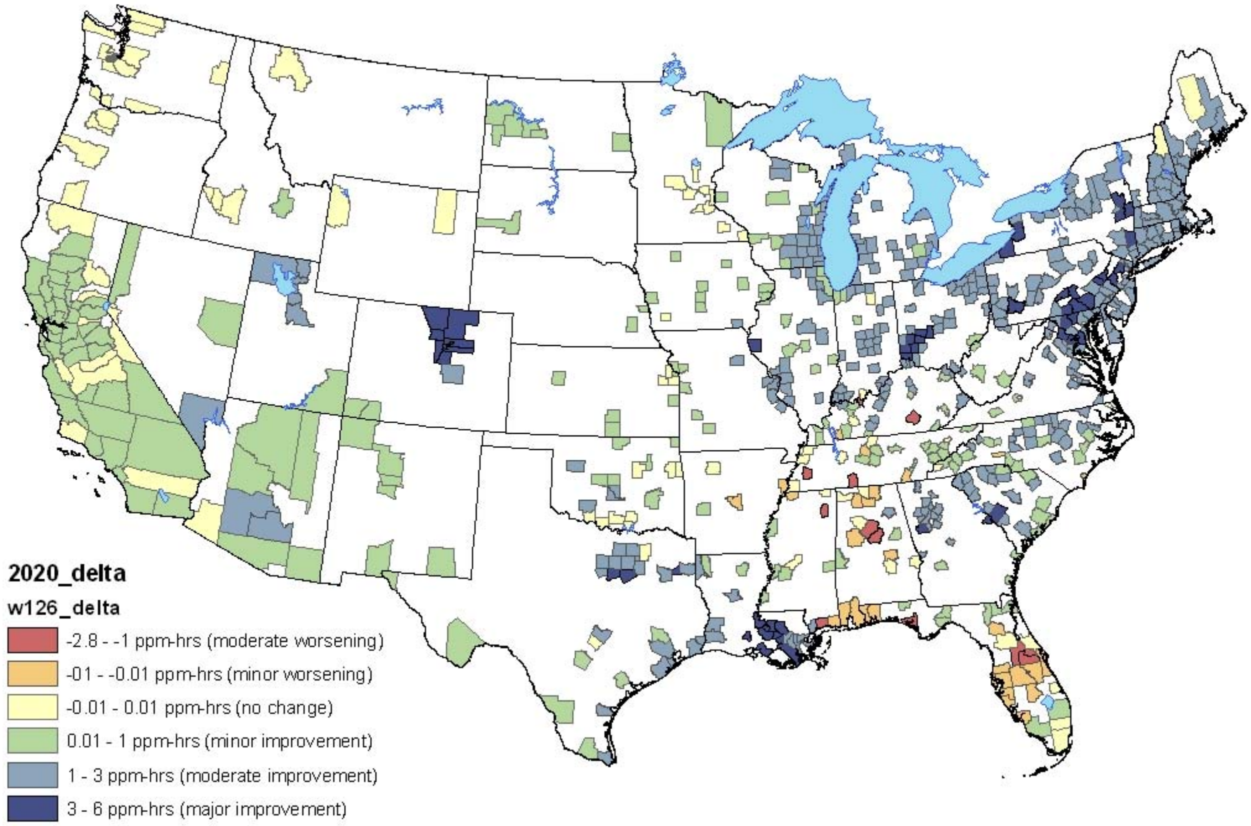
* As these projections are limited to counties with existing ozone monitoring data, there might be other non-monitored areas that would exceed the secondary standard while attaining the primary standard. There are 705 currently monitored counties with sufficient data for this analysis. It is important to note that the modeled hypothetical RIA controls did not fully attain the primary standard of 0.070 ppm, especially in Southern California, Houston, Eastern Lake Michigan, and the Northeast corridor. The number of counties that exceed only the secondary standard increase after the hypothetical RIA controls because those counties now attain the primary standard.

Figure S3-2: Projected W126 Levels in the Baseline in 2020*



* Many of the counties projected to exceed the alternate secondary standard levels are in the South Coast and San Joaquin areas of California, which are not required to attain the primary standards by 2020.

Figure S3-3: Change in Projected W126 Levels from the Hypothetical RIA controls in 2020*



*All of the counties projected to experience minor or moderate worsening due to the hypothetical RIA controls in 2020 are located in areas well below the alternate secondary standard levels. Because the hypothetical RIA controls were designed to reduce ozone concentrations in areas that exceeded the primary standard, those areas are also projected to experience minor to major improvements in W126 levels in 2020. It is important to note that the modeled hypothetical RIA controls did not fully attain the primary standard of 0.070 ppm, especially Southern California, Houston, Eastern Lake Michigan, and the Northeast corridor.

Figure S3-4: Counties Projected to Exceed the Selected Primary and Secondary Standards in the Baseline in 2020*

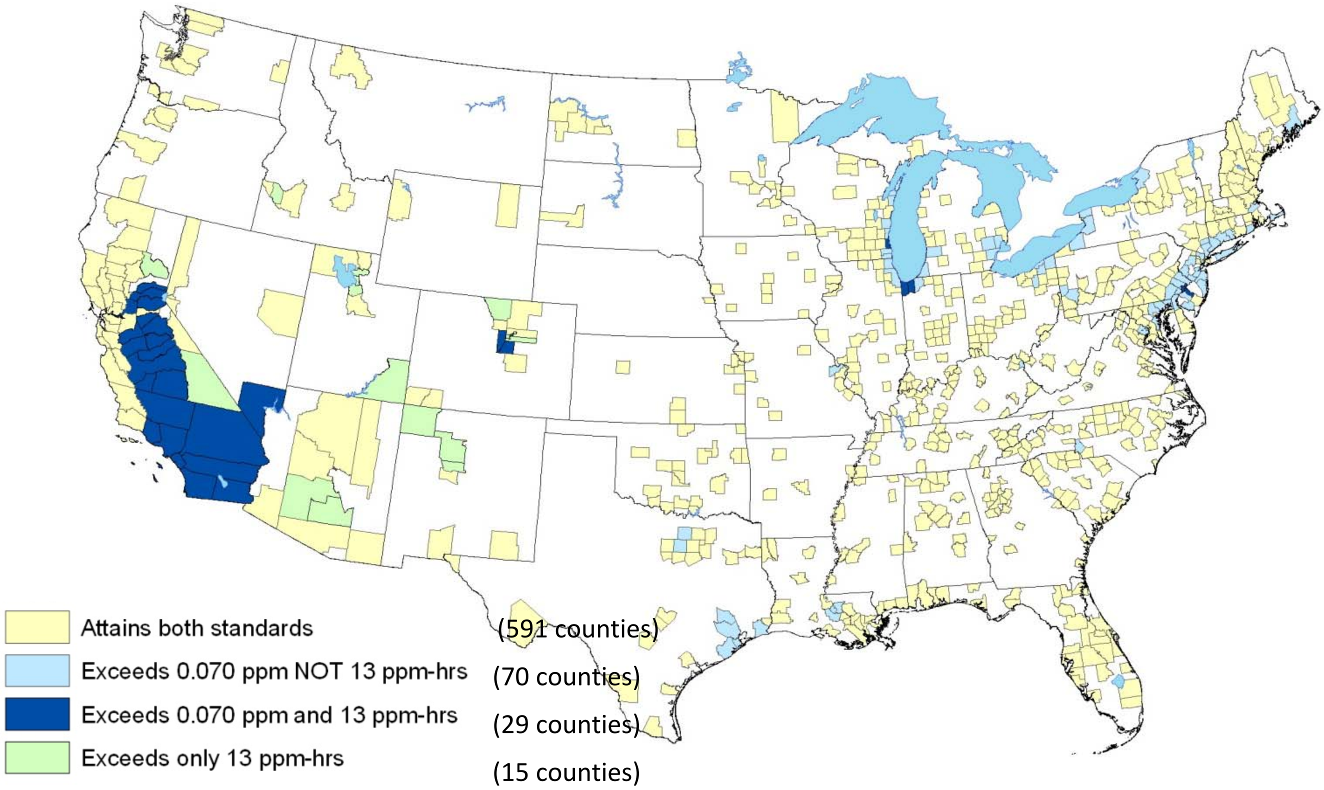
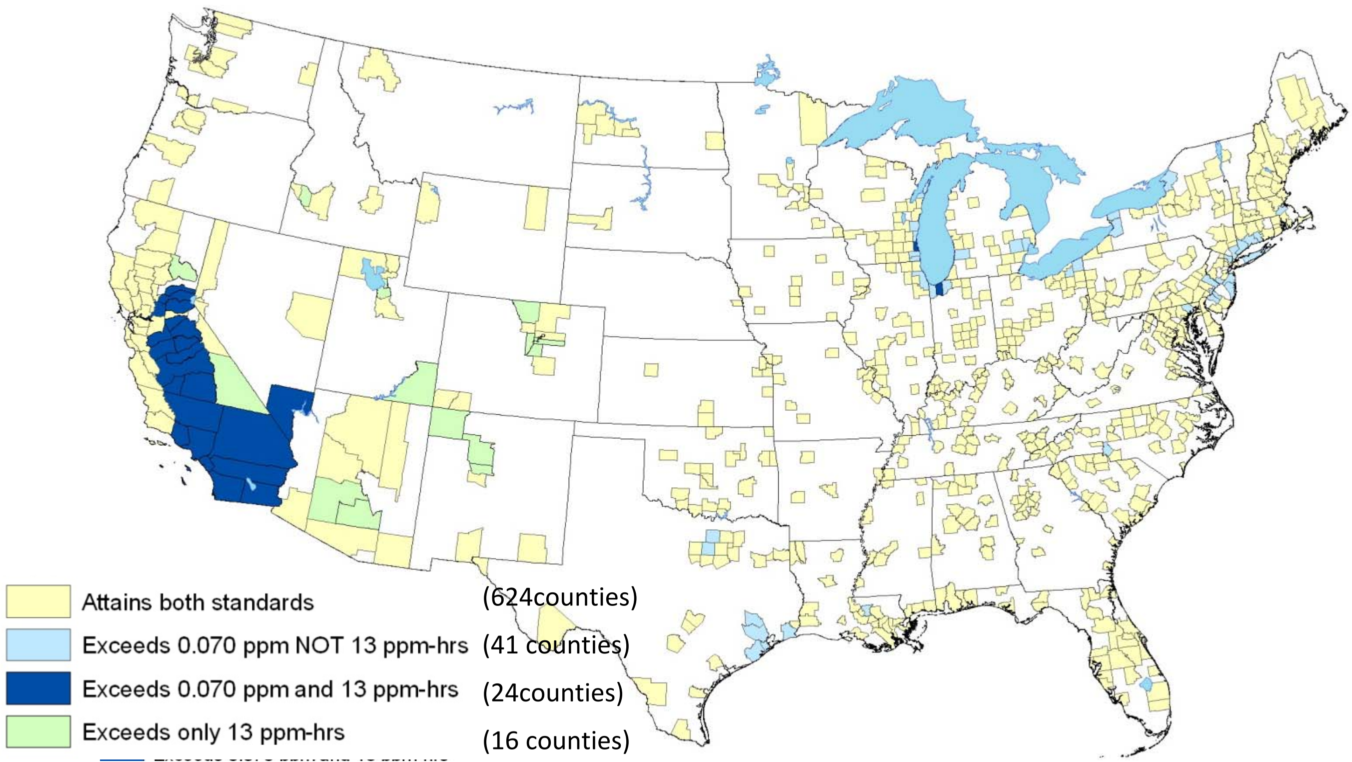


Figure S3-5: Counties Projected to Exceed the Selected Primary and Secondary Standards after Hypothetical RIA Controls in 2020*



* Many of the counties projected to exceed the secondary standard are in the South Coast and San Joaquin areas of California, which are not required to attain the primary standards by 2020. The number of counties that exceed only the secondary standard increase after the hypothetical RIA controls because those counties now attain the primary standard.

As noted above, this analysis only projected W126 levels in 2020 where ozone monitors currently exist. Due to the lack of more complete monitor coverage in many rural areas, this analysis might not be an accurate reflection of ozone concentrations in non-monitored, rural counties where sensitive vegetation, important ecosystems, or other areas of national public interest could be located. Many counties that contain high elevation, rural or remote sites tend to have flatter ozone concentration distributions. These areas may not reflect the typical urban and near-urban pattern of low morning and evening ozone concentrations with a high mid-day peak, but instead maintain relatively flat patterns with many concentrations in the mid-range (e.g., 0.05-0.09 ppm) for extended periods. Therefore, the potential for disconnect between 8-hour average and cumulative, seasonal form is greater. Additional rural, high elevation areas important for vegetation that are not currently monitored would likely experience similar ozone exposure patterns (U.S. EPA, 2007b). This is an important caveat because: (1) the biological database stresses the importance of cumulative, seasonal exposures in determining plant response; (2) plants have not been specifically tested for the importance of daily maximum 8-hour ozone concentrations in relation to plant response; and (3) the effects of attainment of a 8-hour standard in upwind urban areas on rural air quality distributions cannot be characterized with confidence due to the lack of monitoring data in rural and remote areas (U.S. EPA, 2007b).

Thus far, we have not expressly considered the question of whether it would be more difficult to attain the secondary standard than the primary or what levels of controls would be required to attain the secondary standard. Based on the existing air quality modeling from the 2008 Ozone NAAQS RIA, we have examined how W126 values might change in response to the hypothetical RIA control strategy designed to attain the primary standard. Based on projected W126 ozone levels before and after the implementation of the hypothetical RIA control strategy in 2020, there is some evidence that it may indeed be harder to attain the secondary standard in some areas. As an example, the hypothetical RIA control scenario reduces the number of counties exceeding a primary NAAQS of 0.070 ppm by about 34%; whereas the same control scenario reduces the number of counties exceeding a secondary NAAQS of 13 ppm-hours by only 9%.

The air quality modeling for the 2008 RIA focused on quantifying the impacts and costs of attaining the primary standard. Because the form of the secondary standard is calculated by summing the daily ozone concentrations over a three-month period, it is possible that mitigation strategies may be different for a secondary ozone standard than for the primary ozone standard. Initial ambient data analyses and future-year modeling suggest that it may be more difficult to attain the secondary standard in the western U.S. than in the eastern U.S for several reasons. First, ozone concentrations have less variability across days in the western U.S. Second, the meteorological parameters that generally result in lower daily ozone peaks (e.g., clouds, precipitation, frontal passages) occur less frequently in the western States. Lastly, the secondary standard may have larger implications for rural areas currently without monitors as opposed to the urban areas where the primary ozone standard is already a concern. Attainment of the secondary standard may involve more regional and national scale controls than the current local efforts to reduce peak concentrations.

S2.8 Complexities in Quantifying the Costs and Benefits of Attaining a Secondary Ozone NAAQS

Despite recent proposals, EPA has not promulgated a secondary NAAQS with a form, averaging time, and level that is distinct from the health-based primary standard, apart from the secondary NAAQS for PM and SO_2 originally set in the early 1970s. Therefore, prior to this rule, EPA has not conducted a regulatory analysis of a secondary NAAQS. Quantifying the costs and benefits associated with attaining a distinct secondary standard is an exceptionally complex task. We describe these complexities in detail below.

Because of these complexities as well as limited time, resources, and available data within the expedited schedule, we are limited in our ability to quantify the costs and benefits of attaining a distinct secondary NAAQS for ozone. However, we recognize that the regulatory impacts associated with this standard are of interest to many. Therefore, we provide a semi-quantitative assessment in this analysis, including identifying which counties would have an additional requirement to reduce ozone concentrations to attain a secondary standard beyond the reductions needed to attain the primary standard, qualitative descriptions of available pollution control strategies, qualitative benefits of reducing ozone exposure on forests, crops, and ornamental plants, and maps of avoided biomass/yield loss for the currently monitor locations.

S4.3.1 RIA complexities

There are two unresolved RIA issues that complicate a fully quantitative analysis of a secondary standard for ozone. First, it is unclear when an area would need to attain a secondary standard, which makes it difficult to choose an appropriate analysis year for the RIA. Whereas attainment dates for the primary NAAQS are explicitly designated in the CAA, the attainment dates for the secondary NAAQS are required "as expeditiously as practicable" after the nonattainment designation (42 USC §7502(a)(2)). As air quality improves over time from regulations already promulgated, an area would not need as many emission reductions for a later analysis year as the area would need for an earlier analysis year. Assuming an analysis year of 2020 as was assumed for the primary standard would substantially overestimate the costs and benefits associated with attaining the secondary standard. Even if we determined that it was most appropriate to choose an analysis year of 2030, 2040, or even 2050, we are limited to the available modeling data for

2020. Therefore, the choice of an analysis year has a significant effect on the magnitude of the costs and benefits of attaining a secondary standard.

Second, it is unclear whether it is appropriate to include emission reductions that occur as a result of implementing the primary standard in the baseline for the analysis of the secondary standard. This is a critical decision, as it would either improperly ascribe the costs and benefits of the primary NAAQS to the secondary NAAQS or it would violate the requirements of OMB's Circular A-4 to only include promulgated rules in the regulatory baseline. Most of the areas that exceed the secondary standard also exceed the primary standard. As shown in Table S3-2, the hypothetical RIA controls designed to attain the primary standard also reduce the number of counties that exceed the secondary standard. Furthermore, it is likely that full attainment of the primary standard in areas like Southern California or Eastern Lake Michigan would further reduce the number of counties that exceed the secondary standard.

S4.3.2 Air quality data complexities

In addition to unresolved RIA issues, we have limited information available from the available air quality modeling data to inform a secondary standard analysis. As shown in Table S3-2, several counties are projected to not to attain the alternate secondary standard levels in 2020 even after applying controls for the hypothetical RIA control scenario. Estimating the amount of additional reductions (extrapolated tons) needed to attain a secondary standard would require a better understanding of the relationship between emissions reductions and the W126 metric. Our long experience with the primary standard allows us to use simple impact ratios with some confidence in the extrapolated cost analysis for the primary standard. At present, it is not possible to reproduce a similar analysis for the secondary standard. Without the amount of emission reductions required to attain, it is not possible to identify the pollution control measures or the associated costs.

S4.3.3 Monitoring complexities

As described in Section S3.2, the current monitoring network was not designed to adequately reflect W126 levels in many areas of the country, especially the rural west. Therefore, we cannot extrapolate the concentrations beyond the currently monitored counties, and we cannot quantify the potential ozone vegetation impacts in many areas of high ecological value, such as National Parks, wilderness areas, or other areas of sensitive national vegetation and ecosystems. We note, however, that even if additional monitors were deployed, it may prove challenging to completely characterize ozone concentrations in some locations that have not traditionally been areas of focus for ozone network deployment.

S4.3.4 Implementation complexities

Other complexities related to implementation have yet to be resolved. For example, EPA has not yet issued guidance for States to recommend boundaries of nonattainment areas for a seasonal secondary ozone standard. The CAA requires that nonattainment areas include areas that violate the standard as well as nearby areas that contribute to a violation. Based on modeled projections of W126 levels in 2020, many of the areas that would exceed the secondary standard without exceeding the primary standard are located in rural areas. Many of those areas lack significant emission sources of ozone precursors within the area, so the cause of the violation is likely due to longer-range transport of ozone and precursors. Analyses of the origin of the contributing emissions in such areas are unavailable. It is unclear what the appropriate boundaries for these projected nonattainment areas would need to be such that the nearby sources that are contributing to the violation are included but the contributing sources that are not "nearby" are excluded. It is important to note that EPA intends to designate nonattainment areas for the 2011 secondary NAAQS for ozone in 2013 based on the recent air quality monitoring data at that time, not on the 2020 projected levels.

In addition, EPA is in the process of developing rules on how States should implement the secondary ozone standard. One issue that must be addressed from a legal stand point is whether planning for nonattainment areas must be done under the more prescriptive subpart 2 requirements of the CAA, which would require classification (as marginal, moderate, serious, etc) or under the less prescriptive subpart 1 of the CAA. For areas classified under subpart 2, there are certain specific control measures that States must adopt. The CAA language is unclear as to whether subpart 2 applies to nonattainment areas under a secondary standard (although it appears to be clear that the maximum statutory attainment dates in the classification table only apply to the "primary" standard). Therefore, it is unclear whether it is appropriate to include the subpart 2 mandatory measures in this analysis. The agency has never faced this issue in the past for ozone, so this will be addressed in the upcoming rules. Since most, if not all, of the areas that might be designated as nonattainment for the secondary standard would also be in nonattainment for the primary standard, it is unclear whether States would need to adopt additional control measures to attain the secondary standard.

S2.9 Pollution Control Strategies

The pollution control measures that might be adopted to attain the secondary standard overlap substantially with the control measures used to attain the primary standard. The air quality analysis showed that most areas that exceed the secondary standard would also exceed the primary standard. If there are areas that would need additional emission reductions to attain the secondary standard, we have included brief descriptions of some available NOx and VOC controls below.

S3.4.1 Point Source Control Measures

For electrical generating units (EGUs), the primary measures for controlling NOx emissions are selective catalytic reduction (SCR), selective noncatalytic reduction (SNCR), and low-NOx burners (LNB). SCR or SNCR can be applied along with a combustion control to further reduce NOx emissions.

Several types of NOx control technologies exist for nonEGU point sources: SCR, SNCR, natural gas reburn (NGR), coal reburn, and LNB. In some cases, LNB accompanied by flue gas recirculation (FGR) is applicable, such as when fuel-borne NOx emissions are expected to be of greater importance than thermal NOx 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 NOx emissions. All of these control measures are available for application on industrial boilers and other non-EGU point sources.

Besides industrial boilers, other nonEGU point source categories that could install controls include petroleum refineries, kraft pulp mills, cement kilns, stationary internal combustion engines, glass manufacturing, combustion turbines, and incinerators. NOx control measures available for petroleum refineries, particularly process heaters at these plants, include LNB, SNCR, FGR, and SCR along with combinations of these technologies. NOx control measures available for kraft pulp mills include those available to industrial boilers, namely LNB, SCR, SNCR, along with water injection (WI). NOx 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 NOx at such plants. LNB, SCR, and SCR + steam injection (SI) are available measures for combustion turbines. Finally, SNCR is an available control technology at incinerators.

VOC controls include a variety of nonEGU point sources as defined in the emissions inventory. The first control is permanent total enclosure (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 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). The second control is petroleum and solvent evaporation applied to printing and publishing sources as well as to surface coating operations.

S3.4.2 Area Source Control Measures

There are three control measures available for NOx emissions from area sources. The first is RACT (reasonably available control technology) to 25 tpy (LNB). This control is the addition of a low NOx burner to reduce NOx emissions. This control applies to industrial oil, natural gas, and coal combustion sources. The second control is water heaters plus LNB space heaters. This control is based on the installation of low-NOx space heaters and water heaters in commercial and institutional sources for the reduction of NOx emissions. The third control is switching to low sulfur fuel for residential home heating. This control is primarily designed to reduce sulfur dioxide, but has a co-benefit of reducing NOx.

An available control to reduce VOC emissions from area sources is CARB Long-Term Limits. This control, which represents controls available in VOC rules promulgated by the California Air Resources Board, applies to commercial solvents and commercial adhesives, and depends on future technological innovation and market incentive methods to achieve emission reductions. The next most frequently applied control was the use of low or no VOC materials for graphic art source categories. The South Coast Air District's SCAQMD Rule 1168 control applies to wood furniture and solvent source categories sets limits for adhesive and sealant VOC content. The OTC solvent cleaning rule control 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. SCAQMD Limits control establishes VOC content limits for metal coatings along with application procedures and equipment requirements. Switching to Emulsified Asphalts control is a generic control measure replacing VOC-containing cutback asphalt with VOC-free emulsified asphalt. The equipment and maintenance control measure applies to oil and natural gas production. The Reformulation—FIP Rule control measure intends to reach the VOC limits by switching to and/or encouraging the use of low-VOC pesticides and better Integrated Pest Management (IPM) practices.

S3.4.3 Mobile Source Control Measures

The NOx control measures available to onroad mobile sources include retrofits of diesel engines, reduction of long duration heavy duty truck idling, continuous inspection and maintenance programs and commuter programs. For nonroad sources, retrofits of diesel engines and engine rebuilds are available. The VOC control measures available to onroad and nonroad mobile sources include the listed controls for NOx plus reduction of Reid vapor pressure in gasoline engines.

S3.4.4 Control Measures beyond the Identified Control Measures Database

Below is a list of controls beyond those in our identified control measures database that are under development and not widely available as yet. There are major uncertainties associated with each of these measures.

- *Enhanced LDAR for Fugitive Leaks: This control measure is a more stringent program to reduce leaks of fugitive VOC emissions from chemical plants and refineries that presumes that an existing LDAR program already is in operation.*
- *Flare Gas Recovery: This control measure is a condenser that can recover 98 percent of the VOC emitted by flares that emit 20 tons per year or more of the pollutant.*
- *Cooling Towers: This control measure is continuous monitoring of VOC from the cooling water return to a level of 10 ppb. This monitoring is accomplished by using a continuous flow monitor at the inlet to each cooling tower. There is not a general estimate of CE for this measure; one is to apply a continuous flow monitor until VOC emissions have reached a level of 1.7 tons/year for a given cooling tower.*[16]
- *Wastewater Drains and Separators: This control measure includes an inspection and maintenance program to reduce VOC emissions from wastewater drains and water seals on drains. This measure is a more stringent version of measures that underlie existing NESHAP requirements for such sources.*
- *Work Practices or Use of Low VOC Coatings: The control measure is either application of work practices (e.g., storing VOC-containing cleaning materials in closed containers, minimizing spills) or using coatings that have much lower VOC content. These measures, which are of relatively low cost compared to other VOC area source controls, can apply to a variety of processes, both for non-EGU point and area sources, in different industries and is defined in the proposed control techniques guidelines (CTG) for paper, film and foil coatings, metal furniture coatings, and large appliance coatings published by the US EPA in July 2007.*[17] *The estimated CE expected to be achieved by either of these control measures is 90 percent.*

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[16] Bay Area Air Quality Management District (BAAQMD). Proposed Revision of Regulation 8, Rule 8: Wastewater Collection Systems. Staff Report, March 17, 2004.

[17] U.S. Environmental Protection Agency. Consumer and Commercial Products: Control Techniques Guidelines in Lieu of Regulations for Paper, Film, and Foil Coatings; Metal Furniture Coatings; and Large Appliance Coatings. 40 CFR 59. July 10, 2007. Available on the Internet at http://www.epa.gov/ttncaaa1/t1/fr_notices/ctg_ccp092807.pdf. It should be noted that this CTG became final in October 2007.

S2.10 Benefits of Reducing Ozone Effects on Vegetation and Ecosystems[18]

Air pollution can affect the environment and affect ecological systems, leading to changes in the ecological community and influencing the diversity, health, and vigor of individual species (U.S. EPA, 2006). Ozone causes discernible injury to a wide array of vegetation (U.S. EPA, 2006; Fox and Mickler, 1996). Sensitivity to ozone is highly variable across plant species, with over 65 plant species identified as "ozone-sensitive", many of which occur in state and national parks and forests.[19] In terms of forest productivity and ecosystem diversity, ozone may be the pollutant with the greatest potential for regional-scale forest impacts (U.S. EPA, 2006). Studies have demonstrated repeatedly that ozone concentrations commonly observed in polluted areas can have substantial impacts on plant function (De Steiguer et al., 1990; Pye, 1988).

When ozone is present in the air, it can enter the leaves of plants, where it can cause significant cellular damage. Like carbon dioxide (CO_2) and other gaseous substances, ozone enters plant tissues primarily through the stomata in leaves in a process called "uptake" (Winner and Atkinson, 1986). Once sufficient levels of ozone (a highly reactive substance), or its reaction products, reaches the interior of plant cells, it can inhibit or damage essential cellular components and functions, including enzyme activities, lipids, and cellular membranes, disrupting the plant's osmotic (i.e., water) balance and energy utilization patterns (U.S. EPA, 2006; Tingey and Taylor, 1982). With fewer resources available, the plant reallocates existing resources away from root growth and storage, above ground growth or yield, and reproductive processes, toward leaf repair and maintenance, leading to reduced growth and/or reproduction. Studies have shown that plants stressed in these ways may exhibit a general loss of vigor, which can lead to secondary impacts that modify plants' responses to other environmental factors. Specifically, plants may become more sensitive to other air pollutants, or more susceptible to disease, pest infestation, harsh weather (e.g., drought, frost) and other environmental stresses, which can all produce a loss in plant vigor in ozone-sensitive species that over time may lead to premature plant death. Furthermore, there is evidence that ozone can interfere with the formation of mycorrhiza, essential symbiotic fungi associated with the roots of most terrestrial plants, by reducing the amount of carbon available for transfer from the host to the symbiont (U.S. EPA, 2006).

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

[18] It is important to note that these vegetation benefits are contingent upon the secondary standard being the controlling standard. In other words, if the primary standard is controlling in all areas, there would not be any additional vegetation benefits beyond those due to the primary standard.
[19] Appendix S3A contains a list of plant species identified as "ozone-sensitive".

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, 2006; Winner, 1994). After injuries have occurred, plants may be capable of repairing the damage to a limited extent (U.S. EPA, 2006). 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, 2006). 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, 2006, 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, 2006).

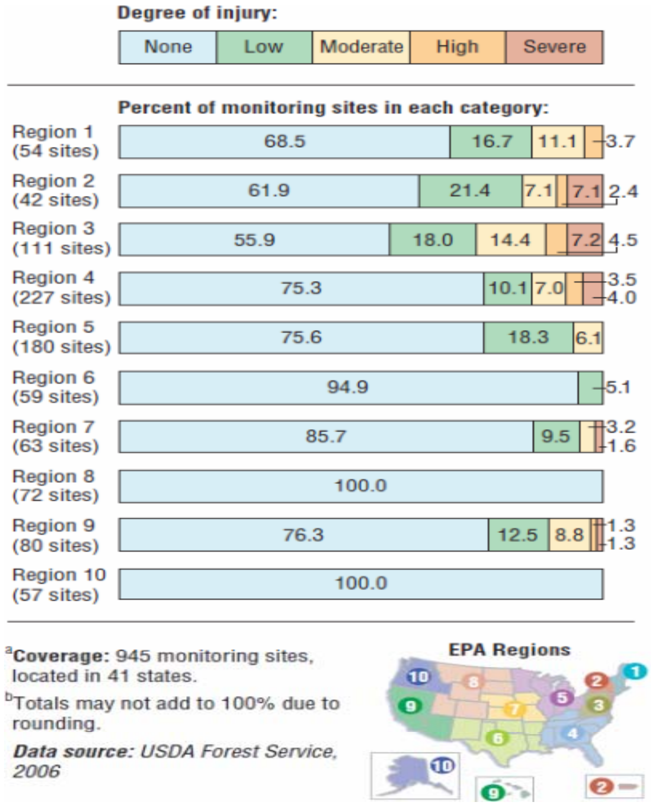
a. Ozone Effects on Forests

Ozone has been shown in numerous studies to have a strong, negative effect on the health of a variety of commercial and ecologically important forest tree species throughout the U.S. (U.S. EPA, 2007b). In the U.S., this data comes from the U.S. Department of Agriculture (USDA) Forest Service Forest Inventory and Analysis (FIA) program. As part of its Phase 3 program (formerly known as Forest Health Monitoring), FIA looks for visible foliar injury of ozone-sensitive forest plant species at each ground monitoring site across the country (excluding woodlots and urban trees) that meets certain minimum criteria. Because ozone injury is cumulative over the course of the growing season, examinations are conducted in July and August, when ozone concentrations and associated injury are typically highest.

Monitoring of ozone injury to plants by the U.S. Forest Service has expanded over the last 15 years from monitoring sites in 10 states in 1994 to nearly 1,000 monitoring sites in 41 states in 2002. Since 2002, the monitoring program has further expanded to 1,130 monitoring sites in 45 states. Figure S3-6 shows the results of this monitoring program for the year 2002 broken down by U.S. EPA Regions.[20] Figure S3-7 identifies the counties that were included in Figure S3-6, and provides the county-level data regarding the presence or absence of ozone-related injury. As shown in Figure S3-7, large geographic areas of EPA Regions 6, 8, and 10 were not included in the assessment. Ozone damage to forest plants is classified using a subjective five-category biosite index based on expert opinion, but designed to be equivalent from site to site. Ranges of biosite values translate to no injury, low or moderate foliar injury (visible foliar injury to highly sensitive or moderately sensitive plants, respectively), and high or severe foliar injury, which would be expected to result in tree-level or ecosystem-level responses, respectively (U.S. EPA, 2006; Coulston, 2004). The highest percentages of observed high and severe foliar injury, which are most likely to be associated with tree or ecosystem-level responses, are primarily found in the Mid-Atlantic and Southeast regions. While the assessment showed considerable regional variation in ozone injury, this assessment targeted different ozone-sensitive species in different parts of the country with varying ozone sensitivity, which contributes to the apparent regional differences. It is important to note that ozone can have other, more significant impacts on forest plants (e.g. reduced biomass growth in trees) prior to showing signs of visible foliar injury (U.S. EPA, 2006).

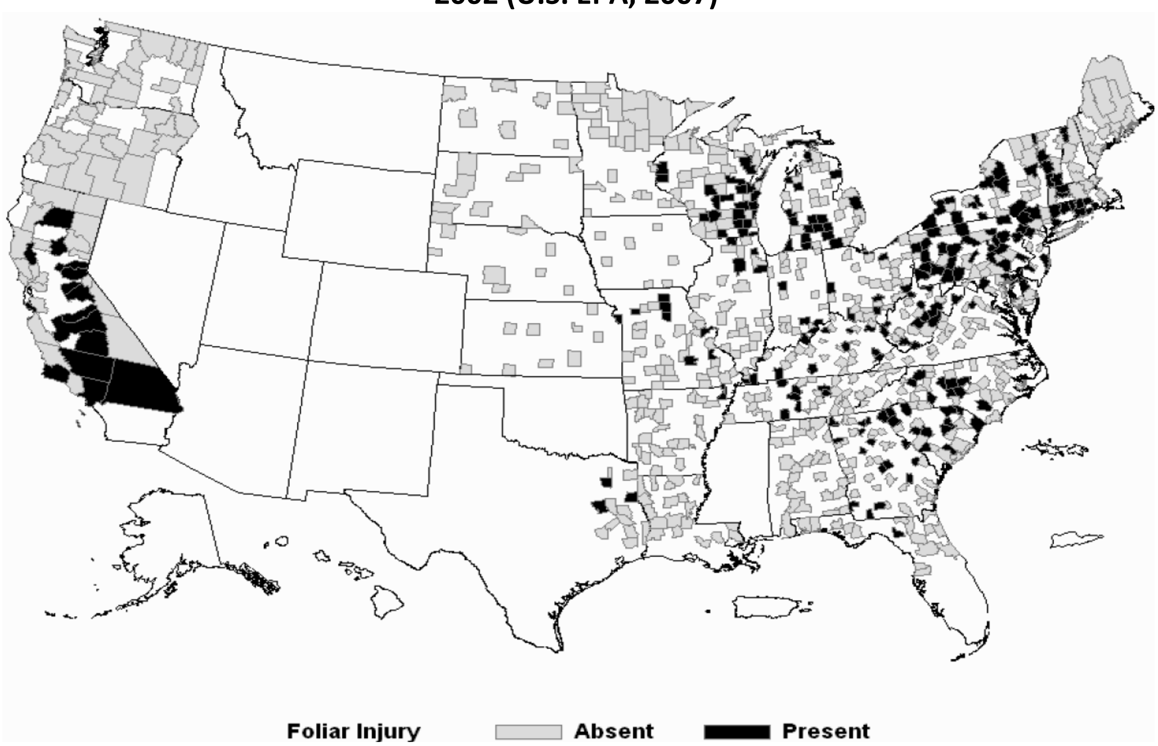
[20] The data are based on averages of all observations collected in 2002, which is the last year for which data are publicly available. For more information, please consult EPA’s 2008 Report on the Environment (U.S. EPA, 2008d).

Figure S3-6: Visible Foliar Injury to Forest Plants from Ozone in U.S. by EPA Regions, 2002[a, b, c]



[c] ***Degree of Injury:*** These categories reflect a subjective index based on expert opinion. Ozone can have other, more significant impacts on forest plants (e.g. reduced biomass growth in trees) prior to showing signs of visible foliar injury.

Figure S3-7: Presence and Absence of Visible Foliar Injury, as measured by U.S. Forest Service, 2002 (U.S. EPA, 2007)

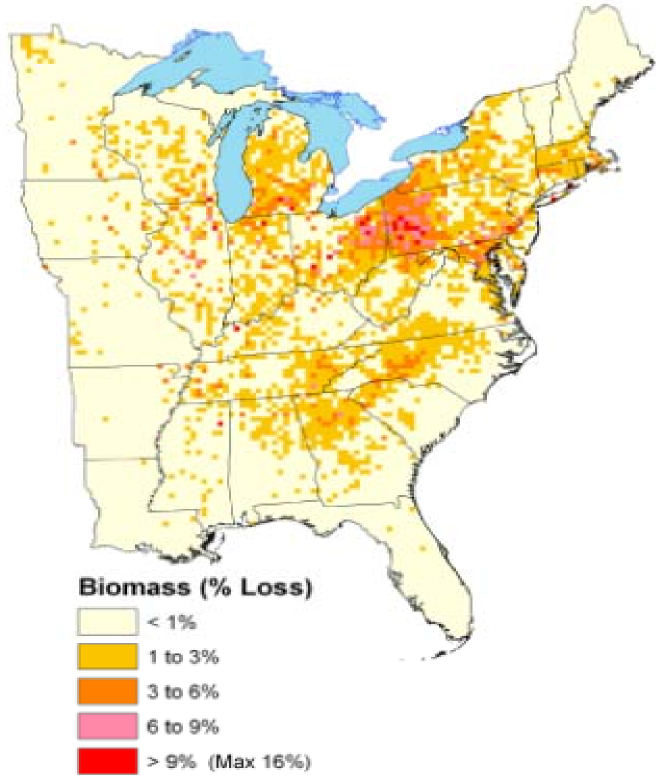


Assessing the impact of ground-level ozone on forests in the U.S involves understanding the risks to sensitive tree species from ambient ozone concentrations and accounting for the prevalence of those species within the forest. As a way to quantify the risks to particular plants from ground-level ozone, scientists have developed ozone-exposure/tree-response functions by exposing tree seedlings to different ozone levels and measuring reductions in growth as "biomass loss." Typically, seedlings are used because they are easy to manipulate and measure their growth loss from ozone pollution. The mechanisms of susceptibility to ozone within the leaves of seedlings and mature trees are identical, and the decreases predicted using the seedlings should be related to the decrease in overall plant fitness for mature trees, but the magnitude of the effect may be higher or lower depending on the tree species (Chappelka and Samuelson, 1998). In areas where certain ozone-sensitive species dominate the forest community, the biomass loss from ozone can be significant. Experts have identified 2% annual biomass loss as a level of concern, which would cause long term ecological harm as the short-term negative effects on seedlings compound to affect long-term forest health (Heck and Cowling, 1997).

Ozone damage to the plants including the trees and understory in a forest can affect the ability of the forest to sustain suitable habitat for associated species particularly threatened and endangered species that have existence value – a nonuse ecosystem service - for the public. Similarly, damage to trees and the loss of biomass can affect the forest's provisioning services in the form of timber for various commercial uses. In addition, ozone can cause discoloration of leaves and more rapid senescence (early shedding of leaves), which could negatively affect fall-color tourism because the fall foliage would be less available or less attractive. Beyond the aesthetic damage to fall color vistas, forests provide the public with many other recreational and educational services that may be impacted by reduced forest health including hiking, wildlife viewing (including bird watching), camping, picnicking, and hunting. Another potential effect of biomass loss in forests is the subsequent loss of climate regulation service in the form of reduced ability to sequester carbon and alteration of hydrologic cycles.

Some of the common tree species in the United States that are sensitive to ozone are black cherry (*Prunus serotina*), tulip-poplar (*Liriodendron tulipifera)*, and eastern white pine (*Pinus strobus)*. Ozone-exposure/tree-response functions have been developed for each of these tree species, as well as for aspen (*Populus tremuliodes*), and ponderosa pine (*Pinus ponderosa*) (U.S. EPA, 2007b). Other common tree species, such as oak (*Quercus* spp*.)* and hickory (*Carya* spp.), have not been studied for ozone sensitivity. Consequently, with knowledge of the range of sensitive species and the level of ozone at particular locations, it is possible to estimate the percentage of biomass loss for each species across their range. As shown in Figure S3-8, current ambient levels of ozone are associated with significant biomass loss across large geographic areas (U.S. EPA, 2009b). However, this information is unavailable for a future analysis year or incremental to a specified control strategy.

Figure S3-8: Estimated Biomass Loss for Black Cherry, Yellow Poplar, Sugar Maple, Eastern White Pine, Virginia Pine, Red Maple, and Quaking Aspen due to Ozone Exposure, 2006-2008 (U.S. EPA, 2009b)*



*This map does not include other tree species that are potentially sensitive to ozone.

According to the Staff Paper, the scientific consensus is that there is no threshold for exposures that cause effects on vegetation (Heck and Cowling 1997, U.S. EPA 2006). It is important to note that biomass loss in tree seedlings is not intended to be a surrogate for expected biomass loss in mature trees of the same species. Studies indicate that mature trees can be more or less sensitive than seedlings depending on the species. Sources of uncertainty include the ozone-exposure/plant-response functions, the tree abundance, and other factors (e.g., soil moisture). Although these factors were not considered in this assessment, they can affect ozone damage (Chappelka and Samuelson, 1998). EPA concluded in the Ozone Criteria Document that significant interactions with acid rain are unlikely (U.S. EPA, 2006).

Since the proposal, we have expanded the analysis of qualitative assessment of ozone impacts on forests. In this analysis, we include quantitative estimates of the tree biomass loss avoided by the primary and secondary standards across the range of the species. In this analysis, we estimate the biomass loss avoided for 6 tree species (i.e., ponderosa pine, red alder, black cherry, quaking aspen, yellow (tulip) poplar, and Virginia pine) in the continental U.S. These species were selected because they met two criteria: (1) the Staff Paper provided a W126-derived exposure-response function, and (2) the Staff Paper listed the species as an ozone-sensitive plant species (U.S. EPA, 2007b). To estimate the biomass loss avoided, we simply used the projected W126 design values in the exposure-response functions and subtracted the difference in biomass loss between the baseline and hypothetical RIA control scenarios. For mapping purposes, we assume that the W126 design value is representative of the W126 levels in the county. We then overlaid a map of the species range to focus on those areas where the species is likely to grow.[21] Though each map shows the geographical range for a species, it does not presume that an individual of that species would be found at every point within its range. Due to uncertainties in extrapolating W126 values, we have confined this analysis to the currently monitored counties. To calculated biomass loss associated with the secondary standard, we simply rolled back the W126 value in only the violating county to just attain the selected secondary standard.

Table S3-6 shows the exposure-response functions used to generate the tree maps. A full list of ozone-sensitive plant species from the Staff Paper is provided in Appendix S3A of this RIA. Figures S3-9 through S3-20 map the biomass loss avoided for each of the selected tree species by the hypothetical RIA controls for the primary standard and by the rollback to the secondary standard. It is important to note that the modeled hypothetical RIA controls did not fully attain the primary standard of 0.070 ppm, so this map underestimates the biomass loss avoided in several areas, especially Southern California, Houston, Eastern Lake Michigan, and the Northeast corridor. It is also important to note that the control strategy is likely to reduce W126 levels over a broader geographic area than just the violating county, so this map underestimates regional biomass loss avoided. Because we deliberately chose assumptions that underestimate tree biomass loss avoided, we have minimized potential uncertainty, and we have high confidence that the benefits are at least as high as those shown in the maps. Due to time and resource limitations, we were unable to monetize the benefits associated with avoiding tree biomass loss in this analysis. As mentioned above, these tree species provide several valuable ecosystem services, including timber, recreational/tourism, existence value, and climate and hydrologic regulation.

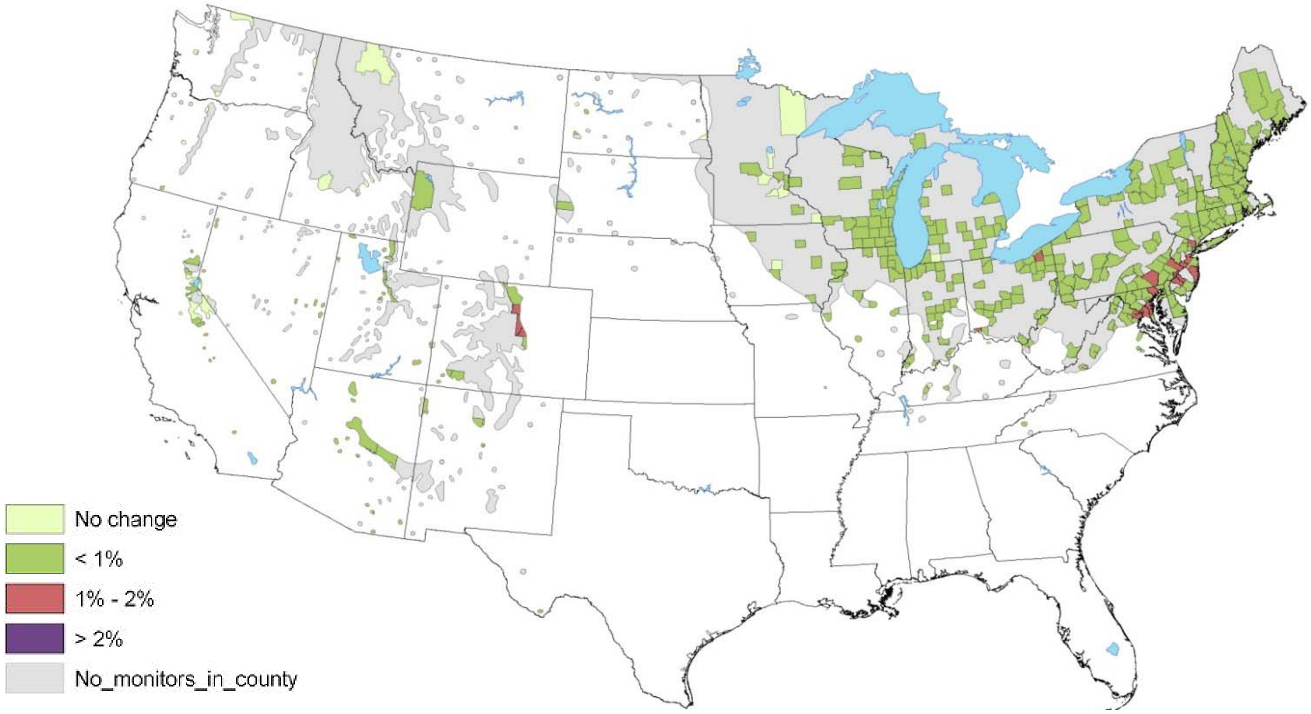
Table S3-6: Biomass Loss Functions for Trees

Species	Exposure-Response Function
Ponderosa Pine	1-exp(-1*(W126/159.63)^1.190)
Red Alder	1-exp(-1*(W126/179.06)^1.2377)
Black Cherry	1-exp(-1*(W126/38.92)^0.9921)
Quaking Aspen	1-exp(-1*(W126/109.81)^1.2198)
Virginia Pine	1-exp(-1*('W126/1714.64)^1)
Yellow (Tulip) Poplar	1-exp(-1*(W126/51.38)^2.0889)

*All functions are from Table 7F-3 of the Staff Paper (U.S. EPA, 2007b). Each function represents the median composite function for tree seedlings.

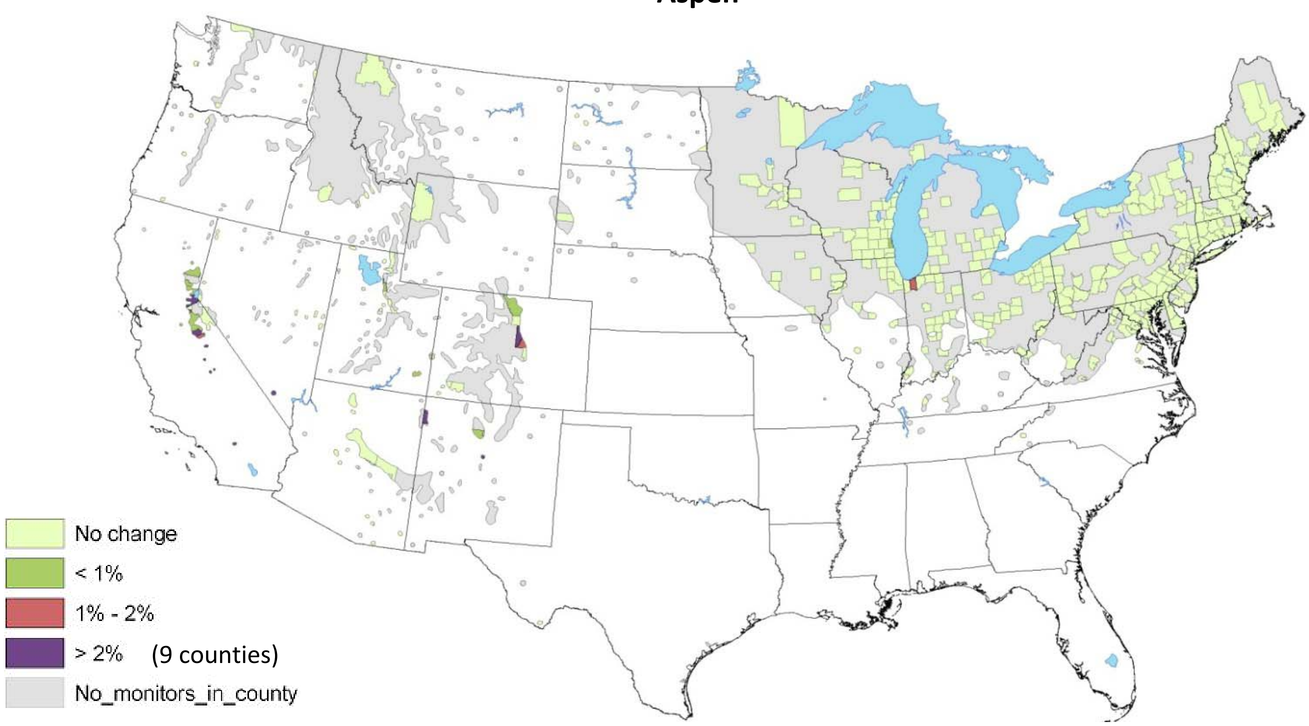
1.1 [21] The species geographic ranges are identical to those in the Staff Paper (U.S. EPA, 2007b)., and are from "Atlas of United States Trees" by Elbert L. Little, Jr, available on the Internet at http://esp.cr.usgs.gov/data/atlas/little/.

Figure S3-9: Biomass Loss Avoided by Primary Standard in 2020 for Quaking Aspen*



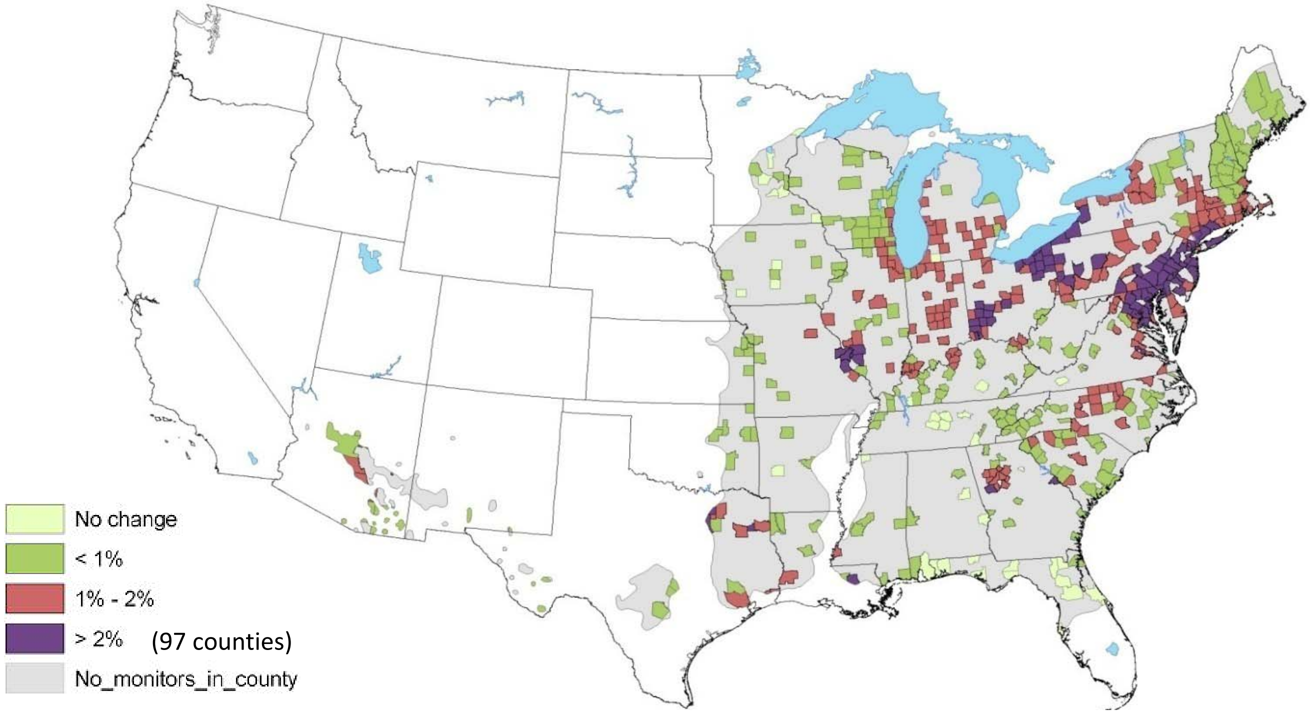
* It is important to note that the modeled hypothetical RIA controls did not fully attain the primary standard of 0.070 ppm, so this map underestimates the biomass loss avoided in several areas, especially Southern California, Houston, Eastern Lake Michigan, and the Northeast corridor. Experts have identified 2% annual biomass loss as a level of concern, which would cause long term ecological harm as the short-term negative effects on seedlings compound to affect long-term forest health. Though each map shows the geographical range for a species, it does not presume that an individual of that species would be found at every point within its range.

Figure S3-10: Additional Biomass Loss Avoided by Secondary Standard of 13 ppm-hrs in 2020 for Quaking Aspen*



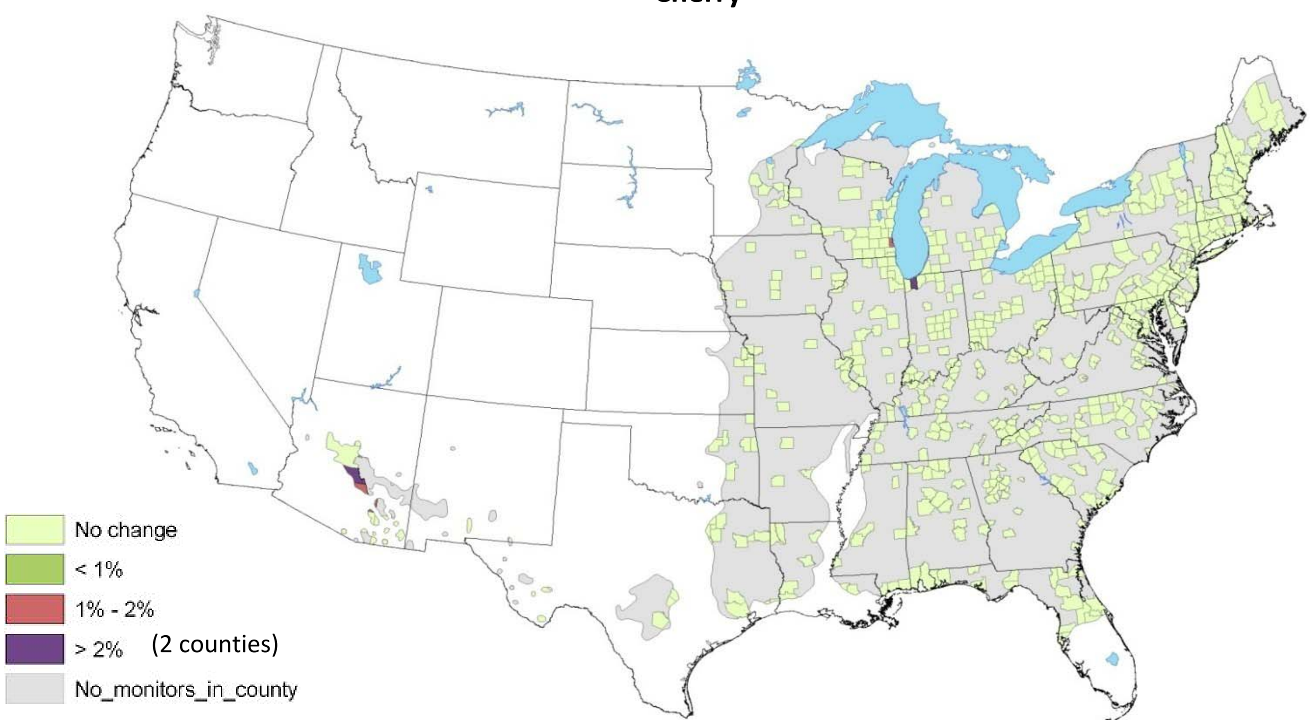
* It is important to note that the control strategy is likely to reduce W126 levels over a broader geographic area than just the violating county, so this map underestimates regional biomass loss avoided. Experts have identified 2% annual biomass loss as a level of concern, which would cause long term ecological harm as the short-term negative effects on seedlings compound to affect long-term forest health. Though each map shows the geographical range for a species, it does not presume that an individual of that species would be found at every point within its range.

Figure S3-11: Biomass Loss Avoided by Primary Standard in 2020 for Black Cherry*



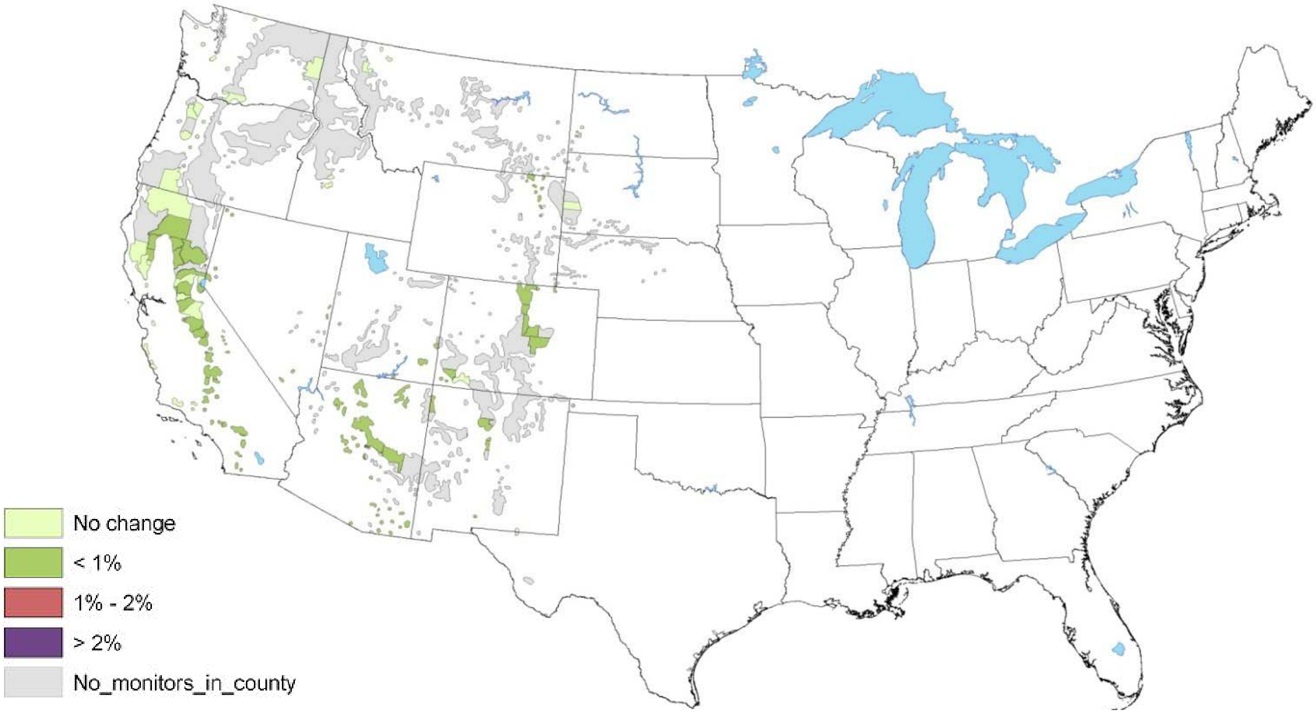
* It is important to note that the modeled hypothetical RIA controls did not fully attain the primary standard of 0.070 ppm, so this map underestimates the biomass loss avoided in several areas, especially Southern California, Houston, Eastern Lake Michigan, and the Northeast corridor. Experts have identified 2% annual biomass loss as a level of concern, which would cause long term ecological harm as the short-term negative effects on seedlings compound to affect long-term forest health. Though each map shows the geographical range for a species, it does not presume that an individual of that species would be found at every point within its range.

Figure S3-12: Additional Biomass Loss Avoided by Secondary Standard of 13 ppm-hrs in 2020 for Black Cherry*



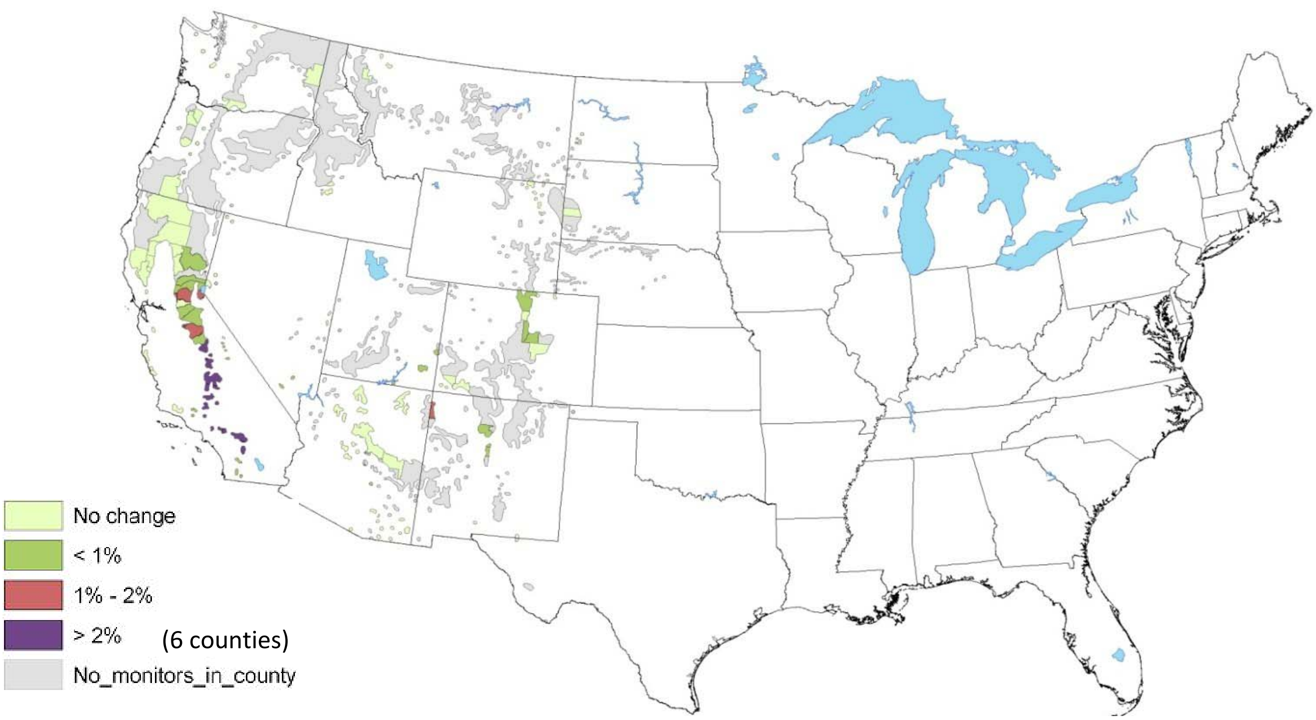
* It is important to note that the control strategy is likely to reduce W126 levels over a broader geographic area than just the violating county, so this map underestimates regional biomass loss avoided. Experts have identified 2% annual biomass loss as a level of concern, which would cause long term ecological harm as the short-term negative effects on seedlings compound to affect long-term forest health. Though each map shows the geographical range for a species, it does not presume that an individual of that species would be found at every point within its range.

Figure S3-13: Biomass Loss Avoided by Primary Standard in 2020 for Ponderosa Pine*



* It is important to note that the modeled hypothetical RIA controls did not fully attain the primary standard of 0.070 ppm, so this map underestimates the biomass loss avoided in several areas, especially Southern California, Houston, Eastern Lake Michigan, and the Northeast corridor. Experts have identified 2% annual biomass loss as a level of concern, which would cause long term ecological harm as the short-term negative effects on seedlings compound to affect long-term forest health. Though each map shows the geographical range for a species, it does not presume that an individual of that species would be found at every point within its range.

Figure S3-14: Additional Biomass Loss Avoided by Secondary Standard of 13 ppm-hrs in 2020 for Ponderosa Pine*



* It is important to note that the control strategy is likely to reduce W126 levels over a broader geographic area than just the violating county, so this map underestimates regional biomass loss avoided. Experts have identified 2% annual biomass loss as a level of concern, which would cause long term ecological harm as the short-term negative effects on seedlings compound to affect long-term forest health. Though each map shows the geographical range for a species, it does not presume that an individual of that species would be found at every point within its range.

Figure S3-15: Biomass Loss Avoided by Primary Standard in 2020 for Red Alder*

* It is important to note that the modeled hypothetical RIA controls did not fully attain the primary standard of 0.070 ppm, so this map underestimates the biomass loss avoided in several areas, especially Southern California, Houston, Eastern Lake Michigan, and the Northeast corridor. Experts have identified 2% annual biomass loss as a level of concern, which would cause long term ecological harm as the short-term negative effects on seedlings compound to affect long-term forest health. Though each map shows the geographical range for a species, it does not presume that an individual of that species would be found at every point within its range.

Figure S3-16: Additional Biomass Loss Avoided by Secondary Standard of 13 ppm-hrs in 2020 for Red Alder*

* It is important to note that the control strategy is likely to reduce W126 levels over a broader geographic area than just the violating county, so this map underestimates regional biomass loss avoided. Experts have identified 2% annual biomass loss as a level of concern, which would cause long term ecological harm as the short-term negative effects on seedlings compound to affect long-term forest health. Though each map shows the geographical range for a species, it does not presume that an individual of that species would be found at every point within its range.

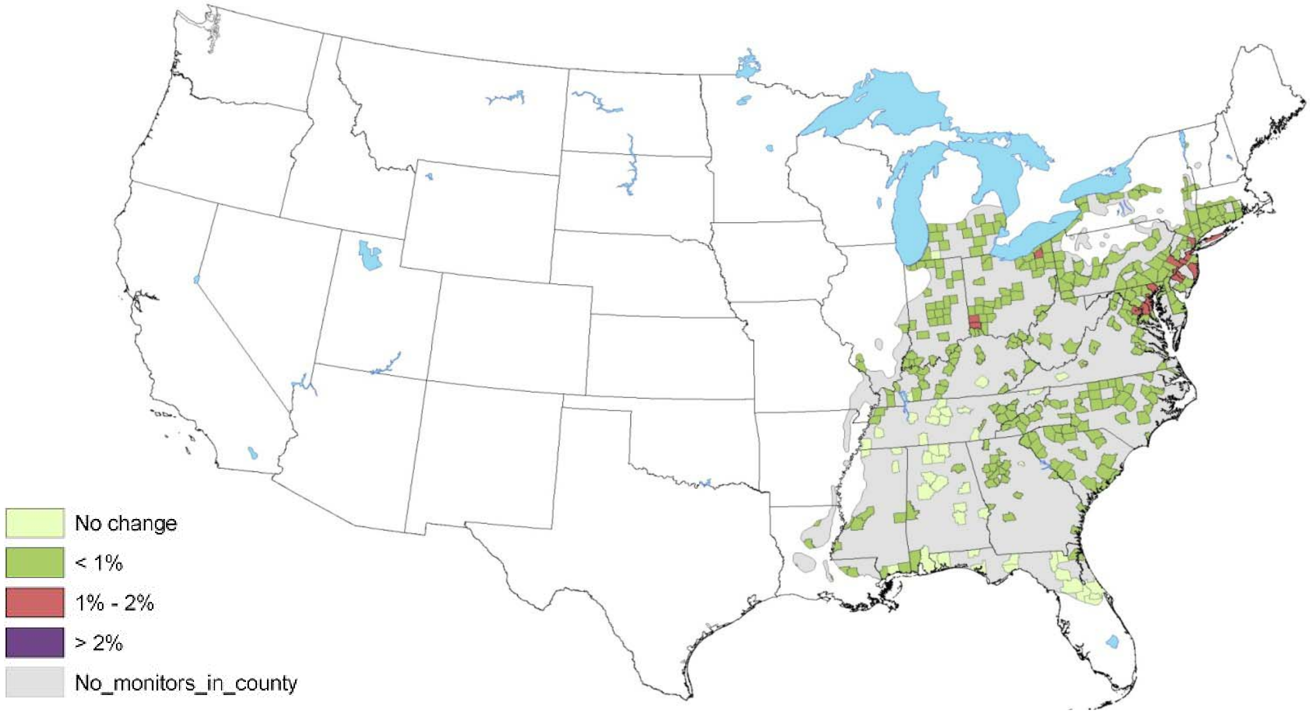
Figure S3-17: Biomass Loss Avoided by Primary Standard in 2020 for Virginia Pine*

* It is important to note that the modeled hypothetical RIA controls did not fully attain the primary standard of 0.070 ppm, so this map underestimates the biomass loss avoided in several areas, especially Southern California, Houston, Eastern Lake Michigan, and the Northeast corridor. Experts have identified 2% annual biomass loss as a level of concern, which would cause long term ecological harm as the short-term negative effects on seedlings compound to affect long-term forest health. Though each map shows the geographical range for a species, it does not presume that an individual of that species would be found at every point within its range.

Figure S3-18: Additional Biomass Loss Avoided by Secondary Standard of 13 ppm-hrs in 2020 for Virginia Pine*

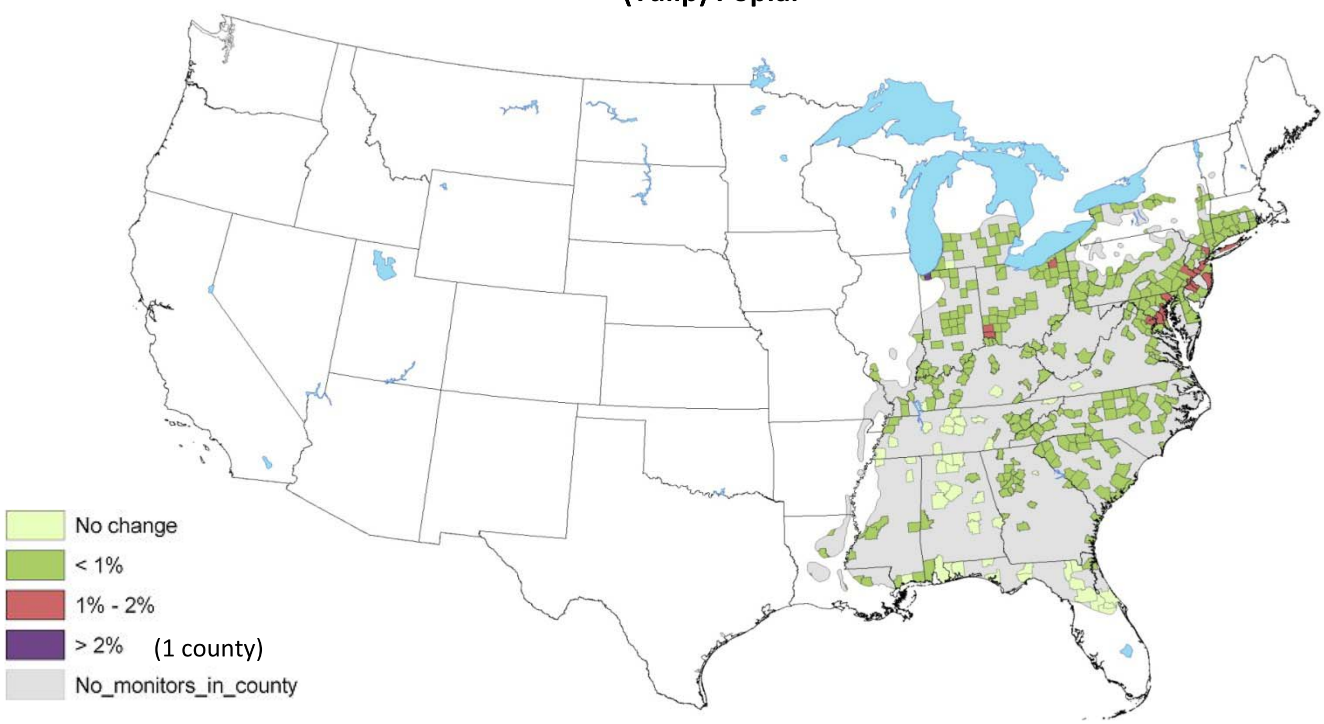
* It is important to note that the control strategy is likely to reduce W126 levels over a broader geographic area than just the violating county, so this map underestimates regional biomass loss avoided. Experts have identified 2% annual biomass loss as a level of concern, which would cause long term ecological harm as the short-term negative effects on seedlings compound to affect long-term forest health. Though each map shows the geographical range for a species, it does not presume that an individual of that species would be found at every point within its range.

Figure S3-19: Biomass Loss Avoided by Primary Standard in 2020 for Yellow (Tulip) Poplar*



* It is important to note that the modeled hypothetical RIA controls did not fully attain the primary standard of 0.070 ppm, so this map underestimates the biomass loss avoided in several areas, especially Southern California, Houston, Eastern Lake Michigan, and the Northeast corridor. Experts have identified 2% annual biomass loss as a level of concern, which would cause long term ecological harm as the short-term negative effects on seedlings compound to affect long-term forest health. Though each map shows the geographical range for a species, it does not presume that an individual of that species would be found at every point within its range.

Figure S3-20: Additional Biomass Loss Avoided by Secondary Standard of 13 ppm-hrs in 2020 for Yellow (Tulip) Poplar*



* It is important to note that the control strategy is likely to reduce W126 levels over a broader geographic area than just the violating county, so this map underestimates regional biomass loss avoided. Experts have identified 2% annual biomass loss as a level of concern, which would cause long term ecological harm as the short-term negative effects on seedlings compound to affect long-term forest health. Though each map shows the geographical range for a species, it does not presume that an individual of that species would be found at every point within its range.

b. Ozone Effects on Crops

Laboratory and field experiments have shown reductions in yields for agronomic crops exposed to ozone, including vegetables (e.g., lettuce) and field crops (e.g., cotton and wheat). Damage to crops from ozone exposures includes yield losses (i.e., in terms of weight, number, or size of the plant part that is harvested), as well as changes in crop quality (i.e., physical appearance, chemical composition, or the ability to withstand storage) (U.S. EPA, 2007b). The most extensive field experiments, conducted under the National Crop Loss Assessment Network (NCLAN) examined 15 species and numerous cultivars. The NCLAN results show that "several economically important crop species are sensitive to ozone levels typical of those found in the United States" (U.S. EPA, 2006). In addition, economic studies have shown reduced economic benefits as a result of predicted reductions in crop yields, directly affecting the amount and quality of the provisioning service provided by the crops in question, associated with observed ozone levels (Kopp et al, 1985; Adams et al., 1986; Adams et al., 1989). In addition, visible foliar injury by itself can reduce the market value of certain leafy crops (such as spinach, lettuce). According to the Staff Paper, there has been no evidence that crops are becoming more tolerant of ozone (U.S. EPA, 2007b). Using the Agriculture Simulation Model (AGSIM) (Taylor, 1994) to calculate the agricultural benefits of reductions in ozone exposure, U.S. EPA estimated that attaining a W126 standard of 13 ppm-hr would produce monetized benefits of approximately $400 million to $620 million (inflated to 2006 dollars) (U.S. EPA, 2007b).

According to the Staff Paper, the scientific consensus is that there is no threshold for exposures that cause effects on vegetation (Heck and Cowling 1997, U.S. EPA 2006). Sources of uncertainty include the ozone-exposure/plant-response functions, soil moisture/irrigation, fertilization, and other factors. Agricultural systems are heavily managed and vulnerable to adverse impacts from a variety of other factors (e.g., weather, insects, disease), which can overshadow the ozone-related effects. Additional research is needed to better understand the nature and significance of interactive effects of ozone with other plant stressors (U.S. EPA, 2007b).

Since the proposal, we have expanded the analysis of qualitative assessment of ozone impacts on crops. In this analysis, we include quantitative estimates of the crop yield loss avoided by the primary and secondary standards across the crop production areas for 3 crops (i.e., cotton, soybean, and winter wheat) in the continental U.S. These crops were selected because they met three criteria: (1) the Staff Paper provided a W126-derived exposure-response function, (2) the Staff Paper listed the crops as an ozone-sensitive plant species (U.S. EPA, 2007b), and (3) the Staff paper included maps of the crop production areas. To estimate the biomass loss avoided, we simply used the projected W126 design values in the exposure-response functions and subtracted the difference in yield loss between the two scenarios. For mapping purposes, we assume that the W126 design value is representative of the W126 levels in the county. We then overlaid a map of the crop production area to focus on those areas where the species is likely to be grown.[22] Due to uncertainties in extrapolating W126 values, we have confined this analysis to the currently monitored counties. To calculated biomass loss associated with the secondary standard, we simply rolled back the W126 value in only the violating county to just attain the selected secondary standard.

Table S3-6 shows the exposure-response functions used to generate the crop maps. A full list of ozone-sensitive crops from the Staff Paper is provided in Appendix S3A of this RIA. Figures S3-21 through S3-26 map the crop yield loss avoided for each of the selected crops by hypothetical RIA controls for the primary standard and by the rollback to the secondary standard. It is important to note that the modeled hypothetical RIA controls did not fully attain the primary standard of 0.070 ppm, so this map underestimates the crop yield loss avoided in several areas, especially Southern California, Houston, Eastern Lake Michigan, and the Northeast corridor. It is also important to note that the control strategy is likely to reduce W126 levels over a broader geographic area than just the violating county, so this map underestimates regional crop yield loss. Because we deliberately chose assumptions that underestimate crop yield loss, we have minimized potential uncertainty, and we have high confidence that the benefits are at least as high as those shown in the maps. Due to time and resource limitations, we were unable to monetize the benefits associated with avoiding crop yield loss in this analysis. As mentioned above, these crop species provide several valuable ecosystem services, including especially food and fiber production.

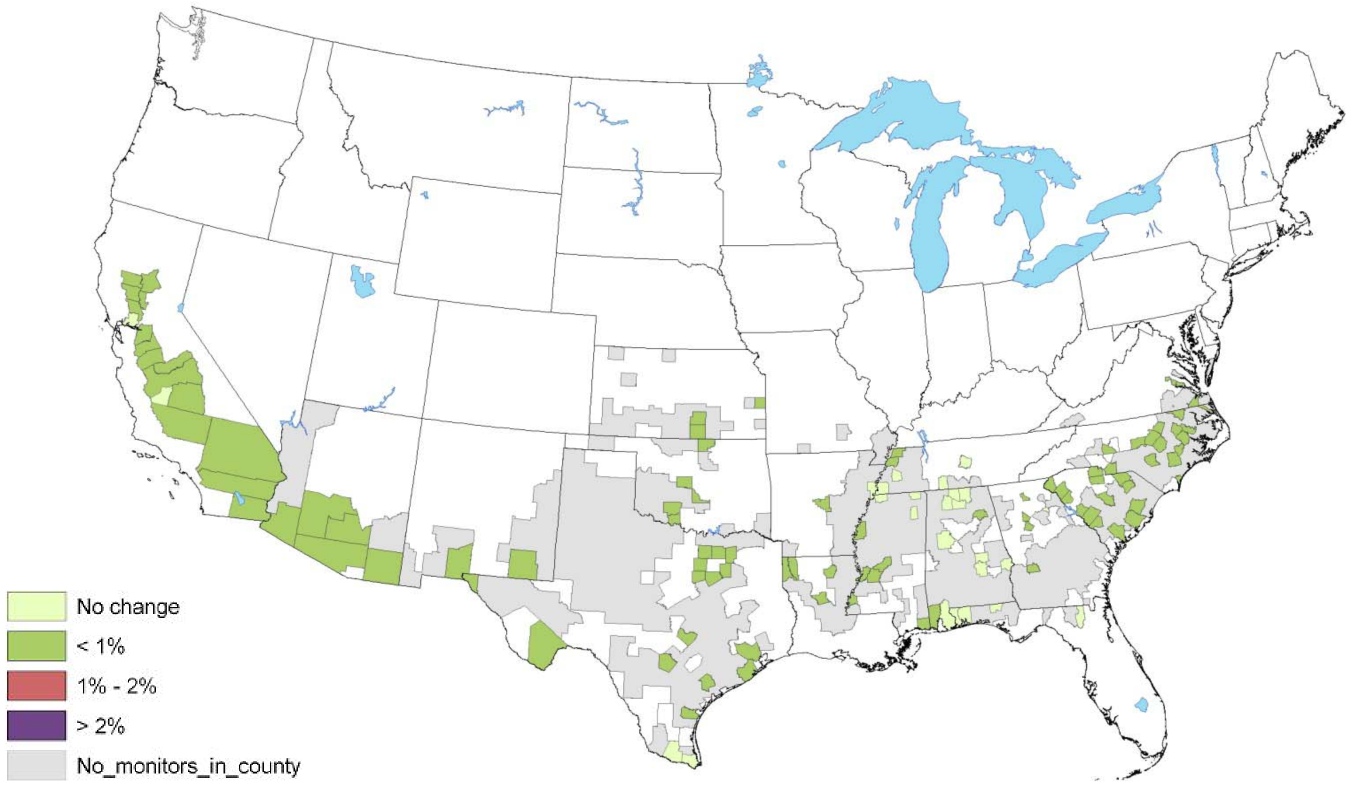
Table S3-6: Yield Loss Functions for Selected Crops

Crop	Exposure-Response Function
Cotton	1-exp(-1*(W126/96.1)^1.482)
Soybean	1-exp(-1*(W126/110.2)^1.359)
Winter Wheat	1-exp(-1*(W126/53.4)^2.367)

*All functions are from Table 7F-1 of the Staff Paper (U.S. EPA, 2007b). Each function represents the median function.

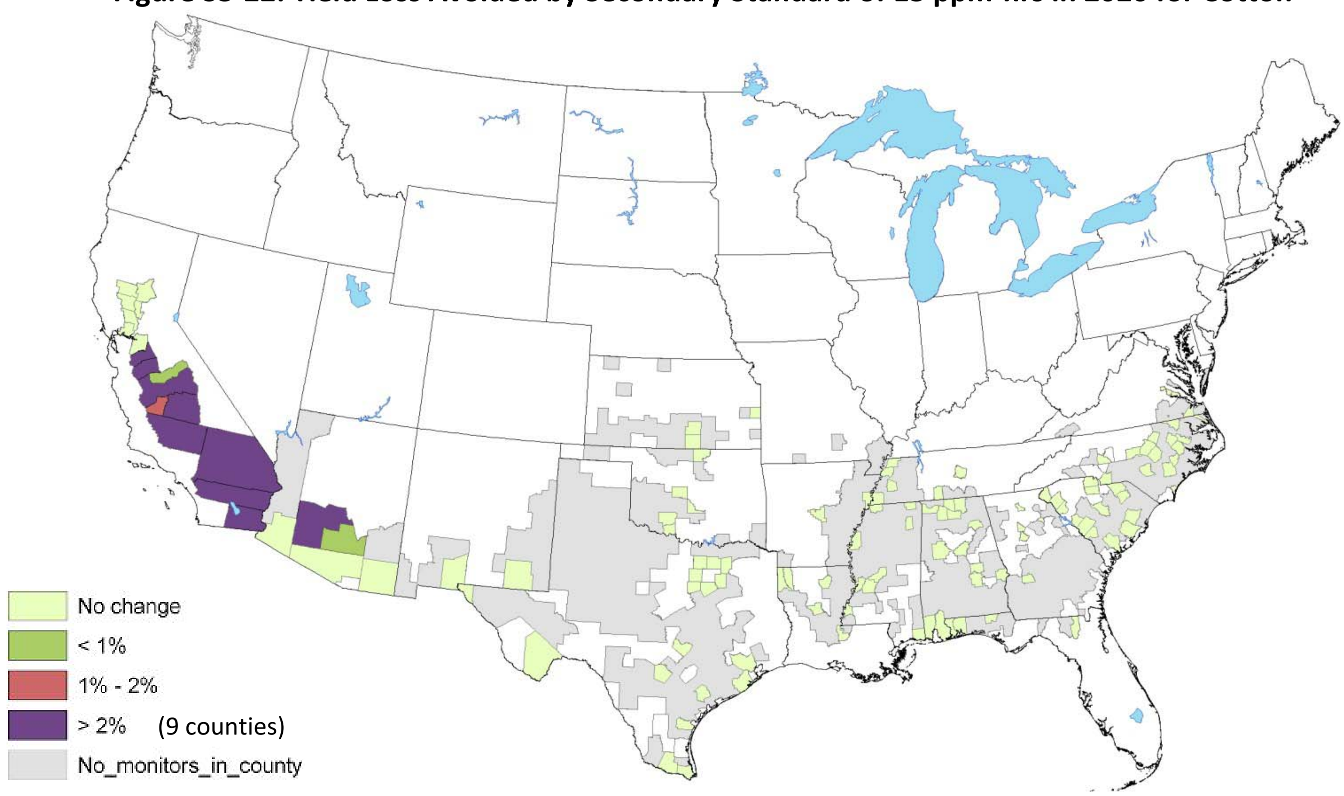
[22] Crop production areas are identical to those in the Staff Paper (U.S. EPA, 2007b) and were derived from the 2002 Census of Agriculture and from NASS 2001 County Crop Data. For more details on the crop production areas, please consult U.S. EPA (2007c).

Figure S3-21: Yield Loss Avoided by Primary Standard in 2020 for Cotton*



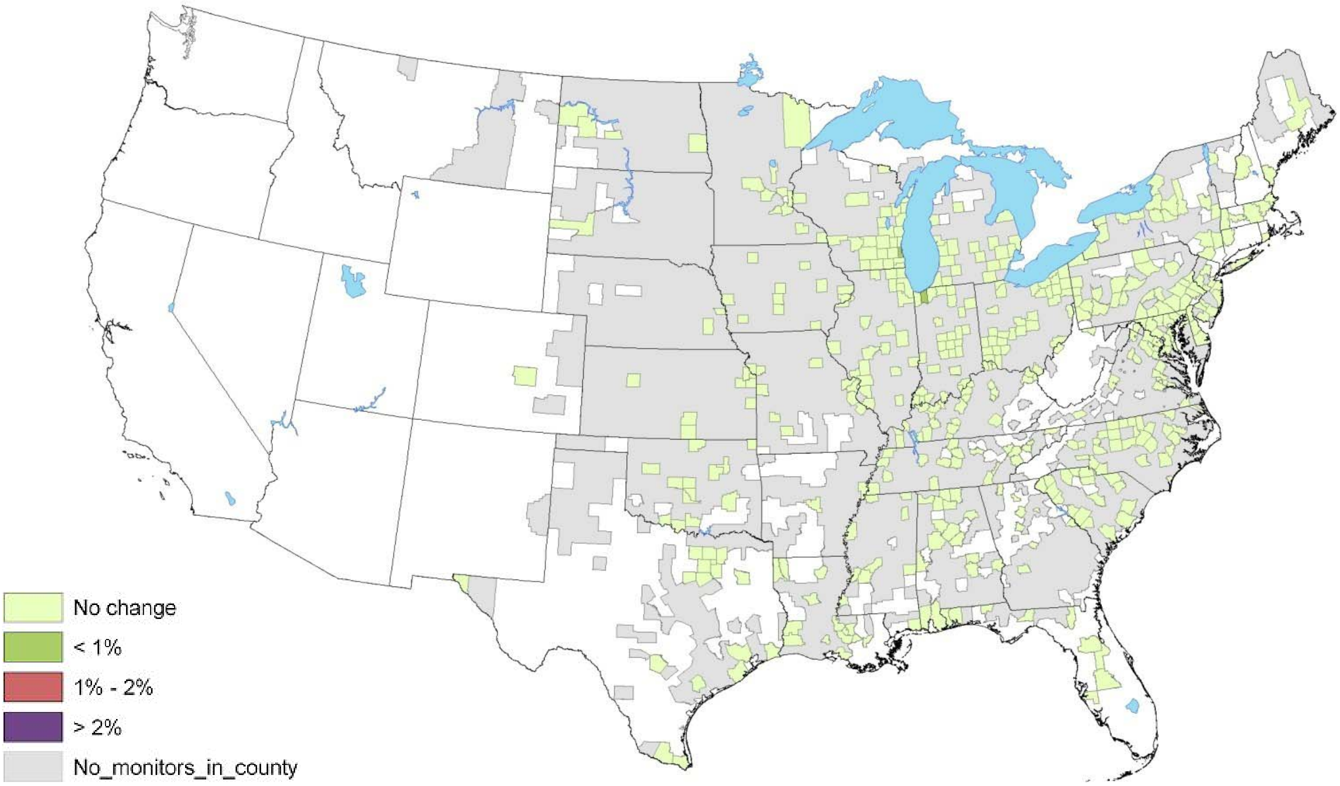
*It is important to note that the modeled hypothetical RIA controls did not fully attain the primary standard of 0.070 ppm, so this map underestimates the yield loss avoided in several areas, especially Southern California, Houston, Eastern Lake Michigan, and the Northeast Corridor.

Figure S3-22: Yield Loss Avoided by Secondary Standard of 13 ppm-hrs in 2020 for Cotton*



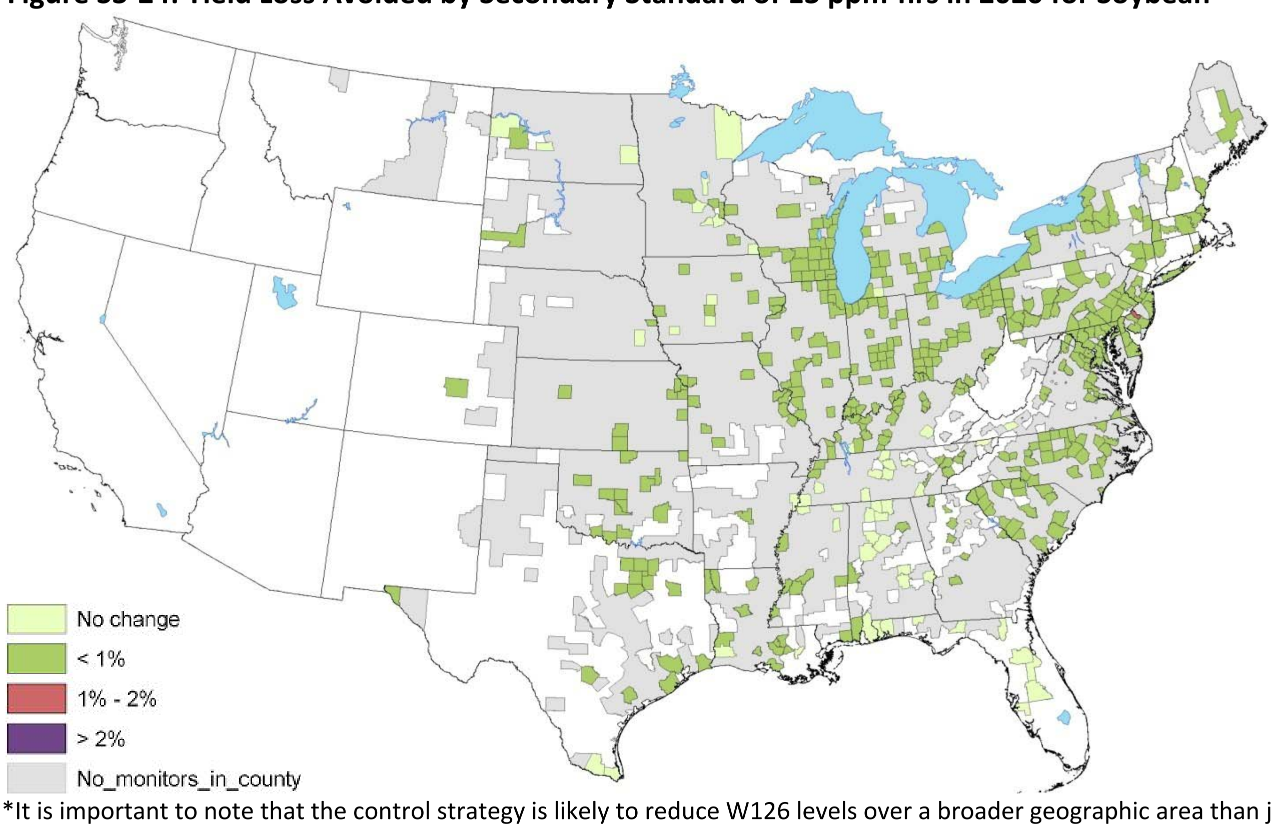
*It is important to note that the control strategy is likely to reduce W126 levels over a broader geographic area than just the violating county, so this map underestimates the regional yield loss avoided.

Figure S3-23: Yield Loss Avoided by Primary Standard in 2020 for Soybean*



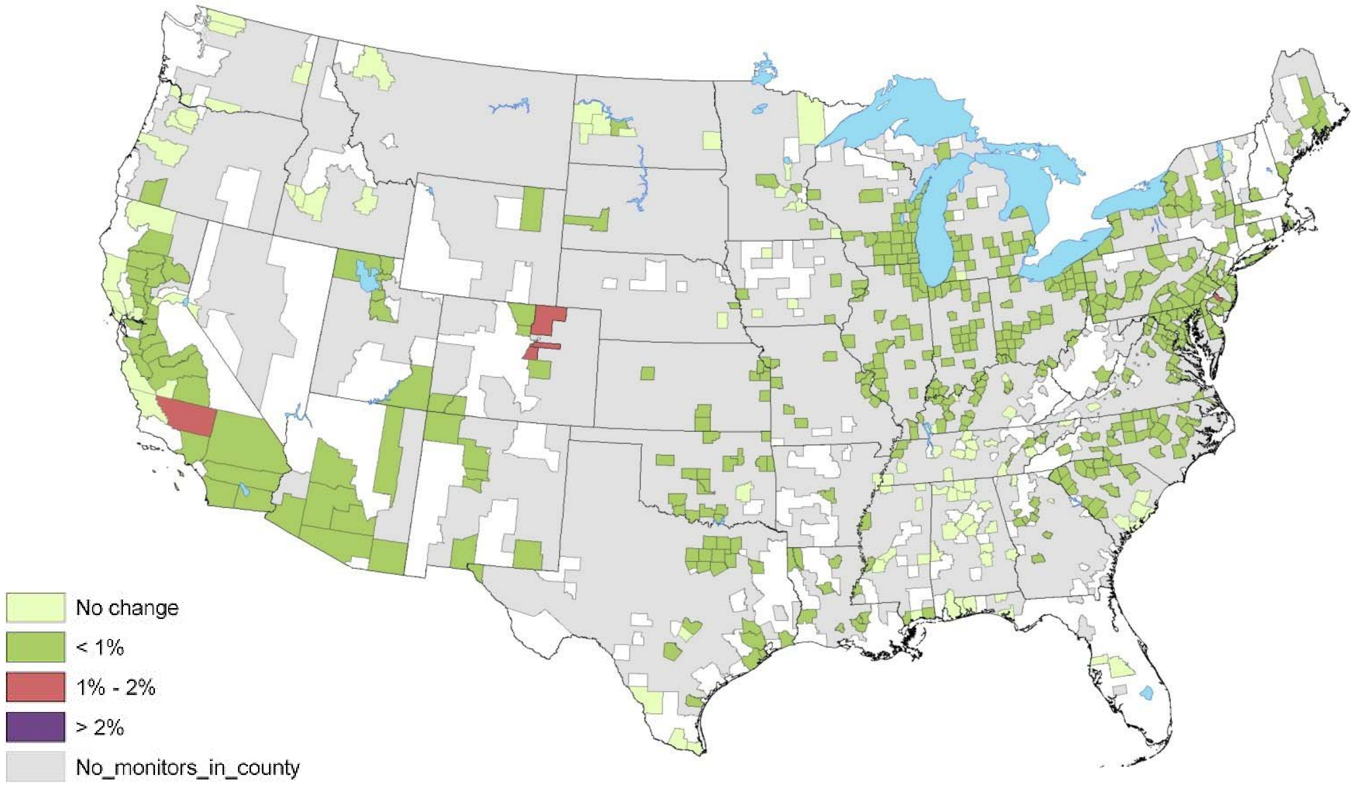
*It is important to note that the modeled hypothetical RIA controls did not fully attain the primary standard of 0.070 ppm, so this map underestimates the yield loss avoided in several areas, especially Southern California, Houston, Eastern Lake Michigan, and the Northeast Corridor.

Figure S3-24: Yield Loss Avoided by Secondary Standard of 13 ppm-hrs in 2020 for Soybean*



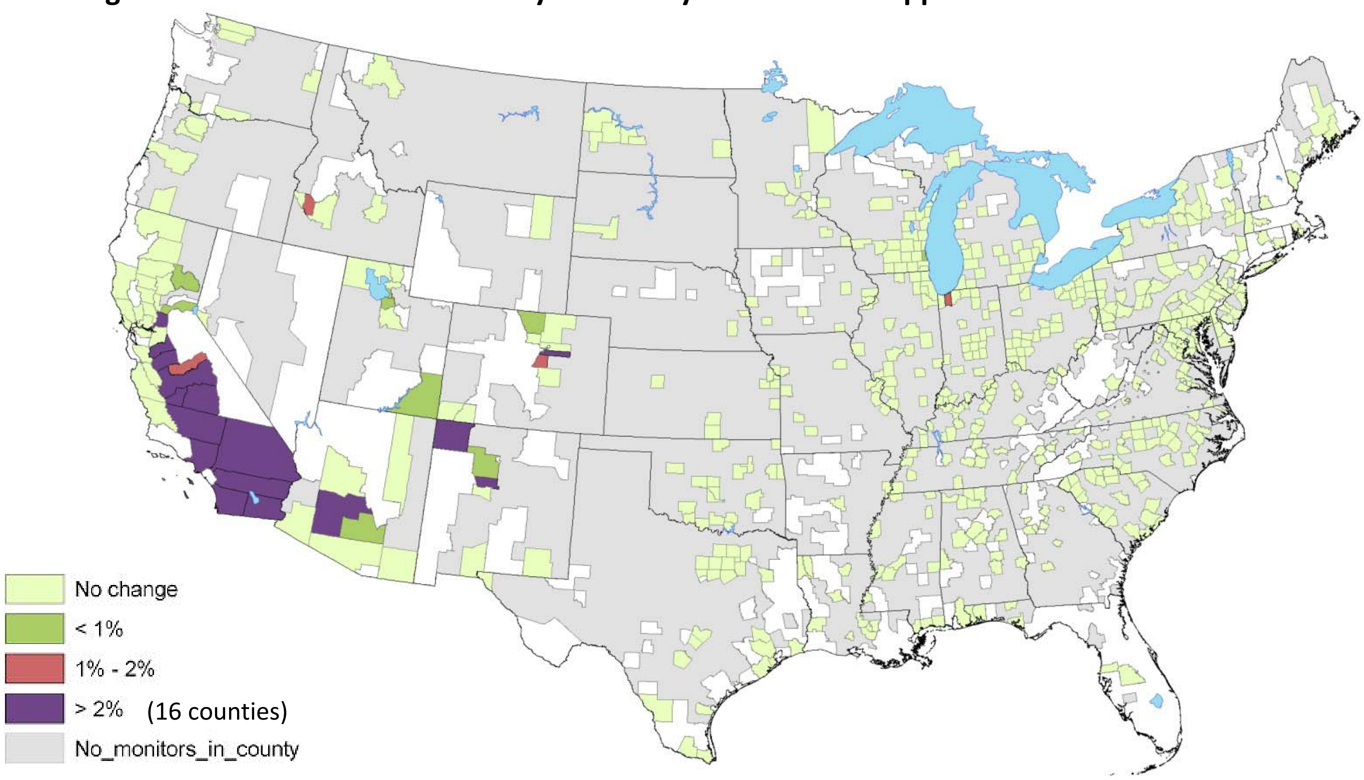
*It is important to note that the control strategy is likely to reduce W126 levels over a broader geographic area than just the violating county, so this map underestimates the regional yield loss avoided.

Figure S3-25: Yield Loss Avoided by Primary Standard in 2020 for Winter Wheat*



*It is important to note that the modeled hypothetical RIA controls did not fully attain the primary standard of 0.070 ppm, so this map underestimates the yield loss avoided in several areas, especially Southern California, Houston, Eastern Lake Michigan, and the Northeast Corridor.

Figure S3-26: Yield Loss Avoided by Secondary Standard of 13 ppm-hrs in 2020 for Winter Wheat*



*It is important to note that the control strategy is likely to reduce W126 levels over a broader geographic area than just the violating county, so this map underestimates the regional yield loss avoided.

c. Ozone Effects on Ornamental Plants

Urban ornamental plants are an additional vegetation category likely to experience some degree of negative effects associated with exposure to ambient ozone levels. A variety of ornamental species have been listed as sensitive to ozone (Abt Associates, 1995). Because ozone causes visible foliar injury, the aesthetic value of ornamental plants (such as petunia, geranium, and poinsettia) in urban landscapes would be reduced (U.S. EPA, 2007b). Sensitive ornamental species would require more frequent replacement and/or increased maintenance (fertilizer or pesticide application) to maintain the desired appearance because of exposure to ambient ozone (U.S. EPA, 2007b). In addition, many businesses rely on healthy-looking vegetation for their livelihoods (e.g., horticulturalists, landscapers, Christmas tree growers, farmers of leafy crops, etc.). The ornamental landscaping industry is a multi-billion dollar industry that affects both private property owners/tenants and governmental units responsible for public areas (Abt Associates, 1995). Preliminary data from the 2007 Economic Census indicate that the landscaping services industry, which is primarily engaged in providing landscape care and maintenance services and installing trees, shrubs, plants, lawns, or gardens, was valued at $53 billion (U.S. Census Bureau, 2010). Therefore, urban ornamentals represent a potentially large unquantified benefit category. This aesthetic damage may affect the enjoyment of urban parks by the public and homeowners' enjoyment of their landscaping and gardening activities. In addition, homeowners may experience a reduction in home value or a home may linger on the market longer due to decreased aesthetic appeal. In the absence of adequate exposure-response functions and economic damage functions for the potential range of effects relevant to ornamental plants, we cannot conduct a quantitative analysis to estimate these effects.

S2.11 Additional Co-benefits

1.3

1.4 In addition to the direct benefits on vegetation that the secondary ozone NAAQS is intended to produce, there are other co-benefits associated with reducing ambient ozone concentrations and ozone precursor pollutants. It is important to note that these additional benefits are contingent upon the secondary standard being the controlling standard. In other words, if the primary standard is controlling in all areas, there would not be any additional benefits beyond those attributable to implementation of the primary standard. For areas where additional control measures are needed to attain the secondary standard beyond those needed to attain the primary standard, there would be additional benefits associated with those emission reductions. These additional benefits are described below.

S4.6.1 Qualitative Human Health Co-benefits

1.4.1.1.1 Reducing ozone concentrations is associated with significant human health benefits, including avoiding mortality and respiratory morbidity. Researchers have associated ozone exposure with adverse health effects in numerous toxicological, clinical and epidemiological studies (U.S. EPA, 2006a). These health effects include respiratory morbidity such as fewer asthma attacks, hospital and ER visits, school loss days, as well as premature mortality.[23]

NOx is an ozone precursor, and reducing NOx emissions would also reduce health effects associated with NO_2 exposure. Following an extensive evaluation of health evidence from epidemiologic and laboratory studies, the Integrated Science Assessment (ISA) for Nitrogen Dioxide concluded that there is a likely causal relationship between respiratory health effects and short-term exposure to NO_2 (U.S. EPA, 2008b). Persons with preexisting respiratory disease, children, and older adults may be more susceptible to the effects of NO_2 exposure. The NO_2 ISA identified four short-term morbidity endpoints as a "likely causal relationship": asthma exacerbation, respiratory-related emergency department visits, and respiratory-related hospitalizations. The NO_2 ISA also 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. The differing evidence and associated strength of the evidence for these different effects is described in detail in the NO_2 ISA.

1.4.1.1.2 Furthermore, NO_X and VOCs are precursors to $PM_{2.5}$ as well as ozone. Reducing exposure to $PM_{2.5}$ is associated with significant human health benefits, including avoiding mortality and respiratory morbidity.[24] Researchers have associated $PM_{2.5}$- exposure with adverse health effects in numerous toxicological, clinical and epidemiological studies (U.S. EPA, 2009). These health effects include premature mortality for adults and infants, cardiovascular morbidity such as heart attacks, hospital admissions, and respiratory morbidity such as fewer asthma attacks, bronchitis, hospital and ER visits, work loss days, restricted activity days, and respiratory symptoms.[25]

[23] See Chapter 6 of the 2008 Ozone RIA, the updated benefits analysis in Section 3 of this supplemental for additional information on the ozone-related health effects associated with attaining the primary standard.

[24] See Chapter 6 of the 2008 Ozone RIA, the updated benefits analysis in Section 3 of this supplemental for additional information on the $PM_{2.5}$-related health effects associated with attaining the primary standard.

[25] See Chapter 6 of the 2008 Ozone RIA, the updated benefits analysis in Section 3 of this supplemental for additional information on the ozone-related health effects associated with attaining the primary standard.

S4.6.2 Qualitative Welfare Co-benefits

In addition to impacts on vegetation, ozone can also impact other welfare categories, including damage to certain manmade materials (e.g., elastomers, textile fibers, dyes, paints, and pigments) and climate interactions. The amount of damage to actual in-use materials and the economic consequences of that damage are poorly characterized, however, and the scientific literature contains very little new information to adequately quantify estimates of materials damage from photochemical oxidants (U.S. EPA, 2007b). Ozone is a well-known greenhouse gas, and the overall body of scientific evidence suggests that high concentrations of ozone on the regional scale could have a discernable influence on climate, leading to surface temperature and hydrological cycle changes (U.S. EPA, 2006).

1.4.1.1.3

1.4.1.1.4 NOx is an ozone precursor, and reducing NOx emissions would also reduce adverse welfare effects from acidic deposition, nutrient enrichment, and visibility impairment. Deposition of nitrogen causes acidification, which can cause a loss of biodiversity of fishes, zooplankton, and macro invertebrates in aquatic ecosystems, as well as a decline in sensitive tree species, such as red spruce (*Picea rubens*) and sugar maple (*Acer saccharum*) in terrestrial ecosystems. In the northeastern United States, the surface waters affected by acidification are a source of food for some recreational and subsistence fishermen and for other consumers and support several cultural services, including aesthetic and educational services and recreational fishing. Biological effects of acidification in terrestrial ecosystems are generally linked to aluminum toxicity, which can cause reduced root growth, which restricts the ability of the plant to take up water and nutrients. These direct effects can, in turn, increase the sensitivity of these plants to stresses, such as droughts, cold temperatures, insect pests, and disease leading to increased mortality of canopy trees. Terrestrial acidification affects several important ecological services, including declines in habitat for threatened and endangered species (cultural), declines in forest aesthetics (cultural), declines in forest productivity (provisioning), and increases in forest soil erosion and reductions in water retention (cultural and regulating). (U.S. EPA, 2008c)

Deposition of nitrogen is also associated with aquatic and terrestrial nutrient enrichment. In estuarine waters, excess nutrient enrichment can lead to eutrophication. Eutrophication of estuaries can disrupt an important source of food production, particularly fish and shellfish production, and a variety of cultural ecosystem services, including water-based recreational and aesthetic services. Terrestrial nutrient enrichment is associated with changes in the types and number of species and biodiversity in terrestrial systems. Excessive nitrogen deposition upsets the balance between native and nonnative plants, changing the ability of an area to support biodiversity. When the composition of species changes, nonnative grasses can fuel more frequent and more intense wildfires. (U.S. EPA, 2008c)

Reducing NOx and the secondary formation of $PM_{2.5}$ would reduce visibility impairment throughout the U.S. Fine particles with significant light-extinction efficiencies include sulfates, nitrates, organic carbon, elemental carbon, and soil (Sisler, 1996). These suspended particles and gases degrade visibility by scattering and absorbing light. Higher visibility impairment levels in the East are due to generally higher concentrations of fine particles, particularly sulfates, and higher average relative humidity levels. Visibility has direct significance to people's enjoyment of daily activities and their overall sense of wellbeing. Good visibility increases the quality of life where individuals live and work, and where they engage in recreational activities.

S2.12 References

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2 APPENDIX S3A: OZONE SENSITIVE PLANTS (FROM U.S. EPA, 2007)

Allegheny blackberry *Rubus allegheniensis*
American elder *Sambucus canadensis*
American hazelnut *Corylus americana*
American sycamore *Platanus occidentalis*
Basswood *Tilia Americana*
Big-leaf aster *Aster macrophyllus*
Black cherry *Prunus serotina*
Black huckleberry *Gaylussacia baccata*
Black locust *Robinia pseudoacacia*
Black poplar *Populus balsamifera trichocarpa*
Blue elderberry *Sambucus mexicana*
Box elder *Acer negundo*
California black oak *Quercus kelloggii*
Chokecherry *Prunus virginiana*
Common milkweed *Asclepias syriaca*
Cottonwood *Populus deltoids*
Crown-beard *Verbesina occidentalis*
Cutleaf coneflower *Rudbeckia laciniata*
Dogbane, Indian hemp *Apocynum cannibinum*
Evening primrose *Oenothera elata*
Goldenrod *Solidago altissima*
Gooding's willow *Salix goodingii*
Green ash *Fraxinus pennsylvanica*
Groundnut *Apios americana*
Huckleberry *Vaccinium membranaceum*
Jack pine *Pinus banksiana*
Jeffrey pine *Pinus jeffreyi*
Loblolly pine *Pinus taeda*
Maleberry *Lyonia ligustrina*
Monterey pine *Pinus radiata*
Mountain dandelion *Krigia montana*
Mugwort *Artemisia douglasiana*
Ninebark *Physocarpus capitatus*
Northern fox grape *Vitis labrusca*
Ohio Buckeye, Horse chestnut *Aesculus glabra*
Pacific ninebark *Physocarpus malvaceum*
Paper birch *Betula papyrifera*
Pinus ponderosa *Pinus ponderosa*
Pitch pine *Pinus rigida*
Poke milkweed *Asclepias exaltata*
Ponderosa pine *Pinus ponderosa*
Quaking aspen *Populus tremuloides*
Red alder *Alnus rubra*
Red elderberry *Sambucus racemosa*
Redbud *Cercis Canadensis*
Saskatoon serviceberry *Amelanchier alnifolia*
Sassafras *Sassafras albidum*
Scouler's willow *Salix scouleriana*
Serviceberry *Amelanchier alnifolia*
Silver wormwood *Artemisia ludoviciana*
Single-leaf ash *Fraxinus anomala*
Skunkbush *Rhus trilobata*
Smooth cordgrass *Spartina alterniflora*
Snowberry *Symphoricarpos albus*
Speckled alder *Alnus rugosa*
Spreading dogbane *Apocynum androsaemifolium*
Swamp milkweed *Asclepias incarnata*
Sweet mock orange *Philadelphus coronarius*
Sweetgum *Liquadambar styraciflua*
Table-mountain pine *Pinus pungens*
Tall milkweed *Asclepias exaltata*
Thimbleberry *Rubus parviflorus*
Thornless blackberry *Rubus canadensis*
Tree-of-heaven *Ailanthus altissima*
Twinberry *Lonicera involucrata*
Virgin's bower *Clematis virginiana*
Virginia creeper *Parthenocissus quinquefolia*
Virginia pine *Prunus virginiana*
White ash *Fraxinus americana*
White snakeroot *Eupatorium rugosum*
White stem blazingstar *Mentzelia albicaulis*
Whorled aster *Aster acuminatus*
Winged sumac *Rhus copallina*
Yellow-poplar *Liriodendron tulipifera*

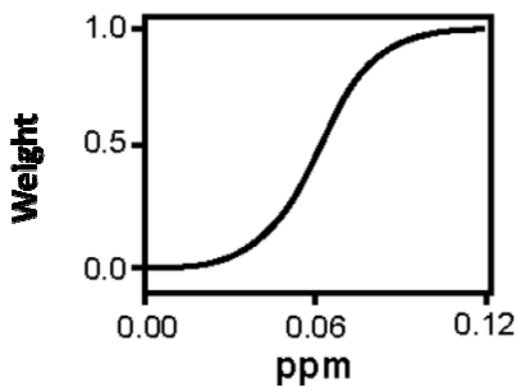
Ozone Sensitive Crops
Cotton
Peanuts
Potatoes
Soybeans
Tobacco
Winter Wheat

Sand blackberry *Rubus cuneifolius*

3 APPENDIX S3B: CALCULATING THE W126 INDEX

Steps in calculating W126 value for a particular site:

1. Measure O_3 concentrations for each hour within 12-hour daylight period (8 am to 8 pm)
2. Weight each hourly O_3 concentration to get a W126 value: lower concentrations receive less weight than higher concentrations
3. Add the 12 weighted hourly W126 values to calculate daily W126 value for each day
4. Sum daily W126 values within each month to get a monthly W126 value
5. Identify the consecutive 3-month period whose sum of monthly W126 values produces the highest W126 index value. This maximum consecutive 3-month sum = seasonal W126 value for that site (in ppm-hrs)



Example of weighting over 5-hour period:

Hourly O_3 (primary)	Weight	W126 (ppm-hrs)
0.03	0.01	0.00
0.05	0.11	0.01
0.06	0.30	0.02
0.08	0.84	0.07
0.10	1.0	0.10
	SUM:	**0.20**

Daily value = sum of values over 12 daylight hours