United States Environmental Protection Agency

Final Regulatory Impact Analysis (RIA) for the NO_2 National Ambient Air Quality Standards (NAAQS)

January 2010

U.S. Environmental Protection Agency
Office of Air Quality Planning and Standards
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Research Triangle Park, North Carolina

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Executive Summary - NO_2 NAAQS RIA

ES.1 Overview

This Regulatory Impact Analysis (RIA) provides illustrative estimates of the incremental costs and monetized human health benefits of attaining a revised short-term Nitrogen Dioxide (NO_2) National Ambient Air Quality Standard (NAAQS) within the current community-wide monitoring network of 409 monitors. Because this analysis only considers counties with NO_2 monitors, the possibility exists that there may be many more potential nonattainment areas than have been analyzed in this RIA.

The final NAAQS is a new short-term NO_2 standard based on the 3-year average of the 98th percentile of 1-hour daily maximum concentrations, establishing a new standard of 100 ppb. We also analyzed a lower level of 80 parts per billion (ppb) and an upper level of 125 ppb. It is important to reiterate that this analysis does not attempt to estimate attainment or nonattainment for any areas of the country other than those counties currently served by one of the 409 monitors in the current network. Chapter 2 explains that the current network is focused on community-wide ambient levels of NO_2, and not near-roadway levels, which may be significantly higher. The final rule also contains requirements for an NO_2 monitoring network that would include monitors near major roadways. We recognize that once a network of near-roadway monitors is put in place, more areas could find themselves exceeding the new hourly NO_2 NAAQS. However for this RIA analysis, we lack sufficient data to predict which additional counties might exceed the new NAAQS after implementation of a near-roadway monitoring network if they do not currently have a monitor. (Regional scale models such as the Community Multi-scale Air Quality Modeling System (CMAQ) do not provide a sufficient level of sub-grid detail to estimate near-road concentrations, and local-scale models such as AERMOD cannot model large regions with appropriate characterization of the near-road component of ambient air quality).

In this RIA, we projected current area-wide monitor values to future year monitor values directly, using future year CMAQ modeling outputs that take into account expected changes in emissions from 2006 to 2020. Because a near-roadway monitoring network does not currently exist, it was not possible to do this same direct projection into the future for near-roadway peaks. Because short-term peak exposures may occur near roadways, we conducted an analysis to approximate such peak exposures. This analysis relies on current and future estimated air quality concentrations at area-wide monitors, making adjustments to future year projections using derived estimates of the relationship between future year area-wide air quality peaks and current near-roadway peaks. This analysis, which effectively extrapolates future year near-roadway air quality from projected area-wide concentrations, represents a screening level approximation with significant additional uncertainties.

The RIA for the proposed NAAQS included an analysis based on community level exposure, represented by the current area-wide monitoring network. Because the final NAAQS is based on expected near-roadway (peak) exposures, the RIA for the final NAAQS focuses on the near-roadway analysis (which was included in the RIA for the proposed NAAQS as an alternative analysis). It is important to note that no current monitors in the (area-wide) network are projected to violate either the final NAAQS level of 100 ppb, or the lower bound of 80 ppb, in 2020, assuming a baseline of no additional control beyond the controls expected from rules that are already in place (including the current $PM_{2.5}$ and ozone NAAQS).[1] As noted above, we recognize that once a network of near-roadway monitors is put in place, more areas could find themselves exceeding the new hourly NO_2 NAAQS.

This RIA chiefly serves two purposes. First, it provides the public with an estimate of the expected costs and benefits of attaining a new NO_2 NAAQS. Second, it fulfills the requirements of Executive Order 12866 and the guidelines of OMB Circular A-4.[2] These documents present guidelines for EPA to assess the benefits and costs of the selected regulatory option, as well as one less stringent and one more stringent option. As stated above, we chose 80 ppb as an analytic lower bound, and 125 ppb as an analytic upper bound.

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

The prohibition against the consideration of cost in the setting of the primary air quality standard, however, does not mean that costs or other economic considerations are unimportant or should be ignored. The Agency believes that consideration of costs and benefits is essential to making efficient, cost effective decisions for implementation of these standards. The impacts of cost and efficiency are considered by states during this process, as they decide what timelines, strategies, and policies are most appropriate. This RIA is intended to inform the public about the potential costs and benefits associated with a hypothetical scenario that may

[1] For this RIA, we chose an analysis year of 2020. Although the actual attainment year is likely to be 2017, time and resource limitations dictated use of pre-existing model runs, which all focused on 2020. In addition, we do not have emission inventory projections for 2017; such projections are done for 5-year intervals.

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

result when a new NO_2 standard is implemented, but is not relevant to establishing the standards themselves.

ES.2 Summary of Analytic Approach for the Analysis of Approximated Future Near-Roadway NO_2 Exceedances of Target NAAQS

Our assessment of the NO_2 NAAQS and lower and upper bounds includes several key elements, including specification of baseline NO_2 emissions and concentrations; development of illustrative control strategies to attain the standard in 2020; and analyses of the control costs and health benefits of reaching each level. Additional information on the methods employed by the Agency for this RIA is presented below.

Overview of Baseline Emissions Forecast and Baseline NO_2 Concentrations

The baseline emissions and concentrations for this RIA are based on NOx emissions data from the 2002 National Emissions Inventory (NEI), and baseline NO_2 concentration values from 2005-2007 across the community-wide monitoring network. We used results from the community multi-scale air quality model (CMAQ) simulations from the ozone NAAQS RIA to calculate the expected reduction in ambient NO_2 concentrations between the 2002 base year and 2020. More specifically, design values (i.e. air quality concentrations at each monitor) were calculated for 2020 using monitored air quality concentrations from 2002 and modeled air quality projections for 2020, countywide emissions inventory data for 2002 and 2005-7, and emissions inventory projections for 2020. These data were used to create ratios between emissions and air quality, and those ratios (relative response factors, or RRFs) were used to estimate air quality monitor design values for 2020.

Because a near-roadway monitoring network does not currently exist, it was not possible to do the same direct projection into the future for near-roadway peaks as was done for the area-wide analysis in the proposal RIA, to analyze the standard levels of 80 ppb, 100 ppb, and 125 ppb (98^{th} percentile value). Therefore, the near-roadway analysis represents a much more uncertain screening level approximation of future year near-roadway air quality. We first select “area-wide” monitors to adjust to approximate near-roadway conditions. The monitors included in this analysis are those considered to be representative of “area-wide” conditions; i.e. those monitors to which it would be appropriate to apply the gradient to scale from area-wide to near-roadway conditions. To reflect the expected roadway gradient discussed in the preamble to the final rule (i.e., near road monitors can be between 30% to 100% greater than the area wide monitors), we adjust our estimated design values at area-wide locations for the future year of 2020 by 130%, 165%, and 200%. The analytic method we used to determine the 2020 design values and the tons needed to attain the alternate standard levels incorporates the near roadway gradient adjustment with a modification to future CMAQ air quality levels. While the modification is conceptually sound, it is a relatively new methodology. We discuss the methodology in detail in chapter 2.

Development of Illustrative Control Strategies

For the final RIA, we analyzed the impact that additional emissions controls would have on predicted ambient NO_2 concentrations, incremental to the baseline set of controls. Thus the modeled analysis for a revised standard focuses specifically on incremental improvements beyond the current standards, and uses control options that might be available to states for application by 2020. The hypothetical modeled control strategy presented in this RIA is one illustrative option for achieving emissions reductions to move towards a national attainment of a tighter standard. It is not a recommendation for how a tighter NO_2 standard should be implemented, and states will make all final decisions regarding implementation strategies once a final NAAQS has been set.

Generally, we expect that many states would be able to attain the NO_2 NAAQS without the addition of new controls beyond those already being planned for the attainment of existing $PM_{2.5}$ and ozone standards by the year 2020. As States develop their plans for attaining these existing standards, they are likely to consider adding controls to reduce NOx, as NOx is a precursor to both $PM_{2.5}$ and ozone. These controls will also directly help areas meet a tighter NO_2 standard.

Analysis of Benefits

Our analysis of the benefits associated with the NO_2 NAAQS includes the ancillary benefits of reducing concentrations of particulate matter (PM). Because NO_x is also a precursor to $PM_{2.5,}$ reducing NO_x emissions in the projected non-attainment areas will also reduce $PM_{2.5}$ formation, human exposure, and the incidence of $PM_{2.5}$-related health effects. In this analysis, we estimated the co-benefits of reducing $PM_{2.5}$ exposure for the alternative standards.

Due to analytical limitations, it was not possible to provide a comprehensive estimate of $PM_{2.5}$-related benefits. Instead, we used the “benefit-per-ton” method to estimate these benefits. The $PM_{2.5}$ benefit-per-ton estimates provide the total monetized human health benefits (the sum of premature mortality and premature morbidity) of reducing one ton of a $PM_{2.5}$ precursor from a specified source category. For this analysis, the $PM_{2.5}$ co-benefits only represent NO_x emission reductions from the mobile sector because data limitations in the control strategy preclude estimating co-emission reductions from directly emitted $PM_{2.5}$ or $PM_{2.5}$ precursors. We assume that all fine particles, regardless of their chemical composition, are equally potent. These estimates reflect EPA's most current interpretation of the scientific literature on $PM_{2.5}$ and mortality, including our updated benefits methodology (i.e., a no-threshold model that calculates incremental benefits down to the lowest modeled $PM_{2.5}$ air quality levels and incorporates two technical updates) compared to the estimates in previous RIAs that did not include these changes. EPA has used a similar technique in previous RIAs, including the recent Ozone NAAQS RIA (U.S. EPA, 2008a) and Portland Cement NESHAP RIA (U.S. EPA, 2009). For the near-roadway benefits, we were unable to estimate NO_2 benefits based on the data available for this analysis. This is discussed further in Chapter 4. Although this benefit in unquantified in this analysis, the area-wide analysis for the proposed NAAQS RIA showed that the monetized NO_2 benefits accounted for only 2% of the total monetized benefits.

Analysis of Costs

Because this analysis examines emissions and air quality approximating near-roadway conditions, we assume that unspecified controls are applied to mobile source emissions. We have estimated that the annualized average cost of controls to attain the NO_2 NAAQS would be in the range of $3,000 to $6,000 per ton. This estimate is based upon previous estimates of controls for mobile sources.

For onroad and nonroad mobile sources, costs, in terms of dollars per ton emissions reduced, were applied to emission reductions calculated for the onroad and nonroad mobile sectors that had previously been generated using the National Mobile Inventory Model (NMIM). NMIM is an EPA model for estimating air emissions from highway vehicles and nonroad mobile equipment. NMIM uses current versions of EPA's model for onroad mobile sources, MOBILE6, and nonroad mobile sources, NONROAD, to calculate emission inventories.[1]

ES.3. Results from Screening Level Near-Roadway Analysis

Air Quality and Emissions

For the revised standard of 100 ppb and the less stringent level of 125 ppb there were no projected exceedances in 2020. For the more stringent level of 80 ppb, exceedances

[1] More information regarding the National Mobile Inventory Model (NMIM) can be found at http://www.epa.gov/otaq/nmim.htm

totaling were projected in 4 counties, with 21,230 tons of emissions reductions needed for attainment.

Benefits and Costs

Tables ES-1 and ES-2 present the counties in nonattainment, tons of NOx reduction, costs, and benefits for compliance with the NO_2 NAAQS in 2020 for this near-roadway analysis, using the near road gradient adjustment at discount rates of 3% and 7% respectively. The selected standard of 100 ppb at the mean expected gradient of 65% is highlighted.

Table ES-1: 2020 Benefit Cost Comparison (in millions of 2006$, 3% discount rate for Benefits only)

	Standard Level	# Counties in Nonattainment	Tons of NOx Reduction	Total Costs *			Total Benefits **			Net Benefits		
30% Gradient	80 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
	100 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
	125 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
65% Gradient	80 ppb	1	680	$5.6	to	$7.7	$3.5	to	$8.6	-$4.1	to	$3.0
	100 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
	125 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
100% Gradient	80 ppb	4	21,000	$67	to	$130	$110	to	$270	-$21	to	$200
	100 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
	125 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6

* Total Cost estimates are shown as a range from $3,000/ton to $6,000/ton. Results include monitoring costs of $3.6m. Costs estimates were only available for a 3% discount rate. All estimates have been rounded to two significant figures.
**Total Benefit estimates are actually $PM_{2.5}$ co-benefits, shown as a range from Pope et al to Laden et al, at a 3% discount rate, using no-threshold functions, assuming NOx emission reductions from the mobile sector.

Table ES-2: 2020 Benefit Cost Comparison (in millions of 2006$, 7% discount rate)

	Standard Level	# Counties in Nonattainment	Tons of NOx Reduction	Total Costs *			Total Benefits **			Net Benefits		
30% Gradient	80 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
	100 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
	125 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
65% Gradient	80 ppb	1	680	$5.6	to	$7.7	$3.2	to	$7.8	-$4.5	to	$2.1
	100 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
	125 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
100% Gradient	80 ppb	4	21,000	$67	to	$130	$100	to	$240	-$31	to	$180
	100 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
	125 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6

* Total Cost estimates are shown as a range from $3,000/ton to $6,000/ton. Results include monitoring costs of $3.6m. Costs estimates were only available for a 3% discount rate. All estimates have been rounded to two significant figures.

**Total Benefit estimates are actually $PM_{2.5}$ co-benefits, shown as a range from Pope et al to Laden et al, at a 3% discount rate, using no-threshold functions, assuming NOx emission reductions from the mobile sector.

ES.4. Caveats and Limitations

General

- Due to the absence of a near-roadway monitoring network, this is a screening level analysis with several simplifying assumptions. It is provided to give a rough projection of the costs and benefits of attaining a revised NO_2 standard based on a yet to be established monitoring network.

- This analysis does not take into account a large variety of localized conditions specific to individual monitors; instead, the analysis attempts to account for some local parameters by adjusting future design values based on average localized impacts near roads from onroad emissions.

- The process of adjusting from a specific 12 km CMAQ receptor to a near-road air quality estimate represents an uncertain approximation at the specific monitor level.

- This analysis is an approximation in that it derives future year (2020) peak air quality concentrations in specific locations by relying on CMAQ estimates that are averages over a 12 km grid square.

- This analysis cannot predict air quality in locations for which there is no current NO_2 monitor, or where current monitoring data is incomplete. There are 142 CBSAs for which we are proposing to add new near-road monitors. Of these, 73 either have no existing monitor in the CBSA, or have a monitor with data not complete enough to include in the near-roadway analysis. In these CBSAs, extrapolation to near-roadway levels is not possible.

- This analysis assumes area-wide monitors remain in the same location; however concentrations are adjusted to reflect near-roadway conditions.

- This analysis omits certain unquantified effects due to lack of data, time and resources. These unquantified endpoints include NO_2 health effects, ozone co-benefits, ecosystem effects, and visibility.

Air Quality Data, Modeling and Emissions

- Current $PM_{2.5}$ and Ozone Controls in Baseline: Our 2020 analysis year baseline assumes that States will put in place the necessary control strategies to attain the current $PM_{2.5}$ and ozone standards. Some of the control strategies employed as part of the ozone RIA, in particular, were unspecified. As States develop their plans for attaining these standards, their NOx control strategies may differ significantly from our analysis.

- Use of Existing CMAQ Model Runs: This analysis represents a screening level analysis. We did not conduct new regional scale modeling specifically targets to NO_2; instead we relied upon impact ratios developed from model runs used in the analysis underlying the ozone NAAQS.

- Analysis Year of 2020: Data limitations necessitated the choice of an analysis year of 2020, as opposed to the presumptive implementation year of 2017. Emission inventory projections are available for 5-year increments; i.e. we have inventories for 2015 and 2020, but not 2017. In addition, the CMAQ model runs upon which we relied were also based on an analysis year of 2020.

- Limited monitoring network: For the current monitoring community-wide monitoring network, the universe of monitors exceeding the target NAAQS levels is very small. Once a network of near-roadway monitors is put in place, there could be more potential nonattainment areas than have been analyzed in this RIA.

- Actual State Implementation Plans May Differ from our Simulation: In order to reach attainment with each selected NAAQS, each state will develop its own implementation plan implementing a combination of emissions controls that may differ from those simulated in this analysis. This analysis therefore represents an approximation of the emissions reductions that would be required to reach attainment and should not be treated as a precise estimate.

- Climate change impacts of NOx or NO_2 emissions, which have not been extensively studied with regard to their impacts on net warming, are only now beginning to be investigated. Since work on this issue is only beginning, an analysis of the quantified impacts of reduction in NO_2 on climate cannot yet be provided.

Costs

- There are some unquantified costs that are not adequately captured in this illustrative analysis. These costs include the costs of federal and State administration of control programs, which we believe are less than the alternative of States developing approvable SIPs, securing EPA approval of those SIPs, and Federal/State enforcement. Additionally, control measure costs referred to as "no cost" may require limited government agency resources for administration and oversight of the program not included in this analysis; those costs are generally outweighed by the saving to the industrial, commercial, or private sector. The Agency also did not consider transactional costs and/or effects on labor supply in the illustrative analysis.

- Known control costs used were derived from data on a variety of known controls, and not based on any one specific control strategy tailored to specific geographic areas that may violate the NAAQS.

Benefits

- There are many uncertainties associated with the health impact functions used in this modeling effort. These include: within study variability; across study variation; the application of concentration-response (C-R) functions nationwide; extrapolation of impact functions across population; and various uncertainties in the C-R function, including causality and thresholds. These uncertainties may under- or over-estimate benefits.

- This analysis is for the year 2020, and projecting key variables introduces uncertainty. Inherent in any analysis of future regulatory programs are uncertainties in projecting atmospheric conditions and source level emissions, as well as population, health baselines, incomes, technology, and other factors.

- This analysis omits certain unquantified effects due to lack of data, time and resources. These unquantified endpoints include other health effects, ecosystem effects, and visibility. EPA will continue to evaluate new methods and models and select those most appropriate for estimating the benefits of reductions in air pollution. Enhanced collaboration between air quality modelers, epidemiologists, toxicologists, ecologists, and economists should result in a more tightly integrated analytical framework for measuring benefits of air pollution policies.

- $PM_{2.5}$ mortality co-benefits represent a substantial proportion of total monetized benefits (over 90%), and these estimates are subject to a number of assumptions and uncertainties.

  1. $PM_{2.5}$ co-benefits were derived through benefit per-ton estimates, which do not reflect local variability in population density, meteorology, exposure, baseline health incidence rates, or other local factors that might lead to an over-estimate or under-estimate of the actual benefits of controlling directly emitted fine particulates.

  2. We assume that all fine particles, regardless of their chemical composition, are equally potent in causing premature mortality. This is an important assumption, because $PM_{2.5}$ produced via transported precursors emitted from EGUs may differ significantly from direct $PM_{2.5}$ released from diesel engines and other industrial sources, but no clear scientific grounds exist for supporting differential effects estimates by particle type.

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

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

Chapter 1: Introduction and Background

Synopsis

This document estimates the incremental costs and monetized human health benefits of attaining a revised primary nitrogen dioxide (NO_2) National Ambient Air Quality Standard (NAAQS) nationwide. This document contains illustrative analyses that consider limited emission control scenarios that states, tribes and regional planning organizations might implement to achieve a revised NO_2 NAAQS. EPA weighed the available empirical data and photochemical modeling to make judgments regarding the proposed attainment status of certain urban areas in the future. According to the Clean Air Act, EPA must use health-based criteria in setting the NAAQS and cannot consider estimates of compliance cost. This Regulatory Impact Analysis (RIA) is intended to provide the public a sense of the benefits and costs of meeting new alternative NO_2 NAAQS, and to meet the requirements of Executive Order 12866 and OMB Circular A-4 (described below in Section 1.2.2).

This RIA provides illustrative estimates of the incremental costs and monetized human health benefits of attaining a revised primary NO_2 National Ambient Air Quality Standard (NAAQS) in 2020 within the current network of 409 monitors. The final rule adds a new short-term (1-hour exposure) standard, in addition to the current annual average standard. It is important to note that there may be many more potential nonattainment areas than have been analyzed in this RIA. The Integrated Science Assessment (ISA) and Risk and Exposure Assessment (REA), discussed in section 1.3 below, summarize available monitoring information, noting elevated short-term NO_2 concentrations near roads with high traffic volumes, with significant gradients relative to areas further away. Therefore there may be near-roadway locations that are currently not served by an NO_2 monitor, but which may have relatively high NO_2 concentrations at peak times.

1.1 Background

Two sections of the Clean Air Act (“Act”) govern the establishment and revision of NAAQS. Section 108 (42 U.S.C. 7408) directs the Administrator to identify pollutants which “may reasonably be anticipated to endanger public health or welfare,” and to issue air quality criteria for them. These air quality criteria are intended to “accurately reflect the latest scientific knowledge useful in indicating the kind and extent of all identifiable effects on public health or welfare which may be expected from the presence of [a] pollutant in the ambient air.” NO_2 is one of six pollutants for which EPA has developed air quality criteria.

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

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

1.2 Role of the Regulatory Impact Analysis in the NAAQS Setting Process

1.2.1 Legislative Roles

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

The prohibition against the consideration of cost in the setting of the primary air quality standard, however, does not mean that costs or other economic considerations are unimportant or should be ignored. The Agency believes that consideration of costs and benefits are essential to making efficient, cost effective decisions for implementation of these standards. The impact of cost and efficiency are considered by states during this process, as they decide what timelines, strategies, and policies make the most sense. This RIA is intended to inform the public about the potential costs and benefits that may result when a new NO_2 standard is implemented, but is not relevant to establishing the standards themselves.

1.2.2 Role of Statutory and Executive Orders

There are several statutory and executive orders that dictate the manner in which EPA considers rulemaking and public documents. This document is separate from the NAAQS decision making process, but there are several statutes and executive orders that still apply to any public documentation. The analysis required by these statutes and executive orders is presented in Chapter 9.

EPA presents this RIA pursuant to Executive Order 12866 and the guidelines of OMB Circular A-4.[1] These documents present guidelines for EPA to assess the benefits and costs of the selected regulatory option, as well as one less stringent and one more stringent option. OMB circular A-4 also requires both a benefit-cost and a cost-effectiveness analysis for rules where health is the primary effect. Within this RIA we provide a benefit-cost analysis. Methodological and data limitations prevent us from performing a cost-effectiveness analysis and a meaningful more formal uncertainty analysis for this RIA.

The final NAAQS is a new short-term NO_2 standard based on the 3-year average of the 98^{th} percentile of 1-hour daily maximum concentrations, establishing a new standard of 100 ppb. We also analyzed a lower level of 80 parts per billion (ppb) and an upper level of 125 ppb. It is important to reiterate that this analysis does not attempt to estimate attainment or nonattainment for any areas of the country other than those counties currently served by one of the 409 monitors in the current network.

1.2.3 Market Failure or Other Social Purpose

OMB Circular A-4 indicates that one of the reasons a regulation such as the NAAQS may be issued is to address market failure. The major types of market failure include: externality, market power, and inadequate or asymmetric information. Correcting market failures is one reason for regulation, but it is not the only reason. Other possible justifications include improving the function of government, removing distributional unfairness, or promoting privacy and personal freedom.

An externality occurs when one party's actions impose uncompensated benefits or costs on another party. Environmental problems are a classic case of externality. For example, the smoke from a factory may adversely affect the health of local residents while soiling the property in nearby neighborhoods. If bargaining was costless and all property rights were well

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

defined, people would eliminate externalities through bargaining without the need for government regulation. From this perspective, externalities arise from high transaction costs and/or poorly defined property rights that prevent people from reaching efficient outcomes through market transactions.

Firms exercise market power when they reduce output below what would be offered in a competitive industry in order to obtain higher prices. They may exercise market power collectively or unilaterally. Government action can be a source of market power, such as when regulatory actions exclude low-cost imports. Generally, regulations that increase market power for selected entities should be avoided. However, there are some circumstances in which government may choose to validate a monopoly. If a market can be served at lowest cost only when production is limited to a single producer of local gas and electricity distribution services, a natural monopoly is said to exist. In such cases, the government may choose to approve the monopoly and to regulate its prices and/or production decisions. Nevertheless, it should be noted that technological advances often affect economies of scale. This can, in turn, transform what was once considered a natural monopoly into a market where competition can flourish.

Market failures may also result from inadequate or asymmetric information. Because information, like other goods, is costly to produce and disseminate, an evaluation will need to do more than demonstrate the possible existence of incomplete or asymmetric information. Even though the market may supply less than the full amount of information, the amount it does supply may be reasonably adequate and therefore not require government regulation. Sellers have an incentive to provide information through advertising that can increase sales by highlighting distinctive characteristics of their products. Buyers may also obtain reasonably adequate information about product characteristics through other channels, such as a seller offering a warranty or a third party providing information.

There are justifications for regulations in addition to correcting market failures. A regulation may be appropriate when there are clearly identified measures that can make government operate more efficiently. In addition, Congress establishes some regulatory programs to redistribute resources to select groups. Such regulations should be examined to ensure that they are both effective and cost-effective. Congress also authorizes some regulations to prohibit discrimination that conflicts with generally accepted norms within our society. Rulemaking may also be appropriate to protect privacy, permit more personal freedom or promote other democratic aspirations.

From an economics perspective, setting an air quality standard is a straightforward case of addressing an externality, in this case where entities are emitting pollutants, which cause health and environmental problems without compensation for those suffering the problems. Setting a standard with a reasonable margin of safety attempts to place the cost of control on those who emit the pollutants and lessens the impact on those who suffer the health and environmental problems from higher levels of pollution.

1.2.4 Illustrative Nature of the Analysis

This NO_2 NAAQS RIA is an illustrative analysis that provides useful insights into a limited number of emissions control scenarios that states might implement to achieve a revised NO_2 NAAQS. Because states are ultimately responsible for implementing strategies to meet any revised standard, the control scenarios in this RIA are necessarily hypothetical in nature. They are not forecasts of expected future outcomes. Important uncertainties and limitations are documented in the relevant portions of the analysis.

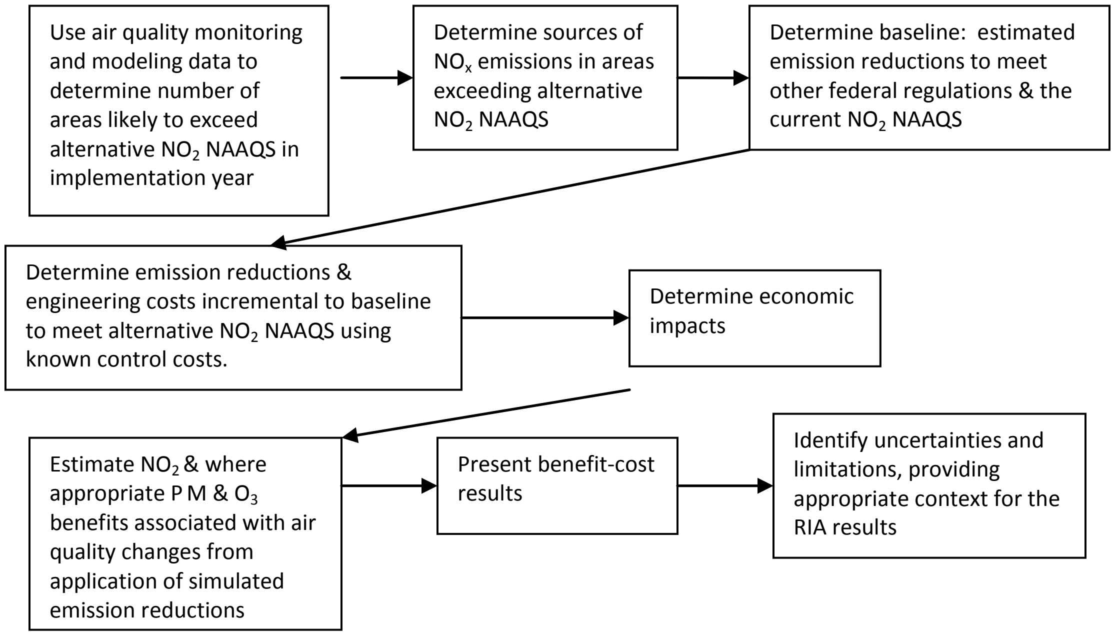
The illustrative goals of this RIA are somewhat different from other EPA analyses of national rules, or the implementation plans states develop, and the distinctions are worth brief mention. This RIA does not assess the regulatory impact of an EPA-prescribed national or regional rule such as the Clean Air Interstate Rule, nor does it attempt to model the specific actions that any state would take to implement a revised NO_2 standard. This analysis attempts to estimate the costs and human and welfare benefits of cost-effective implementation strategies which might be undertaken to achieve national attainment of new standards. These hypothetical strategies represent a scenario where states use one set of cost-effective controls to attain a revised NO_2 NAAQS. Because states—not EPA—will implement any revised NAAQS, they will ultimately determine appropriate emissions control scenarios. State implementation plans would likely vary from EPA's estimates due to differences in the data and assumptions that states use to develop these plans.

The illustrative attainment scenarios presented in this RIA were constructed with the understanding that there are inherent uncertainties in projecting emissions and controls. Despite these limitations, EPA has used the best available data and methods to produce this RIA.

1.3 Overview and Design of the RIA

This Regulatory Impact Analysis evaluates the costs and benefits of hypothetical national strategies to attain several potential revised primary NO_2 standards. The document is intended to be straightforward and written for the lay person with a minimal background in chemistry, economics, and/or epidemiology. Figure 1-1 provides an illustration of the process

Figure 1-1: The Process Used to Create this RIA



1.3.1 Baseline and Years of Analysis

It is important to note that no current monitors in the (area-wide) network are projected to violate the final NAAQS level of 100 ppb in 2020, assuming a baseline of no additional control beyond the controls expected from rules that are already in place (including the current $PM_{2.5}$ and ozone NAAQS). The analysis year for this regulatory impact analysis is 2020, which approximates the required attainment year under the Clean Air Act.[2] For purposes of this analysis, we assess attainment by 2020 for all areas. Some areas for which we assume 2020 attainment may in fact need more time to meet one or more of the analyzed standards, while others will need less time. This analysis does not prejudge the attainment dates that will ultimately be assigned to individual areas under the Clean Air Act.

The methodology first estimates what baseline NO_2 levels might look like in 2020 with existing Clean Air Act programs, including application of controls to meet the current NO_2 NAAQS, various rules addressing mobile source emissions, various maximum achievable control technology (MACT) standards, and the revised particulate matter (PM) and ozone (O_3) NAAQS

[2] Although the actual attainment year is likely to be 2017, time and resource limitations dictated use of pre-existing model runs, which all focused on 2020. In addition, we do not have emission inventory projections for 2017; such projections are done for 5-year intervals.

standards. It is important to note that as a result of these rules, NOx emissions nationally are expected to decrease about 48% over the period for this analysis (2002-2020).[3] The analysis then predicts the change in NO_2 levels following the application of additional controls to reach tighter alternative standards. This allows for an analysis of the incremental change between the current standard and alternative standards. Since NO_2 is a precursor of both ozone and PM, it is important that we account for the impact on NO_2 concentrations of both the NO_2 controls used in the hypothetical control scenario in the ozone NAAQS RIA, and the NO_2 and PM controls used in the hypothetical control scenario in the PM NAAQS RIA, so as to avoid double counting the benefits and costs of these controls.

1.3.2 Control Scenarios Considered in this RIA

The final NAAQS is a new short-term NO_2 standard based on the 3-year average of the 98th percentile of 1-hour daily maximum concentrations, establishing a new standard of 100 ppb. We also analyzed a lower level of 80 parts per billion (ppb) and an upper level of 125 ppb.

In this RIA, we projected current area-wide monitor values to future year monitor values directly, using future year CMAQ modeling outputs that take into account expected changes in emissions from 2006 to 2020. Because a near-roadway monitoring network does not currently exist, it was not possible to do this same direct projection into the future for near-roadway peaks. Because short-term peak exposures may occur near roadways, we conducted an analysis to approximate such peak exposures. This analysis relies on current and future estimated air quality concentrations at area-wide monitors, making adjustments to future year projections using derived estimates of the relationship between future year area-wide air quality peaks and current near-roadway peaks. The area-wide air quality peaks are adjusted using a gradient multiplier to simulate the monitors being near the road. The concentrations are further adjusted, using relationships between onroad and total emissions (all sources) to account for the fact that as the monitors are made "near-road" monitors, they will be affected more by onroad emissions reductions than if they were area-wide monitors. This analysis, which effectively extrapolates future year near-roadway air quality from projected area-wide concentrations, represents a screening level approximation with significant additional uncertainties.

[3] The NO_2 NAAQS is based on 2002V3 inventories and projections to 2020. Data summaries can be found at: http://www.epa.gov/ttn/chief/emch/index.html#2002. See the compressed Excel workbook for 2002 and 2020 "2020cc-2002cc_20070925.zip".

1.3.3 Evaluating Costs and Benefits

Because the final NAAQS is based on expected near-roadway (peak) exposures, the RIA for the final NAAQS focuses on the near-roadway analysis (which was included in the RIA for the proposed NAAQS as an alternative analysis). For the final RIA, we analyzed the impact that additional emissions controls would have on predicted ambient NO_2 concentrations, incremental to the baseline set of controls. Thus the modeled analysis for a revised standard focuses specifically on incremental improvements beyond the current standards, and uses control options that might be available to states for application by 2020.

Although no current monitors in the (area-wide) network are projected to violate the final NAAQS level of 100 ppb in 2020, assuming a baseline of no additional control beyond the controls expected from rules that are already in place (including the current $PM_{2.5}$ and ozone NAAQS). We recognize that once a network of near-roadway monitors is put in place, more areas could find themselves exceeding the new hourly NO_2 NAAQS. This methodology enabled us to evaluate nationwide costs and benefits of attaining a tighter NO_2 standard using hypothetical strategies.[4]

To streamline this RIA, this document refers to several previously published documents, including two technical documents EPA produced to prepare for promulgation of the NO_2 NAAQS. The first was the Integrated Science Assessment created by EPA's Office of Research and Development (U.S. EPA, 2007), which presented the latest available pertinent information on atmospheric science, air quality, exposure, health effects, and environmental effects of NO_2. The second was a risk and exposure assessment (REA) (U.S. EPA, 2008) for various standard levels. The REA also includes staff conclusions and recommendations to the Administrator regarding potential revisions to the standards.

1.4 NO_2 Standard Alternatives Considered

EPA has performed an illustrative analysis of the potential costs and human health and visibility benefits of nationally attaining the final NO_2 NAAQS of 100 ppb, as well as a lower bound of 80 ppb and an upper bound of 125 ppb. Note that our projections indicated no counties in 2020 that would have ambient 1-hour peak levels as high as the final NAAQS standard of 100 ppb in 2020, assuming a baseline of no additional control beyond the controls expected from rules that are already in place (including the current $PM_{2.5}$ and ozone NAAQS),

[4] Because the secondary NO_2 NAAQS is under development in a separate regulatory process, no additional costs and benefits were calculated in this RIA.

and *solely within the bounds of the existing monitoring network*. The benefit and cost estimates below are calculated incremental to a 2020 baseline that incorporates air quality improvements achieved through the projected implementation of existing regulations and full attainment of the existing ozone and PM National Ambient Air Quality Standards (NAAQS). The baseline also includes the MACT program, the clean air interstate rule (CAIR), and implementation of current consent decrees, all of which would help many areas move toward attainment of the proposed NO_2 standard.

1.5 References

U.S. Environmental Protection Agency (U.S. EPA). 1970. Clean Air Act. 40 CFR 50.

U.S. Environmental Protection Agency (U.S. EPA). 2007. Review of the National Ambient Air Quality Standards for NO_2: Integrated Science Assessment. Office of Air Quality Planning and Standards, RTP, NC, available at http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=194645.

U.S. Environmental Protection Agency (U.S. EPA). 2008. Review of the National Ambient Air Quality Standards for NO_2: Risk and Exposure Assessment. Office of Air Quality Planning and Standards, RTP, NC, available at http://www.epa.gov/ttn/naaqs/standards/nox/s_nox_cr_rea.html.

Chapter 2: Air Quality Analysis

Synopsis

This chapter describes the NOx emissions, NO_2 monitoring network, and approach used to calculate 2020 baseline near-roadway NO_2 design values and the amount of emissions reductions needed to attain alternative levels of the 1-hour NO_2 NAAQS. We first describe data on NO_2 emission sources contained in available EPA emission inventories. We then provide an overview of data sources for air quality measurement, and finally the methodology used to project NO_2 levels to 2020. For a more in-depth discussion of NO_2 emissions and air quality data, see the Integrated Science Assessment for the NO_2 NAAQS (EPA, 2007a).

2.1 Sources of NO_2

The primary data source for this discussion is the National Emissions Inventory (NEI) for 2002 (USEPA, 2007b). Ambient levels of NO_2 are the product of both direct NO_2 emissions and emissions of other NOx (e.g., NO), which can then be converted to NO_2 through reaction with ozone. Nationally, anthropogenic sources account for approximately 87% of total NOx emissions. (Apart from these anthropogenic sources, there are also natural sources of NOx including microbial activity in soils, lightning, and wildfires.)

Stationary sources (e.g., electrical utilities and industry) account for about 40% of the national anthropogenic NOx emissions in the 2002 NEI. The main stationary sources of NOx emissions in the 2002 NEI are combustion-related emissions and industrial process-related emissions. Table 2-1 presents emissions estimates for stationary sources grouped into descriptive categories. Presence and relative position of a source category on this list does not necessarily provide an indication of the significance of the emissions from individual sources within the source category. A source category, for example, may be composed of many small (i.e., low-emitting) sources, or of just a few very large (high-emitting) sources.

Mobile sources (both on-road and non-road) account for about 60% of the national anthropogenic NOx emissions in the 2002 NEI. Highway vehicles represent the major mobile source component. In the United States, approximately half the mobile source emissions are contributed by diesel engines and half are emitted by gasoline-fueled vehicles and other sources.

As a result of Clean Air Act requirements, emissions standards promulgated for many source categories have taken effect since 2002, including numerous mobile source standards for gasoline and diesel vehicles/engines, and are projected to result in much lower emissions of both direct NO_2 and other NOx at the current time or in the near future.

Table 2-1. NOx Sources (2002 NEI)

NOx Source Category	Emissions (tons/year)
Electric Utility Fuel Combustion	3,792,292
Industrial Fuel Combustion	1,897,944
Fuel Combustion, other	730,259
Chemical and Allied Product Manufacturing	60,901
Metals Processing	66,173
Petroleum and Related Industries	358,223
Industrial Processes, other	482,007
Solvent Utilization	4,365
Storage and Transport	16,109
Waste Disposal and Recycling	145,678
Highway Vehicles	6,491,821
Off-highway Vehicles	6,027,085
Miscellaneous Source Categories	270,913
Total	**20,343,770**

2.2 Air Quality Monitoring Data

2.2.1 Background on NO_2 monitoring network

From its inception in the late 1970's through the present (2008), the NO_2 network has remained relatively stable with regard to the number of monitoring sites (see memo by Watkins, 2008). As of October 2008, there were 409 NOx monitors within the U.S. actively reporting NO_2 data to the air quality system AQS. The NO_2 network was originally deployed to support implementation of the NO_2 NAAQS established in 1971. The first requirements for NO_2 monitoring were issued in May 1979. At that time, 40 CFR Part 58, Appendix D, section 3.5 stated:

> "Nitrogen Dioxide NAMS [National Ambient Monitoring Stations, now a defunct term] will be required in those areas of the country which have a population greater than 1,000,000. These areas will have two NO_2 NAMS. It is felt that stations in these major metropolitan areas would provide sufficient data for a national analysis of the data, and also because NO_2 problems occur in areas of greater than 1,000,000. Within urban areas requiring [NO_2] NAMS, two permanent monitors are sufficient. The first station (category (a), middle scale or neighborhood scale) would be to measure the photochemical production of NO_2 and would best be located in that part of the urban

> area where the emission density of NOx is the highest. The second station (category (b) urban scale), would be to measure the NO_2 produced from the reaction of NO with O_3 and should be downwind of the area peak NOx emission areas."

In the October, 2006 monitoring rule, these NO_2 monitoring requirements were removed from the CFR due in part to the absence of any NO_2 non-attainment problems under the current standards. In the 2006 rule, EPA rewrote 40 CFR Part 58, Appendix D, section 4.3 to state that:

> "There are no minimum requirements for the number of NO_2 monitoring sites. Continued operation of existing SLAMS [State and Local Ambient Monitoring Station] NO_2 sites using FRM [Federal Reference Method] or FEM [Federal Equivalent Method] is required until discontinuation is approved by the EPA Regional Administrator. Where SLAMS NO_2 monitoring is ongoing, at least one NO_2 site in the area must be located to measure the maximum concentration of NO_2."

As noted earlier, the size of the NO_2 network has been fairly stable through time, even though an actual requirement for state and local air agencies to monitor NO_2, other than for Photochemical Assessment Monitoring Stations (PAMS) or Prevention of Significant Deterioration (PSD), was removed in the 2006 monitoring rule. The maintenance of the NO_2 monitoring network has been driven by several factors, including the need to support ozone modeling and forecasting, the need to track PM precursors, and a general desire on the part of states to continue to understand trends in ambient NO_2.

To characterize the current NO_2 network, staff has reviewed the NO_2 network meta-data. The data reviewed are those available from AQS in October 2008, for monitors reporting data in 2008. The meta-data fields are typically created by state and local agencies when a monitor site is initiated, moved, or re-characterized. While these files are useful for characterizing specific monitors, there is some uncertainty surrounding this meta-data given that there is no routine or enforced process for updating or correcting meta-data fields. With this uncertainty in mind, staff has compiled information on the monitoring objectives and measurement scales for monitors in the NO_2 network.

The monitor objective meta-data field describes the purpose of the monitor. For example the purpose of a particular monitor could be to characterize health effects, photochemical activity, transport, and/or welfare effects. As of October 2008, there were 489 records of NO_2 monitor objective values (some monitors have multiple monitor objectives). Table 2-2 lists the distribution of monitoring objectives across the network. There are 12 categories of monitor objectives for NO_2 monitors within AQS. The "other" category is for sites likely addressing a state or local need outside of the routine objectives, and the "unknown" category represents missing meta-data. The remaining categories stem directly from categorizations of site types within CFR. In 40 CFR Part 58 Appendix D, there are six examples of NO_2 site types:

1. Sites located to determine the highest concentration expected to occur in the area covered by the network (Highest Concentration).
2. Sites located to measure typical concentrations in areas of high population (Population Exposure).
3. Sites located to determine the impact of significant sources or source categories on air quality (Source Oriented).
4. Sites located to determine general background concentration levels (General Background).
5. Sites located to determine the extent of regional pollutant transport among populated areas; and in support of secondary standards (Regional Transport).
6. Sites located to measure air pollution impacts on visibility, vegetation damage, or other welfare-based impacts (Welfare Related Impacts).

The remaining four categories represent available site types for Photochemical Assessment Monitoring Stations (PAMS) network. These PAMS site types are described in 40 CFR Part 58 Appendix D:

1. Type 1 sites are established to characterize upwind background and transported ozone and its precursor concentrations entering the area and will identify those areas which are subjected to transport (Upwind Background).
2. Type 2 sites are established to monitor the magnitude and type of precursor emissions in the area where maximum precursor emissions are expected to impact and are suited for the monitoring of urban air toxic pollutants (Maximum Precursor Impact).
3. Type 3 sites are intended to monitor maximum ozone concentrations occurring downwind from the area of maximum precursor emissions (Maximum Ozone Concentration).
4. Type 4 sites are established to characterize the downwind transported ozone and its precursor concentrations exiting the area and will identify those areas which are potentially contributing to overwhelming transport in other areas (Extreme Downwind).

Table 2-2: NO_2 Network Distribution of Monitor Objectives.

NO_2 Monitor Objective	Number of Monitor Objective Records	Percent Distribution
Population Exposure	177	36.20
Highest Concentration	58	11.86
General Background	51	10.43
Max. Precursor Impact (PAMS Type 2 Site)	21	4.29
Source Oriented	19	3.89
Upwind Background (PAMS Type 1 Site)	18	3.68
Regional Transport	12	2.45
Other	9	1.84
Max. Ozone Concentration (PAMS Type 3 Site)	8	1.64
Extreme Downwind (PAMS Type 4 Site)	3	0.61
Welfare Related Impacts	1	0.20
Unknown	112	22.90
Totals:	**489**	**100%**

The meta-data for the NO_2 network also indicate the measurement scale represented by each particular monitor. The definitions of measurement scales can be found in 40 CFR Part 58, Appendix D, Section 1 "Monitoring Objectives and Spatial Scales." This part of the regulation spells out what data from a monitor can represent in terms of air volumes associated with area dimensions:

Microscale - 0 to 100 meters
Middle Scale - 100 to 500 meters
Neighborhood Scale - 500 meters to 4 kilometers
Urban Scale - 4 to 50 kilometers
Regional Scale - 50 kilometers up to 1000km

There are 386 NO_2 monitor records in AQS with available measurement scale data. Table 2-3 shows the measurement scale distribution across all NO_2 sites form the available data in AQS of monitors reporting data in 2008.

Table 2-3: NO_2 Network Distribution across Measurement Scales.

Measurement Scale	Number of Measurement Scale Records	Percent Distribution
Microscale	3	0.78
Middle Scale	23	5.96
Neighborhood	212	54.92
Urban Scale	119	30.83
Regional Scale	29	7.51
Totals:	**386**	**100%**

Many of the monitors used in the analyses presented here, especially for the near-road adjustment calculations, are defined as area-wide monitors. These are monitors that would meet the following criteria:

- Neighborhood, urban, or regional scale (based on measurement scale)
- Not a site identified as being operated by industry
- If the monitor is a neighborhood scale monitor, its monitor objective is not highest concentration and its dominant source is not a point source.

The criteria above will be used to identify monitors to adjust for near-road conditions in Section 2.3.2.2. More details about monitor classification can be found in Appendix 2.

In summary, the NO_2 network is primarily targeting public health and photochemical process monitoring objectives. Nearly half of the monitor objective records are directly targeting public health through the population exposure (36.2%) and highest concentration (11.8%) categories alone. The other categories serve to inform public health concerns, but also address photochemistry issues where NOx serves as a precursor to ozone. Further, it appears that approximately 10% of NO_2 monitors are in place to serve the PAMS network. In reality, a large majority of sites likely could serve both public health and photochemistry related objectives due to their proximity to urban areas. The exceptions would likely be categories such as upwind background, extreme downwind, regional transport, and possibly maximum O_3 concentration. These four categories only represent approximately 7% of the NO_2 network, and have a higher likelihood of being rural and regional in scale.

2.2.2 Trends in and characterizations of ambient concentrations of NO_2

As noted above, NO_2 is monitored largely in urban areas and, therefore, data from the NO_2 monitoring network is generally more representative of urban areas than rural areas. According to monitoring data, nationwide levels of ambient NO_2 (annual average) decreased 41% between 1980 and 2006 (ISA, Figure 2.4-15). Between 2003 and 2005, national mean concentrations of NO_2 were about 15 ppb for averaging periods ranging from a day to a year. The average daily maximum hourly NO_2 concentrations were approximately 30 ppb. These values are about twice as high as the 24-h averages. The highest maximum hourly concentrations (~200 ppb) between 2003 and 2005 are more than a factor of ten higher than the mean hourly or 24-h concentrations (ISA, Figure 2.4-13). The monthly highest levels of NO_2 in the United States can be found in and around Los Angeles, in the Midwest, and in the Northeast. Local maxim around Denver, CO, Salt Lake City, UT, and El Paso, TX can also be found (ISA, Figure 2.4-14) Policy-relevant background concentrations, which are those concentrations that would occur in the United States in the absence of anthropogenic emissions in continental North America (defined here as the United States, Canada, and Mexico), are estimated to range from only 0.1 ppb to 0.3 ppb on an annual basis (ISA, section 2.4.6).

Ambient levels of NO_2 exhibit both seasonal and diurnal variation. In southern cities, such as Atlanta, higher concentrations are found during winter, consistent with the lowest mixing layer heights being found during that time of the year. Lower concentrations are found during summer, consistent with higher mixing layer heights and increased rates of photochemical oxidation of NO_2. For cities in the Midwest and Northeast, such as Chicago and New York City, higher levels tend to be found from late winter to early spring with lower levels occurring from summer though the fall. In Salt Lake City, higher concentrations tend to be found in winter in association with winter temperature inversions. In Los Angeles the highest levels tend to occur from autumn though early winter and the lowest levels from spring though early summer. Mean and peak concentrations in winter can be up to a factor of two larger than in the summer at sites in Los Angeles. In terms of daily variability, NO_2 levels typically peak during the morning rush hours. Monitor siting plays a key role in evaluating diurnal variability as monitors located further away from traffic will show cycles that are less pronounced over the course of a day than monitors located closer to traffic.

2.2.3 Uncertainty Associated with the Ambient NO_2 Monitoring Method

As has been acknowledged by the Agency and the scientific community for some time, the most prevalently used measurement method for estimating ambient NO_2 levels (i.e., subtraction of NO from a measure of total NOx) is subject to interference by NOx oxidation products. Limited evidence from some studies suggests that these interferences could result in an overestimate of NO_2 levels by roughly 20 to 25% at typical ambient levels. However, smaller relative errors are estimated to occur in measurements taken near strong NOx sources since most of the mass emitted as NO or NO_2 would not yet have been further oxidized. Relatively larger errors appear in locations more distant from strong local NOx sources. Two additional sources of uncertainty in NO_2 measurements can result from monitor siting. First, many NO_2 monitors are located above ground level in the cores of large cities. Because most sources of NO_2 are near ground level (i.e., combustion emissions from traffic), higher levels NO_2 concentrations exist near ground level and lower levels being detected at the elevated monitors. One comparison has found an average of a 2.5-fold increase in NO_2 concentration measured at 4 meters above the ground compared to 15 meters above the ground. The ISA notes that levels are likely even higher at elevations below 4 meters (ISA, section 2.5.3.3). Second, NO_2 monitors are currently sited to determine annual regional levels rather than to capture small-scale variability in NO_2 concentrations near sources such as roadway traffic. Significant gradients in NO_2 concentrations near roadways have been observed in several studies, and NO_2 concentrations have been found to be negatively correlated with distance from roadway and traffic volume (ISA, section 2.5.3.2).

2.3 Air Quality Analysis

The principle objective of this air quality analysis is to estimate 2020 design values[1] that reflect maximum concentrations, compare these estimates to alternative levels of the NO_2 NAAQS, and determine the emission reductions required to reduce NO_2 air quality concentrations to below these various levels. Two challenges exist: estimating future levels given reductions from promulgated control programs and determining these future levels in locations where we expect maximum short term concentrations to occur. The first challenge is typical of RIA analyses and the second is unique to NO_2 because the monitoring network is not currently optimized to represent maximum short term levels. Such levels are expected to occur near roads but the monitoring network, while urban in its orientation, is oriented to area-wide measurements. In order to overcome the absence of a current near road monitoring network, we have used scientific literature on the gradients between near road levels and those locations at various distances from roads to estimate near road levels. In other words, we are adjusting NO_2 levels from area wide locations to attempt to approximate near road conditions.

The alternative levels of the NO_2 NAAQS being analyzed are 80, 100, and 125 ppb based on design values calculated using the 3-year average of the 98^{th} percentile 1-hour daily maximum concentrations based on the monitoring network described in section 2.2 with adjustments for a near-road network. The projected 2020 baseline NO_2 design values are used to identify 2020 nonattainment counties and to calculate, for each such county, the amount of reduction in NO_2 concentration necessary to attain the alternative levels of the NAAQS. This section also describes the approach for calculating "ppb NO_2 concentration per ton NOx

[1] A design value is a statistic that describes the air quality status of a given area relative to the level of the National Ambient Air Quality Standards (NAAQS). http://www.epa.gov/airtrends/values.html

emissions" ratios that are used to estimate the amount of NOx emissions reductions that may be needed to provide for attainment of the alternative NO_2 standards. As described below, the air quality analysis relies on NO_2 predictions from simulations of the Community Multiscale Air Quality (CMAQ) model coupled with ambient 2005-2007 design values and emissions data to project 2020 NO_2 design value concentrations and the "ppb per ton" ratios. A description of CMAQ is provided in the Ozone NAAQS RIA Air Quality Modeling Platform Document (U.S. EPA, 2008a).

2.3.1 2005-2007 Design Values

The form of the final NO_2 standard is the 3-year average of the 98^{th} concentration of the daily 1-hour maximum concentration for each year using measurements from the monitoring network described in Section 2.2. The first step in calculating the 3-year 2005-2007 design values is to identify the maximum 1-hour concentration for every day during the three years 2005 through 2007. Next, the 98^{th} percentile concentration of these daily 1-hour maximum concentrations is calculated for each year. The 98^{th} percentile concentrations for each year are averaged to determine a 3-year average concentration. Only monitors that had valid measurements for at least 75% of the day, 75% of the days in a quarter, and all 4 quarters for all three years were considered to have sufficient data completeness to be representative and were thereby included in the analysis[2]. In 2007, there were 435 monitors (259 counties) for NO_2 nationwide. Of those 435 monitors, 256 monitors (160 counties) met the criteria described above. Appendix 2a contains the complete list of 2005-2007 design values used in calculation of the 2020 design values. Note that Hawaiian monitors were excluded from the air quality analysis because there was no CMAQ data over Hawaii. This decreased the number of monitors and counties used in the analysis to 255 monitors and 159 counties

In Figure 2-1, the Core Based Statistical Areas' (CBSA) with populations greater 350,000 people are shown along with the number of monitors in each CBSA (CBSAs outside the continental U.S. are not included). Those with zero monitors have no monitors because: 1) no monitor was in the CBSA or 2) the monitors in the CBSA did not meet the completeness criteria described above. The number of monitors in Figure 2-1 represents 210 of the 255 monitors. The remaining 45 monitors were either in CBSAs with populations less than 350,000 people or not located in a CBSA. Figure 2-2 shows the population of the CBSAs shown in Figure 2-1. Figure 2-3 shows the population of the CBSAs within several population categories for CBSAs with population greater than 350,000 people. Shown are populations for CBSAs with monitors in the 2005-2007 period (green bars), those that have monitors but were excluded due to data completeness (yellow bars) and those CBSAs currently not monitored (orange bars). Also shown

[2] Email from Rhonda Thompson to James Thurman, January 22, 2009.

in each bar are the number of CBSAs in each population category. As can be seen by Figure 2-3, approximately 160 million people are in CBSAs that have at least one monitor in 2005-2007. Also, CBSAs with populations greater than 1 million people are represented in the analyses presented here. The large urban centers such as New York, Los Angeles, and Chicago are represented. Notable CBSAs not included in the analyses are: Detroit, Baltimore, Las Vegas and Seattle. While Detroit, Baltimore, and Las Vegas do have monitors, they were excluded due to incomplete data in 2005-2007, Seattle is currently the largest CBSA without monitors. As part of the new monitoring requirements, Seattle will have at least two monitors as the population of the CBSA is over 2.5 million.

Table 2a-1 in Appendix 2a lists the CBSAs with and the number of monitors from each area used in the analysis and Table 2a-2 lists the CBSAs with populations greater than 350,000 people not included in the analyses. In Table 2a-2 in Appendix A, the CBSA area for each of the 255 monitors is also listed.

Figure 2-1: Number of monitors per CBSA for CBSAs with 2007 population greater than 350,000 people

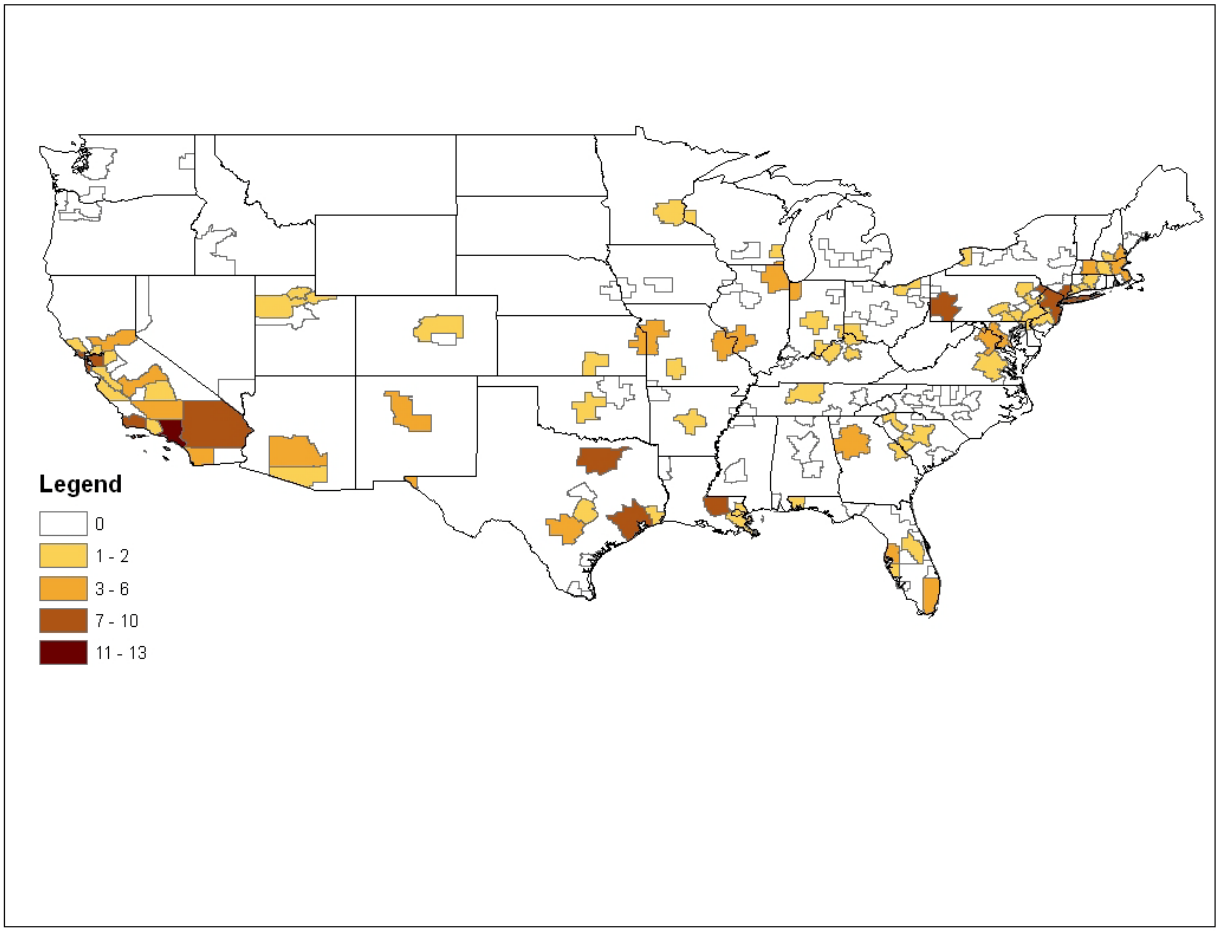


Figure 2-2: Populations of CBSAs with 2007 populations greater than 350,000 people

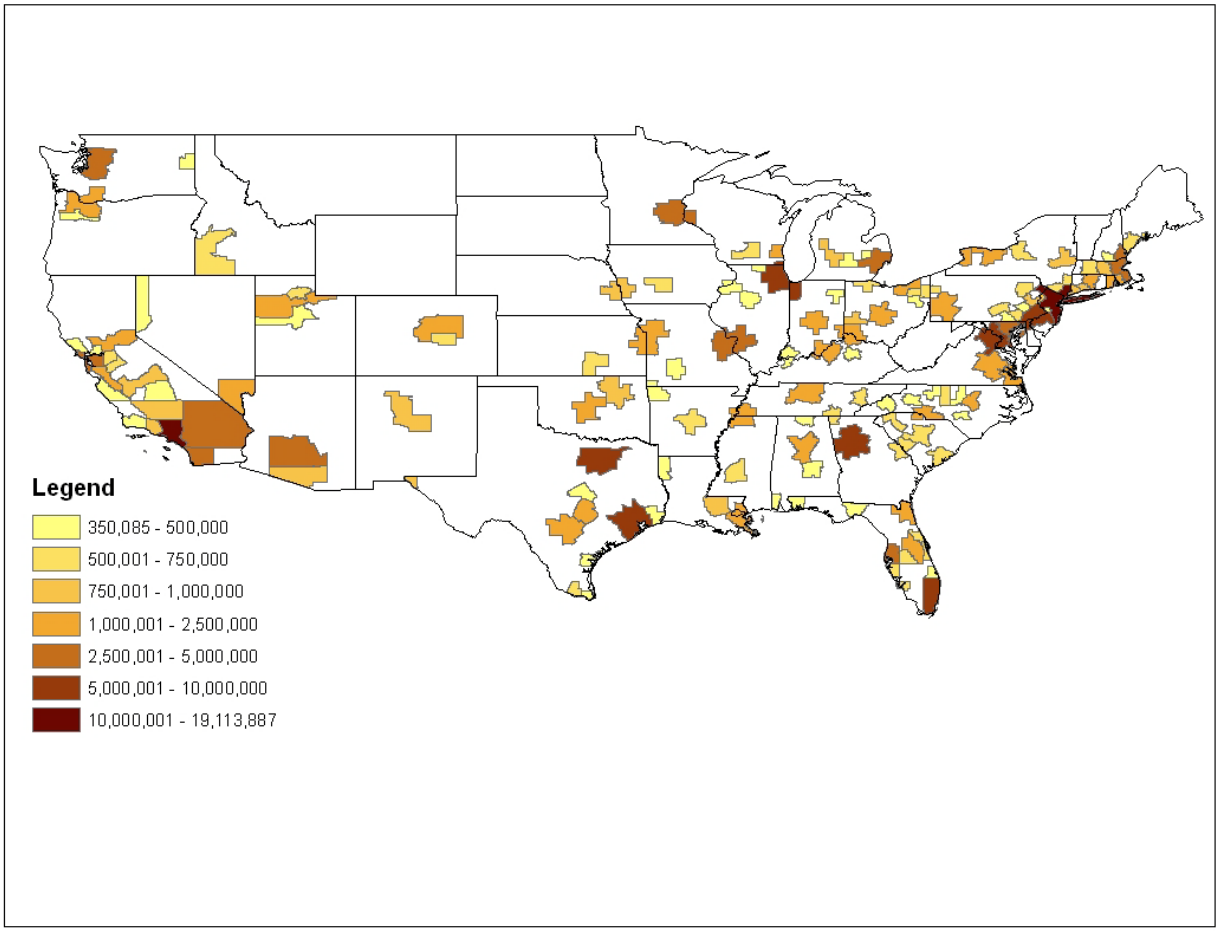
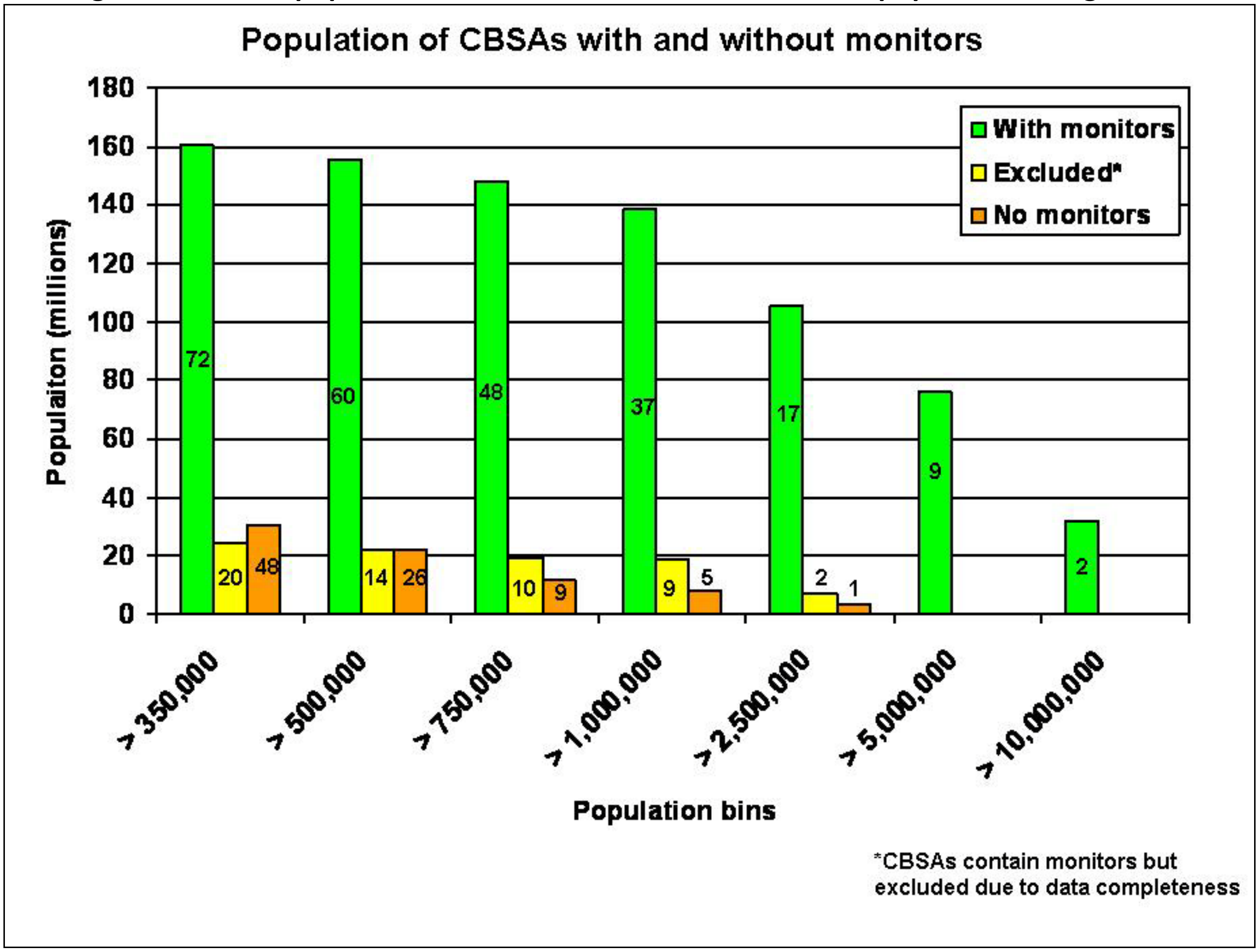


Figure 2-3: Total population and number of CBSAs for several population categories



2.3.2 Calculation of 2020 Projected Design Values

The 2020 baseline design values were calculated in a two step process. First, the 2005-2007 design values, which represented area-wide design values, were projected to 2020 using CMAQ concentrations and county-level emissions. This yielded a 2020 area-wide design value. Second, the projected 2020 area-wide design values were adjusted to simulate near-road concentrations. This adjustment involves two steps: (1) using concentrations gradients at distances from roadways from the scientific literature (i.e., 30, 65, and 100%); and an adjustment to account for the greater efficacy of onroad controls to near-road monitors in the future. This section describes the processing in the projection of 2005-2007 design values to 2020 near-road design values.

2.3.2.1 Calculation of 2020 area-wide design values

The 2020 baseline area-wide design values were determined using CMAQ concentrations for 2002 and 2020 and county emissions for 2002, 2006, and 2020. The CMAQ daily 1-hour maximum concentrations from 2002 and 2020 were used to calculate a relative response factor (RRF). The daily 1-hour maximum NO_2 concentrations in 2002 and 2020 were obtained from CMAQ runs performed for the ozone RIA (U.S. EPA, 2008b). The modeled NOx emissions in the CMAQ runs reflect reductions from federal programs including the Clean Air Interstate Rule (EPA, 2005a), the Clean Air Mercury Rule (EPA, 2005b), the Clean Air Visibility Rule (EPA, 2005c), the Clean Air Nonroad Diesel Rule (EPA, 2004), the Light-Duty Vehicle Tier 2 Rule (EPA, 1999), the Heavy Duty Diesel Rule (EPA, 2000); proposed rules for Locomotive and Marine Vessels (EPA, 2007c) and for Small Spark-Ignition Engines (EPA, 2007d); and national, 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[3].

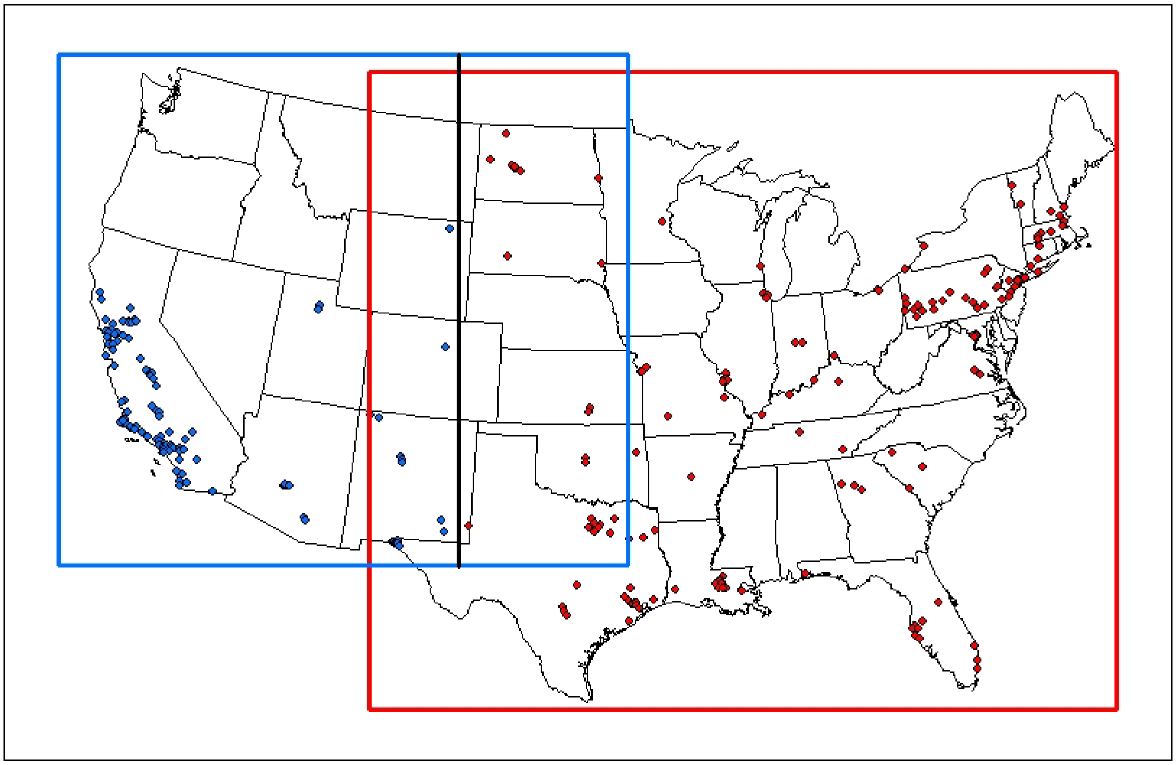
In brief, these CMAQ runs were performed at 12 km horizontal resolution for two modeling domains which, collectively, cover the lower 48 States and adjacent portions of Canada and Mexico. The boundaries of these two domains are shown in Figure 2-4. For 2020 we used CMAQ-predicted NO_2 concentrations from the Ozone NAAQS RIA control case. The CMAQ output represents concentrations based on emissions needed to attain an ozone standard of 0.070 ppm. We will refer to these concentrations and associated emissions as

[3] It should be noted that the emission reductions modeled for the PM2.5 and Ozone standards represent one possible control scenario, while the actual control strategies and resulting levels of emission reductions will be determined as part of the process of developing and implementing state implementation plans over the coming years. We should also note that since the finalization of these recent NAAQS standards, several of the proposed mobile source rules mentioned above have been finalized with updated analyses showing slightly greater levels of expected NOx reductions.

2020_070. As is standard analytic practice used in other RIAs previously, in order to align the base year modeled NO_2 data with the mid-point of the 2005-2007 design value period, we used the relationship between 2002 and 2006 NOx emissions used to estimate the 2002 NO_2 model-predicted concentrations to 2006. In addition to NOx emissions for the modeled 2002_070 (base emissions used in the projected 2020 0.070 ppm standard case) scenario, we calculated emissions for the 2020 baseline scenario, based on an emissions forecast described in Chapter 4 of the ozone RIA (EPA, 2008b). We refer to this inventory as 2020_075. This inventory contains emission reductions for 21 counties that did not meet the 0.070 ppm standard or less stringent 0.075 ppm standard (EPA, 2008b). In these 21 counties, across the board reductions of 30%, 60%, and 90% were made in the areas encompassing parts of California, Texas (centered on Houston), the Midwest (Chicago and Detroit areas), and the Northeast (portions of eastern Pennsylvania, New York, New Jersey, Maryland, Delaware, and Connecticut. These reductions were made to in an attempt to attain the 0.070 ppm standard. These are referred to as Phase I areas in the ozone RIA and can be seen in Figure 4.1 of the ozone RIA (EPA, 2008b. The RRF values and emissions were used to forecast 2020 design values and the amount of residual nonattainment at each monitored location.

The following are the steps used in calculating 2020 baseline NO_2 design values from the 2005-2007 monitor design values and CMAQ NO_2 concentrations for the 2002 and 2020_070 scenarios. Ambient monitored data were assigned to CMAQ grid cells using ArcGIS. Since there were areas of the country where the eastern and western domains overlapped, monitors in these overlapping areas were assigned to the eastern or western grid cells by using a "combined grid." This combined grid was a mesh of the eastern and western domains, with overlapping areas assigned eastern grid cells or western grid cells based on the location relative to the dividing line shown in Figure 2-4. Figure 2-4 also shows the assignment of monitors to the two domains. An example of monitors in both domains was the El Paso County monitors. These monitors were assigned to the western domain.

Figure 2-4: CMAQ 12 km domains and monitors used in air quality analyses. The western domain is outlined in blue and the eastern domain outlined in red. The black vertical line denotes the dividing line to assign monitoring sites to either the eastern or western domains. Monitors in red were assigned to the eastern domain and monitors in blue were assigned to the western domain.



The steps in projecting the 2020 area-wide design values were:

1. Beginning with 12-km CMAQ output, we calculated daily 1-hour maximum concentrations for each grid cell for 2002 and 2020_070 model output. For each grid cell, the top 10 daily 1-hour maximum concentrations for 2002 were averaged (C_{2002}). For 2020_070, the daily 1-hour maximum concentrations for the same calendar days corresponding to the top ten days in 2002 were also averaged (C_{2020_070}).

2. Relative response factors (RRF_C) were calculated by dividing the average of the 2020_070 concentrations by the average of the 2002 concentrations from step 1 (Equation 2.1).

$$RRF_C = \frac{C_{2020\_070}}{C_{2002}} \tag{2.1}$$

3. Monitors were assigned 2002, 2006, 2020_070, and 2020_075 county-wide emissions for the counties in which they were located. The 2020 baseline area-wide design values (i.e., using 2020_075 scenario emissions) were calculated by:

    a. An emissions relative response factor ($RRF_{E:2020\_070}$) was calculated to represent the emission changes from 2002 (E_{2002}) to 2020_070 (E_{2020_070}) as

$$RRF_{E:2020\_070} = \frac{E_{2020\_070}}{E_{2002}} \tag{2.2}$$

    Where E_{2020_070} are the 2020_070 county emissions, E_{2002} are the 2002 county emissions used in the modeling to yield the concentrations used in Steps 1 and 2. The emissions relative response factor is essentially the magnitude of 2020 emissions relative to 2002. If $RRF_{E:2020\_070}$ equals 0.9, that means the 2020_070 emissions are 90% of the 2002 emissions.

    b. We then calculated an emissions relative response factor ($RRF_{E:2020\_075}$) for emissions changes from 2006 (E_{2006}) to 2020_075 (E_{2020_075}) as

$$RRF_{E:2020\_075} = \frac{E_{2020\_075}}{E_{2006}} \tag{2.3}$$

    c. By assuming that the ratio of reduction in concentrations and reduction in emissions from 2002 to 2020_070 would be equal for a change from 2006 to 2020_075,

$$\frac{1 - RRF_C}{1 - RRF_{E:2020\_070}} = \frac{1 - RRF_{C:2020\_075}}{1 - RRF_{E:2020\_075}} \tag{2.4a}$$

    we calculated a concentration RRF for 2020_075 ($RRF_{C:2020\_075}$) as

$$RRF_{C:2020\_075} = 1 - \left[\left(\left(1 - RRF_C\right) \Big/ \left(1 - RRF_{E:2020\_070}\right) \right) \times \left(1 - RRF_{E:2020\_075}\right) \right] \tag{2.4b}$$

    A concentration RRF for 2020_075 must be calculated from this relationship because we do not have modeled 2006 concentrations or 2020 concentrations under the 0.075 ppm scenario.

d. Using the results from above, a 2020 area-wide 98th percentile design value (DV_{2020_075}) was calculated by multiplying the 2020_075 concentration RRF by the monitor's 2005-2007 98th percentile design values ($DV_{2005\text{-}07}$) by the concentration RRF ($RRF_{C:2020\_075}$) calculated in Equation 2.4b

$$DV_{2020\_075} = RRF_{C:2020\_075} \times DV_{2005-07} \qquad (2.5)$$

4. Once 2020_075 98th percentile design values were calculated, changes in concentrations relative to emissions (ppb/ton) between 2020_075 and 2006 were calculated as:

$$ppb / ton = \left(DV_{2020\_075} - DV_{2005-2007}\right) \Big/ \left(E_{2020\_075} - E_{2006}\right) \qquad (2.6)$$

2.3.2.2 Near-road adjustment of area-wide design values

Once 2020 area-wide design values were calculated, they were adjusted to simulate near-road concentrations.

2.3.2.2.1 Identification of monitors for adjustment

To identify monitors that, accounting for the gradient in concentrations away from the roadway, could inform near-road conditions, OAQPS used (1) monitor characteristics (i.e., metadata) in the AQS database, (2) visual inspection by using Google Earth geospatial software, and (3) the condition that only Core Based Statistical Areas (CBSAs) with populations of 350,000 or greater would be required to have at least one maximum concentration site near roadways consistent with the final NO_2 NAAQS rulemaking.

We first select "area-wide" monitors to adjust to approximate near-roadway conditions. The monitors included in this analysis are those considered to be representative of "area-wide" conditions; i.e. those monitors to which it would be appropriate to apply the gradient to scale from area-wide to near-roadway conditions. Specifically, we did not select monitors that are microscale or middle scale, source oriented, non-EPA (one federal monitor in Yosemite National Park), or those affected by a dominant source, including roadways, in this analysis[4].

[4] This process excluded no monitoring sites; it merely identified those monitors relevant to adjust for a near-roadway approximation. Monitors not selected for adjustment were still included in the overall analysis.

Next, to address the limitations of the monitors' metadata, we conducted a visual inspection and geospatial analysis using Google Earth of the remaining monitors. The analysis reviewed where the site was physically located in an urban area, checked its proximity to major roads (such as interstates, freeways, and major arterial roads), and its proximity to identifiable sources such as industrial complexes and facilities, commercial facilities (such as trucking depots), or proximity to other area sources (such as airports or shipping ports).

Finally, we did not scale up any sites that were not in CBSAs with a population of 350,000 or greater to be consistent with the population based thresholds that trigger minimum required near-road monitors in the NO_2 NAAQS and monitoring package.

Using the list of area-wide monitors appropriate for near-roadway adjustment, we included only those monitors with sufficient data completeness to estimate a 2020 design value (see Section 2.3.1 for details). One hundred seventy-three monitors were considered appropriate for near-road adjustment and eighty-two were considered inappropriate for scaling up. For more details about the monitor selection methodology see Appendix 2a, and for the full list of monitors with criteria see Table 2-3a of Appendix 2a.

2.3.2.2.2 Adjustment methodology

Reflecting scientific literature on the roadway gradient discussed in the final NAAQs rule's preamble (i.e., near road monitors can be from 30% to 100% greater than the area wide monitors), we adjust our estimated design values at area-wide locations for the future year of 2020 by 130%, 165%, and 200% (30%, 65%, and 100% gradients respectively).

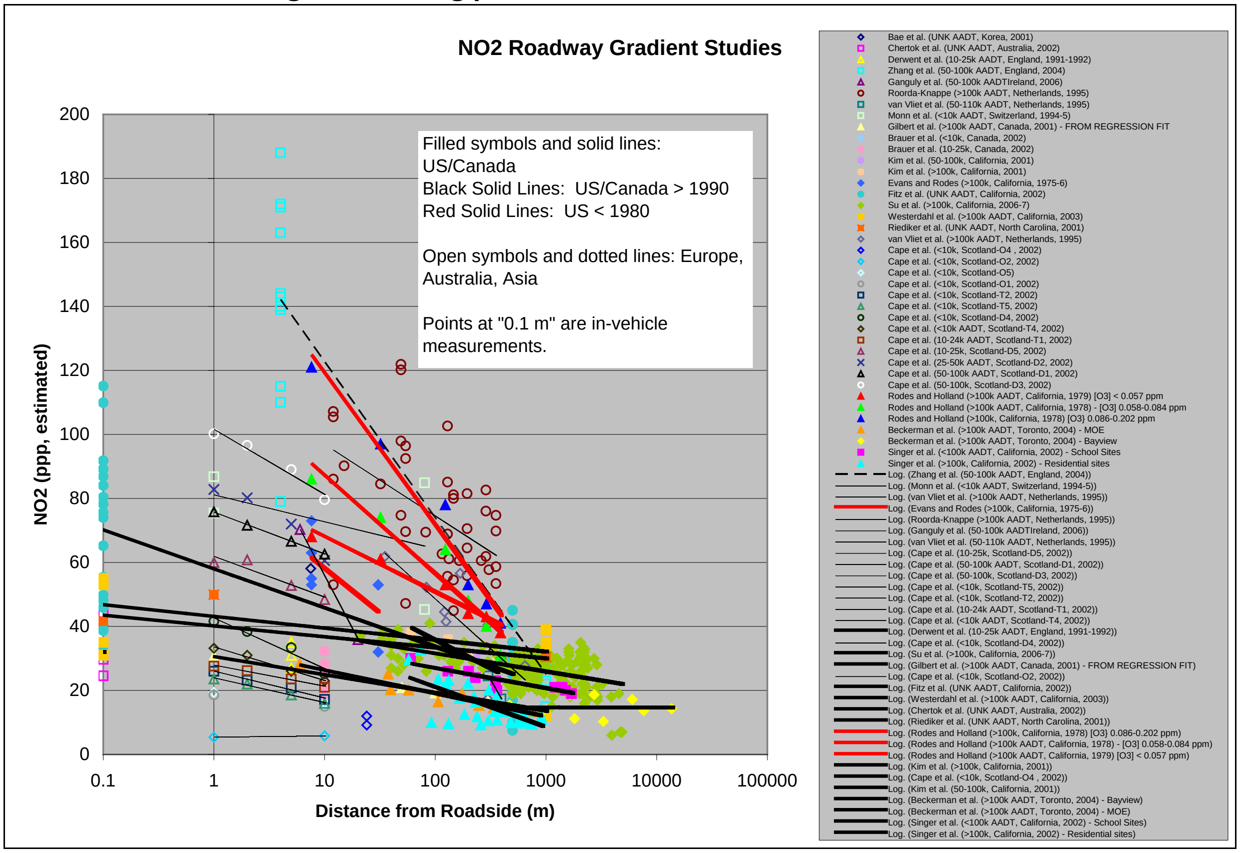
One significant limitation of attempting to approximate near road conditions by simply multiplying by the gradient alone is that the range may not account for the expected future design values near roads (i.e., we believe this approach may over-estimate future design values near roads and may suggest that the future nonattainment problem is worse than it might be, and that the costs and benefits of addressing the residual nonattainment problem in the future are greater than they will actually be). This potential overestimation results from the fact that CMAQ averages the reductions from all sources over the 12km grid which effectively smoothes the concentration changes of source-specific emissions reductions that would have a greater effect at any specific location within the grid, e.g., mobile source emissions reductions near roads. We presume that future near-roadway peaks are reduced more than future area-wide peaks because (1) the near-road proxy monitors are by definition located near the roadway; and (2) on-road mobile source emission reductions between 2006 and 2020 are expected to be significant due to a number of previously-cited Federal mobile source regulations. This suggests that we should consider an appropriate adjustment of the 2020 design values at 'near roadway' proxy monitors to account for the dilution of mobile emission reductions across entire grid squares by CMAQ.

To adjust for the fact that air quality peak design values near roadways will be affected more significantly by mobile source emission reductions than will air quality peak design values in area-wide locations, we start with the design values adjusted to account for the near road gradients described previously and , based on available data, we calculated a relative effectiveness metric for each county reflecting the greater efficacy of mobile source emissions reductions (i.e., ppb/ton) at those locations than predicted by CMAQ for area wide monitor locations. We then apply the resulting national average metric (1.20) across all monitors calculated above to adjust the 2020 design values at the 'near-roadway' proxy monitors consistently.

Reviews of roadway studies indicate that a second adjustment is also reasonable. An analysis of U.S. studies before 1980 and U.S. and Canadian studies after 1990 indicate that the slope of the concentration gradient from the roadway becomes less steep with time (Figure 2-5). The red lines are the concentration gradients for U.S. studies before 1980, while the black lines are concentration gradients for U.S. and Canadian studies after 1990. The black lines are flatter than the red lines, indicating that with time, concentration gradients may decrease. Average NO2 concentrations for US and Canadian studies from 1970-1979 and 2000-2009 for AADT > 100K, show concentration gradient changes from approximately 65% (1970-79) to approximately 30% (2000-2009) for concentrations > 200 m from road when compared to concentrations < 50 m from road. In other words, in the 1970-1979 period, concentrations near the road (50 m) would be 165% higher than concentrations farther from the road (200 m). In 2000-2009, concentrations near the road (50 m) would be 130% higher than concentrations farther from the road (200 m). The difference between the gradients in this context is then approximately 20%. Therefore, the change in gradients with time provides justification for our use of a factor of 1.2 to adjust the required reductions in roadside emissions downward.[5]

[5] That the two adjustment factors have the same value is coincidental.

Figure 2-5: Log plots of NO_2 vs. distance from roadside



While we believe this approach is conceptually sound, it is a new methodology developed out of necessity to complete this assessment for near-roadway monitor locations in the absence of such a monitoring network and based on limited data and modeling results, i.e., information not designed to address near-road situations. Furthermore, the use of a national average adjustment as opposed to a county-specific adjustment makes the adjustment more straight forward but does result in some specific under- and over-adjustments at particular locations.

Following is the methodology to develop the national adjustment factor, 1.20 for the adjustment of the 2020 area-wide design values to near-road design values. The national adjustment factor is based on the use of the 98th percentile design values for 2005-2007 and 2020. The following calculations were performed for monitors that were appropriate for scaling:

1. First we calculated the 2005-2007 ($DV_{on:2005\text{-}2007}$) and 2020 ($DV_{on:2020}$) onroad components of the 2005-2007 and 2020 98th percentile area-wide design values by

multiplying the area-wide design values by the ratio of county onroad ($E_{onroad:2006}$ and $E_{onroad:2020}$) to county total emissions ($E_{total:2006}$ and $E_{total::2020}$) for each scalable monitor:

$$DV_{on:2005-2007} = DV_{2005-2007} \times \frac{E_{onroad:2006}}{E_{total:2006}} \tag{2.9}$$

$$DV_{on:2020} = DV_{2020} \times \frac{E_{onroad:2020}}{E_{total:2020}} \tag{2.10}$$

The county emissions for both 2006 and 2020 are the county emissions used to calculate the 2020 area-wide design values as described in Section 2.3.2.1. The 2020 emissions are the 2020 emissions used to meet the 0.075 ppm ozone standard [See Chapter 4 of the ozone RIA (EPA, 2008)].

2. After calculating the onroad components of the area-wide design values for 2005-2007 and 2020, the onroad ppb/ton estimate, ppb/ton_{onroad}, was calculated as:

$$ppb/ton_{onroad} = \frac{DV_{on:2020} - DV_{on:2005-2007}}{E_{on:2020} - E_{on:2006}} \tag{2.11}$$

3. Next, the ratio of onroad to total ppb/ton metric, $Ratio_{ppb/ton}$ was calculated as:

$$Ratio_{ppb/ton} = \frac{ppb/ton_{onroad}}{ppb/ton_{total}} \tag{2.12}$$

Where ppb/ton_{onroad} is as defined above and ppb/ton_{total} is defined as in Equation 2.6 of Section 2.3.2.1.

4. Finally, we calculate the national average of $Ratio_{ppb/ton}$ across all monitors appropriate for scale-up as

$$\frac{\sum_{i=1}^{N} Ratio_{ppb/toni}}{N} = 1.2 \tag{2.13}$$

Where N is the number of monitors appropriate for scale-up

To simplify the analysis, we used the average $Ratio_{ppb/ton}$ in step 4 above across all scalable monitors in the final adjustment for the near-road proxy monitors. The national average ratio was calculated as 1.2, meaning that onroad emissions reductions were approximately 20% more effective at reducing near-roadway concentrations than total emission reductions in the county.

After calculating the national average ratio in step 3, the final near-roadway adjusted 2020 design values were calculated as:

$$DV_{NR:GRAD} = \frac{DV_{2020:} \times GRAD}{1.2} \tag{2.14}$$

Where $DV_{NR2:GRAD}$ is the 2020 near-roadway adjusted concentration for each gradient with GRAD equal to 1.3, 1.65, or 2 (i.e., reflecting 30%, 65%, or 100% increase respectively), and DV_{2020} is the 2020 area-wide design value for the 98th percentile. The 1.2 factor in the denominator is the national average ratio calculated in Equation 2.13. For the eighty-two monitors that were not deemed appropriate for adjustment, the near-road design value was set equal to the 2020 area-wide design value.

Once the near-roadway design values were calculated for 2020 for each of the three gradient increases (30%, 65%, and 100%), residual concentration improvements needed to result in levels below the NAAQS were calculated for three alternative levels of the standard (in ppb): 80, 100, and 125. Nonattainment was calculated as:

$$NA_{GRAD:AS} = DV_{NR:GRAD} - AS \tag{2.15}$$

Where $NA_{GRAD:AS}$ is the residual nonattainment (ppb) for GRAD equal to 30, 65, or 100% increase for alternative standard AS of 80, 100, or 125 ppb and $DV_{NR:GRAD}$ is the 2020 near-roadway adjusted design value for the 30%, 65%, or 100% increase for the 98th percentile. For locations exceeding a particular alternative standard AS, the mobile tons needed to reach attainment are calculated as:

$$Tons_{GRAD:AS} = \frac{NA_{GRAD:AS}}{\left(ppb / ton \times 1.2\right)} \tag{2.16}$$

Where $Tons_{GRAD:AS}$ are the tons needed for attainment of alternative standard for the near-roadway increase of 30%, 65%, or 100%, $NA_{GRAD:AS}$ is defined in Equation 2.15 above, and ppb/ton is the total (all county emissions) ppb/ton for the 98th percentile design value as calculated in Equation 2.8. The total ppb/ton is multiplied by 1.20 in Equation 2.16 to approximate the onroad ppb/ton based on the national average ratio of onroad ppb/ton to total ppb/ton calculated in Equation 2.13. While, each monitor had its own value of onroad ppb/ton estimates as calculated in Equation 2.11, in order to maintain consistency with the 1.2 adjustment factor (the ratio of onroad ppb/ton to total ppb/ton), the ppb/ton estimate for each monitor was multiplied by 1.2 to approximate the onroad ppb/ton. For locations below a particular alternative standard, AS, tons for control were not calculated and additional emission controls were not needed.

A complete list of 2020 projected design values by monitor can be found in Table 2-1a of Appendix 2a.

2.4 Results

2.4.1 Nonattainment of alternative standards

Figure 2-6 shows the projected design values for 2020 for the 98th percentile NO_2 design value concentrations for the most extreme case, 100% gradient. Shown are the highest projected design values for each county. Counties in white were below the lowest alternative standard, 80 ppb. It should be noted all of the non-adjusted monitors were below 80 ppb. Table 2-4 shows the number of monitors and counties that exceeded the alternative standards for the three gradient increases.

Figure 2-6: 2020 98th percentile design values for the 100% gradient increase

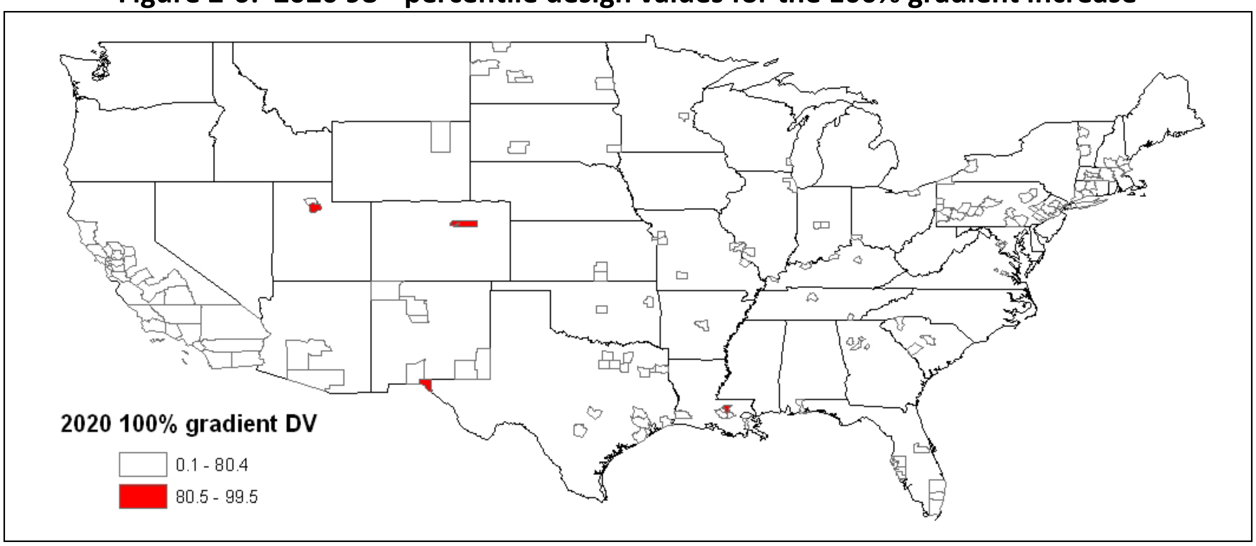


Table 2-4: Summary of 2020 near-road design values exceeding alternative standards for gradient increases

Gradient (%)	Alternative standard	Number of Monitors	Number of Counties
30	80	0	0
	100	0	0
	125	0	0
65	80	1	1
	100	0	0
	125	0	0
100	80	5	4
	100	0	0
	125	0	0

The one county that exceeded 80 ppb for the 65% increase was Adams County, CO with a design value of 82.0 ppb and we estimated a reduction in onroad emissions of 676 tons were needed to attain 80 ppb. The counties that after adjustment for the 100% gradient had NO_2 ambient concentrations projected to be above 80 ppb are shown in Table 2-5.

Table 2-5: Nonattainment counties for 80 ppb for 100% gradient. Onroad mobile tons needed for attainment are also listed

			Tons for control
CO	Adams	99.5	9,861
TX	El Paso	95.8	8,643
UT	Salt Lake	89.0	4,088
LA	East Baton Rouge	80.8	456

While the counties in Table 2-5 were predicted to be in nonattainment in 2020 after adjusting to near-road monitors, there were other sources or events influencing the concentrations before near-road adjustments. In Adams County, CO, the monitor was near a large EGU source and several non-EGU point sources. In El Paso County, TX the monitors were near the international border between the U.S. and Mexico. El Paso is explained in more detail in Section 2.4.2.2. Salt Lake County, UT appeared to be influenced by seasonal inversions, which can lead to higher surface concentrations. In East Baton Rouge, LA, the violating monitor was in the downtown area and located near several non-EGU point sources.

It should be noted that different values of the gradient may be more appropriate for some monitors than other values of the gradient. Many of the monitors may be more influenced by stationary sources than onroad sources or the distance from the roadway may justify the use of a lower gradient. For example, the Charles City County, VA monitor is not located near major roads (within 1 mile), so the 30% gradient may be more appropriate to apply than 65% or 100%. Also, one monitor in Los Angeles County is near the Long Beach Port and Long Beach Municipal Airport. The monitor is located within 500 m of the nearest roadway and most likely already has an influence from the road, so the 30% or 65% gradient may be more appropriate than 100%. However, it should be noted that neither of these monitors exceeded 80 ppb in 2020 when the 100% gradient was applied.

2.4.2 Special cases

After projection of 2005-2007 design values to 2020, some notable results were seen. This section describes the reasons for those values.

2.4.2.1 Non-calculated projected design values

For sixteen monitors (eleven counties), the projected 2020 design values were not calculated for the 98th percentile concentrations (see 2020 concentrations denoted by “*” Table 2a-3 in Appendix 2a). Ten of the counties were in California and one in Pennsylvania. These counties were in regions that were not forecast to meet the 0.070 or 0.075 ozone standard as described in Chapter 4 of the ozone RIA (U.S. EPA, 2008b). These counties received across the board reductions in NOx in addition to the reductions included in the 0.070 ozone analysis. . In the California counties, the 2020_075 emissions were 20% of the 2020_070 emissions, while in Pennsylvania, the 2020_075 emissions were 13% of the 2020_070 emissions. For more details about the emissions reduction see Chapter 4 of the ozone RIA (U.S. EPA, 2008b). Concentrations could not be calculated because 2020_075 emissions were so low that the methodology described in Section 2.3.2.1 did not produce reasonable results. All of the monitors in question were already below the lowest alternative standard of 80 ppb in 2005-2007, so these monitors should not have issues with nonattainment.

2.4.2.2 El Paso County

El Paso County represents a case where future design values for NO_2 above the levels being considered are influenced by international emissions. The 2005-2007 98th percentile design values are shown in Figure 2-7. The three monitors in the black circle were the highest monitors. The 2020 98th percentile design values are shown in Figure 2-8. Area-wide and near-road adjusted projected design values are shown. One monitor was not considered appropriate for adjustment and has no near-road design value listed.

Figure 2-7: 2005-2007 98th percentile design values (ppb) for El Paso County

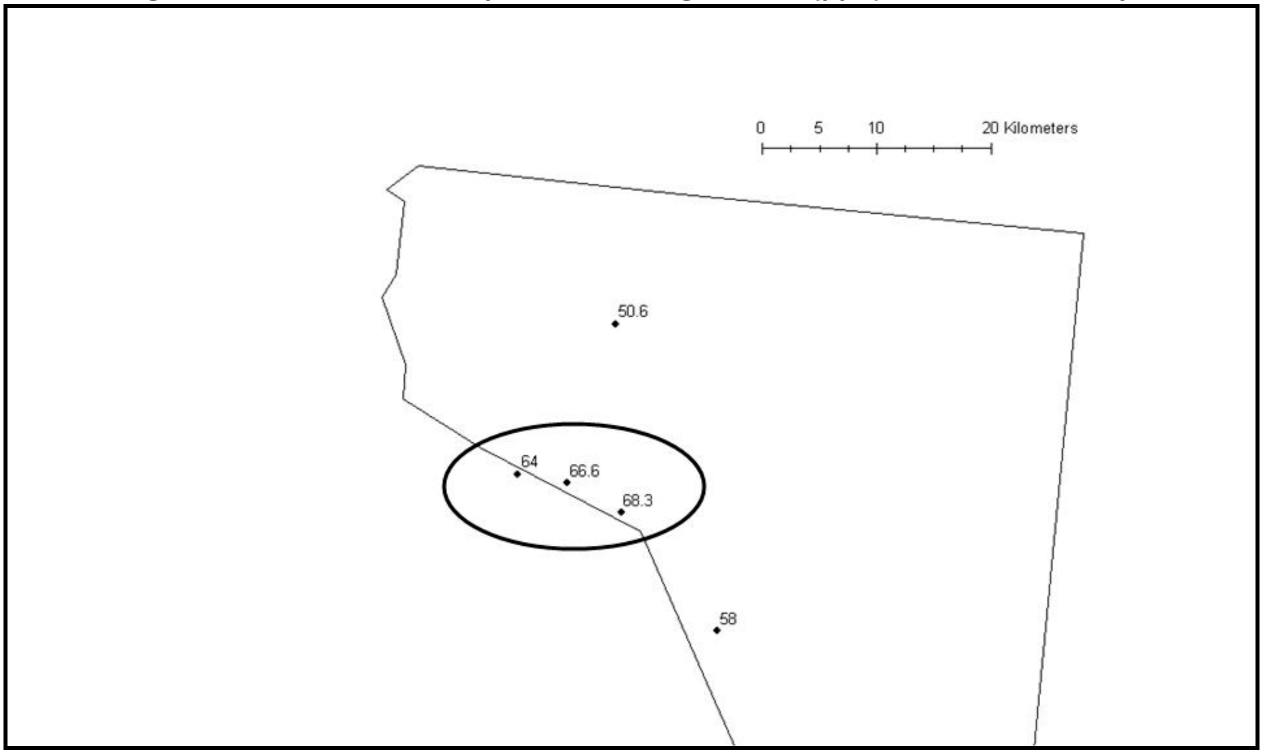
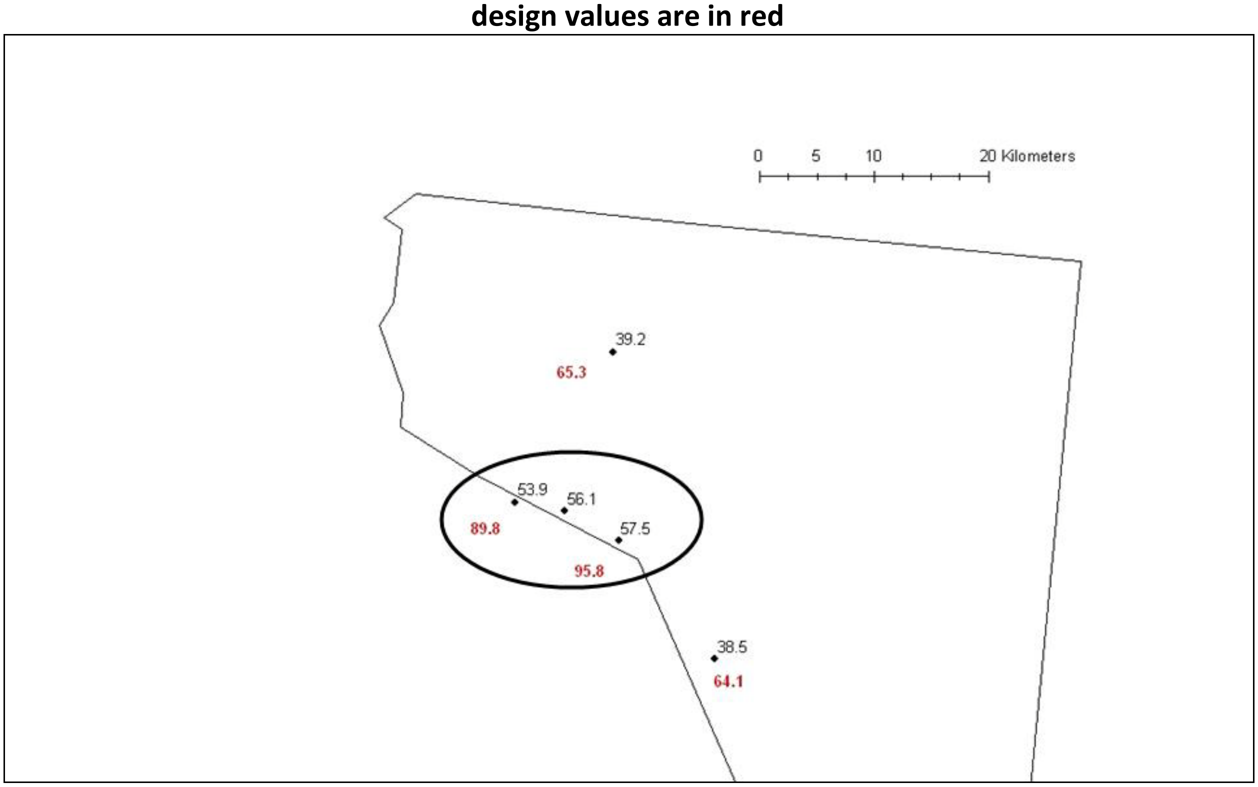
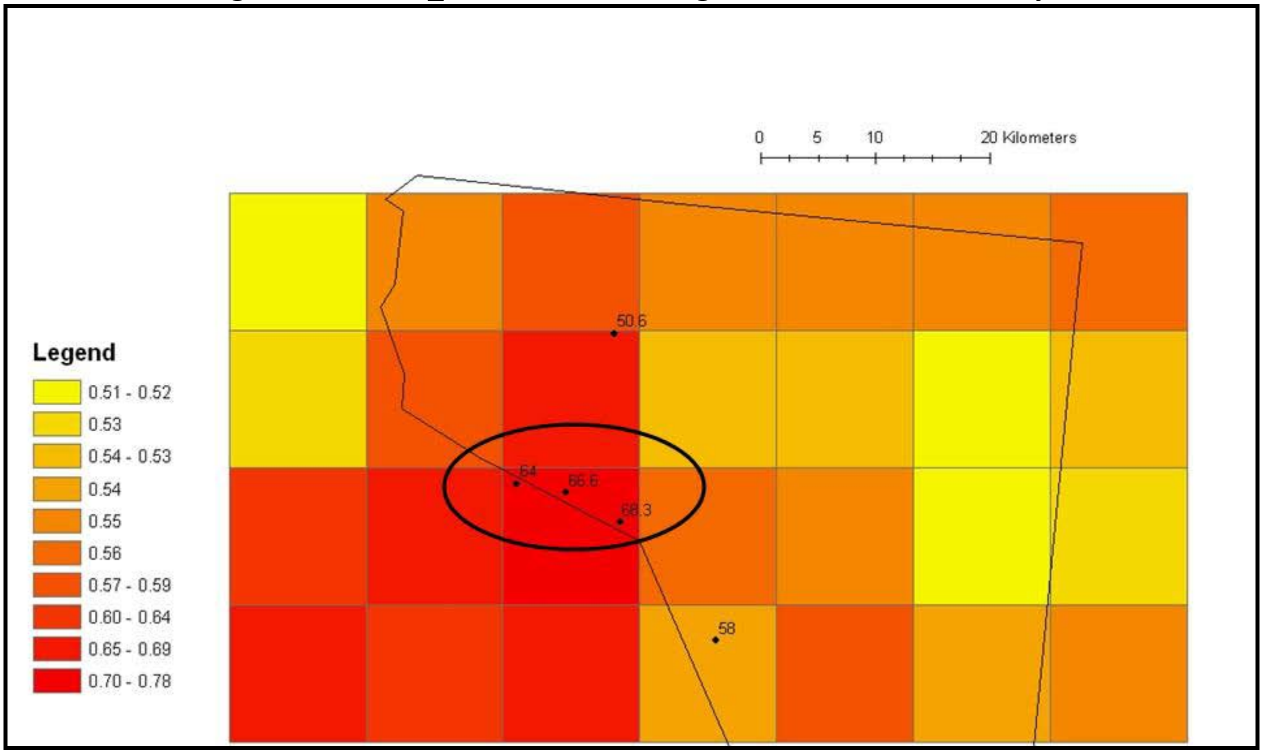


Figure 2-8: 2020 98th percentile design values (ppb) for El Paso County. Area-wide design values are in black and for monitors that were scalable, 100% gradient adjusted near-road design values are in red



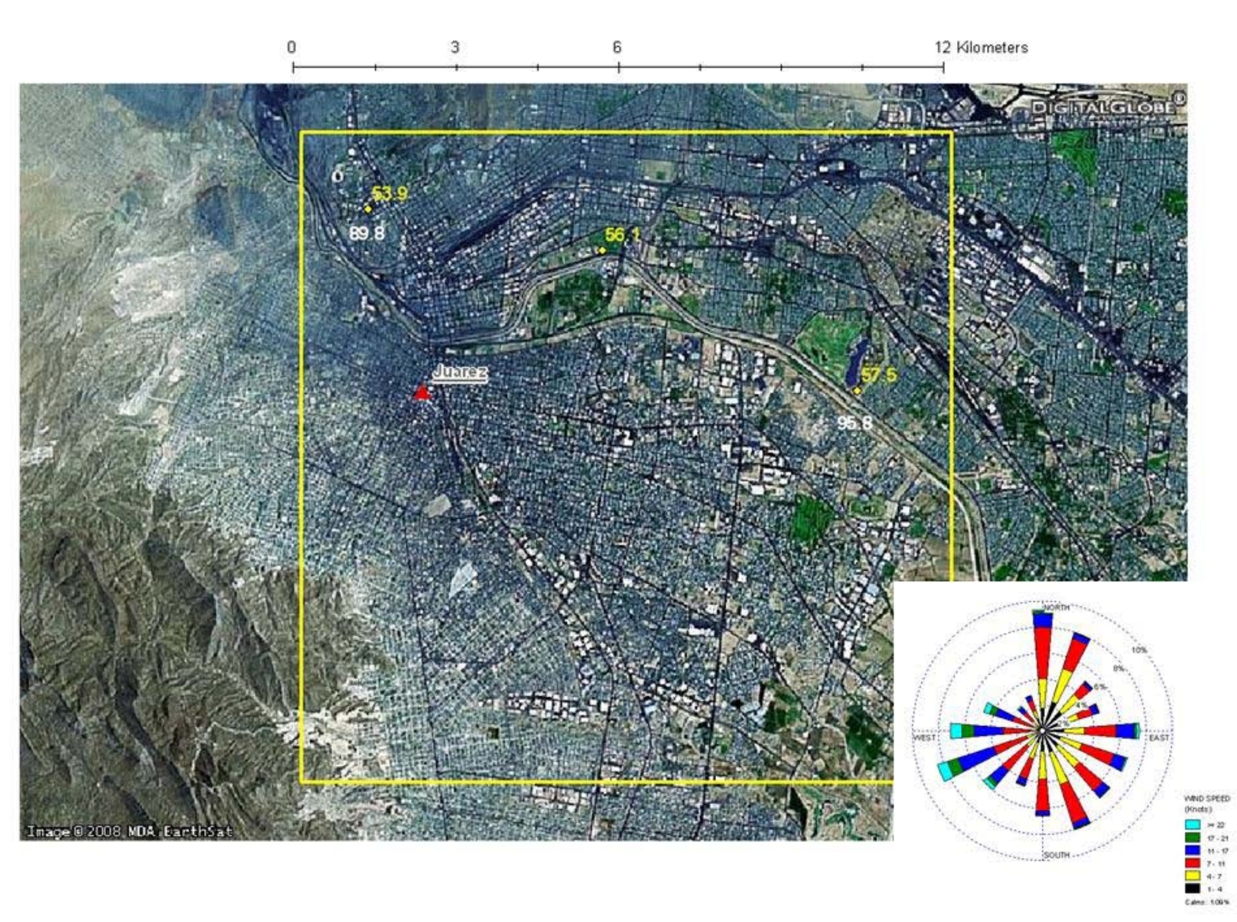
In 2020, two of the near-road design values of near-road adjusted monitors exceeded 80 ppb for the 100% gradient adjustment, 89.8 ppb and 95.8 ppb (Figure 2-8). Examining the average of the top ten daily 1-hour maximum concentrations for 2002 and the average of the daily 1-hour maximum concentrations for the same ten calendar days in 2020 showed that the grid cell containing the top two nonattainment monitors was the highest value among the grid cells in the county containing monitors, 65.6 ppb for 2002 and 51.3 for 2020 (not shown). The resulting RRF was also the highest, 0.78 (Figure 2-9) and the mean daily 1-hour maximum concentration in 2020 was also highest for the county, 31.3 ppb.

Figure 2-9: 2020_070 RRF values for grid cells in El Paso County



Note that these monitors were not only located along the border highway, but they were also very close to the international border with city of Juarez just to the southwest (Figure 2-10). A wind rose from El Paso Airport for 2005-2007 exhibited a relatively high frequency of winds from the east-southeast through west-southwest that would transport pollutants from Juarez toward the three NO_2 monitoring sites across the river in El Paso. The grid cell that contained the two highest monitors is mostly in Mexico. Emissions from across the international border could impact the modeled concentrations of the grid cells containing the monitors. However, for our emission inventories, we do not forecast controls on international emissions over which we have no jurisdiction.

Figure 2-10: Aerial photograph of CMAQ grid cell containing nonattainment monitors for El Paso County. Yellow box is 12 x 12 km grid cell and El Paso 2005-2007 wind rose is shown in lower right corner. Area-wide design values are in yellow and near-road adjusted design values are in white



In summary:

- Two monitors in El Paso County were the highest monitors in the 2005-2007 and 2020 98th percentile design values in the county.
- The grid cell containing the monitors had the highest average of the top 10 daily 1-hour maximum concentrations for 2002 for grid cells containing monitors in El Paso County.
- Also, the monitors' grid cell had the highest average of the 2020_070 daily 1-hour maximum concentrations for the same days as the ten days in the average of the 2002 daily 1-hour maximum concentrations.
- The monitors' grid cell had the highest RRF value for all monitor grid cells in the county.
- Since all of the monitors in the county used the same 2002, 2006, 2020_070, and 2020_075 emissions for emissions RRF calculations (Equations 3.2 and 3.3), the driving factor was the high RRF for the grid cell.

- The grid cell contained international emissions and were not controlled in the 2020_070 inventory, resulting in higher daily 1-hour maximum concentrations when compared to other monitor grid cells.

2.5 Summary

In summary, 2020 NO_2 design value concentrations were projected from 2005-2007 observed design values using CMAQ output from the 2002 and the 2020_070 scenario simulations performed for the ozone NAAQS RIA (U.S. EPA, 2008b). County emissions for 2002, 2006, and 2020 were used in conjunction with the CMAQ output to project the 2005-2007 design values for the 2020 area-wide design values. The 2020 area-wide design values from appropriate monitors were then adjusted to (1) reflect a near-roadway network of monitors using gradient increases of 30%, 65%, and 100%; and (2) to reflect the efficacy of controls on onroad mobile emissions in the future. The 2020 near-roadway concentrations were then compared against three alternative standards of 80, 100, and 125 ppb for each of the three gradient increases. No counties exceeded 80 ppb for the 30% gradient, one county exceeded 80 ppb for the 65% gradient, and four counties exceeded 80ppb, for the 100% gradient. No counties exceeded 100 ppb for any of the three gradients.

2.6 References

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Appendix 2a: Monitor adjustment selection, Roadway field studies, 2005-2007 and projected 2020 Design Values

2a.1 Monitor adjustment selection

OAQPS applied several screening techniques in the effort to select monitors within the NO_2 monitoring network that would be appropriate to simulate what a near-road monitor might record. OAQPS used monitor site characteristics and visual inspection using Google Earth geospatial software to determine which of the monitor sites were appropriate to simulate near-road monitors. We then screened that list of monitors so that only those located in Core Based Statistical Areas (CBSAs) with populations of 350,000 or greater, which corresponds to the proposed population threshold in the NO_2 NAAQS and monitoring proposal package, would be scaled-up.

All NO_2 monitoring sites that are used for comparison to the NAAQS report their data to the Air Quality System (AQS). Each monitoring site has a profile in AQS containing metadata pertaining to the monitor, including where the monitor is located, the monitoring objective, the scale of representativeness, and whether it is thought to be influenced by a particular type of emission source, among other data metrics. Although, the metadata in AQS are informative, we must note that AQS metadata should be used with caution as there are no formal requirements for the responsible state and local air monitoring agencies that operate the monitoring network to quality assure or update metadata at any frequency.

In conjunction with the language in the NO_2 NAAQS and monitoring proposal package, this exercise was intended to only use “area-wide” monitors to simulate near-road concentrations. Area-wide monitors are monitors that are not significantly influenced by point, area, or mobile sources, meaning they typically do not represent the maximum concentration that may be attributable to a source or sources. Further, area-wide sites and are sited to represent neighborhood, urban, and regional spatially representative scales. To identify which sites in the NO_2 network were suitable to classify as an “area-wide” site, we screened sites utilizing three particular AQS metadata metrics: 1) monitor objective, 2) spatial (measurement) scale, and 3) dominant source.

The monitor objective meta-data field describes what the data from the monitor are intended to characterize. The focus of the data presented is to show the nature of the network in terms of its attempt to generally characterize health effects, photochemical activity, transport, or welfare effects. There are 11 categories of monitor objective for a NO_2 monitor within AQS. The first six categories listed below stem directly from categorizations of site types within the Code of Federal Regulations (CFR). In 40 CFR Part 58 Appendix D, there are seven examples of NO_2 site types:

1. Sites located to determine the highest concentration expected to occur in the area covered by the network (Highest Concentration).
2. Sites located to measure typical concentrations in areas of high population (Population Exposure).
3. Sites located to determine the impact of significant sources or source categories on air quality (Source Oriented).
4. Sites located to determine general background concentration levels (General Background).
5. Sites located to determine the extent of regional pollutant transport among populated areas; and in support of secondary standards (Regional Transport).
6. Sites located to measure air pollution impacts on visibility, vegetation damage, or other welfare-based impacts (Welfare Related Impacts).
7. Sites with unspecified or non-routine monitor objectives (Other).

The remaining four categories available are a result of updating the AQS database. In the more recent upgrade to AQS, the data handlers inserted the available site types for the Photochemical Assessment Monitoring Stations (PAMS) network. These PAMS site types are spelled out in 40 CFR Part 58 Appendix D:

1. Type 1 sites are established to characterize upwind background and transported ozone and its precursor concentrations entering the area and will identify those areas which are subjected to transport (Upwind Background).
2. Type 2 sites are established to monitor the magnitude and type of precursor emissions in the area where maximum precursor emissions are expected to impact and are suited for the monitoring of urban air toxic pollutants (Max. Precursor Impact).
3. Type 3 sites are intended to monitor maximum ozone concentrations occurring downwind from the area of maximum precursor emissions (Max. Ozone Concentration).
4. Type 4 sites are established to characterize the downwind transported ozone and its precursor concentrations exiting the area and will identify those areas which are potentially contributing to overwhelming transport in other areas (Extreme Downwind).

It should be noted that any particular monitor can have multiple monitor objectives. For this screening exercise, we selected one reported monitor objective based on a hierarchy to represent an individual monitor. The hierarchy used was to select, in order of priority: 1) source oriented, 2) high concentration, 3) population exposure, or 4) general background, if they existed at a site with multiple monitoring objectives. So, for example, any monitor with "source oriented" among multiple objectives was classified as "source oriented".

The spatial (measurement) scales are also defined in 40 CFR Part 58, Appendix D. This regulation language spells out what data from a monitor can represent in terms of air volumes associated with area dimensions where:

> Microscale – Defines the concentration in air volumes associated with area dimensions ranging from several meters up to about 100 meters.
>
> Middle scale – Defines the concentration typical of areas up to several city blocks in size, with dimensions ranging from about 100 meters to 0.5 kilometers.
>
> Neighborhood scale – Defines concentrations within some extended area of the city that has relatively uniform land use with dimensions in the 0.5 to 4.0 kilometers range.
>
> Urban scale – Defines concentrations within an area of city-like dimensions, on the order of 4 to 50 kilometers. Within a city, the geographic placement of sources may result in there being no single site that can be said to represent air quality on an urban scale. The neighborhood and urban scales have the potential to overlap in applications that concern secondarily formed or homogeneously distributed air pollutants.
>
> Regional scale – Defines usually a rural area of reasonably homogeneous geography without large sources, and extends from tens to hundreds of kilometers.

Therefore the meta-data records for the NO_X network in AQS indicate what the measurement scale of a particular monitor represents. It is important to note that a monitor can only have one measurement scale, as opposed to the possibility of a single monitor having multiple monitor objectives.

The "dominant source" metric in AQS allows responsible state and local air monitoring agencies to identify, if applicable, what type of emission source may be the dominant source influencing the measurements at a particular site. There are three choices for the dominant source category: 1) Point, 2) Area, and 3) Mobile. It should be noted that not all NO_2 monitor records have a value in the dominant source field, either because the responsible state and local monitoring agency does not believe any particular type of source is influencing a particular site, or because the information was simply not entered into the database.

For the first screening to identify area-wide NO_2 monitoring sites, we chose to exclude all sites that met one or more of the following criteria based on AQS metadata:

- Any microscale site (measurement scale)
- Any middle scale site (measurement scale)
- Any source oriented site (monitor objective)
- Any site with the following combination of metadata: Highest Concentration, Neighborhood scale, and Point source dominated
  (monitor objective/measurement scale/dominant source)
- Any site identified as being operated by industry, as these sites are usually micro or middle scale, source oriented sites.

As a result of the first screening, of the original 255 sites used in the area-wide design value calculations in Section 2.3.2.1 of Chapter 2, sixteen were excluded from scaling due to negative design value calculations. For the sixteen sites (eleven counties), the projected 2020 design values were not calculated for the 98th percentile concentrations. Ten of the counties were in California and one in Pennsylvania. These were counties that were in regions that were not forecast to meet the 0.075 ozone standard as described in Chapter 4 of the ozone RIA (U.S. EPA, 2008b). These counties received across the board reductions in NOx in addition to the reductions included in the 0.070 ozone analysis. In the California counties, the 2020_075 emissions were 20% of the 2020_070 emissions, while in Pennsylvania, the 2020_075 emissions were 13% of the 2020_070 emissions. For more details about the emissions reduction see Chapter 4 of the ozone RIA (U.S. EPA, 2008b). Concentrations could not be calculated because 2020_075 emissions were so low that the methodology described in Section 3.3.1 did not produce reasonable results. Most of the sites in question were already below the lowest alternative standard of 65 ppb in 2005-2007, so these monitors should not have issues with nonattainment. After exclusion of the 16 sites and sites based on AQS metadata (22 sites), 217 sites remained in use for the second screening process.

The second screening process was by visual inspection and geospatial analysis using Google Earth of the top eleven NO_2 sites, ranked by estimated ppb/ton and two other monitor sites located in counties with multiple monitoring sites that had higher estimated ppb/ton values. The analysis reviewed where the site was physically located in an urban area, checked its proximity to major roads (such as interstates, freeways, and major arterial roads), and its proximity to identifiable sources such as industrial complexes and facilities, commercial facilities (such as trucking depots), or proximity to other area sources (such as airports or shipping ports). As a result, three more sites were excluded from the pool of NO_2 sites that were to be allowed to be scaled-up to simulate near-road monitoring sites.

The final screening was to remove any sites that were not in CBSAs with a population of 350,000 or greater. This was done to match the proposed population-based thresholds that trigger minimum required near-road monitors in the NO_2 NAAQS and monitoring proposal package. This screening removed 41monitors, leaving 181 monitors to use in the simulation.

2a.2 2005-2007 and 2020 design values

Table 2a-1 lists the CBSAs of monitors used in the analyses. Also listed in Table 2a-1 are population and number of monitors per CBSA. Table 2a-2 lists the CBSAs with populations greater than 350,000 people that do not have monitors in the analyses. The reasons for no monitors is also given. Those with the reason "Monitors excluded due to data completeness" have monitors but the monitors did not meet the completeness criteria discussed in Section 2.3.1 of Chapter 2. Table 2a-3 lists the 2005-2007 design values used in projecting 2020 design values. 2020 design values denoted by "*" were monitors where a projected design value could not be calculated. 2020 design values for various values of the near-road gradient are shown. For monitors that were not justified to scale up, the 2020 design values are equal to the 2020 area-wide design value. Monitors determined to be appropriate for scale up are listed as "SCALE UP" in the scale up column of Table 2a-1. The reasons for no scale up of the 2020 design values are given for the negative design values ("NO SCALE UP: NEGATIVE"), visual inspection ("NO SCALE UP: VISUAL NEAR ROAD"), population ("NO SCALE UP: POP < 350K), and due to AQS metadata (various reasons).

Table 2a-1: Number of monitors per CBSA

CBSA	TYPE	2007 Population	Monitors
New York-Northern New Jersey-Long Island, NY-NJ-PA	Metropolitan	19,113,887	8
Los Angeles-Long Beach-Santa Ana, CA	Metropolitan	13,192,758	13
Chicago-Naperville-Joliet, IL-IN-WI	Metropolitan	9,747,870	4
Dallas-Fort Worth-Arlington, TX	Metropolitan	6,118,183	8
Philadelphia-Camden-Wilmington, PA-NJ-DE-MD	Metropolitan	5,930,083	2
Houston-Sugar Land-Baytown, TX	Metropolitan	5,620,734	10
Miami-Fort Lauderdale-Miami Beach, FL	Metropolitan	5,607,038	3
Washington-Arlington-Alexandria, DC-VA-MD-WV	Metropolitan	5,451,302	5
Atlanta-Sandy Springs-Marietta, GA	Metropolitan	5,322,915	3
Boston-Cambridge-Quincy, MA-NH	Metropolitan	4,515,779	5
San Francisco-Oakland-Fremont, CA	Metropolitan	4,316,905	9
Phoenix-Mesa-Scottsdale, AZ	Metropolitan	4,163,757	5
Riverside-San Bernardino-Ontario, CA	Metropolitan	4,152,464	7
Minneapolis-St. Paul-Bloomington, MN-WI	Metropolitan	3,313,789	1
San Diego-Carlsbad-San Marcos, CA	Metropolitan	3,064,142	5
St. Louis, MO-IL	Metropolitan	2,833,676	5
Tampa-St. Petersburg-Clearwater, FL	Metropolitan	2,765,528	4
Denver-Aurora, CO	Metropolitan	2,469,929	1
Pittsburgh, PA	Metropolitan	2,404,190	7
Cleveland-Elyria-Mentor, OH	Metropolitan	2,150,129	2
Sacramento--Arden-Arcade--Roseville, CA	Metropolitan	2,141,388	6
Cincinnati-Middletown, OH-KY-IN	Metropolitan	2,118,580	1
Orlando-Kissimmee, FL	Metropolitan	2,098,102	1
Kansas City, MO-KS	Metropolitan	1,997,567	3
San Antonio, TX	Metropolitan	1,985,996	3
San Jose-Sunnyvale-Santa Clara, CA	Metropolitan	1,829,059	1
Indianapolis-Carmel, IN	Metropolitan	1,701,870	2
Austin-Round Rock, TX	Metropolitan	1,569,880	1
Milwaukee-Waukesha-West Allis, WI	Metropolitan	1,534,473	1
Nashville-Davidson--Murfreesboro, TN	Metropolitan	1,507,461	1
Louisville-Jefferson County, KY-IN	Metropolitan	1,247,196	1
Richmond, VA	Metropolitan	1,215,134	2
Hartford-West Hartford-East Hartford, CT	Metropolitan	1,203,355	1
Oklahoma City, OK	Metropolitan	1,198,114	2
Buffalo-Niagara Falls, NY	Metropolitan	1,152,143	1
New Orleans-Metairie-Kenner, LA	Metropolitan	1,084,072	1
Salt Lake City, UT	Metropolitan	1,073,432	1
Tucson, AZ	Metropolitan	976,521	2
Bridgeport-Stamford-Norwalk, CT	Metropolitan	918,315	1
Fresno, CA	Metropolitan	915,824	5
New Haven-Milford, CT	Metropolitan	852,576	1
Albuquerque, NM	Metropolitan	833,634	3
Oxnard-Thousand Oaks-Ventura, CA	Metropolitan	827,163	2
Allentown-Bethlehem-Easton, PA-NJ	Metropolitan	808,151	2
CBSA	**TYPE**	**2007 Population**	**Monitors**
Worcester, MA	Metropolitan	806,147	1
Bakersfield, CA	Metropolitan	796,111	5
Baton Rouge, LA	Metropolitan	762,905	9

El Paso, TX	Metropolitan	751,891	5
Columbia, SC	Metropolitan	719,810	1
Sarasota-Bradenton-Venice, FL	Metropolitan	716,099	2
Stockton, CA	Metropolitan	694,530	1
Springfield, MA	Metropolitan	693,880	3
Little Rock-North Little Rock, AR	Metropolitan	673,404	1
Greenville, SC	Metropolitan	608,312	1
Wichita, KS	Metropolitan	599,959	2
Scranton--Wilkes-Barre, PA	Metropolitan	556,812	2
Augusta-Richmond County, GA-SC	Metropolitan	541,258	1
Harrisburg-Carlisle, PA	Metropolitan	535,228	2
Ogden-Clearfield, UT	Metropolitan	518,302	1
Lancaster, PA	Metropolitan	503,871	1
Santa Rosa-Petaluma, CA	Metropolitan	483,728	1
Pensacola-Ferry Pass-Brent, FL	Metropolitan	462,147	1
Lexington-Fayette, KY	Metropolitan	450,105	1
Visalia-Porterville, CA	Metropolitan	431,643	1
Vallejo-Fairfield, CA	Metropolitan	426,952	1
Salinas, CA	Metropolitan	425,924	1
York-Hanover, PA	Metropolitan	422,449	1
Santa Barbara-Santa Maria, CA	Metropolitan	422,299	9
Manchester-Nashua, NH	Metropolitan	414,036	1
Springfield, MO	Metropolitan	413,710	1
Beaumont-Port Arthur, TX	Metropolitan	392,826	1
Trenton-Ewing, NJ	Metropolitan	371,660	1
Erie, PA	Metropolitan	283,041	1
San Luis Obispo-Paso Robles, CA	Metropolitan	267,623	3
Santa Cruz-Watsonville, CA	Metropolitan	264,678	1
Merced, CA	Metropolitan	256,700	1
Sioux Falls, SD	Metropolitan	221,466	1
Burlington-South Burlington, VT	Metropolitan	211,172	1
Longview, TX	Metropolitan	203,587	1
Las Cruces, NM	Metropolitan	202,485	2
Lake Charles, LA	Metropolitan	199,974	1
Tyler, TX	Metropolitan	196,814	1
Fargo, ND-MN	Metropolitan	194,208	1
El Centro, CA	Metropolitan	170,210	1
Yuba City, CA	Metropolitan	166,165	1
Madera, CA	Metropolitan	149,180	1
Johnstown, PA	Metropolitan	147,230	1
State College, PA	Metropolitan	145,418	1
Napa, CA	Metropolitan	137,087	1
Altoona, PA	Metropolitan	126,760	1
Farmington, NM	Metropolitan	125,916	2
Owensboro, KY	Metropolitan	112,941	1
CBSA	**TYPE**	**2007 Population**	**Monitors**
Cleveland, TN	Metropolitan	111,646	1
Paducah, KY-IL	Micropolitan	97,571	1
New Castle, PA	Micropolitan	92,154	1
Ukiah, CA	Micropolitan	90,385	2

Indiana, PA	Micropolitan	89,830	1
Marshall, TX	Micropolitan	64,971	1
Rutland, VT	Micropolitan	64,432	1
Hobbs, NM	Micropolitan	56,428	1
Carlsbad-Artesia, NM	Micropolitan	51,269	2
Tahlequah, OK	Micropolitan	46,332	1
Gillette, WY	Micropolitan	37,981	1
No CBSA	NA	NA	8

Table 2a-2: CBSAS with populations greater than 350,000 people not included in analyses

CBSA	TYPE	2007 Population	Reason for no monitoring
Detroit-Warren-Livonia, MI	Metropolitan	4,561,522	Monitors excluded due to data completeness
Seattle-Tacoma-Bellevue, WA	Metropolitan	3,327,901	Not currently monitored
Baltimore-Towson, MD	Metropolitan	2,699,671	Monitors excluded due to data completeness
Portland-Vancouver-Beaverton, OR-WA	Metropolitan	2,162,868	Monitors excluded due to data completeness
Las Vegas-Paradise, NV	Metropolitan	1,893,507	Monitors excluded due to data completeness
Columbus, OH	Metropolitan	1,780,581	Not currently monitored
Virginia Beach-Norfolk-Newport News, VA-NC	Metropolitan	1,691,070	Monitors excluded due to data completeness
Providence-New Bedford-Fall River, RI-MA	Metropolitan	1,639,860	Monitors excluded due to data completeness
Charlotte-Gastonia-Concord, NC-SC	Metropolitan	1,621,635	Monitors excluded due to data completeness
Jacksonville, FL	Metropolitan	1,359,173	Monitors excluded due to data completeness
Memphis, TN-MS-AR	Metropolitan	1,307,699	Monitors excluded due to data completeness
Birmingham-Hoover, AL	Metropolitan	1,115,659	Not currently monitored
Rochester, NY	Metropolitan	1,054,376	Not currently monitored
Raleigh-Cary, NC	Metropolitan	1,023,620	Not currently monitored
Tulsa, OK	Metropolitan	919,698	Monitors excluded due to data completeness
Albany-Schenectady-Troy, NY	Metropolitan	861,146	Not currently monitored
Dayton, OH	Metropolitan	848,761	Not currently monitored
Omaha-Council Bluffs, NE-IA	Metropolitan	842,715	Not currently monitored
Grand Rapids-Wyoming, MI	Metropolitan	788,817	Not currently monitored
McAllen-Edinburg-Mission, TX	Metropolitan	732,166	Not currently monitored
Akron, OH	Metropolitan	707,682	Not currently monitored
Greensboro-High Point, NC	Metropolitan	691,871	Not currently monitored
Poughkeepsie-Newburgh-Middletown, NY	Metropolitan	684,296	Not currently monitored
Knoxville, TN	Metropolitan	675,798	Not currently monitored
Toledo, OH	Metropolitan	667,360	Not currently monitored
Syracuse, NY	Metropolitan	653,964	Not currently monitored
Cape Coral-Fort Myers, FL	Metropolitan	634,375	Not currently monitored
Charleston-North Charleston, SC	Metropolitan	628,187	Monitors excluded due to data completeness
Colorado Springs, CO	Metropolitan	616,432	Not currently monitored
Youngstown-Warren-Boardman, OH-PA	Metropolitan	590,887	Not currently monitored
Boise City-Nampa, ID	Metropolitan	587,526	Not currently monitored
Lakeland, FL	Metropolitan	581,653	Not currently monitored
Madison, WI	Metropolitan	557,650	Not currently monitored
Palm Bay-Melbourne-Titusville, FL	Metropolitan	557,320	Not currently monitored
Des Moines-West Des Moines, IA	Metropolitan	540,397	Monitors excluded due to data completeness
CBSA	**TYPE**	**2007 Population**	**Reason for no monitoring**
Jackson, MS	Metropolitan	539,724	Not currently monitored
Portland-South Portland-Biddeford, ME	Metropolitan	529,286	Monitors excluded due to data completeness
Modesto, CA	Metropolitan	529,038	Monitors excluded due to data completeness
Deltona-Daytona Beach-Ormond Beach, FL	Metropolitan	517,851	Not currently monitored
Chattanooga, TN-GA	Metropolitan	508,709	Not currently monitored
Provo-Orem, UT	Metropolitan	489,312	Monitors excluded due to data completeness
Durham, NC	Metropolitan	477,119	Not currently monitored
Lansing-East Lansing, MI	Metropolitan	469,278	Not currently monitored
Winston-Salem, NC	Metropolitan	464,838	Monitors excluded due to data completeness
Spokane, WA	Metropolitan	453,859	Not currently monitored
Flint, MI	Metropolitan	448,530	Not currently monitored
Fayetteville-Springdale-Rogers, AR-MO	Metropolitan	438,460	Not currently monitored

Corpus Christi, TX	Metropolitan	428,222	Not currently monitored
Reno-Sparks, NV	Metropolitan	425,289	Monitors excluded due to data completeness
Port St. Lucie-Fort Pierce, FL	Metropolitan	422,461	Not currently monitored
Fort Wayne, IN	Metropolitan	412,381	Not currently monitored
Canton-Massillon, OH	Metropolitan	411,749	Not currently monitored
Mobile, AL	Metropolitan	409,542	Not currently monitored
Asheville, NC	Metropolitan	407,274	Not currently monitored
Reading, PA	Metropolitan	406,222	Monitors excluded due to data completeness
Brownsville-Harlingen, TX	Metropolitan	395,867	Not currently monitored
Shreveport-Bossier City, LA	Metropolitan	393,854	Not currently monitored
Salem, OR	Metropolitan	383,801	Not currently monitored
Huntsville, AL	Metropolitan	380,907	Not currently monitored
Davenport-Moline-Rock Island, IA-IL	Metropolitan	380,003	Monitors excluded due to data completeness
Peoria, IL	Metropolitan	375,672	Not currently monitored
Killeen-Temple-Fort Hood, TX	Metropolitan	374,779	Not currently monitored
Hickory-Lenoir-Morganton, NC	Metropolitan	364,397	Not currently monitored
Montgomery, AL	Metropolitan	363,598	Not currently monitored
Tallahassee, FL	Metropolitan	362,802	Not currently monitored
Fayetteville, NC	Metropolitan	353,650	Not currently monitored
Evansville, IN-KY	Metropolitan	351,661	Monitors excluded due to data completeness
Rockford, IL	Metropolitan	350,085	Not currently monitored

Table 2a-3: NO_2 2005-2007 and 2020 gradient adjusted (30%, 65%, and 100%) projected 98th percentile design values (ppb)

						2020		
State	**County**	**CBSA**	**Site**	**Scale up**	**2005-07**	**30%**	**65%**	**100%**
AZ	Maricopa	Phoenix-Mesa-Scottsdale, AZ	19	SCALE-UP	68.0	37.0	47.0	57.0
AZ	Maricopa	Phoenix-Mesa-Scottsdale, AZ	3002	SCALE-UP	70.3	36.6	46.4	56.3
AZ	Maricopa	Phoenix-Mesa-Scottsdale, AZ	3003	SCALE-UP	60.3	27.5	34.9	42.3
AZ	Maricopa	Phoenix-Mesa-Scottsdale, AZ	3010	NO SCALE UP: MIDDLE SCALE	83.3		41.9	
AZ	Maricopa	Phoenix-Mesa-Scottsdale, AZ	9997	SCALE-UP	64.0	33.3	42.3	51.3
AZ	Pima	Tucson, AZ	1011	SCALE-UP	47.0	25.1	31.9	38.6
AZ	Pima	Tucson, AZ	1028	SCALE-UP	46.6	22.8	29.0	35.1
AR	Pulaski	Little Rock-North Little Rock, AR	7	SCALE-UP	50.0	26.0	33.0	40.0
CA	Alameda	San Francisco-Oakland-Fremont, CA	7	SCALE-UP	48.3	3.2	4.1	5.0
CA	Alameda	San Francisco-Oakland-Fremont, CA	1001	SCALE-UP	49.0	17.6	22.4	27.1
CA	Contrasta	San Francisco-Oakland-Fremont, CA	2	SCALE-UP	38.6	0.4	0.5	0.6
CA	Contrasta	San Francisco-Oakland-Fremont, CA	1002	SCALE-UP	33.0	3.3	4.2	5.1
CA	Contrasta	San Francisco-Oakland-Fremont, CA	1004	SCALE-UP	43.6	13.6	17.3	21.0
CA	Contrasta	San Francisco-Oakland-Fremont, CA	3001	SCALE-UP	43.6	14.4	18.2	22.1
CA	Fresno	Fresno, CA	7	SCALE-UP	62.6	25.1	31.9	38.6
CA	Fresno	Fresno, CA	8	SCALE-UP	62.3	22.1	28.0	34.0
CA	Fresno	Fresno, CA	242	SCALE-UP	44.6	8.1	10.3	12.5
CA	Fresno	Fresno, CA	4001	SCALE-UP	45.0	10.9	13.8	16.8
CA	Fresno	Fresno, CA	5001	SCALE-UP	59.8	25.7	32.7	39.6
CA	Imperial	El Centro, CA	5	NO SCALE UP: POP < 350K	75.0		8.0	
CA	Kern	Bakersfield, CA	7	SCALE-UP	42.6	16.4	20.9	25.3
CA	Kern	Bakersfield, CA	10	SCALE-UP	65.3	31.9	40.5	49.1
CA	Kern	Bakersfield, CA	14	SCALE-UP	63.3	30.9	39.3	47.6
CA	Kern	Bakersfield, CA	5001	SCALE-UP	38.0	8.0	10.1	12.3
CA	Kern	Bakersfield, CA	6001	SCALE-UP	64.3	41.9	53.2	64.5
CA	Los Angeles	Los Angeles-Long Beach-Santa Ana, CA	2	SCALE-UP	82.3	15.7	19.9	24.1
CA	Los Angeles	Los Angeles-Long Beach-Santa Ana, CA	16	SCALE-UP	77.3	14.7	18.7	22.6
CA	Los Angeles	Los Angeles-Long Beach-Santa Ana, CA	113	SCALE-UP	63.1	37.7	47.8	58.0

State **County** **CBSA** **Site** **Scale up** **2005-07** **2020**

State	County	CBSA	Site	Scale up	2005-07	2020		
						30%	65%	100%
CA	Los Angeles	Los Angeles-Long Beach-Santa Ana, CA	1002	SCALE-UP	75.0	7.4	9.4	11.5
CA	Los Angeles	Los Angeles-Long Beach-Santa Ana, CA	1103	SCALE-UP	83.6	24.3	30.9	37.5
CA	Los Angeles	Los Angeles-Long Beach-Santa Ana, CA	1201	SCALE-UP	60.6	23.2	29.5	35.8
CA	Los Angeles	Los Angeles-Long Beach-Santa Ana, CA	1301	SCALE-UP	79.0	44.3	56.2	68.1
CA	Los Angeles	Los Angeles-Long Beach-Santa Ana, CA	1701	SCALE-UP	79.6	8.4	10.7	13.0
CA	Los Angeles	Los Angeles-Long Beach-Santa Ana, CA	2005	SCALE-UP	73.0	6.7	8.5	10.3
CA	Los Angeles	Los Angeles-Long Beach-Santa Ana, CA	4002	SCALE-UP	74.0	51.5	65.4	79.3
CA	Los Angeles	Los Angeles-Long Beach-Santa Ana, CA	6012	SCALE-UP	61.3	0.9	1.2	1.5
CA	Los Angeles	Los Angeles-Long Beach-Santa Ana, CA	9033	NO SCALE UP: MIDDLE SCALE	57.0		6.8	
CA	Madera	Madera, CA	4	NO SCALE UP: NEGATIVE	41.3		*	
CA	Marin	San Francisco-Oakland-Fremont, CA	1	SCALE-UP	45.0	25.4	32.3	39.1
CA	Mendocino	Ukiah, CA	8	NO SCALE UP: NEGATIVE	31.6		*	
CA	Mendocino	Ukiah, CA	9	NO SCALE UP: POP < 350K	27.3		0.1	
CA	Merced	Merced, CA	3	NO SCALE UP: POP < 350K	43.0		4.0	
CA	Monterey	Salinas, CA	1003	NO SCALE UP: NEGATIVE	37.0		*	
CA	Napa	Napa, CA	3	NO SCALE UP: POP < 350K	41.3		10.6	
CA	Orange	Los Angeles-Long Beach-Santa Ana, CA	5001	SCALE-UP	73.3	33.4	42.4	51.5
CA	Placer	Sacramento--Arden-Arcade--Roseville, CA	6	NO SCALE UP: NEGATIVE	57.0		*	
CA	Riverside	Riverside-San Bernardino-Ontario, CA	5001	NO SCALE UP: NEGATIVE	50.0		*	

						30%	65%	100%
CA	Riverside	Riverside-San Bernardino-Ontario, CA	8001	SCALE-UP	64.3	21.3	27.0	32.8
CA	Riverside	Riverside-San Bernardino-Ontario, CA	9001	NO SCALE UP: MIDDLE SCALE	53.0		8.1	
CA	Sacramento	Sacramento--Arden-Arcade--Roseville, CA	6	SCALE-UP	47.0	5.6	7.1	8.6
CA	Sacramento	Sacramento--Arden-Arcade--Roseville, CA	10	SCALE-UP	54.3	19.9	25.3	30.6
CA	Sacramento	Sacramento--Arden-Arcade--Roseville, CA	12	SCALE-UP	35.0	2.9	3.7	4.5
CA	Sacramento	Sacramento--Arden-Arcade--Roseville, CA	13	SCALE-UP	55.6	21.5	27.3	33.1
CA	San Bernardino	Riverside-San Bernardino-Ontario, CA	1	NO SCALE UP: NEGATIVE	72.0		*	
CA	San Bernardino	Riverside-San Bernardino-Ontario, CA	306	NO SCALE UP: NEGATIVE	65.6		*	
CA	San Bernardino	Riverside-San Bernardino-Ontario, CA	2002	SCALE-UP	80.0	0.3	0.4	0.5
CA	San Bernardino	Riverside-San Bernardino-Ontario, CA	9004	SCALE-UP	70.6	2.9	3.7	4.5
CA	San Diego	San Diego-Carlsbad-San Marcos, CA	1	SCALE-UP	60.6	12.4	15.8	19.1
CA	San Diego	San Diego-Carlsbad-San Marcos, CA	6	NO SCALE UP: HIGHESTNC; NEIGHBORHOOD; POINT	61.1		11.5	
CA	San Diego	San Diego-Carlsbad-San Marcos, CA	1002	SCALE-UP	59.6	6.2	7.9	9.6
CA	San Diego	San Diego-Carlsbad-San Marcos, CA	1006	NO SCALE UP: NEGATIVE	42.6		*	
CA	San Diego	San Diego-Carlsbad-San Marcos, CA	1008	SCALE-UP	62.3	9.4	11.9	14.5
CA	San Francisco	San Francisco-Oakland-Fremont, CA	5	SCALE-UP	54.6	29.4	37.4	45.3
							2020	
State	**County**	**CBSA**	**Site**	**Scale up**	**2005-07**	**30%**	**65%**	**100%**

State	County	CBSA	Site	Scale up	2005-07	2020		
CA	San Joaquin	Stockton, CA	1002	SCALE-UP	58.0	20.0	25.4	30.8
CA	San Luis Obispo	San Luis Obispo-Paso Robles, CA	3001	NO SCALE UP: POP < 350K	35.3		6.4	
CA	San Luis Obispo	San Luis Obispo-Paso Robles, CA	4002	NO SCALE UP: POP < 350K	30.3		2.9	
CA	San Luis Obispo	San Luis Obispo-Paso Robles, CA	8001	NO SCALE UP: POP < 350K	44.3		6.4	
CA	San Mateo	San Francisco-Oakland-Fremont, CA	1001	SCALE-UP	50.0	28.3	36.0	43.6
CA	Santa Barbara	Santa Barbara-Santa Maria, CA	8	SCALE-UP	31.6	6.3	8.1	9.8
CA	Santa Barbara	Santa Barbara-Santa Maria, CA	1013	NO SCALE UP: NEGATIVE	8.0		*	
CA	Santa Barbara	Santa Barbara-Santa Maria, CA	1014	NO SCALE UP: NEGATIVE	6.6		*	
CA	Santa Barbara	Santa Barbara-Santa Maria, CA	1018	NO SCALE UP: INDUSTRIAL	26.0		2.7	
CA	Santa Barbara	Santa Barbara-Santa Maria, CA	1021	NO SCALE UP: NEGATIVE	19.6		*	
CA	Santa Barbara	Santa Barbara-Santa Maria, CA	1025	NO SCALE UP: INDUSTRIAL	14.6		2.7	
CA	Santa Barbara	Santa Barbara-Santa Maria, CA	2004	NO SCALE UP: NEGATIVE	30.0		*	
CA	Santa Barbara	Santa Barbara-Santa Maria, CA	2011	SCALE-UP	37.0	18.5	23.5	28.5
CA	Santa Barbara	Santa Barbara-Santa Maria, CA	4003	NO SCALE UP: NEGATIVE	8.3		*	
CA	Santa Clara	San Jose-Sunnyvale-Santa Clara, CA	5	SCALE-UP	57.3	33.9	43.0	52.1
CA	Santa Cruz	Santa Cruz-Watsonville, CA	3	NO SCALE UP: NEGATIVE	24.3		*	
CA	Solano	Vallejo-Fairfield, CA	4	SCALE-UP	43.0	18.3	23.2	28.1
CA	Sonoma	Santa Rosa-Petaluma, CA	3	SCALE-UP	39.3	6.8	8.6	10.5
CA	Sutter	Yuba City, CA	3	NO SCALE UP: NEGATIVE	50.1		*	
CA	Tulare	Visalia-Porterville, CA	2002	SCALE-UP	58.6	11.1	14.1	17.1
CA	Ventura	Oxnard-Thousand Oaks-Ventura, CA	2002	SCALE-UP	47.6	0.9	1.2	1.5
CA	Ventura	Oxnard-Thousand Oaks-Ventura, CA	3001	SCALE-UP	40.6	1.4	1.7	2.1
CA	Yolo	Sacramento--Arden-Arcade--Roseville, CA	4	SCALE-UP	37.6	7.0	8.9	10.8
CO	Adams	Denver-Aurora,	3001	SCALE-UP	74.3	64.6	82.0	99.5
CT	Fairfield	Bridgeport-Stamford-Norwalk, CT	9003	SCALE-UP	56.6	3.4	4.4	5.3
CT	Hartford	Hartford-West Hartford-East Hartford, CT	1003	SCALE-UP	51.8	13.6	17.3	21.0
CT	New Haven	New Haven-Milford, CT	27	SCALE-UP	68.3	24.1	30.6	37.1
DC	Washington	Washington-Arlington-Alexandria, DC-VA-MD-WV	25	SCALE-UP	56.0	26.5	33.6	40.8

						30%	65%	100%
DC	Washington	Washington-Arlington-Alexandria, DC-VA-MD-WV	41	SCALE-UP	63.0	27.0	34.3	41.6
DC	Washington	Washington-Arlington-Alexandria, DC-VA-MD-WV	43	SCALE-UP	60.6	26.0	33.0	40.0
FL	Broward	Miami-Fort Lauderdale-Miami Beach, FL	8002	SCALE-UP	54.0	34.5	43.8	53.1
FL	Escambia	Pensacola-Ferry Pass-Brent, FL	4	SCALE-UP	33.6	20.3	25.8	31.3
FL	Hillsborough	Tampa-St. Petersburg-Clearwater, FL	81	SCALE-UP	33.0	23.8	30.2	36.6
FL	Hillsborough	Tampa-St. Petersburg-Clearwater, FL	1065	SCALE-UP	38.6	31.2	39.6	48.0
FL	Hillsborough	Tampa-St. Petersburg-Clearwater, FL	3002	SCALE-UP	32.0	19.1	24.3	29.5
FL	Manatee	Sarasota-Bradenton-Venice, FL	4012	SCALE-UP	31.3	12.3	15.6	19.0
FL	Miami-Dade	Miami-Fort Lauderdale-Miami Beach, FL	27	SCALE-UP	48.0	22.3	28.3	34.3
FL	Orange	Orlando-Kissimmee, FL	2002	SCALE-UP	44.3	17.1	21.7	26.3
FL	Palm Beach	Miami-Fort Lauderdale-Miami Beach, FL	1004	NO SCALE UP: MIDDLE SCALE	46.0		20.5	
FL	Pinellas	Tampa-St. Petersburg-Clearwater, FL	18	SCALE-UP	39.6	21.1	26.8	32.5
FL	Sarasota	Sarasota-Bradenton-Venice, FL	1006	SCALE-UP	27.6	12.0	15.2	18.5
GA	Fulton	Atlanta-Sandy Springs-Marietta, GA	48	SCALE-UP	73.0	34.7	44.1	53.5
GA	Paulding	Atlanta-Sandy Springs-Marietta, GA	3	SCALE-UP	25.0	13.3	16.9	20.5
GA	Rockdale	Atlanta-Sandy Springs-Marietta, GA	1	SCALE-UP	29.6	16.6	21.1	25.6
IL	Cook	Chicago-Naperville-Joliet, IL-IN-WI	63	NO SCALE UP: MIDDLE SCALE	100.0		17.8	
IL	Cook	Chicago-Naperville-Joliet, IL-IN-WI	76	SCALE-UP	63.6	12.4	15.8	19.1
IL	Cook	Chicago-Naperville-Joliet, IL-IN-WI	3103	NO SCALE UP: MIDDLE SCALE	74.6		37.9	
IL	Cook	Chicago-Naperville-Joliet, IL-IN-WI	4002	SCALE-UP	68.3	17.3	22.0	26.6
IL	St Clair	St. Louis, MO-IL	10	SCALE-UP	50.3	33.1	42.0	51.0

							2020	
State	**County**	**CBSA**	**Site**	**Scale up**	**2005-07**	**30%**	**65%**	**100%**
IN	Hendricks	Indianapolis-Carmel, IN	2	NO SCALE UP: INDUSTRIAL	41.0		7.4	
IN	Marion	Indianapolis-Carmel, IN	73	SCALE-UP	47.6	26.2	33.2	40.3

							2020	
State	County	CBSA	Site	Scale up	2005-07	30%	65%	100%
KS	Sedgwick	Wichita, KS	10	SCALE-UP	46.5	29.6	37.6	45.6
KS	Sumner	Wichita, KS	2	SCALE-UP	27.0	16.1	20.4	24.8
KS	Wyandotte	Kansas City, MO-KS	21	SCALE-UP	57.0	29.4	37.4	45.3
KY	Daviess	Owensboro, KY	5	NO SCALE UP: POP < 350K	34.6		15.2	
KY	Fayette	Lexington-Fayette, KY	12	SCALE-UP	53.0	32.9	41.8	50.6
KY	Jefferson	Louisville-Jeffersonunty, KY-IN	1021	SCALE-UP	51.5	16.1	20.4	24.8
KY	Mc Cracken	Paducah, KY-IL	1024	NO SCALE UP: POP < 350K	43.5		14.7	
LA	Ascension	Baton Rouge, LA	4	SCALE-UP	43.0	41.1	52.2	63.3
LA	Calcasieu	Lake Charles, LA	8	NO SCALE UP: POP < 350K	39.3		35.8	
LA	East Baton Rouge	Baton Rouge, LA	3	SCALE-UP	56.3	49.0	62.2	75.5
LA	East Baton Rouge	Baton Rouge, LA	9	SCALE-UP	58.0	52.5	66.6	80.8
LA	East Baton Rouge	Baton Rouge, LA	13	NO SCALE UP: MICROSCALE	22.3		16.4	
LA	East Baton Rouge	Baton Rouge, LA	1001	SCALE-UP	42.0	37.8	47.9	58.1
LA	Iberville	Baton Rouge, LA	7	SCALE-UP	27.6	24.9	31.6	38.3
LA	Iberville	Baton Rouge, LA	9	SCALE-UP	30.6	27.9	35.4	43.0
LA	Iberville	Baton Rouge, LA	12	SCALE-UP	40.3	37.7	47.8	58.0
LA	Jefferson	New Orleans-Metairie-Kenner, LA	1001	SCALE-UP	52.0	40.6	51.5	62.5
LA	West Baton Rouge	Baton Rouge, LA	1	SCALE-UP	53.0	49.2	62.5	75.8
MA	Essex	Boston-Cambridge-Quincy, MA-NH	2006	SCALE-UP	43.3	29.0	36.8	44.6
MA	Essex	Boston-Cambridge-Quincy, MA-NH	5005	SCALE-UP	40.6	24.2	30.8	37.3
MA	Hampden	Springfield, MA	8	NO SCALE UP: HIGHESTNC; NEIGHBORHOOD; POINT	43.3		26.3	
MA	Hampden	Springfield, MA	16	SCALE-UP	46.6	28.7	36.4	44.1
MA	Hampshire	Springfield, MA	4002	SCALE-UP	32.6	19.3	24.6	29.8
MA	Suffolk	Boston-Cambridge-Quincy, MA-NH	2	NO SCALE UP: MICROSCALE	57.0		31.8	
MA	Suffolk	Boston-Cambridge-Quincy, MA-NH	42	SCALE-UP	50.3	30.4	38.6	46.8
MA	Worcester	Worcester, MA	23	SCALE-UP	45.0	28.2	35.8	43.5
MN	Anoka	Minneapolis-St. Paul-Bloomington, MN-WI	1002	SCALE-UP	44.0	34.0	43.1	52.3
MO	Clay	Kansas City, MO-KS	5	SCALE-UP	39.0	25.6	32.5	39.5
MO	Greene	Springfield, MO	36	SCALE-UP	52.0	31.8	40.4	49.0
MO	Jackson	Kansas City, MO-KS	34	SCALE-UP	59.6	36.7	46.6	56.5
MO	St Charles	St. Louis, MO-IL	1002	SCALE-UP	37.0	18.8	23.9	29.0

State	County	CBSA	Site	Scale up	2005-07	2020 30%	2020 65%	2020 100%
MO	Ste Genevieve		5	NO SCALE UP: POP < 350K	19.6		13.0	
MO	St Louis	St. Louis, MO-IL	4	SCALE-UP	45.0	24.5	31.2	37.8
MO	St Louis	St. Louis, MO-IL	3001	SCALE-UP	49.3	26.4	33.5	40.6
MO	St Louis	St. Louis, MO-IL	86	SCALE-UP	62.0	43.9	55.8	67.6
NH	Hillsborough	Manchester-Nashua, NH	20	SCALE-UP	44.3	28.3	36.0	43.6
NH	Rockingham	Boston-Cambridge-Quincy, MA-NH	14	SCALE-UP	39.0	22.2	28.1	34.1
NJ	Essex	New York-Northern New Jersey-Long Island, NY-NJ-PA	1003	SCALE-UP	74.0	24.3	30.9	37.5
NJ	Hudson	New York-Northern New Jersey-Long Island, NY-NJ-PA	6	SCALE-UP	69.3	32.9	41.8	50.6
NJ	Mercer	Trenton-Ewing, NJ	5	SCALE-UP	48.6	17.1	21.7	26.3
NJ	Middlesex	New York-Northern New Jersey-Long Island, NY-NJ-PA	11	SCALE-UP	55.6	23.7	30.1	36.5
NJ	Morris	New York-Northern New Jersey-Long Island, NY-NJ-PA	3001	SCALE-UP	41.6	17.8	22.6	27.5
NJ	Union	New York-Northern New Jersey-Long Island, NY-NJ-PA	4	SCALE-UP	80.6	40.4	51.2	62.1
NM	Bernalillo	Albuquerque, NM	23	SCALE-UP	56.0	40.6	51.5	62.5
NM	Bernalillo	Albuquerque, NM	24	SCALE-UP	48.0	34.7	44.1	53.5
NM	Dona Ana	Las Cruces, NM	21	NO SCALE UP: POP < 350K	49.6		30.5	
NM	Dona Ana	Las Cruces, NM	22	NO SCALE UP: POP < 350K	44.0		25.2	
NM	Eddy	Carlsbad-Artesia, NM	1004	NO SCALE UP: POP < 350K	30.3		28.6	
NM	Eddy	Carlsbad-Artesia, NM	1005	NO SCALE UP: POP < 350K	22.6		20.3	
NM	Lea	Hobbs, NM	8	NO SCALE UP: POP < 350K	45.3		43.9	
NM	Sandoval	Albuquerque, NM	1003	SCALE-UP	46.6	32.8	41.6	50.5
NM	San Juan	Farmington, NM	9	NO SCALE UP: POP < 350K	42.3		40.8	
NM	San Juan	Farmington, NM	1005	NO SCALE UP: POP < 350K	47.3		42.4	
NY	Erie	Buffalo-Niagara Falls, NY	5	NO SCALE UP: VISUAL NEAR ROAD	79.0		44.7	
NY	New York	New York-Northern New Jersey-Long Island, NY-NJ-PA	56	NO SCALE UP: MIDDLE SCALE	78.3		22.9	
NY	Queens	New York-Northern New Jersey-Long Island, NY-NJ-PA	124	SCALE-UP	68.6	25.2	32.0	38.8

State	County	CBSA	Site	Scale up	2005-07	2020 30%	2020 65%	2020 100%
NY	Suffolk	New York-Northern New Jersey-Long Island, NY-NJ-PA	9	SCALE-UP	44.6	9.5	12.1	14.6
ND	Burke		4	NO SCALE UP: POP < 350K	13.0		10.7	
ND	Cass	Fargo, ND-MN	1004	NO SCALE UP: POP < 350K	37.3		19.1	
ND	Mc Kenzie		2	NO SCALE UP: POP < 350K	7.0		4.8	
ND	Mercer		4	NO SCALE UP: POP < 350K	21.6		16.9	
ND	Mercer		102	NO SCALE UP: POP < 350K	21.0		16.4	
ND	Mercer		124	NO SCALE UP: POP < 350K	23.0		17.8	
ND	Oliver		2	NO SCALE UP: POP < 350K	21.0		16.3	
OH	Cuyahoga	Cleveland-Elyria-Mentor, OH	60	NO SCALE UP: VISUAL NEAR ROAD	62.0		40.4	
OH	Cuyahoga	Cleveland-Elyria-Mentor, OH	70	SCALE-UP	59.0	37.3	47.4	57.5
OH	Hamilton	Cincinnati-Middletown, OH-KY-IN	40	SCALE-UP	60.3	30.8	39.1	47.5
OK	Cherokee	Tahlequah, OK	9002	NO SCALE UP: POP < 350K	38.3		22.4	
OK	Oklahoma	Oklahoma City, OK	33	SCALE-UP	53.3	31.8	40.4	49.0
OK	Oklahoma	Oklahoma City, OK	1037	SCALE-UP	43.0	23.9	30.3	36.8
PA	Allegheny	Pittsburgh, PA	8	SCALE-UP	49.6	37.2	47.3	57.3
PA	Allegheny	Pittsburgh, PA	10	SCALE-UP	63.6	47.7	60.6	73.5
PA	Allegheny	Pittsburgh, PA	1005	SCALE-UP	46.3	32.5	41.2	50.0
PA	Beaver	Pittsburgh, PA	14	SCALE-UP	48.3	27.7	35.2	42.6
PA	Blair	Altoona, PA	801	NO SCALE UP: POP < 350K	50.6		23.4	
PA	Bucks	Philadelphia-Camden-Wilmington, PA-NJ-DE-MD	12	SCALE-UP	53.6	8.9	11.4	13.8
PA	Cambria	Johnstown, PA	11	NO SCALE UP: POP < 350K	43.6		23.1	
PA	Centre	Statellege, PA	100	NO SCALE UP: POP < 350K	38.0		17.5	
PA	Dauphin	Harrisburg-Carlisle, PA	401	SCALE-UP	51.0	4.8	6.1	7.5
PA	Erie	Erie, PA	3	NO SCALE UP: POP < 350K	54.0		26.6	
PA	Indiana	Indiana, PA	4	NO SCALE UP: POP < 350K	33.0		12.1	
PA	Lackawanna	Scranton--Wilkes-Barre, PA	2006	SCALE-UP	47.3	4.7	6.0	7.3
PA	Lancaster	Lancaster, PA	7	SCALE-UP	46.0	9.2	11.6	14.1
PA	Lawrence	New Castle, PA	15	NO SCALE UP: POP < 350K	49.0		33.5	
PA	Lehigh	Allentown-Bethlehem-Easton, PA-NJ	4	SCALE-UP	47.3	9.9	12.6	15.3
PA	Luzerne	Scranton--Wilkes-Barre, PA	1101	SCALE-UP	44.3	3.9	4.9	6.0

PA	Montgomery	Philadelphia-Camden-Wilmington, PA-NJ-DE-MD	13	SCALE-UP	54.0	11.9	15.1	18.3
PA	Northampton	Allentown-Bethlehem-Easton, PA-NJ	25	SCALE-UP	47.3	7.6	9.7	11.8
PA	Perry	Harrisburg-Carlisle, PA	301	NO SCALE UP: NEGATIVE	24.0		*	
PA	Washington	Pittsburgh, PA	5	SCALE-UP	43.0	27.0	34.3	41.6
PA	Washington	Pittsburgh, PA	5001	SCALE-UP	29.6	17.7	22.5	27.3
PA	Westmoreland	Pittsburgh, PA	8	SCALE-UP	43.0	28.4	36.1	43.8
PA	York	York-Hanover, PA	8	SCALE-UP	57.3	4.4	5.6	6.8
SC	Aiken	Augusta-Richmondunty, GA-SC	3	NO SCALE UP: SOURCE ORIENTED	23.3		8.8	
SC	Greenville	Greenville, SC	9	NO SCALE UP: NON-REGULATORY	43.6		20.5	
SC	Richland	Columbia, SC	7	SCALE-UP	49.6	15.3	19.5	23.6
SD	Jackson		1	NO SCALE UP: POP < 350K	7.6		4.8	
SD	Minnehaha	Sioux Falls, SD	7	NO SCALE UP: POP < 350K	33.0		17.8	
TN	Bradley	Cleveland, TN	102	NO SCALE UP: POP < 350K	37.3		16.8	
TN	Davidson	Nashville-Davidson--Murfreesboro, TN	11	SCALE-UP	55.6	21.2	26.9	32.6
TX	Bexar	San Antonio, TX	46	NO SCALE UP: MICROSCALE	54.6		32.2	
TX	Bexar	San Antonio, TX	52	SCALE-UP	25.0	13.3	16.9	20.5
TX	Bexar	San Antonio, TX	59	NO SCALE UP: SOURCE ORIENTED	33.6		16.5	
TX	Brazoria	Houston-Sugar Land-Baytown, TX	1016	NO SCALE UP: MIDDLE SCALE	26.3		3.9	
TX	Dallas	Dallas-Fort Worth-Arlington, TX	69	SCALE-UP	58.0	34.2	43.4	52.6
TX	Dallas	Dallas-Fort Worth-Arlington, TX	75	SCALE-UP	45.0	25.3	32.1	39.0
TX	Denton	Dallas-Fort Worth-Arlington, TX	34	SCALE-UP	38.6	21.0	26.6	32.3
							2020	
State	**County**	**CBSA**	**Site**	**Scale up**	**2005-07**	**30%**	**65%**	**100%**
TX	El Paso	El Paso, TX	37	SCALE-UP	64.0	58.3	74.1	89.8
TX	El Paso	El Paso, TX	44	NO SCALE UP: VISUAL NEAR ROAD	66.6		56.1	
TX	El Paso	El Paso, TX	55	SCALE-UP	68.3	62.2	79.0	95.8
TX	El Paso	El Paso, TX	57	SCALE-UP	58.0	41.7	52.9	64.1
TX	El Paso	El Paso, TX	58	SCALE-UP	50.6	42.4	53.9	65.3
TX	Gregg	Longview, TX	1	NO SCALE UP: POP < 350K	29.3		18.9	
TX	Harris	Houston-Sugar Land-Baytown, TX	26	NO SCALE UP: MIDDLE SCALE	52.0		34.5	

TX	Harris	Houston-Sugar Land-Baytown, TX	29	SCALE-UP	35.6	16.4	20.9	25.3
TX	Harris	Houston-Sugar Land-Baytown, TX	47	SCALE-UP	60.3	28.9	36.7	44.5
TX	Harris	Houston-Sugar Land-Baytown, TX	75	SCALE-UP	61.8	43.4	55.1	66.8
TX	Harris	Houston-Sugar Land-Baytown, TX	1034	SCALE-UP	56.3	42.3	53.7	65.1
TX	Harris	Houston-Sugar Land-Baytown, TX	1035	SCALE-UP	58.3	43.8	55.6	67.5
TX	Harris	Houston-Sugar Land-Baytown, TX	1039	SCALE-UP	46.6	27.3	34.6	42.0
TX	Harris	Houston-Sugar Land-Baytown, TX	1050	NO SCALE UP: MIDDLE SCALE	34.0		22.1	
TX	Harrison	Marshall, TX	2	NO SCALE UP: POP < 350K	23.0		15.9	
TX	Hunt	Dallas-Fort Worth-Arlington, TX	1006	SCALE-UP	34.3	15.7	19.9	24.1
TX	Jefferson	Beaumont-Port Arthur, TX	22	SCALE-UP	29.6	15.0	19.1	23.1
TX	Kaufman	Dallas-Fort Worth-Arlington, TX	5	SCALE-UP	31.3	17.7	22.5	27.3
TX	Montgomery	Houston-Sugar Land-Baytown, TX	78	NO SCALE UP: MIDDLE SCALE	37.3		19.9	
TX	Smith	Tyler, TX	7	NO SCALE UP: POP < 350K	25.3		14.9	
TX	Tarrant	Dallas-Fort Worth-Arlington, TX	1002	SCALE-UP	59.6	30.9	39.3	47.6
TX	Tarrant	Dallas-Fort Worth-Arlington, TX	3009	SCALE-UP	43.6	28.4	36.1	43.8
TX	Tarrant	Dallas-Fort Worth-Arlington, TX	3011	SCALE-UP	46.3	25.2	32.0	38.8
TX	Travis	Austin-Round Rock, TX	20	SCALE-UP	28.3	13.3	16.9	20.5
UT	Davis	Ogden-Clearfield, UT	4	SCALE-UP	65.0	39.0	49.5	60.0
UT	Salt Lake	Salt Lake City, UT	3006	SCALE-UP	63.6	57.8	73.4	89.0
VT	Chittenden	Burlington-South Burlington, VT	14	NO SCALE UP: POP < 350K	44.4		27.0	
VT	Rutland	Rutland, VT	2	NO SCALE UP: POP < 350K	44.5		19.6	
VA	Charles City	Richmond, VA	2	SCALE-UP	61.0	49.1	62.4	75.6

						2020		
State	**County**	**CBSA**	**Site**	**Scale up**	**2005-07**	**30%**	**65%**	**100%**
VA	Fairfax	Washington-Arlington-Alexandria, DC-VA-MD-WV	1005	SCALE-UP	51.6	25.3	32.1	39.0
VA	Fairfax	Washington-Arlington-Alexandria, DC-VA-MD-WV	5001	SCALE-UP	53.6	24.0	30.5	37.0
VA	Richmond	Richmond, VA	24	SCALE-UP	59.5	38.0	48.2	58.5
WI	Milwaukee	Milwaukee-Waukesha-West Allis, WI	26	SCALE-UP	51.0	5.4	6.8	8.3
WY	Campbell	Gillette, WY	123	NO SCALE UP: POP < 350K	11.6		9.3	

Chapter 3: Emissions Controls Analysis – Design and Analytical Results

Synopsis

The revised NO2 standard is 100 parts per billion (ppb), calculated from the average of the 98th percentile of 1-hour daily maximum concentrations from three consecutive years. OMB Circular A-4 requires the RIA to contain, in addition to analysis of the impacts of the revised NAAQS of 100 ppb, analysis of a level more stringent and a level less stringent than the NAAQS. For this analysis, we chose a more stringent level of 80 ppb and a less stringent level of 125 ppb.

This chapter documents the illustrative emission control strategy we applied to simulate attainment with the revised NO_2 NAAQS and the two additional levels being analyzed. Section 3.1 describes the approach we followed to select emission controls to simulate attainment. Section 3.2 describes the emission control measures identified as appropriate for this illustrative control strategy. Section 3.3 summarizes the emission reductions estimated as necessary to meet the revised NAAQS and the two additional levels included in the analysis. Section 3.4 includes the estimated costs of controls for each area projected to exceed one or more of the levels of the analysis. Section 3.5 discusses key limitations in the approach we used to estimate the control strategies for each alternative standard.

3.1 Designing the Control Strategy Analysis

It is important to note that this analysis does not attempt to estimate attainment or nonattainment for any areas of the country other than those counties currently served by one of the 409 monitors in the current network. Chapter 2 explains that the current network is focused on community-wide ambient levels of NO_2, and not near-roadway levels, which may be significantly higher. The revised standard contains requirements for an NO_2 monitoring network that will include monitors near major roadways. We recognize that once a network of near-roadway monitors is in place, more areas could find themselves exceeding the new hourly NO_2 NAAQS. However for this RIA analysis, we lack sufficient data to predict which counties beyond the current network might exceed the revised NAAQS after implementation of a near-roadway monitoring network. Therefore we lack a credible analytic path to estimating costs and benefits for counties outside of the current NO2 monitoring network. This analysis relies on current and future estimated air quality concentrations at area-wide monitors, making adjustments to future year projections using derived estimates of the relationship between future year area-wide air quality peaks and current near-roadway peaks.

As part of our economic analysis of the revised NO_2 standard, our 2020 analysis baseline assumes that States will put in place the necessary control strategies to attain the previous $PM_{2.5}$ and ozone standards. The cost of these control strategies was included in the RIAs for those rulemakings. We do not include the cost of those controls in this analysis, in order to prevent counting the cost of installing and operating the controls twice. Of course, the health and environmental benefits resulting from installation of those controls were attributed to attaining those standards and are not counted again for the analysis of this NO_2 standard.

The first step in the control strategy analysis was to identify the geographic areas projected to exceed the revised standard or one of the additional levels in the time period for which attainment is required. We based this assessment on monitor design values projected to the year 2020 and adjusted to simulate levels for a near-road monitoring network, as discussed in Chapter 2. (Prior to near-roadway adjustment, all the monitor design values in the current network were below the alternative NAAQS levels). After identifying the geographic areas, we estimated the amount of NOx emission reductions necessary to bring the areas into attainment with the three levels being analyzed. The process for estimating the necessary emission reductions is described in Chapter 2. Because of the focus of the revised NAAQS on near-road issues, we chose to apply mobile source control measures to achieve the necessary emission reductions. The types of measures appropriate for such an analysis are described in the next section. Finally, we determined the cost of control, as discussed in Section 3.4.

3.2 Control Measures

Because this analysis primarily concerns meeting the requirements for the near-road monitoring network, the control strategy is focused on control measures that reduce emissions from onroad and nonroad mobile sources. Onroad mobile sources are mobile sources that travel on roadways. These sources include automobiles, buses, trucks, and motorcycles traveling on roads and highways. Nonroad mobile sources are any combustion engine that travels by other means than roadways. These sources include railroad locomotives; marine vessels; aircraft; off-road motorcycles; snowmobiles; pleasure craft; and farm, construction, industrial and lawn/garden equipment.

Local onroad and nonroad mobile source control measures that are effective in reducing emissions of NOx include:

- Diesel Retrofits (Onroad)
- Diesel Retrofits and Engine Rebuilds (Nonroad)
- Elimination of Long Duration Idling (Onroad)
- Continuous Inspection and Maintenance (Onroad)

Information describing these measures, the effectiveness of each, and the role of EPA's National Mobile Inventory Model (NMIM) in calculating reductions for the diesel retrofit measures, is contained in Chapter 3 of the document "Final Ozone NAAQS Regulatory Impact Analysis"[1]. Each of these measures reduces emissions of NOx which has the co-benefit of reducing secondary formation of PM2.5. In addition, diesel retrofits and elimination of long duration idling reduce direct emissions of PM2.5.

3.3 Estimated Emission Reductions

As described in Chapter 2, air quality design values from the current monitoring network were projected to the year 2020 and were adjusted to estimate the range of levels that might occur at near-road monitor locations at gradients of 30 percent, 65 percent, and 100 percent. These adjusted design values were also adjusted to reflect the change in influence from mobile source emissions projected for the year 2020. Finally, these adjusted design values were used to estimate the level of emission reductions necessary to meet the 3 levels of the standard included in this analysis.

For the revised standard of 100 ppb and the less stringent level of 125 ppb there were no projected exceedances in 2020. For the more stringent level of 80 ppb, exceedances were projected in 4 counties. The counties and their estimated emission reductions are presented in Table 3.1. For this illustrative analysis, we identified several mobile source control measures that would be appropriate for achieving the necessary emission reductions, as described in section 3.2. These measures currently are not required in the geographic areas listed in Table 3.1 and could be implemented in those areas as part of a local control strategy for reducing emissions.

Table 3-1: NOx Emission Reductions (tons/yr) by County in 2020 for More Stringent Level of 80 ppb[a]

County	ST	Tons reduced (30% Gradient)	Tons reduced (65% Gradient)	Tons reduced (100% Gradient)
Adams Co	CO		680	8,070
East Baton Rouge Par	LA			460
El Paso Co	TX			8,600
Salt Lake Co	UT			4,100

[a] All estimates rounded to two significant figures.

[1] http://www.epa.gov/ttn/ecas/regdata/RIAs/452_R_08_003.pdf (beginning on page 3a-12).

3.4 Costs of Mobile Source Controls

Because this analysis examines emissions and air quality approximating near-roadway conditions, we believe it is appropriate to focus analysis of controls on mobile sources. For the purposes of this analysis EPA reviewed existing cost effectiveness estimates for a number of federal on-road and non-road regulations that have been promulgated in the last several years. These regulations include the Tier 2 regulation for light-duty motor vehicles, the 2007 highway heavy duty rules, the Tier 4 non-road equipment rule, the locomotive/marine rule, and the small spark ignition equipment rule. EPA also reviewed the cost effectiveness estimates for the mobile source controls that were applied in the RIA for the 2008 ozone NAAQS. That RIA included cost effectiveness estimates for mobile source controls that included retrofits for on-road vehicles and non-road equipment, elimination of long duration truck idling, continuous inspection and maintenance of light-duty vehicles, the introduction of plug-in hybrid vehicles into the national vehicle fleet, more stringent requirements for aftermarket replacement catalytic converters, commuter programs to reduce vehicle miles travelled and vehicle trips, and improved emission control systems for new vehicles.

As summarized in Table 3.2 the majority of these controls have costs of between $1,000 and $5,000 per ton of NOx or NOx+non-methane hydrocarbons. There are some exceptions. Several of the measures produce fuel savings that offset the cost of the control equipment or vehicle and any operating expenses; therefore, these measures produce NOx reductions at no cost. Some non-road retrofits, particularly for agricultural equipment, are more expensive. However, this type of equipment would not be the primary focus of an attainment strategy for the NO_2 NAAQS under a near roadway monitoring scenario. Retrofits of class 6 and 7 heavy duty vehicles and commuter programs also have higher costs per ton. However, these do not provide large emission reductions. Finally, the estimated cost per ton of NOx reductions from improvements in the emissions control systems for new motor vehicles is also higher. However, as noted in the RIA for 2008 ozone NAAQS, this is a very rough estimate of the cost of these controls. Only one method for achieving the desired level of emissions was considered. A much more detailed analysis would be required to develop a representative cost for such future controls on new vehicles.

The purpose of this analysis is to develop an estimate of the average cost per ton of NOx reductions that would be needed to bring projected nonattainment areas into compliance with the revised NO_2 NAAQS. Based on the estimates in these recent RIAs it is evident that there remain mobile source control strategies that provide emissions reductions in the range of $1,000 to $5,000 per ton of NOx. However, we also recognize that the costs of controls will likely increase as additional control measures are implemented. We anticipate that nonattainment areas would employ a mixture of controls that fall within the range of $1,000 to $5,000 per ton and some additional controls that have higher costs per ton. Given the screening nature of this analysis we have estimated that the annualized average cost of controls to attain the NO_2 NAAQS would be in the range of $3,000 to $6,000 per ton. This estimate is based upon knowledge of the cost of mobile source controls included in previous analyses, especially for the control measures listed in Section 3.2, which are generally based on a three percent discount rate. A discount rate of seven percent was not available for all estimates provided in Table 3.2.

Table 3-2: Estimated $/ton Costs of NOx Emissions Reductions from Recent RIAs

SOURCE CATEGORY[a]	NO_X COST/TON	NOTES
C3 Marine Coordinated Strategy NPRM, 2009	510	a
Nonroad Small Spark-Ignition Engines 73 FR 59034, October 8, 2008	330-1,200	a, b, c
Stationary Diesel (CI) Engines (71 FR 39154, July 11, 2006)	580 – 20,000	a
Locomotives and C1/C2 Marine (Both New and Remanufactured) (73 FR 25097, May 6, 2008)	730	a, b
Heavy Duty Nonroad Diesel Engines (69 FR 38957, June 29, 2004)	1,100	a, b
2007 Highway Heavy Duty Rule (66 FR 5001, January 18, 2001)	2,200	a, b
Tier 2 (Page VI-18 of the Tier 2 RIA)	2,047	b, d
Continuous Light-duty Vehicle Inspection and Maintenance (2008 ozone RIA Appendix 5a pages 5a-7 – 5a-9)	0	
Eliminate Long Duration Truck Idling (2008 ozone RIA Appendix 5a pages 5a-9 – 5a-10)	0	
Plug-in Hybrid Vehicles (2008 ozone RIA Appendix 7a pages 7a-4 – 7a-96)	0	
Retrofit Class 8b Trucks (2008 ozone RIA Appendix 5a pages 5a-6 – 5a-7)	1,100-2,500	
Retrofit Class 6 & 7 Trucks (2008 ozone RIA Appendix 5a pages 5a-6 – 5a-7)	5,600-14,100	
Retrofit Non-road Equipment – SCR (2008 ozone RIA Appendix 5a pages 5a-6 – 5a-7)	2,600-10,400	
Retrofit Non-road Equipment – Rebuild/Upgrade (2008 ozone RIA Appendix 5a pages 5a-6 – 5a-7)	1,000-4,900	
Improve Aftermarket Replacement Catalytic Converters (2008 ozone RIA Appendix 7a pages 7a-6 – 7a-8)	3,700	
Commuter Programs (2008 ozone RIA Appendix 5a pages 5a-10 – 5a-11)	19,200	
Improve Catalyst Efficiency for New Light-duty Vehicles (2008 ozone RIA Appendix 7a pages 7a-3 – 7a-4)	17,500	

[a] Table presents aggregate program-wide cost/ton over 30 years, discounted at a 3 percent NPV, except for Stationary CI Engines and Locomotive/Marine retrofits, for which annualized costs of control for individual sources are presented. All figures are in 2006 U.S. dollars per short ton.
[b] Includes NO_X plus non-methane hydrocarbons (NMHC). NMHC are also ozone precursors, thus some rules set combined NO_X+NMHC emissions standards. NMHC are a small fraction of the overall reductions so aggregate cost/ton comparisons are still reasonable.
[c] Low end of range represents costs for marine engines with credit for fuel savings, high end of range represents costs for other nonroad SI engines without credit for fuel savings.
[d] Discounted aggregate cost effectiveness.

To calculate the engineering costs for this screening-level near-roadway analysis we multiplied the tons needed from Section 3.3 by the lower and upper ends of the range of \$3,000 to \$6,000/ton (2006\$). Cost estimates are provided in Table 3.3. Note that due to the screening level nature of this analysis, we did not examine local conditions for each of these areas and apply known control measures.

Table 3-3: Total Costs (millions of 2006\$) by County in 2020 for More Stringent Level of 80 ppb[a]

County	ST	Tons reduced (30% Gradient)	Tons reduced (65% Gradient)	Tons reduced (100% Gradient)
Adams Co	CO		\$2.0 to \$4.1	\$24 to \$48
East Baton Rouge Par	LA			\$1.4 to \$2.7
El Paso Co	TX			\$26 to \$52
Salt Lake Co	UT			\$12 to \$25

[a] Total Cost estimates are shown as a range of annualized costs from \$3,000/ton to \$6,000/ton. Results do not include monitoring costs, estimated to be \$3.6m for the U.S. Costs estimates were only available for a 3% discount rate.

3.5 Key Limitations

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

- *Current PM2.5 and Ozone Controls in Baseline*: Our 2020 analysis year baseline assumes that States will put in place the necessary control strategies to attain the current $PM_{2.5}$ and ozone standards. There is a significant level of uncertainty in the control strategies assumed to be employed in these RIAs. As States develop their plans for attaining these standards, their NOx control strategies may differ significantly from our analysis.

- *Use of Existing CMAQ Model Runs*: This analysis represents a screening level analysis. We did not conduct new regional scale modeling specifically targets to

NO_2; instead we relied upon impact ratios developed from model runs used in the analysis underlying the ozone NAAQS.

- *Analysis Year of 2020*: Data limitations necessitated the choice of an analysis year of 2020, as opposed to the presumptive implementation year of 2017. Emission inventory projections are available for 5-year increments; i.e. we have inventories for 2015 and 2020, but not 2017. In addition, the CMAQ model runs upon which we relied were also based on an analysis year of 2020.

Chapter 4: Benefits Analysis Approach and Results

Synopsis

EPA estimated the monetized human health benefits of reducing cases of morbidity and premature mortality among populations exposed to NO_2 and $PM_{2.5}$ for alternate levels of the NO_2 NAAQS standard. In this analysis, we examined alternate standard levels of 80 ppb, 100 ppb, and 125 ppb with near-roadway gradients of 30%, 65%, and 100% to simulate the effect of a near-roadway monitoring network at a 98th percentile. For the selected standard of 100 ppb, there would be zero costs and benefits as we project all areas to attain this standard without additional controls. However, we present the benefits results for the more stringent alternatives in this chapter. These estimates reflect EPA's most current interpretation of the scientific literature on $PM_{2.5}$ and mortality, including our updated benefits methodology (i.e., a no-threshold model that calculates incremental benefits down to the lowest modeled $PM_{2.5}$ air quality levels and incorporates two technical updates) compared to the estimates in previous RIAs that did not include these changes. These benefits are incremental to an air quality baseline that reflects attainment with the recent National Ambient Air Quality Standards (NAAQS) for Ozone (U.S. EPA, 2008a) and $PM_{2.5}$ (U.S. EPA, 2006c). Higher or lower estimates of benefits are possible using other assumptions. Methodological limitations and a lack of air quality data prevented EPA from monetizing the benefits from several important benefit categories, including health benefits of reduced NO_2 exposure near roadways, co-benefits from reduced ozone exposure, ecosystem effects from nitrogen deposition, and improvements in visibility.

4.1 NO_2 Health Benefits

Epidemiological researchers have associated NO_2 exposure with adverse health effects in numerous toxicological, clinical and epidemiological studies, as described in the Integrated Science Assessment for Oxides of Nitrogen - Health Criteria (Final Report) (U.S. EPA, 2008c; hereafter, "NO_2 ISA"). The NO_2 ISA provides a comprehensive review of the current evidence of health and environmental effects of NO_2. The Risk and Exposure Assessment for NO_2 summarizes the NO_2 ISA conclusions regarding health effects from NO_2 exposure as follows (U.S. EPA, 2008e; Section 4.2.1):

> "The ISA concludes that, taken together, recent studies provide scientific evidence that is sufficient to infer a likely causal relationship between short-term NO_2 exposure and adverse effects on the respiratory system (ISA, section 5.3.2.1). This finding is supported by the large body of recent epidemiologic evidence as well as findings from

human and animal experimental studies. These epidemiologic and experimental studies encompass a number of endpoints including [Emergency Department (ED)] visits and hospitalizations, respiratory symptoms, airway hyperresponsiveness, airway inflammation, and lung function. Effect estimates from epidemiologic studies conducted in the United States and Canada generally indicate a 2-20% increase in risks for ED visits and hospital admissions and higher risks for respiratory symptoms (ISA, section 5.4)."

Previous reviews of the NO_2 primary NAAQS, completed in 1985 and 1996, did not include a quantitative benefits assessment for NO_2 exposure. A number of adverse health effects have been found to be associated with NO_2 exposure, but only a subset are ready to be quantified with a dose-response relationship for a benefits analysis due to limitations in understanding for some of these health endpoints. As part of this analysis, we identified those endpoints with sufficient evidence to support a quantified concentration-response relationship using the information presented in the NO_2 ISA, which contains an extensive literature review for several health endpoints related to NO_2 exposure. Because the ISA only included studies published or accepted for publication through December 2007, we also performed supplemental literature searches in the online search engine PubMed® to identify relevant studies published between January 2008, and the present.[1] Based on our review of this information, we identified four short-term morbidity endpoints that the NO_2 ISA identified as "sufficient to infer a likely causal relationship": asthma exacerbation, respiratory-related emergency department visits, and respiratory-related hospitalizations. In addition, there are other endpoints potentially linked to NO_2 exposure, but which are not yet ready to quantify with concentration-response functions in a benefits analysis, such as pulmonary function and other categories of hospitalizations and emergency department visits. The differing evidence and associated strength of the evidence for these different effects is described in detail in the NO_2 ISA.

The NO_2 ISA concluded that the relationship between short-term NO_2 exposure and premature mortality was "suggestive but not sufficient to infer a causal relationship" because it is difficult to attribute the mortality risk effects to NO_2 alone. Therefore, our current decision is not to quantify premature mortality from NO_2 exposure despite evidence suggesting a positive association (U.S. EPA, 2008a, Section 3.3.2). Although the NO_2 ISA stated that studies consistently reported a relationship between NO_2 exposure and mortality, the effect was

[1] We identified one additional study (O'Conner et al., 2008) as part of this analysis that was published after the cut-off date for inclusion in the NO_2 ISA. For more information regarding the studies identified, please see the study summaries provided in Appendix 4a of this RIA.

generally smaller than that for other pollutants such as PM. We may revisit this decision in future benefits assessment for NO_2.

When identifying concentration-response functions, we reviewed the scientific evidence regarding the presence of thresholds in the concentration-response functions for NO_2-related health effects to determine whether the function is approximately linear across the relevant concentration range. The NO_2 ISA concluded that, "[t]hese results do not provide adequate evidence to suggest that nonlinear departures exist along any part of this range of NO_2 exposure concentrations." Therefore, we do not believe that there is sufficient justification to incorporate thresholds in the concentration-response function for NO_2-related health effects.

We were unable to estimate the health benefits of reduced NO_2 exposure in this near-roadway analysis because we do not have fine-scale air quality modeling data available for this analysis, and we cannot speculate on the exact location of near-roadway monitors that do not yet exist. Without knowing the specific monitor location, it is difficult to estimate the near-roadway exposure for nearby populations because the gradient can be highly variable. Because benefits estimation is highly dependent on the number of people exposed to various concentrations and because all of the epidemiology studies rely on the current area-wide monitoring network, we were unable to estimate the NO_2 health benefits for this analysis. Therefore, this analysis only quantifies and monetizes the $PM_{2.5}$ co-benefits associated with those reductions in NO_2 required to meet alternate standard levels. Although it is not appropriate for estimating near-roadway exposures for this particular analysis, we retain the methodology for estimating area-wide NO_2 health benefits in Appendix 4a.

4.2 $PM_{2.5}$ Health Co-Benefits

Because NO_2 is also a precursor to $PM_{2.5,}$ reducing NO_2 emissions in the projected non-attainment areas would also reduce $PM_{2.5}$ formation, human exposure and the incidence of $PM_{2.5}$-related health effects. In this analysis, we estimated the co-benefits of reducing $PM_{2.5}$ exposure for the alternative standards. Due to analytical limitations, it was not possible to provide a comprehensive estimate of $PM_{2.5}$-related benefits. Instead, we used the "benefit-per-ton" method to estimate these benefits (Fann et al., 2009). The $PM_{2.5}$ benefit-per-ton methodology incorporates key assumptions described in detail below. These $PM_{2.5}$ benefit-per-ton estimates provide the total monetized human health benefits (the sum of premature mortality and premature morbidity) of reducing one ton of $PM_{2.5}$ from a specified source. EPA has used the benefit per-ton technique in previous RIAs, including the recent Ozone NAAQS RIA (U.S. EPA, 2008a) and SO_2 NAAQS RIA (U.S. EPA, 2009e). Table 4-1 shows the quantified and unquantified benefits captured in those benefit-per-ton estimates. Please see Chapter 2 of this RIA for more information on the tons of emission reductions calculated for the control strategy.

Table 4-1: Human Health and Welfare Effects of $PM_{2.5}$

Pollutant / Effect	Quantified and Monetized in Primary Estimates	Unquantified Effects Changes in:
$PM_{2.5}$	Adult premature mortality Bronchitis: chronic and acute Hospital admissions: respiratory and cardiovascular Emergency room visits for asthma Nonfatal heart attacks (myocardial infarction) Lower and upper respiratory illness Minor restricted-activity days Work loss days Asthma exacerbations (asthmatic population) Infant mortality	Subchronic bronchitis cases Low birth weight Pulmonary function Chronic respiratory diseases other than chronic bronchitis Non-asthma respiratory emergency room visits Visibility Household soiling

Consistent with the Portland Cement NESHAP (U.S. EPA, 2009a), the benefits estimates utilize the concentration-response functions as reported in the epidemiology literature, as well as the 12 functions obtained in EPA's expert elicitation study as a sensitivity analysis.

- One estimate is based on the concentration-response (C-R) function developed from the extended analysis of American Cancer Society (ACS) cohort, as reported in Pope et al. (2002), a study that EPA has previously used to generate its primary benefits estimate. When calculating the estimate, EPA applied the effect coefficient as reported in the study without an adjustment for assumed concentration threshold of 10 $\mu g/m^3$ as was done in recent (post-2006) Office of Air and Radiation RIAs.
- One estimate is based on the C-R function developed from the extended analysis of the Harvard Six Cities cohort, as reported by Laden et al (2006). This study, published after the completion of the Staff Paper for the 2006 $PM_{2.5}$ NAAQS, has been used as an alternative estimate in the $PM_{2.5}$ NAAQS RIA and $PM_{2.5}$ co-benefits estimates in RIAs completed since the $PM_{2.5}$ NAAQS. When calculating the estimate, EPA applied the effect coefficient as reported in the study without an adjustment for assumed concentration threshold of 10 $\mu g/m^3$ as was done in recent (post 2006) RIAs.
- Twelve estimates are based on the C-R functions from EPA's expert elicitation study (IEc, 2006; Roman et al., 2008) on the $PM_{2.5}$ -mortality relationship and interpreted for benefits analysis in EPA's final RIA for the $PM_{2.5}$ NAAQS. For that study, twelve experts (labeled A through L) provided independent estimates of the $PM_{2.5}$ -mortality concentration-response function. EPA practice has been to develop independent estimates of $PM_{2.5}$ -mortality estimates corresponding to the

concentration-response function provided by each of the twelve experts, to better characterize the degree of variability in the expert responses.

The effect coefficients are drawn from epidemiology studies examining two large population cohorts: the American Cancer Society cohort (Pope et al., 2002) and the Harvard Six Cities cohort (Laden et al., 2006).[2] These are logical choices for anchor points in our presentation because, while both studies are well designed and peer reviewed, there are strengths and weaknesses inherent in each, which we believe argues for using both studies to generate benefits estimates. Previously, EPA had calculated benefits based on these two empirical studies, but derived the range of benefits, including 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).[3] Within this assessment, we include the benefits estimates derived from the concentration-response function provided by each of the twelve experts to better characterize the uncertainty in the concentration-response function for mortality and the degree of variability in the expert responses. Because the experts used these cohort studies to inform their concentration-response functions, benefits estimates using these functions generally fall between results using these epidemiology studies (see Figure 4-9). In general, the expert elicitation results support the conclusion that the benefits of $PM_{2.5}$ control are very likely to be substantial.

Readers interested in reviewing the methodology for creating the benefit-per-ton estimates used in this analysis can consult Fann et al. (2009). As described in the documentation for the benefit per-ton estimates cited above, national per-ton estimates are developed for selected pollutant/source category combinations. The per-ton values calculated therefore apply only to tons reduced from those specific pollutant/source combinations (e.g., NO_2 emitted from electric generating units; NO_2 emitted from mobile sources). Our estimate of $PM_{2.5}$ co-control benefits is therefore based on the total emissions controlled by sector and multiplied by this per-ton value. For this analysis, the $PM_{2.5}$ co-benefits only represent NO_x emission reductions from the mobile sector because data limitations in the control strategy preclude estimating co-emission reductions from directly emitted $PM_{2.5}$ or $PM_{2.5}$ precursors. Each of the illustrative control measures reduces emissions of NO_x, and the diesel retrofits and elimination of long duration idling would also reduce direct emissions of $PM_{2.5}$.[4] We were unable to quantify the direct $PM_{2.5}$ emission reductions in this analysis. We assume that all fine particles, regardless of their chemical composition, are equally potent.

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[2] These two studies specify multi-pollutant models that control for SO_2, among other co-pollutants.

[3] Please see the Section 5.2 of the Portland Cement RIA in Appendix 5A for more information regarding the change in the presentation of benefits estimates.

[4] For more information regarding the illustrative control strategies, please consult Chapter 3 of this RIA.

The benefit-per-ton coefficients in this analysis were derived using modified versions of the health impact functions used in the PM NAAQS Regulatory Impact Analysis. Specifically, this analysis uses the benefit-per-ton estimates first applied in the Portland Cement NESHAP RIA (U.S. EPA, 2009a), which incorporated three updates: a new population dataset, an expanded geographic scope of the benefit-per-ton calculation, and the functions directly from the epidemiology studies without an adjustment for an assumed threshold.[5] Removing the threshold assumption is a key difference between the method used in this analysis of PM-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.

EPA strives to use the best available science to support our benefits analyses, and we recognize that interpretation of the science regarding air pollution and health is dynamic and evolving. Based on our review of the body of scientific literature, EPA applied the no-threshold model in this analysis. EPA's Integrated Science Assessment (2009c), which was reviewed by EPA's Clean Air Scientific Advisory Committee (U.S. EPA-SAB, 2009a; U.S. EPA-SAB, 2009b), concluded that the scientific literature consistently finds that a no-threshold log-linear model most adequately portrays the PM-mortality concentration-response relationship while recognizing potential uncertainty about the exact shape of the concentration-response function.[6] Although this document does not necessarily represent agency policy, it provides a basis for reconsidering the application of thresholds in $PM_{2.5}$ concentration-response functions used in EPA's RIAs.[7] It is important to note that while CASAC provides advice regarding the science associated with setting the National Ambient Air Quality Standards, typically other scientific advisory bodies provide specific advice regarding benefits analysis.[8]

Because the benefits are sensitive to the assumption of a threshold, we also provide a sensitivity analysis using the previous methodology (i.e., a threshold model at 10 $\mu g/m^3$ without the two technical updates) as a historical reference. Table 4-6 shows the sensitivity of an assumed threshold on the monetized results, with and without an assumed threshold at 10 $\mu g/m^3$.

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[5] The benefit-per-ton estimates have also been updated since the Cement RIA to incorporate a revised VSL, as discussed on the next page.

[6] It is important to note that uncertainty regarding the shape of the concentration-response function is conceptually distinct from an assumed threshold. An assumed threshold (below which there are no health effects) is a discontinuity, which is a specific example of non-linearity.

[7] The final PM ISA, which will have undergone the full agency scientific review process, is scheduled to be completed in late December 2009.

[8] In the Portland Cement RIA (U.S. EPA, 2009a), EPA solicited comment on the use of the no-threshold model for benefits analysis within the preamble of that proposed rule. The comment period for the Portland Cement proposed NESHAP closed on September 4, 2009 (Docket ID No. EPA–HQ–OAR–2002–0051 available at http://www.regulations.gov). EPA is currently reviewing those comments.

As is the nature of Regulatory Impact Analyses (RIAs), the assumptions and methods used to estimate air quality benefits evolve over time to reflect the Agency's most current interpretation of the scientific and economic literature. For a period of time (2004-2008), the Office of Air and Radiation (OAR) valued mortality risk reductions using a value of statistical life (VSL) estimate derived from a limited analysis of some of the available studies. OAR arrived at a VSL using a range of $1 million to $10 million (2000$) consistent with two meta-analyses of the wage-risk literature. The $1 million value represented the lower end of the interquartile range from the Mrozek and Taylor (2002) meta-analysis of 33 studies. The $10 million value represented the upper end of the interquartile range from the Viscusi and Aldy (2003) meta-analysis of 43 studies. The mean estimate of $5.5 million (2000$)[9] was also consistent with the mean VSL of $5.4 million estimated in the Kochi et al. (2006) meta-analysis. However, the Agency neither changed its official guidance on the use of VSL in rule-makings nor subjected the interim estimate to a scientific peer-review process through the Science Advisory Board (SAB) or other peer-review group.

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

Until updated guidance is available, the Agency determined that a single, peer-reviewed estimate applied consistently best reflects the SAB-EEAC advice it has received. Therefore, the Agency has decided to apply the VSL that was vetted and endorsed by the SAB in the Guidelines for Preparing Economic Analyses (U.S. EPA, 2000)[10] while the Agency continues its efforts to update its guidance on this issue. This approach calculates a mean value across VSL estimates derived from 26 labor market and contingent valuation studies published between 1974 and

[9] In this analysis, we adjust the VSL to account for a different currency year (2006$) and to account for income growth to 2020. After applying these adjustments to the $5.5 million value, the VSL is $7.7m.

[10] In the (draft) update of the Economic Guidelines (U.S. EPA, 2008d), EPA retained the VSL endorsed by the SAB with the understanding that further updates to the mortality risk valuation guidance would be forthcoming in the near future. Therefore, this report does not represent final agency policy.

1991. The mean VSL across these studies is $6.3 million (2000$).[11] The Agency is committed to using scientifically sound, appropriately reviewed evidence in valuing mortality risk reductions and has made significant progress in responding to the SAB-EEAC's specific recommendations. The Agency anticipates presenting results from this effort to the SAB-EEAC in Spring 2010 and that draft guidance will be available shortly thereafter.

Table 4-2 provides the unit values used to monetize the benefits of reduced exposure to $PM_{2.5}$. Figure 4-1 illustrates the relative breakdown of the monetized $PM_{2.5}$ health benefits.

Table 4-2: Unit Values used for Economic Valuation of $PM_{2.5}$ Health Endpoints (2006$)*

Health Endpoint	Central Estimate of Value Per Statistical Incidence (2020 income level)	Derivation of Distributions of Estimates
Premature Mortality (Value of a Statistical Life)	$8,900,000	EPA currently recommends a central VSL of $6.3m (2000$) based on a Weibull distribution fitted to 26 published VSL estimates (5 contingent valuation and 21 labor market studies). The underlying studies, the distribution parameters, and other useful information are available in Appendix B of EPA's current Guidelines for Preparing Economic Analyses (U.S. EPA, 2000).
Chronic Bronchitis (CB)	$490,000	The WTP to avoid a case of pollution-related CB is calculated as $WTP_x = WTP_{13} * e^{-\beta*(13-x)}$, where x is the severity of an average CB case, WTP13 is the WTP for a severe case of CB, and $ is the parameter relating WTP to severity, based on the regression results reported in Krupnick and Cropper (1992). The distribution of WTP for an average severity-level case of CB was generated by Monte Carlo methods, drawing from each of three distributions: (1) WTP to avoid a severe case of CB is assigned a 1/9 probability of being each of the first nine deciles of the distribution of WTP responses in Viscusi et al. (1991); (2) the severity of a pollution-related case of CB (relative to the case described in the Viscusi study) is assumed to have a triangular distribution, with the most likely value at severity level 6.5 and endpoints at 1.0 and 12.0; and (3) the constant in the elasticity of WTP with respect to severity is normally distributed with mean = 0.18 and standard deviation = 0.0669 (from Krupnick and Cropper [1992]). This process and the rationale for choosing it is described in detail in the Costs and Benefits of the Clean Air Act, 1990 to 2010 (U.S. EPA, 1999).
Nonfatal Myocardial Infarction (heart attack)		No distributional information available. Age-specific cost-of-illness values reflect lost earnings and direct medical costs over a 5-year on period following a nonfatal MI. Lost earnings estimates are based Cropper and Krupnick (1990). Direct medical costs are based on

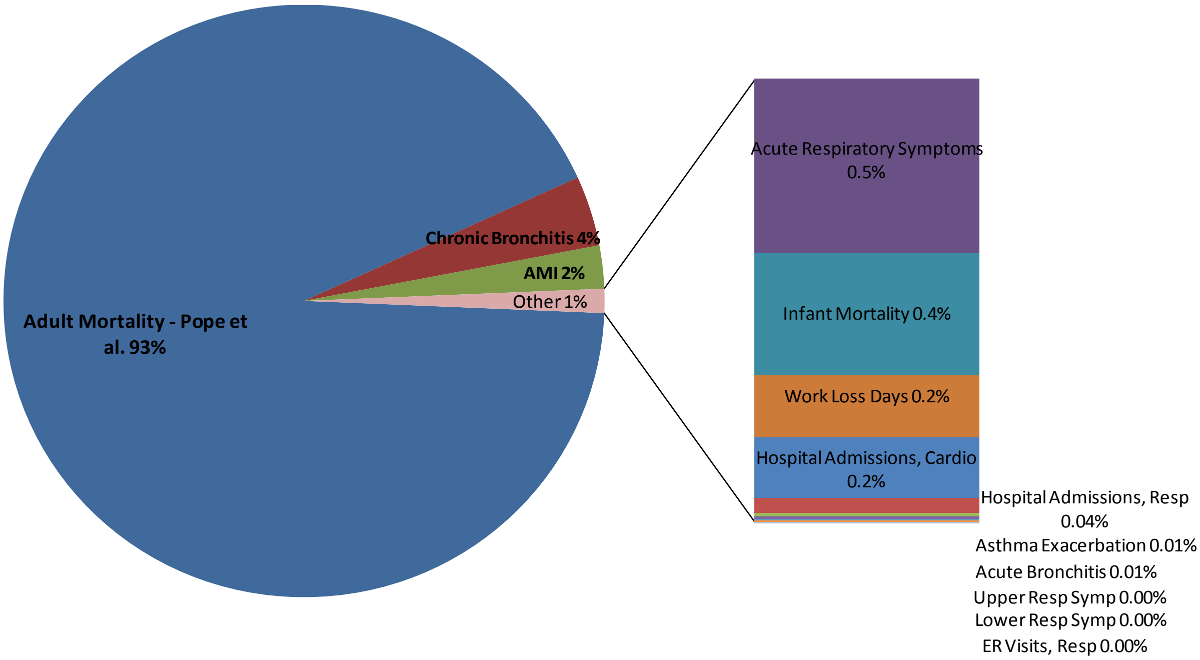
[11] In this analysis, we adjust the VSL to account for a different currency year (2006$) and to account for income growth to 2020. After applying these adjustments to the $6.3 million value, the VSL is $8.9m.

<u>3% discount rate</u>		simple average of estimates from Russell et al. (1998) and Wittels et al. (1990). Lost earnings: Cropper and Krupnick (1990). Present discounted value of 5 years of lost earnings in (2006$): age of onset: at 3%, at 7% 25–44: $11,000, $10,000 45–54: $17,000, $15,000 55–65: $96,000, $86,000 Direct medical expenses: An average of: 1. Wittels et al. (1990) ($130,000—no discounting) 2. Russell et al. (1998), 5-year period ($29,000 at 3%, $27,000 at 7%)
Age 0–24	$80,000	
Age 25–44	$90,000	
Age 45–54	$94,000	
Age 55–65	$170,000	
Age 66 and over	$80,000	
<u>7% discount rate</u>		
Age 0–24	$80,000	
Age 25–44	$88,000	
Age 45–54	$92,000	
Age 55–65	$160,000	
Age 66 and over	$78,000	
Hospital Admissions and ER Visits		
Chronic Obstructive Pulmonary Disease (COPD)	$17,000	No distributional information available. The COI estimates (lost earnings plus direct medical costs) are based on ICD-9 code-level information (e.g., average hospital care costs, average length of hospital stay, and weighted share of total COPD category illnesses) reported in Agency for Healthcare Research and Quality (2000) (www.ahrq.gov).
Asthma Admissions	$8,900	No distributional information available. The COI estimates (lost earnings plus direct medical costs) are based on ICD-9 code-level information (e.g., average hospital care costs, average length of hospital stay, and weighted share of total asthma category illnesses) reported in Agency for Healthcare Research and Quality (2000) (www.ahrq.gov).
All Cardiovascular	$25,000	No distributional information available. The COI estimates (lost earnings plus direct medical costs) are based on ICD-9 code-level information (e.g., average hospital care costs, average length of hospital stay, and weighted share of total cardiovascular category illnesses) reported in Agency for Healthcare Research and Quality (2000) (www.ahrq.gov).
All respiratory (ages 65+)	$25,000	No distributions available. The COI point estimates (lost earnings plus direct medical costs) are based on ICD-9 code level information (e.g., average hospital care costs, average length of hospital stay, and weighted share of total COPD category illnesses) reported in Agency for Healthcare Research and Quality, 2000 (www.ahrq.gov).
All respiratory (ages 0–2)	$10,000	No distributions available. The COI point estimates (lost earnings plus direct medical costs) are based on ICD-9 code level information (e.g., average hospital care costs, average length of hospital stay, and weighted share of total COPD category illnesses) reported in Agency for Healthcare Research and Quality, 2000 (www.ahrq.gov).
Emergency Room Visits for Asthma	$370	No distributional information available. Simple average of two unit COI values: (1) $400 (2006$), from Smith et al. (1997) and (2) $340 (2006$), from Stanford et al. (1999).

Respiratory Ailments Not Requiring Hospitalization		
Upper Respiratory Symptoms (URS)	$31	Combinations of the three symptoms for which WTP estimates are available that closely match those listed by Pope et al. result in seven different "symptom clusters," each describing a "type" of URS. A dollar value was derived for each type of URS, using mid-range estimates of WTP (IEc, 1994) to avoid each symptom in the cluster and assuming additivity of WTPs. In the absence of information surrounding the frequency with which each of the seven types of URS occurs within the URS symptom complex, we assumed a uniform distribution between $11 and $50 (2006$).
Lower Respiratory Symptoms (LRS)	$19	Combinations of the four symptoms for which WTP estimates are available that closely match those listed by Schwartz et al. result in 11 different "symptom clusters," each describing a "type" of LRS. A dollar value was derived for each type of LRS, using mid-range estimates of WTP (IEc, 1994) to avoid each symptom in the cluster and assuming additivity of WTPs. The dollar value for LRS is the average of the dollar values for the 11 different types of LRS. In the absence of information surrounding the frequency with which each of the 11 types of LRS occurs within the LRS symptom complex, we assumed a uniform distribution between $8 and $29 (2006$).
Asthma Exacerbations	$53	Asthma exacerbations are valued at $49 (2006$) per incidence, based on the mean of average WTP estimates for the four severity definitions of a "bad asthma day," described in Rowe and Chestnut (1986). This study surveyed asthmatics to estimate WTP for avoidance of a "bad asthma day," as defined by the subjects. For purposes of valuation, an asthma exacerbation is assumed to be equivalent to a day in which asthma is moderate or worse as reported in the Rowe and Chestnut (1986) study. The value is assumed have a uniform distribution between $19 and $83 (2006$).
Acute Bronchitis	$440	Assumes a 6-day episode, with the distribution of the daily value specified as uniform with the low and high values based on those recommended for related respiratory symptoms in Neumann et al. (1994). The low daily estimate of $12 (2006$) is the sum of the mid-range values recommended by IEc for two symptoms believed to be associated with acute bronchitis: coughing and chest tightness. The high daily estimate was taken to be twice the value of a minor respiratory restricted-activity day, or $130 (2006$).
Work Loss Days (WLDs)	Variable	No distribution available. Point estimate is based on county-specific median annual wages divided by 50 (assuming 2 weeks of vacation) and then by 5—to get median daily wage. U.S. Year 2000 Census, compiled by Geolytics, Inc.
Minor Restricted Activity Days (MRADs)	$63	Median WTP estimate to avoid one MRAD from Tolley et al. (1986). Distribution is assumed to be triangular with a minimum of $26 and a maximum of $97 (2006$). Range is based on assumption that value should exceed WTP for a single mild symptom (the highest estimate for a single symptom—for eye irritation—is $19 (2006$)) and be less than that for a WLD. The triangular distribution acknowledges that the actual value is likely to be closer to the point estimate than either extreme.

*All estimates rounded to two significant figures. All values have been inflated to reflect values in 2006 dollars and 2020 income levels.

Figure 4-1: Breakdown of Monetized $PM_{2.5}$ Health Benefits using Mortality Function from Pope et al *



*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.

Because epidemiology studies have indicated that there is a lag between exposure to $PM_{2.5}$ and premature mortality, the discount rate has a substantial effect on the final monetized benefits. Therefore, we provide the PM co-benefit results using both discount rates in Table 4-5, and we test the sensitivity of the results to discount rates of 3% and 7% in Table 4-7.

In this analysis, we examined alternate standard levels of 80 ppb, 100 ppb, and 125 ppb with near-roadway gradients of 30%, 65%, and 100% at a 98th percentile. As there are no areas that are projected to not attain the following standard levels, these standard levels would have zero costs and benefits: 30% gradient, 100 ppb with any gradient, and 125 ppb with any gradient. Therefore, we have not presented these alternative standards in the results shown below. We provide the benefit-per-ton estimates used in this analysis in Table 4-3. To be consistent with the cost analysis, we only used the benefit-per-ton estimate corresponding to NO_X emission reductions from the mobile sector. Table 4-4 provides the health incidences associated with alternate levels of the standard. Table 4-5 shows the monetized results using the two epidemiology-based estimates as well as the 12 expert-based estimates. Table 4-6 shows the monetized results for all standard levels and all gradients at discount rates of 3% and

7%. Figure 4-3 provides an illustrative graphical representation of all 14 of the $PM_{2.5}$ co-benefits, at both a 3 percent and 7 percent discount rate for the most stringent alternative analyzed (80 ppb at 100% gradient). Other standard levels would show a similar distribution of values, albeit with smaller magnitudes.

Table 4-3: $PM_{2.5}$ Benefit-per-ton estimates at discount rates of 3% and 7% (millions of 2006$)[a]

$PM_{2.5}$ Precursor	Benefit per Ton Estimate (Pope)	Benefit per Ton Estimate (Laden)
NO_x Mobile 3% (no-threshold) [b]	$5,200	$13,000
NO_x Mobile 7% (no-threshold) [b]	$4,700	$11,000

[a] Numbers have been rounded to two significant figures. This table includes extrapolated tons, spread across the sectors in proportion to the emissions in the county. $PM_{2.5}$ co-benefit estimates do not include confidence intervals because they are derived using benefit per-ton estimates. For the selected standard of 100 ppb, there would be zero costs and benefits as we project all areas to attain this standard without additional controls.
[b] The benefit-per-ton estimates using thresholds are $4,300 to $9,300 at 3% and $3,900 to $8,400 at 7%. These estimates assume a threshold at 10 μg/m^3, and are provided as a historical reference only.

Table 4-4: Summary of Reductions in Health Incidences from $PM_{2.5}$ Co-Benefits to Attain Alternate Standard Levels in 2020*

	80 ppb (65% gradient)	80 ppb (100% gradient)
Avoided Premature Mortality		
Pope	0	10
Laden	1	30
Woodruff (Infant Mortality)	0	0
Avoided Morbidity		
Chronic Bronchitis	0	9
Acute Myocardial Infarction	1	20
Hospital Admissions, Respiratory	0	3
Hospital Admissions, Cardiovascular	0	6
Emergency Room Visits, Respiratory	0	10
Acute Bronchitis	1	20
Work Loss Days	60	2,000
Asthma Exacerbation	8	200
Acute Respiratory Symptoms	300	10,000
Lower Respiratory Symptoms	8	300
Upper Respiratory Symptoms	6	200

*All estimates are for the analysis year (2020) and are rounded to whole numbers with two significant figures. All fine particles are assumed to have equivalent health effects, but each $PM_{2.5}$ precursor pollutant has a different propensity to form $PM_{2.5}$. For the selected standard of 100 ppb, there would be zero costs and benefits as we project all areas to attain this standard without additional controls.

Table 4-5: All $PM_{2.5}$ Co-Benefits Estimates to Attain Alternate Standard Levels in 2020 at discount rates of 3% and 7% (in millions of 2006$)*

	80 ppb (65%)		80 ppb (100%)	
	3%	7%	3%	7%
Benefit-per-ton Coefficients Derived from Epidemiology Literature				
Pope et al.	$3.5	$3.2	$110	$100
Laden et al.	$8.6	$7.8	$270	$240
Benefit-per-ton Coefficients Derived from Expert Elicitation				
Expert A	$9.1	$8.2	$290	$260
Expert B	$7.0	$6.3	$220	$200
Expert C	$6.9	$6.3	$220	$200
Expert D	$4.9	$4.5	$150	$140
Expert E	$11	$10	$350	$320
Expert F	$6.3	$5.7	$200	$180
Expert G	$4.2	$3.8	$130	$120
Expert H	$5.2	$4.7	$160	$150
Expert I	$6.9	$6.2	$220	$200
Expert J	$5.6	$5.1	$180	$160
Expert K	$1.4	$1.3	$45	$42
Expert L	$5.2	$4.7	$160	$150

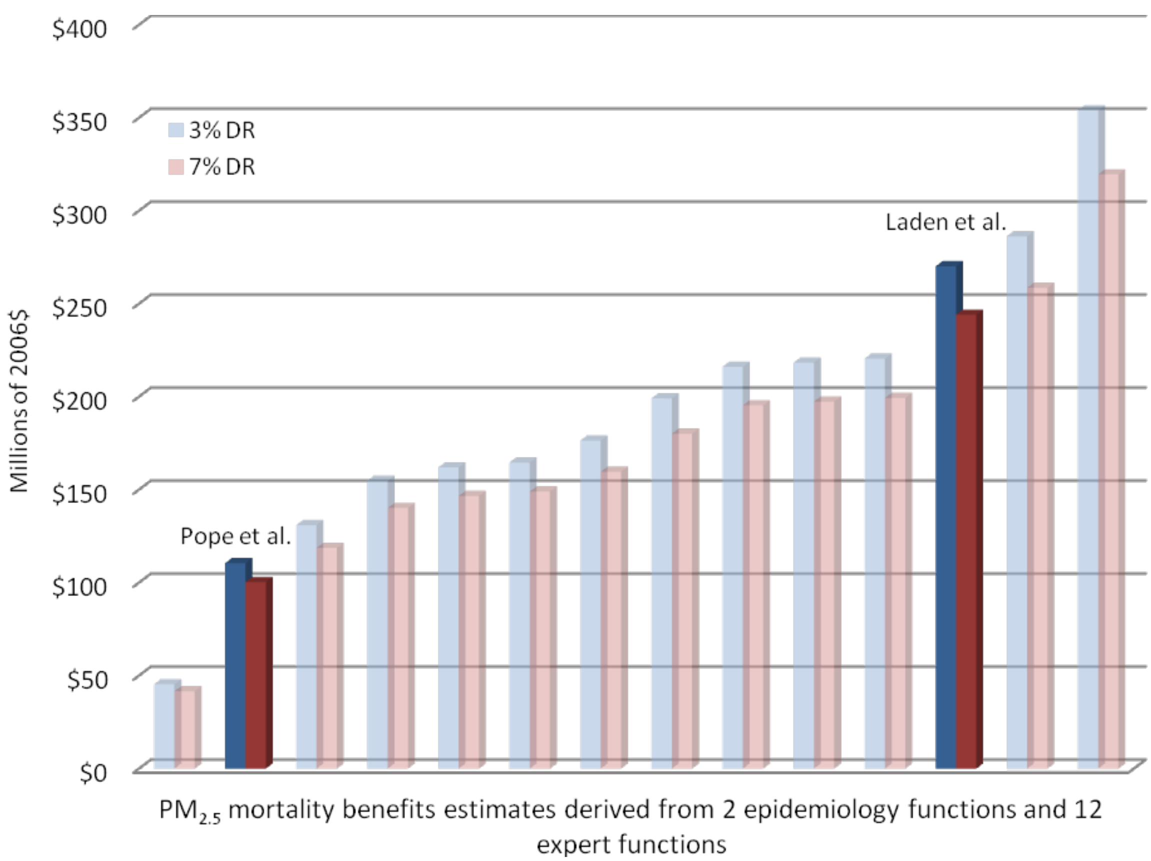
*All estimates are rounded to two significant figures. Estimates do not include confidence intervals because they were derived through the benefit-per-ton technique described above. The benefits estimates from the Expert Elicitation are provided as a reasonable characterization of the uncertainty in the mortality estimates associated with the concentration-response function. For the selected standard of 100 ppb, there would be zero costs and benefits as we project all areas to attain this standard without additional controls.

Table 4-6: $PM_{2.5}$ Co-benefits Estimates to Attain Alternate Standard Levels in 2020 at discount rates of 3% and 7% (in millions of 2006$)*

	Standard Level	Total Benefits 3%	Total Benefits 7%
30% Gradient	80 ppb	$0 to $0	$0 to $0
	100 ppb	$0 to $0	$0 to $0
	125 ppb	$0 to $0	$0 to $0
65% Gradient	80 ppb	$3.5 to $8.6	$3.2 to $7.8
	100 ppb	$0 to $0	$0 to $0
	125 ppb	$0 to $0	$0 to $0
100% Gradient	80 ppb	$110 to $270	$100 to $240
	100 ppb	$0 to $0	$0 to $0
	125 ppb	$0 to $0	$0 to $0

*All estimates are rounded to two significant figures. Total benefits estimates are actually $PM_{2.5}$ co-benefits, shown as a range from Pope et al. to Laden et al. For the selected standard of 100 ppb, there would be zero costs and benefits as we project all areas to attain this standard without additional controls.

Figure 4-3: Monetized $PM_{2.5}$ Co-Benefits of Attaining 80 ppb (100% gradient)



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This graph shows the estimated co-benefits in 2020 using the no-threshold model at discount rates of 3% and 7% using effect coefficients derived from 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 for an alternative standard of 80 ppb at a 100% near-roadway gradient. 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. Other gradients would show a similar distribution of values, albeit with smaller magnitudes. For the selected standard of 100 ppb, there would be zero costs and benefits as we project all areas to attain this standard without additional controls.

We performed a couple of sensitivity analyses on the benefits results to assess the sensitivity of the primary results to various data inputs and assumptions. We then changed each default input one at a time and recalculated the total monetized benefits to assess the percent change from the default. We present the results of this sensitivity analysis in Table 4-6. We indicated each input parameter, the value used as the default, and the values for the sensitivity analyses, and then we provide the total monetary benefits for each input and the percent change from the default value. We show the sensitivity analysis for the most stringent alternative analyzed (80 ppb at 100% gradient) in Table 4-7, but other standard levels would show similar sensitivity to these perturbations, albeit with smaller magnitudes. For the selected standard of 100 ppb, there would be zero costs and benefits as we project all areas to attain this standard without additional controls.

Table 4-7: Sensitivity Analyses for $PM_{2.5}$ Health Co-Benefits for 80 ppb alternative standard (100% gradient)

		Total $PM_{2.5}$ Benefits (millions of 2006$)	% Change from Default
Threshold Assumption (with Epidemiology Study)	No Threshold (Pope)	$110	N/A
	No Threshold (Laden)	$270	N/A
	Threshold (Pope)*	$120	-17%
	Threshold (Laden)*	$250	-27%
Discount Rate (with Epidemiology Study)	3% (Pope)	$110	N/A
	3% (Laden)	$270	N/A
	7% (Pope)	$100	-10%
	7% (Laden)	$240	-10%

* The threshold model is not directly comparable to the no-threshold model. The threshold estimates do not include two technical updates, and they are based on data for 2015, instead of 2020. Directly comparable estimates are not available. For the selected standard of 100 ppb, there would be zero costs and benefits as we project all areas to attain this standard without additional controls.

4.3 Ozone Co-benefits

Because NO_2 is also a precursor to ozone, reducing NO_2 emissions in the projected non-attainment areas would also reduce ozone formation, human exposure and the incidence of ozone-related health effects. 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, 2006a). Epidemiological 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. In addition, there are substantial benefits that would occur from reducing ozone exposure to vegetation (U.S. EPA, 2007). Unfortunately, due to data and resource limitations, we were unable to quantify the health and vegetation effects of reduced ozone exposure that are expected to occur as a result of NO_2 emission reductions required to meet alternate standard levels.

In certain areas of the country, reductions in NO_2 emissions cause localized increases in ozone concentrations, which are sometimes referred to as “ozone disbenefits”. In urban cores, which are often dominated by fresh emissions of NOx, the ozone catalysts are removed via the production of nitric acid, which slows the ozone formation rate. Because NO_X is generally depleted more rapidly than VOCs, this effect is usually short-lived and the emitted NO_X can lead to ozone formation later and further downwind. Therefore, the net effect of NO_2 reductions is generally an overall decrease in ozone exposure.

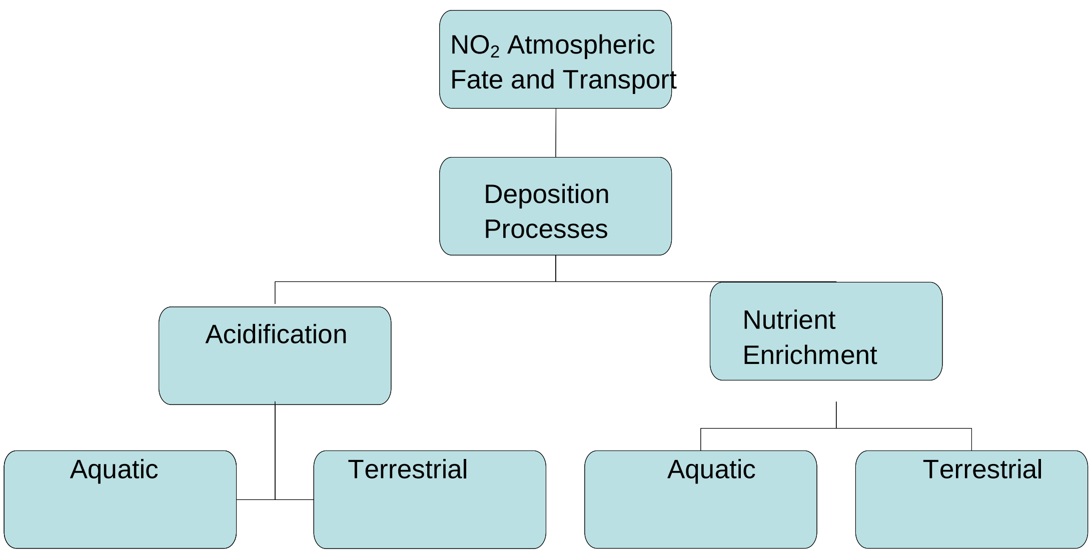
4.4 Unquantified Welfare Benefits

This analysis is limited by the available data and resources. As such, we are not able to quantify several welfare benefit categories in this analysis because we are limited by the available data or resources. In this section, we provide a qualitative assessment of the two largest welfare benefit categories from reduced NO_2 deposition: ecosystem benefits of reducing nitrogen deposition and visibility improvements.

4.4.1 Ecosystem Benefits of Reduced Nitrogen Deposition

Reducing nitrogen deposition has two primary categories of ecosystem benefits – a reduction in acidification and a reduction in excess nutrient enrichment. See the schematic diagram in Figure 4-4. Although there is some evidence that nitrogen deposition may have positive effects on agricultural output through passive fertilization, it is likely that the overall value is very small relative to other health and welfare effects.

Figure 4-4: Schematic of Ecological Effects of Nitrogen Deposition

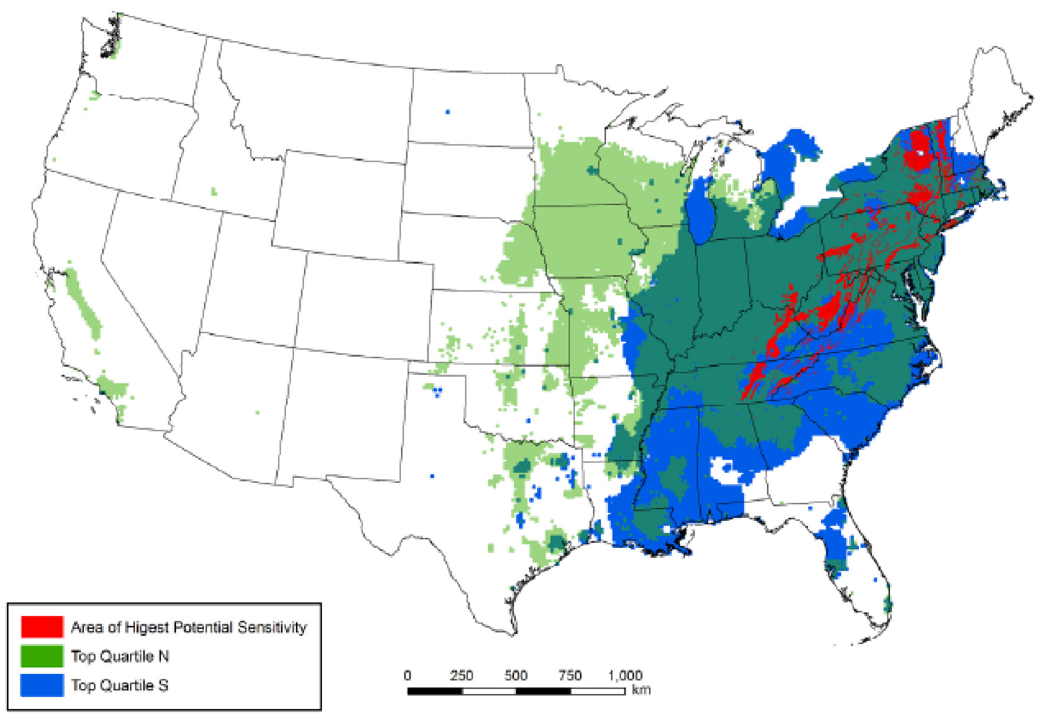


Terrestrial and Aquatic Acidification

Deposition of nitrogen (along with sulfur) causes acidification, which alters biogeochemistry and affects animal and plant life in terrestrial and aquatic ecosystems across the U.S. (U.S. EPA, 2008f). Major effects include a decline in sensitive tree species, such as red spruce (*Picea rubens*) and sugar maple (*Acer saccharum*); and a loss of biodiversity of fishes, zooplankton, and macro invertebrates. The sensitivity of terrestrial and aquatic ecosystems to acidification is predominantly governed by geological characteristics (bedrock, weathering rates, etc.). Biological effects of acidification in terrestrial ecosystems are generally linked to aluminum toxicity and decreased ability of plant roots to take up base cations. Decreases in the acid neutralizing capacity and increases in inorganic aluminum concentration contribute to declines in zooplankton, macro invertebrates, and fish species richness in aquatic ecosystems.

Acidifying deposition has altered major biogeochemical processes in the U.S. by increasing the nitrogen and sulfur content of soils, accelerating nitrate and sulfate leaching from soil to drainage waters, depleting base cations (especially calcium and magnesium) from soils, and increasing the mobility of aluminum. Plants affected by high levels of aluminum from the soil often have reduced root growth, which restricts the ability of the plant to take up water and nutrients, especially calcium (U. S. EPA, 2008f). These direct effects can, in turn, influence the response of these plants to climatic stresses such as droughts and cold temperatures. They can also influence the sensitivity of plants to other stresses, including insect pests and disease (Joslin et al., 1992) leading to increased mortality of canopy trees. Certain ecosystems in the continental U.S. are potentially sensitive to terrestrial acidification (U.S. EPA, 2008f). Figure 4-5 depicts areas across the U.S. that are potentially sensitive to terrestrial acidification.

Figure 4-5: Areas Potentially Sensitive to Terrestrial Acidification (U.S. EPA, 2008f)

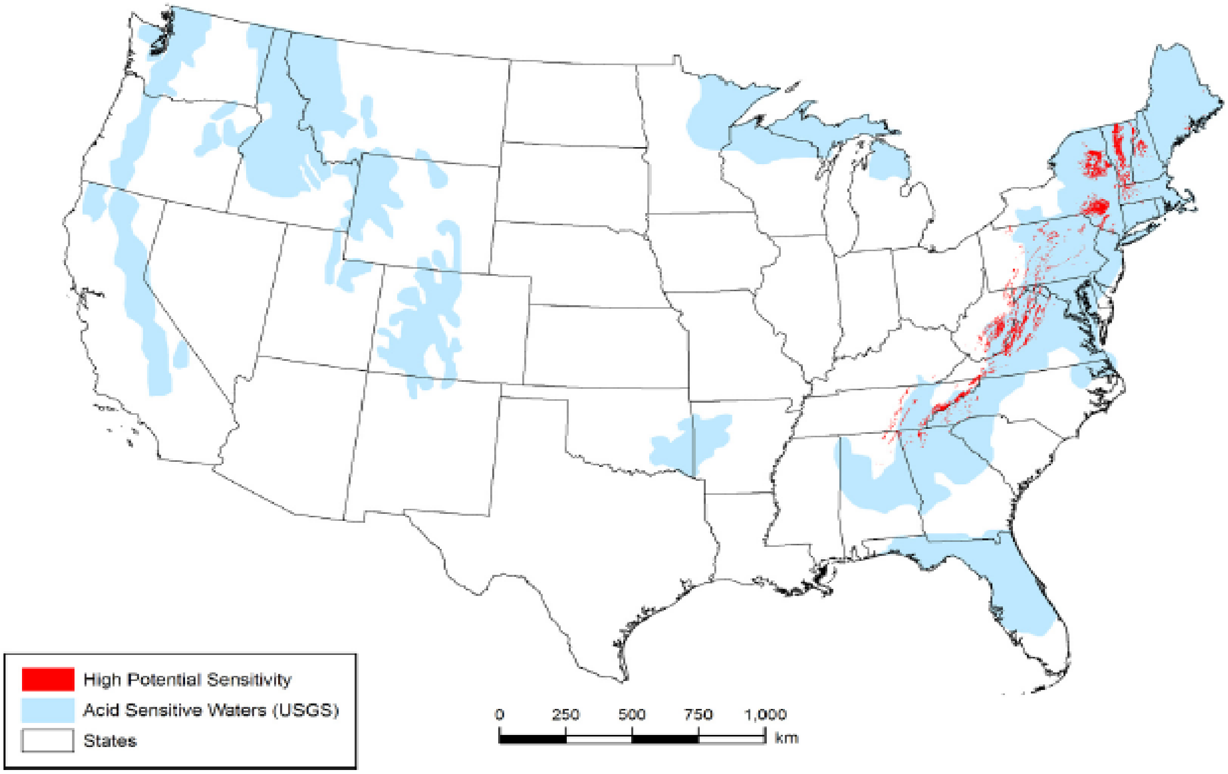


Terrestrial acidification affects several important ecological endpoints, including declines in habitat for threatened and endangered species (cultural), declines in forest aesthetics (cultural), declines in forest productivity (provisioning), and increases in forest soil erosion and reductions in water retention (cultural and regulating) (U.S. EPA, 2009d). Forests in the northeastern United States provide several important and valuable provisioning services in the form of tree products, such as commercial timber and maple syrup. Forests in the northeastern United States are also an important source of cultural ecosystem services—nonuse (i.e., existence value for threatened and endangered species), recreational, and aesthetic services. Forest lands support a wide variety of outdoor recreational activities, including fishing, hiking, camping, off-road driving, hunting, and wildlife viewing. Although it is difficult to estimate the portion of these recreational services that are specifically attributable to forests and to the health of specific tree species, fall color viewing is one recreational activity that is directly dependent on forest conditions. Forests in the northeastern United States also support and provide a wide variety of valuable regulating services, including soil stabilization and erosion control, water regulation, and climate regulation. The total value of these ecosystem services is very difficult to quantify in a meaningful way, as is the reduction in the value of these services associated with total nitrogen and sulfur deposition. As terrestrial acidification contributes to root damages, reduced biomass growth, and tree mortality, all of these services are likely to be affected; however, the magnitude of these impacts is currently very uncertain.

Aquatic acidification effects have been well studied in the U.S. and elsewhere at various trophic levels. These studies indicate that aquatic biota have been affected by acidification at virtually all levels of the food web in acid sensitive aquatic ecosystems. Effects have been most clearly documented for fish, aquatic insects, other invertebrates, and algae. Biological effects are primarily attributable to a combination of low pH and high inorganic aluminum concentrations. Such conditions occur more frequently during rainfall and snowmelt that cause high flows of water and less commonly during low-flow conditions, except where chronic acidity conditions are severe. Biological effects of episodes include reduced fish condition factor, changes in species composition and declines in aquatic species richness across multiple taxa, ecosystems and regions. These conditions may also result in direct fish mortality (Van Sickle et al., 1996). Studies have shown that surface water with acid neutralizing capacity (ANC) values greater than 50 μeq/L tend to protect most fish (i.e., brook trout, others) and other aquatic organisms (U.S. EPA, 2009d).

A number of national and regional assessments have been conducted to estimate the distribution and extent of surface water acidity in the U.S (U.S. EPA, 2008f). As a result, several regions of the U.S. have been identified as containing a large number of lakes and streams that are seriously impacted by acidification. Figure 4-6 illustrates those areas of the U.S. where aquatic ecosystems are at risk from acidification.

Figure 4-6: Areas Potentially Sensitive to Aquatic Acidification (U.S. EPA, 2008f)



Because aquatic acidification primarily affects the diversity and abundance of aquatic biota, it also affects the ecosystem services that are derived from the fish and other aquatic life found in these surface waters (U.S. EPA, 2009d). Although acidification is unlikely to have serious negative effects on water supplies, it can limit the productivity of surface waters as a source of food (i.e., fish). In the northeastern United States, the surface waters affected by acidification are not a major source of commercially raised or caught fish; however, they are a source of food for some recreational and subsistence fishermen and for other consumers. Inland surface waters support several cultural services, including aesthetic and educational services and recreational fishing. Recreational fishing in lakes and streams is among the most popular outdoor recreational activities in the northeastern U.S. In addition, inland surface waters provide a number of regulating services associated with hydrological and climate regulation by providing environments that sustain aquatic food webs. The toxic effects of acidification on fish and other aquatic life disrupt these services, but it is difficult to quantify these services and how they are affected by acidification.

Aquatic Enrichment

One of the main adverse ecological effects resulting from N deposition, particularly in the Mid-Atlantic region of the United States, is the effect associated with nutrient enrichment in estuarine waters. A recent assessment of 141 estuaries nationwide by the National Oceanic and Atmospheric Administration (NOAA) concluded that 19 estuaries (13%) suffered from moderately high or high levels of eutrophication due to excessive inputs of both N and phosphorus, and a majority of these estuaries are located in the coastal area from North Carolina to Massachusetts (NOAA, 2007). For estuaries in the Mid-Atlantic region, the contribution of atmospheric distribution to total N loads is estimated to range between 10% and 58% (Valigura et al., 2001).

Eutrophication in estuaries is associated with a range of adverse ecological effects. The conceptual framework developed by NOAA emphasizes four main types of eutrophication effects—low dissolved oxygen (DO), harmful algal blooms (HABs), loss of submerged aquatic vegetation (SAV), and low water clarity. Low DO disrupts aquatic habitats, causing stress to fish and shellfish, which, in the short-term, can lead to episodic fish kills and, in the long-term, can damage overall growth in fish and shellfish populations. Low DO also degrades the aesthetic qualities of surface water. In addition to often being toxic to fish and shellfish, and leading to fish kills and aesthetic impairments of estuaries, HABs can, in some instances, also be harmful to human health. SAV provides critical habitat for many aquatic species in estuaries and, in some instances, can also protect shorelines by reducing wave strength; therefore, declines in SAV due to nutrient enrichment are an important source of concern. Low water clarity is the result of accumulations of both algae and sediments in estuarine waters. In addition to contributing to declines in SAV, high levels of turbidity also degrade the aesthetic qualities of the estuarine environment.

Estuaries in the eastern United States are an important source of food production, in particular fish and shellfish production. The estuaries are capable of supporting large stocks of resident commercial species, and they serve as the breeding grounds and interim habitat for several migratory species. To provide an indication of the magnitude of provisioning services associated with coastal fisheries, from 2005 to 2007, the average value of total catch was $1.5 billion per year. It is not known, however, what percentage of this value is directly attributable to or dependent upon the estuaries in these states.

Very few studies have developed empirical bioeconomic models to estimate how changes in environmental quality affect fish harvests and the value of these services (Knowler, 2002). One exception is Kahn and Kemp (1985), which estimated a bioeconomic model of commercial and recreational striped bass fishing using annual data from 1965 to 1979, measuring the effects of SAV levels on fish stocks, harvests, and social welfare. They estimated, for example, that a 50% reduction in SAV from levels existing in the late 1970s (similar to current levels [Chesapeake Bay Program, 2008]) would decrease the net social benefits from striped bass by roughly $16 million (in 2007 dollars).

In addition to affecting provisioning services through commercial fish harvests, eutrophication in estuaries may also affect these services through its effects on the demand for seafood. For example, a well-publicized toxic pfiesteria bloom in the Maryland Eastern Shore in 1997, which involved thousands of dead and lesioned fish, led to an estimated $56 million (in 2007 dollars) in lost seafood sales for 360 seafood firms in Maryland in the months following the outbreak (Lipton, 1999).

Estuaries in the United States also provide an important and substantial variety of cultural ecosystem services, including water-based recreational and aesthetic services. The water quality in the estuary directly affects the quality of these experiences.
For example, data from the National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (FHWAR) indicate that, in 2006, 4.8% of the 16 and older population in coastal states from North Carolina to Massachusetts participated in saltwater fishing. The total number of days of saltwater fishing in these states was 26million in 2006. Based on estimates from Kaval and Loomis (2003), the average consumer surplus value for a fishing day was $36(in 2007 dollars) in the Northeast and $87 in the Southeast. Therefore, the total recreational consumer surplus value from these saltwater fishing days was approximately $1.3 billion (in 2007 dollars).

Recreational participation estimates for several other coastal recreational activities are also available for 1999–2000 from the National Survey on Recreation & the Environment (NSRE). As reported in Leeworthy and Wiley (2001), almost 6 million individuals aged 16 and older participated in motorboating in coastal states from North Carolina to Massachusetts, for a total of nearly 63 million days annually during 1999–2000. Using a national daily value estimate of $32 (in 2007 dollars) for motorboating from Kaval and Loomis (2003), the aggregate value of these coastal motorboating outings was $2 billion per year. Almost 7 million participated in birdwatching, for a total of almost 175 million days per year, and more than 3 million participated in visits to non-beach coastal waterside areas, for a total of more than 35 million days per year.

Estuaries and marshes have the potential to support a wide range of regulating services, including climate, biological, and water regulation; pollution detoxification; erosion prevention;

and protection against natural hazards (MEA, 2005c). It is more difficult, however, to identify the specific regulating services that are significantly impacted by changes in nutrient loadings. One potentially affected service is provided by SAV, which can help reduce wave energy levels and thus protect shorelines against excessive erosion. Declines in SAV may, therefore, also increase the risks of episodic flooding and associated damages to near-shore properties or public infrastructure. In the extreme, these declines may even contribute to shoreline retreat, such that land and structures are lost to the advancing waterline.

Terrestrial Enrichment

Terrestrial enrichment occurs when terrestrial ecosystems receive N loadings in excess of natural background levels, either through atmospheric deposition or direct application. Evidence presented in the Integrated Science Assessment (U.S. EPA, 2008) supports a causal relationship between atmospheric N deposition and biogeochemical cycling and fluxes of N and carbon in terrestrial systems. Furthermore, evidence summarized in the report supports a causal link between atmospheric N deposition and changes in the types and number of species and biodiversity in terrestrial systems. Nitrogen enrichment occurs over a long time period; as a result, it may take as much as 50 years or more to see changes in ecosystem conditions and indicators. This long time scale also affects the timing of the ecosystem service changes.

The ecosystem service impacts of terrestrial nutrient enrichment include primarily cultural and regulating services. Concerns focus on a decline in native plants and an increase in nonnative grasses and other species, impacts on the viability of threatened and endangered species, an increase in fire frequency, and a change in a forest's nutrient cycling that may affect surface water quality through nitrate leaching (EPA, 2008). The primary cultural ecosystem services associated with terrestrial ecosystems are recreation, aesthetic, and nonuse values. Below we discuss the possible ecosystem service benefits from reducing N enrichment and provide a general overview of the types and relative magnitude of the benefits. National parks and monuments across the country preserve important terrestrial ecosystems that provide diverse recreational opportunities to the public. Visitors to these parks engage in activities such as camping, hiking, attending educational programs, horseback riding, wildlife viewing, water based recreation, and fishing. The quality of these trips depends in part on the health of the ecosystems and their ability to support the diversity of plants and animals found in important habitats.

The 2006 FHWAR (DOI, 2007) reports on the number of individuals involved in fishing, hunting, and wildlife viewing. Millions of people are involved in just these three activities each year. To take only one state, California, as an example, a day of fishing has an *average* value of

$48 (in 2007 dollars) based on 15 studies (Kaval and Loomis, (2003). For hunting and wildlife viewing in this region, average day values were estimated to be $50and $79from 18 and 23 studies, respectively. Multiplying these average values by the total participation days, the total benefits in 2006 from fishing, hunting, and wildlife viewing away from home were approximately $950 million, $170 million, and $3.5 billion, respectively. In addition, data from California State Parks (2003) indicate that in 2002, 68% of adult residents participated in trail hiking for an average of 24.1 days per year. Applying these same rates to Census estimates of the California adult population in 2007 suggests that there were roughly 453 million days of hiking by residents in California in 2007. According to Kaval and Loomis (2003), the average value of a hiking day in the Pacific Coast region is $25, based on a sample of 49 studies. Multiplying this average day value by the total participation estimate indicates that the aggregate annual benefit for California residents from trail hiking in 2007 was nearly $12 billion.

Beyond the recreational value, native landscapes provide aesthetic services to local residents and homeowners. Aesthetic services not related to recreation include the view of the landscape from houses, as individuals commute, and as individuals go about their daily routine in a nearby community. Studies find that scenic landscapes are capitalized into the price of housing. Studies document the existence of housing price premia associated with proximity to forest and open space (REA, 2009).

Nonuse value, also called existence value or preservation value, encompasses a variety of motivations that lead individuals to place value on environmental goods or services that they do not use. The values individuals place on protecting rare species, rare habitats, or landscape types that they do not see or visit and that do not contribute to the pleasure they get from other activities are examples of nonuse values. While measuring the public's willingness to pay to protect endangered species poses theoretical and technical challenges, it is clear that the public places a value on preserving endangered species and their habitat. Data on charitable donations, survey results, and the time and effort different individuals or organizations devote to protecting species and habitat suggest that endangered species have intrinsic value to people beyond the value derived from using the resource (recreational viewing or aesthetic value) (REA, 2009).

Excessive N deposition upsets the balance between native and nonnative plants, changing the ability of an area to support biodiversity. A change in the composition of species changes fire frequency and intensity, as nonnative grasses fuel more frequent and more intense wildfires. More frequent and intense fires also reduce the ability of native plants to regenerate after a fire and increase the proportion of nonnative grasses (EPA, 2008). Excess N deposition leads to changes in forest structure, such as increased density and loss of root biomass, which in turn can result in more intense fires and water quality problems related to nitrate leaching (EPA, 2008). The terrestrial enrichment case study identified fire regulation as a service that could be impacted by enrichment of terrestrial ecosystems. Wildfires represent a serious threat and cause billions of dollars in damage. Benefits include the value of avoided residential property damages, avoided damages to timber, rangeland, and wildlife resources, avoided losses from fire-related air quality impairments, avoided deaths and injury due to fire, improved outdoor recreation opportunities, and savings in costs associated with fighting the fires and protecting lives and property. Maintaining water quality emerged as a regulating service that can be upset by excessive N. When the soil becomes saturated, nitrates may leach into the surface water and cause acidification.

4.4.2 Visibility Improvements

Reductions in NO_2 emissions and secondary formation of $PM_{2.5}$ due to the alternative standards would improve the level of visibility throughout the United States because these suspended particles and gases degrade visibility by scattering and absorbing light (U.S. EPA, 2009c). Visibility directly affects people's enjoyment of a variety of daily activities. Individuals value visibility both in the places they live and work, in the places they travel to for recreational purposes, and at sites of unique public value, such as the Great Smokey Mountains National Park. Without the necessary air quality data, we were unable to calculate the predicted change in visibility due to control strategy to attain various alternate standard levels. However, in this section, we describe the process by which NO_2 emissions impair visibility and how this impairment affects the public.

Visual air quality (VAQ) is commonly measured as either light extinction, which is defined as the loss of light per unit of distance in terms of inverse megameters (Mm^{-1}) or the deciview (dv) metric (Pitchford and Malm, 1993), which is a logarithmic function of extinction. Extinction and deciviews are physical measures of the amount of visibility impairment (e.g., the amount of "haze"), with both extinction and deciview increasing as the amount of haze increases. Light extinction is the optical characteristic of the atmosphere that occurs when light is either scattered or absorbed, which converts the light to heat. Particulate matter and gases can both scatter and absorb light. Fine particles with significant light-extinction efficiencies include sulfates, nitrates, organic carbon, elemental carbon, and soil (Sisler, 1996). The extent to which any amount of light extinction affects a person's ability to view a scene depends on both scene and light characteristics. For example, the appearance of a nearby object (i.e. a building) is generally less sensitive to a change in light extinction than the appearance of a similar object at a greater distance.

Annual average visibility conditions (reflecting light extinction due to both anthropogenic and non-anthropogenic sources) vary regionally across the U.S. (U.S. EPA, 2009c). The rural East generally has higher levels of impairment than remote sites in the West, with the exception of urban-influenced sites such as San Gorgonio Wilderness (CA) and Point Reyes National Seashore (CA), which have annual average levels comparable to certain sites in the Northeast (U.S. EPA, 2004). Higher visibility impairment levels in the East are due to generally higher concentrations of fine particles, particularly sulfates, and higher average relative humidity levels. While visibility trends have improved in most Class I areas, the recent data show that these areas continue to suffer from visibility impairment. In eastern parks, average visual range has decreased from 90 miles to 15-25 miles, and in the West, visual range has decreased from 140 miles to 35-90 miles (U.S. EPA, 2004; U.S. EPA, 1999).

Visibility has direct significance to people's enjoyment of daily activities and their overall sense of wellbeing (U.S. EPA, 2009c). Good visibility increases the quality of life where individuals live and work, and where they engage in recreational activities. When the necessary AQ data is available, EPA generally considers benefits from these two categories of visibility changes: residential visibility (i.e., the visibility in and around the locations where people live) and recreational visibility (i.e., visibility at Class I national parks and wilderness areas.) In both cases, economic benefits are believed to consist of use values and nonuse values. Use values include the aesthetic benefits of better visibility, improved road and air safety, and enhanced recreation in activities like hunting and bird watching. Nonuse values are based on people's beliefs that the environment ought to exist free of human-induced haze. Nonuse values may be more important for recreational areas, particularly national parks and monuments. In addition, evidence suggests that an individual's WTP for improvements in visibility at a Class I area is influenced by whether it is in the region in which the individual lives, or whether it is somewhere else (Chestnut and Rowe, 1990). In general, people appear to be willing to pay more for visibility improvements at parks and wilderness areas that are "in-region" than at those that are "out-of-region." This is plausible, because people are more likely to visit, be familiar with, and care about parks and wilderness areas in their own part of the country. EPA generally uses a contingent valuation study as the basis for monetary estimates of the benefits of visibility changes in recreational areas (Chestnut and Rowe, 1990). To estimate the monetized value of visibility changes, an analyst would multiply the willingness-to-pay estimates by the amount of visibility impairment, but this information in unavailable for this analysis.

4.5 Limitations and Uncertainties

The National Research Council (NRC) (2002) highlighted the need for EPA to conduct rigorous quantitative analysis of uncertainty in its benefits estimates and to present these estimates to decision makers in ways that foster an appropriate appreciation of their inherent uncertainty. In response to these comments, EPA's Office of Air and Radiation (OAR) is developing a comprehensive strategy for characterizing the aggregate impact of uncertainty in key modeling elements on both health incidence and benefits estimates. Components of that strategy include emissions modeling, air quality modeling, health effects incidence estimation, and valuation.

In this analysis, we use two methods to assess uncertainty quantitatively: sensitivity analysis, and alternate concentration-response functions for PM mortality. We also provide a qualitative assessment for those aspects that we are unable to address quantitatively in this analysis. Each of these analyses is described in detail in the following sections.

This analysis includes many data sources as inputs, including emission inventories, air quality data from models (with their associated parameters and inputs), population data, health effect estimates from epidemiology studies, and economic data for monetizing benefits. Each of these inputs may be uncertain and would affect the benefits estimate. When the uncertainties from each stage of the analysis are compounded, small uncertainties can have large effects on the total quantified benefits. In this analysis, we are unable to quantify the cumulative effect of all of these uncertainties, but we provide the following analyses to characterize many of the largest sources of uncertainty.

4.5.1 Sensitivity analyses

We performed a couple of sensitivity analyses on the benefits results to assess the sensitivity of the primary results to various data inputs and assumptions. We then changed each default input one at a time and recalculated the total monetized benefits to assess the percent change from the default. The results of this sensitivity analysis are available in Table 4-7.

4.5.2 Alternate concentration-response functions for PM mortality

$PM_{2.5}$ mortality co-benefits are the largest benefit category that we monetized in this analysis. To better understand the concentration-response relationship between $PM_{2.5}$ exposure and premature mortality, EPA conducted an expert elicitation in 2006 (Roman et al., 2008; IEc, 2006). In general, the results of the expert elicitation support the conclusion that the benefits of $PM_{2.5}$ control are very likely to be substantial. In previous RIAs, EPA presented benefits estimates using concentration response functions derived from the $PM_{2.5}$ Expert Elicitation as a range from the lowest expert value (Expert K) to the highest expert value (Expert E). However, this approach did not indicate the agency's judgment on what the best estimate of PM benefits may be, and EPA's Science Advisory Board described this presentation as misleading. Therefore, we began to present the cohort-based studies (Pope et al, 2002; and Laden et al., 2006) as our core estimates in the Portland Cement RIA (U.S. EPA, 2009a). Using alternate relationships between $PM_{2.5}$ and premature mortality supplied by experts, higher and lower benefits estimates are plausible, but most of the expert-based estimates fall between the two epidemiology-based estimates (Roman et al., 2008).

In this analysis, we present the results derived from the expert elicitation as indicative of the uncertainty associated with a major component of the health impact functions, and we provide the independent estimates derived from each of the twelve experts to better characterize the degree of variability in the expert responses. In this chapter, we provide the results using the concentration-response functions derived from the expert elicitation in both tabular (Table 4-5) and graphical form (Figure 4-3). Please note that these results 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. Because in this RIA we estimate benefits using benefit-per-ton estimates, technical limitations prevent us from providing the associated credible intervals with the expert functions.

4.5.3 Qualitative assessment of uncertainty and other analysis limitations

Although we strive to incorporate as many quantitative assessments of uncertainty, there are several aspects for which we are only able to address qualitatively. These aspects are important factors to consider when evaluating the relative benefits of the attainment strategies for each of the alternative standards:

1. Because a near-roadway monitoring network does not yet exist, this analysis represents a rough estimate with several simplifying assumptions. This analysis does not take into account a large variety of localized conditions specific to individual monitors; instead, the analysis attempts to account for some local parameters by adjusting future design values based on average localized impacts near roads from on-road emissions. This analysis assumes area-wide monitors remain in the same location; however concentrations are adjusted to reflect near-roadway conditions. This analysis cannot

predict air quality in locations for which there is no current NO_2 monitor, or where current monitoring data is incomplete

2. There are many uncertainties associated with the health impact functions used in this analysis. These include: within study variability (the precision with which a given study estimates the relationship between air quality changes and health effects); across study variation (different published studies of the same pollutant/health effect relationship typically do not report identical findings and in some instances the differences are substantial); the application of C-R functions nationwide (does not account for any relationship between region and health effect, to the extent that such a relationship exists); extrapolation of impact functions across population (we assumed that certain health impact functions applied to age ranges broader than that considered in the original epidemiological study); and various uncertainties in the C-R function, including causality and thresholds. These uncertainties may under- or over-estimate benefits.
3. This analysis is for the year 2020, and projecting key variables introduces uncertainty. Inherent in any analysis of future regulatory programs are uncertainties in projecting atmospheric conditions and source level emissions, as well as population, health baselines, incomes, technology, and other factors.
4. This analysis omits certain unquantified effects due to lack of data, time and resources. These unquantified endpoints include other health effects, ecosystem effects, and visibility. EPA will continue to evaluate new methods and models and select those most appropriate for estimating the benefits of reductions in air pollution. Enhanced collaboration between air quality modelers, epidemiologists, toxicologists, ecologists, and economists should result in a more tightly integrated analytical framework for measuring benefits of air pollution policies.
5. $PM_{2.5}$ co-benefits represent the total monetized benefits for this analysis, and these estimates are subject to a number of assumptions and uncertainties.
    a. $PM_{2.5}$ co-benefits were derived through benefit per-ton estimates, which do not reflect local variability in population density, meteorology, exposure, baseline health incidence rates, or other local factors that might lead to an over-estimate or under-estimate of the actual benefits of controlling directly emitted fine particulates.
    b. We assume that all fine particles, regardless of their chemical composition, are equally potent in causing premature mortality. This is an important assumption, because $PM_{2.5}$ produced via transported precursors emitted from EGUs may differ significantly from direct $PM_{2.5}$ released from diesel engines and other industrial sources, but no clear scientific grounds exist for supporting differential effects estimates by particle type.

c. We assume that the health impact function for fine particles is linear down to the lowest air quality levels modeled in this analysis. Thus, the estimates include health benefits from reducing fine particles in areas with varied concentrations of $PM_{2.5}$, including both regions that are in attainment with fine particle standard and those that do not meet the standard down to the lowest modeled concentrations.
d. To characterize the uncertainty in the relationship between $PM_{2.5}$ and premature mortality (which typically accounts for 85% to 95% of total monetized PM benefits), we include a set of twelve estimates based on results of the expert elicitation study in addition to our core estimates. Even these multiple characterizations, omit the uncertainty in air quality estimates, baseline incidence rates, populations exposed and transferability of the effect estimate to diverse locations. As a result, the reported confidence intervals and range of estimates give an incomplete picture about the overall uncertainty in the $PM_{2.5}$ estimates. This information should be interpreted within the context of the larger uncertainty surrounding the entire analysis. For more information on the uncertainties associated with $PM_{2.5}$ co-benefits, please consult the $PM_{2.5}$ NAAQS RIA (Table 5.5).

4.6 Discussion

For the selected standard of 100 ppb, there would be zero costs and benefits as we project all areas to attain this standard without additional controls. However, we present the results for other more stringent standards that would produce substantial health co-benefits from reducing $PM_{2.5}$ exposure from avoided premature mortality and other morbidity effects.

There are several health benefits categories that we were unable to quantify due to data limitations. Several of these unquantified benefits in this analysis could be substantial, including the health benefits of reduced NO_2 exposure, health benefits of reduced ozone exposure, benefits from improved visibility, and the ecosystem benefits of reduced nitrogen deposition. Because we were unable to estimate NO_2 exposure in order to calculate NO_2 health benefits, this analysis only quantifies and monetizes only the PM component of the total health benefits associated with reducing NO_2 emissions. Despite omitting this important benefits category, we believe that the $PM_{2.5}$ co-benefits capture the majority of the monetized health benefits. The area-wide analysis for 50 ppb in the proposal RIA (U.S. EPA, 2009b) showed that the monetized NO_2 benefits only accounted for 2% of the total monetized benefits, with $PM_{2.5}$ co-benefits accounting for the remainder.

Because NO_X is also a precursor to ozone, reductions in NO_X would also reduce ozone formation and the effects associated with ozone exposure. Unfortunately, we did not have the air quality data available for this analysis to estimate the health effects of reduced ozone exposure as a result of the NO_X emission reductions. As the RIA for the Ozone NAAQS (U.S. EPA, 2008a) demonstrated, the monetized health benefits of reducing ozone exposure can be substantial, up to 40% as much as the $PM_{2.5}$ co-benefits. In addition, there are substantial benefits that would occur from reducing ozone exposure on vegetation (U.S. EPA, 2007). Despite ozone disbenefits that might occur downwind in certain areas of the country due to reductions in NO_2 emissions, the net effect of NO_2 reductions is generally an overall decrease in ozone exposure.

We were unable to estimate the benefits from several welfare benefit categories, including improvements in visibility from reducing light-scattering particles because we lacked the necessary air quality data. Visibility directly affects people's enjoyment of a variety of daily activities. Individuals value visibility both in the places they live and work, in the places they travel to for recreational purposes, and at sites of unique public value, such as the Great Smokey Mountains National Park. Previous RIAs for ozone (U.S. EPA, 2008a) and $PM_{2.5}$ (U.S. EPA, 2006c) indicate that visibility is an important benefit category, and previous efforts to monetize those benefits have only included a subset of visibility benefits, excluding benefits in urban areas and many national and state parks. Even this subset accounted for up to 5% of total monetized benefits in the Ozone NAAQS RIA.

We were also unable to estimate the ecosystem benefits of reduced nitrogen deposition because we lacked the necessary air quality data and we are still developing the methodology to estimate ecosystem benefits. Previous assessments (U.S. EPA, 1999; U.S. EPA, 2005; U.S. EPA, 2008f; U.S. EPA, 2009d) indicate that ecosystem benefits are also an important benefits category, but those efforts were only able to monetize a tiny subset of ecosystem benefits in specific geographic locations, such as recreational fishing effects from lake acidification in the Adirondacks. Although there is some evidence that nitrogen deposition may have positive effects on agricultural output through passive fertilization, it is likely that the overall value is very small relative to other health and welfare effects.

In section 4.2 of this RIA, we discuss the revised presentation using benefits based on Pope et al. and Laden et al. as anchor points instead of the low and high end of the expert elicitation. This change was incorporated in direct response to recommendations from EPA's Science Advisory Board (U.S.EPA-SAB, 2008). Although using benefit-per-ton estimates limited our ability to incorporate all of their suggestions fully, we have incorporated the following recommendations into this analysis:

- Added “bottom line” statements where appropriate
- Clarified that the benefits results shown are not the actual judgments of the experts
- Acknowledged uncertainties exist at each stage of the analytic process, although difficult to quantify when using benefit-per-ton estimates
- Did not use the expert elicitation range to characterize the uncertainty as it focuses on the most extreme judgments with zero weight to all the others,
- Described the rationale for using expert elicitation in the context of the regulatory process (to characterize uncertainty)
- Identified results based on epidemiology studies and expert elicitation separately
- Showed central mass of expert opinion using graphs
- Presented the quantitative results using diverse tables and more graphics

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Appendix 4a - NO_2 Benefits Methodology

4a.1 Introduction

This appendix documents the methodology for estimating and monetizing the health benefits expected from reducing exposure to NO_2. In addition, this appendix includes a brief discussion regarding the key findings from the NO_2 benefits analysis as well as the limitations and areas of uncertainty in our approach. Although this approach was incorporated into the NO_2 NAAQS proposal RIA for the area-wide analysis (U.S. EPA, 2009), this approach was not deemed appropriate for estimating NO_2 exposure at near roadway monitors that do not yet exist. [1] Therefore, this appendix documents a methodological approach for estimating direct NO_2 benefits, and we do not include these results in the NO_2 NAAQS final RIA.

4a.2 Primary Benefits Approach

This section presents our approach for estimating avoided adverse health effects due to NO_2 exposure in humans resulting from achieving alternative scenarios, relative to a baseline concentration of ambient NO_2. First, we summarize the scientific evidence concerning potential health effects of NO_2 exposure, and then we present the health endpoints we selected for our primary benefits estimate. Next, we describe our benefits model, including the key input data and assumptions. Finally, we describe our approach for assigning an economic value to the NO_2 health benefits.

Benefits Scenario

We estimated the economic benefits from annual avoided health effects expected to result from achieving alternative scenarios (the “control scenarios”). We estimated benefits in the control scenarios relative to the incidence of health effects consistent with the ambient NO_2 concentration expected (the “baseline”). Note that this “baseline” reflects emissions reductions and ambient air quality improvements that we anticipate will result from implementation of other air quality rules, including compliance with all relevant rules already promulgated

We compared benefits across three alternative scenarios. Consistent with EPA’s approach for RIA benefits assessments, we estimated the health effects associated with an

[1] $PM_{2.5}$ co-benefits of reducing NO_2 emissions to meet alternate standard levels are quantified and monetized in Chapter 4 of this RIA.

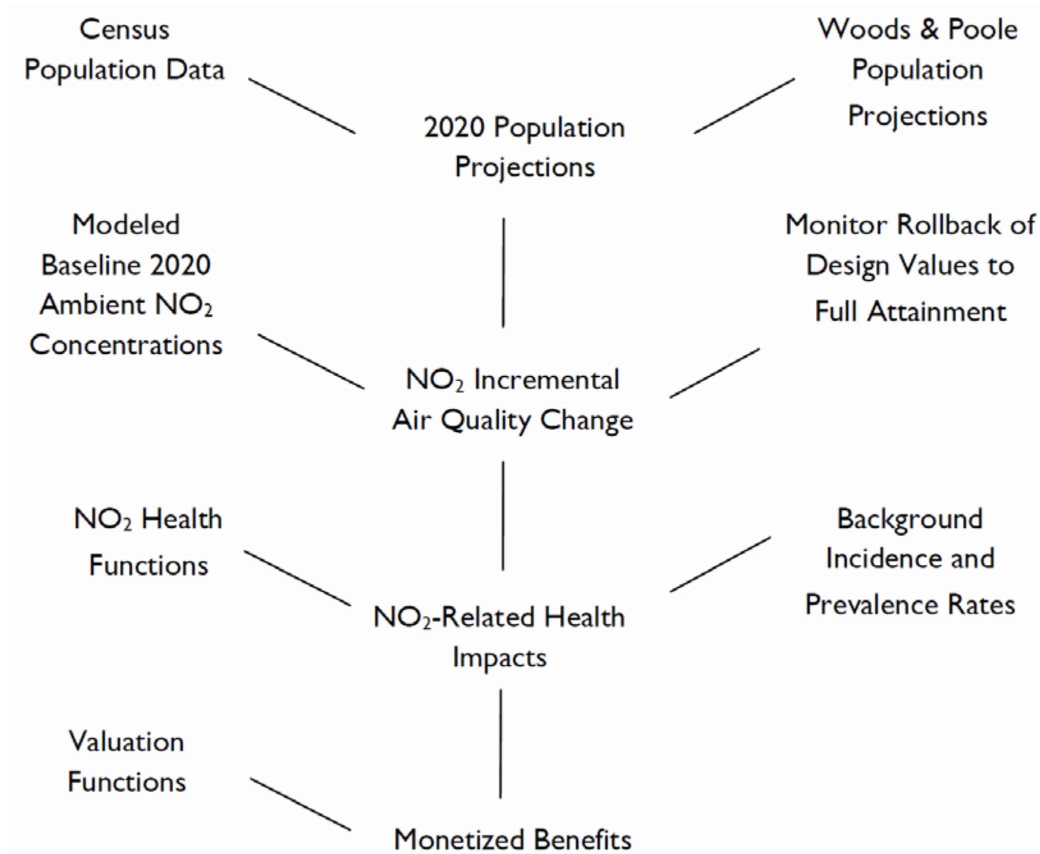
incremental difference in ambient concentrations between a baseline scenario and a pollution control strategy.

4a.3 Overview of analytical framework for benefits analysis

Benefits Model

For the primary benefits analysis, we use the Environmental Benefits Mapping and Analysis Program (BenMAP) to estimate the health benefits occurring as a result of implementing alternative NO_2 NAAQS levels. Although BenMAP has been used extensively in previous RIAs to estimate the health benefits of reducing exposure to $PM_{2.5}$ and ozone, this is the first RIA to use BenMAP to estimate the health benefits of reducing exposure to NO_2. Figure 4a-1 shows the major components of and inputs to the BenMAP model.

Figure 4a-1: Diagram of Inputs to BenMAP model for NO_2 Analysis

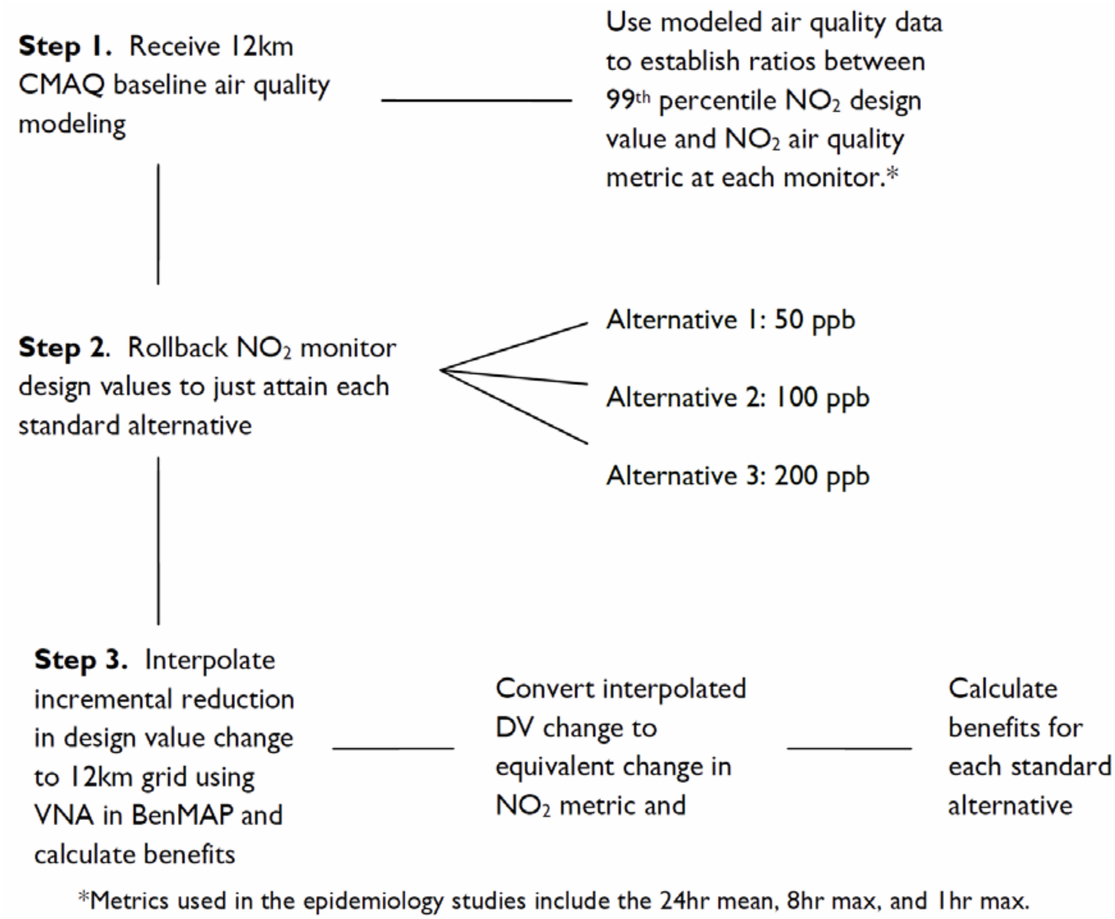


Air Quality Estimates

As shown in Figure 4a-1, the primary input to any benefits assessment is the estimated changes in ambient air quality expected to result from a simulated control strategy or attainment of a particular standard.

The CMAQ air quality model provides projects both design values at NO_2 monitors and air quality concentrations at 12km grid cells. To estimate the benefits of fully attaining the standards in all areas, EPA employed the “monitor rollback” approach to approximate the air quality change resulting from just attaining alternative scenarios at each design value monitor. Figure 4a-2 depicts the steps in the rollback process. The approach described here aims to estimate the change in population exposure associated with attaining an alternate NAAQS. This approach relies on data from the existing NO_2 monitoring network and the inverse distance squared variant of the Veronoi Neighborhood Averaging (VNA) interpolation method to adjust the CMAQ-modeled NO_2 concentrations such that each area just attains each alternative scenario. We believe that the interpolation method using inverse distance squared most appropriately reflects the steep exposure gradient for NO_2 around each monitor (see: EPA, 2008b). A sensitivity analysis for the NO_2 NAAQS proposal RIA (U.S. EPA, 2009) showed that the results are not very sensitive to the interpolation method.

Figure 4a-2: Diagram of Rollback Method



Because the VNA rollback approach interpolates monitor values, it is most reliable in areas with a denser monitoring network. In areas with a sparser monitoring network, there is less observed monitoring data to support the VNA interpolation and we have less confidence in the predicted air quality values further away from the monitors. For this reason, we interpolated air quality values—and estimated health impacts—within the CMAQ grid cells that are located within 30 km of the monitor, assuming that emission changes within this radius would affect the NO_2 concentration at each monitor. Limiting the interpolation to this radius attempts to account for the limitations of the VNA approach and ensures that the benefits and costs analyses consider a consistent geographic area.[2] Therefore, the primary benefits analysis assesses health impacts occurring to populations living in the CMAQ grid cells located within the 30km buffer for the specific geographic areas assumed to not attain the alternate standard levels.

[2] Please see Chapter 3 for more information regarding the technical basis for the 30 km assumption.

4a.4 Estimating Avoided Health Effects from NO_2 Exposure

Selection of Health Endpoints for NO_2

Epidemiological researchers have associated NO_2 exposure with adverse health effects in numerous toxicological, clinical and epidemiological studies, as described in the Integrated Science Assessment for Oxides of Nitrogen - Health Criteria (Final Report) (U.S. EPA, 2008a; hereafter, "NO_2 ISA"). The NO_2 ISA provides a comprehensive review of the current evidence of health and environmental effects of NO_2. The Risk and Exposure Assessment for NO_2 summarizes the NO_2 ISA conclusions regarding health effects from NO_2 exposure as follows (U.S. EPA, 2008b; Section 4.2.1):

> "The ISA concludes that, taken together, recent studies provide scientific evidence that is sufficient to infer a likely causal relationship between short-term NO_2 exposure and adverse effects on the respiratory system (ISA, section 5.3.2.1). This finding is supported by the large body of recent epidemiologic evidence as well as findings from human and animal experimental studies. These epidemiologic and experimental studies encompass a number of endpoints including [Emergency Department (ED)] visits and hospitalizations, respiratory symptoms, airway hyperresponsiveness, airway inflammation, and lung function. Effect estimates from epidemiologic studies conducted in the United States and Canada generally indicate a 2-20% increase in risks for ED visits and hospital admissions and higher risks for respiratory symptoms (ISA, section 5.4)."

Previous reviews of the NO_2 primary NAAQS, completed in 1985 and 1996, did not include a quantitative benefits assessment for NO_2 exposure. As the first health benefits assessment for NO_2 exposure, we build on the methodology and lessons learned from the NO_2 risk and exposure assessment (U.S. EPA, 2008b) and the benefits assessments for the recent $PM_{2.5}$ and O_3 NAAQS (U.S. EPA, 2006a; U.S. EPA, 2008a).

We selected the health endpoints to be consistent with the conclusions of the NO_2 ISA. In general, we follow a weight of evidence approach, based on the biological plausibility of effects, availability of concentration-response functions from well conducted peer-reviewed epidemiological studies, cohesiveness of results across studies, and a focus on endpoints reflecting public health impacts (like hospital admissions) rather than physiological responses (such as changes in clinical measures like Forced Expiratory Volume (FEV1)). The differing evidence and associated strength of the evidence for these different effects is described in detail in the NO_2 ISA.

Although a number of adverse health effects have been found to be associated with NO_2 exposure, this benefits analysis only includes a subset due to limitations in understanding and quantifying the dose-response relationship for some of these health endpoints. In this analysis, we only estimated the benefits for those endpoints with sufficient evidence to support a quantified concentration-response relationship using the information presented in the NO_2 ISA, which contains an extensive literature review for several health endpoints related to NO_2 exposure. Because the ISA only included studies published or accepted for publication through December 2007, we also performed supplemental literature searches in the online search engine PubMed® to identify relevant studies published between January 2008, and the present.[3] Based on our review of this information, we quantified four short-term morbidity endpoints that the NO_2 ISA identified as "sufficient to infer a likely causal relationship": asthma exacerbation, respiratory-related emergency department visits, and respiratory-related hospitalizations.

Table 4a-1 presents the health effects related to NO_2 exposure quantified in this benefits analysis. In addition, the table includes other endpoints potentially linked to NO_2 exposure, but which we are not yet ready to quantify with concentration-response functions.

The NO_2 ISA concluded that the relationship between short-term NO_2 exposure and premature mortality was "suggestive but not sufficient to infer a causal relationship" because it is difficult to attribute the mortality risk effects to NO_2 alone. Therefore, we decided not to quantify premature mortality from NO_2 exposure in this analysis despite evidence suggesting a positive association (U.S. EPA, 2008a, Section 3.3.2). Although the NO_2 ISA stated that studies consistently reported a relationship between NO_2 exposure and mortality, the effect was generally smaller than that for other pollutants such as PM. We may revisit this decision in future benefits assessment for NO_2.

[3] The O'Conner et al. study (2008) is the only study included in this analysis that was published after the cut-off date for inclusion in the NO_2 ISA.

Table 4a-1: Human Health and Welfare Effects of NO_2

Pollutant / Effect	Quantified and Monetized in Primary Estimates [a]	Unquantified Effects [b, c] Changes in:
NO_2/Health	Asthma Hospital Admissions Chronic Lung Disease Hospital Admissions Asthma ER visits Asthma exacerbation Acute Respiratory symptoms	Premature mortality Pulmonary function Other respiratory emergency department visits Other respiratory hospital admissions
NO_2/Welfare		Visibility Commercial fishing and forestry from acidic deposition Recreation in terrestrial and aquatic ecosystems from acid deposition Commercial fishing, agriculture, and forestry from nutrient deposition Recreation in terrestrial and estuarine ecosystems from nutrient deposition Other ecosystem services and existence values for currently healthy ecosystems

[a] Primary quantified and monetized effects are those included when determining the primary estimate of total monetized benefits of the alternative standards.
[b] The categorization of unquantified toxic health and welfare effects is not exhaustive.
[c] Health endpoints in the unquantified benefits column include both a) those for which there is not consensus on causality and those for which causality has been determined but empirical data are not available to allow calculation of benefits.

Selection of Concentration-Response Functions

After identifying the health endpoints to quantify in this analysis, we then selected concentration-response functions drawn from the epidemiological literature identified in the NO_2 ISA. We considered several factors in selecting the appropriate epidemiological studies and concentration-response functions for this benefits assessment.

- First, we considered ambient NO_2 studies that were identified as key studies in the NO_2 ISA (or a more recent study), excluding those affected by the general additive model (GAM) S-Plus issue.[4]
- Second, we judged that studies conducted in the United States are preferable to those conducted outside the United States, given the potential for effect estimates to be affected by factors such as the ambient pollutant mix, the

[4] The S-Plus statistical software is widely used for nonlinear regression analysis in time-series research of health effects. However, in 2002, a problem was discovered with the software's default conversion criteria in the general additive model (GAM), which resulted in biased relative risk estimates in many studies. This analysis does not include any studies that encountered this problem. For more information on this issue, please see U.S. EPA (2002).

placement of monitors, activity patterns of the population, and characteristics of the healthcare system especially for hospital admissions and emergency department visits. We include Canadian studies in sensitivity analyses, when available.

- Third, we only incorporated concentration-response functions for which there was a corresponding valuation function. Currently, we only have a valuation function for asthma-related emergency department visits, but we do not have a valuation function for all-respiratory-related emergency department visits.
- Fourth, we preferred concentration-response functions that correspond to the age ranges most relevant to the specific health endpoint, with non-overlapping ICD-9 codes. We preferred completeness when selecting functions that correspond to particular age ranges and ICD codes. Age ranges and ICD codes associated with the selected functions are identified in Table 4a.2.
- Fifth, we preferred multi-city studies or combined multiple single city studies, when available.
- Sixth, when available, we judged that effect estimates with distributed or cumulative lag structures were most appropriate for this analysis.
- Seventh, when available, we selected NO_2 concentration-response functions based on multi-pollutant models. Studies with multi-pollutant models are identified in Table 4a.2.

These criteria reflect our preferences for study selection, and it was possible to satisfy many of these, but not all. There are trade-offs inherent in selecting among a range of studies, as not all studies met all criteria outlined above. At minimum, we ensured that none of the studies were GAM affected, we selected only U.S. based studies, and we quantified health endpoints for which there was a corresponding valuation function.

We believe that U.S.-based studies are most appropriate studies to use in this analysis to estimate the number of hospital admissions associated with NO_2 exposure because of the characteristics of the ambient air, population, and healthcare system. Using only U.S.-based studies, we are limited to estimating the hospital admissions for asthma (ICD-9 493) and chronic lung disease (ICD-9 490-496) rather than all respiratory-related hospital admission, which is a more complete measure of health impacts. However, there are several Canada-based epidemiology studies that provide a more complete estimate of respiratory hospital admissions (Fung, 2006; Luginaah, 2005; Yang, 2003). Compared to the U.S. based studies, the Canadian studies produce a larger estimate of hospital admissions associated with NO_2 exposure.

When selecting concentration-response functions to use in this analysis, we reviewed the scientific evidence regarding the presence of thresholds in the concentration-response functions for NO_2-related health effects to determine whether the function is approximately linear across the relevant concentration range. The NO_2 ISA concluded that, "[t]hese results do not provide adequate evidence to suggest that nonlinear departures exist along any part of this range of NO_2 exposure concentrations." Therefore, we have not incorporated thresholds in the concentration-response function for NO_2-related health effects in this analysis.

Table 4a-2 shows the studies and health endpoints that we selected for this analysis. Table 4a-3 shows the baseline health data used in combination with these health functions. Following these tables is a description of each of the epidemiology studies used in this analysis.

Table 4a-2: NO_2-Related Health Endpoints Quantified, Studies Used to Develop Health Impact Functions and Sub-Populations to which They Apply

Endpoint	Study	Study Population
Hospital Admissions [b]		
Asthma	Linn et al. (2000)—ICD-9 493	All ages
Chronic Lung Disease	Moolgavkar (2003) —ICD-9 490-496	> 65
Emergency Department Visits		
Asthma	Pooled Estimate: Ito et al. (2007)—ICD-9 493 NYDOH (2006)[c]—ICD-9 493 Peel et al. (2005)—ICD-9 493	All ages
Other Health Endpoints		
Asthma exacerbations	Pooled estimate: O'Connor et al. (2008) (slow play, missed school days, nighttime asthma) [c] Ostro et al. (2001) (cough, cough (new cases), shortness of breath, shortness of breath (new cases), wheeze, wheeze (new cases) [a] Schildcrout et al. (2006) (one or more symptoms)	4 - 12
	Delfino et al. (2002) (one or more symptoms)	13 - 18[a]
Acute Respiratory Symptoms	Schwartz et al. (1994)[c]	7 - 14

[a] The original study populations were 9 to 18 for the Delfino et al. (2002) study, and 8-13 for the Ostro et al. (2001) study. We extended the applied population to facilitate the pooling process, recognizing the common biological basis for the effect in children in the broader age group. See: National Research Council (NRC). 2002. *Estimating the Public Health Benefits of Proposed Air Pollution Regulations*. Washington, DC: The National Academies Press, pg 117.

[b] We recognize that the ICD codes for asthma and chronic lung disease overlap partially, suggesting that our combined estimate of respiratory hospital admissions may be overstated to a small degree. However, we believe that using the other available health impact functions to quantify this endpoint would have resulted in a more biased and uncertain estimate, as these functions failed to meet key selection criteria.

[c] Study specifies a multipollutant model

Table 4a-3: National Average Baseline Incidence Rates used to Calculate NO_2-Related Health Impacts [a]

Endpoint	Source	Notes	Rate per 100 people per year by Age Group						
			<18	18–24	25–34	35–44	45–54	55–64	65+
Respiratory Hospital Admissions	1999 NHDS public use data files [b]	incidence	0.043	0.084	0.206	0.678	1.926	4.389	11.629
Asthma ER visits	2000 NHAMCS public use data files [c]; 1999 NHDS public use data files [b]	incidence	1.011	1.087	0.751	0.438	0.352	0.425	0.232
Minor Restricted Activity Days (MRADs)	Schwartz (1994, table 2)	incidence	0.416	—	—	—	—	—	—
Asthma Exacerbations	Delfino et al. (2002)	Incidence (and prevalence) among asthmatic children		Asthma symptoms			0.157 (0.0567)		
	O'Connor et al. (2008)	Incidence (and prevalence) among asthmatic children		Missed school One or more symptoms Slow play Nighttime asthma			0.057 (0.0567) 0.207 (0.0567) 0.157 (0.0567) 0.121 (0.0567)		
	Ostro et al. (2001)	Incidence (and prevalence) among asthmatic African American children		Cough Cough (new cases) Shortness of breath Shortness of breath (new cases) Wheeze Wheeze (new cases)			0.145 (0.0726) 0.067 (0.0726) 0.074 (0. 0726) 0.037 (0. 0726) 0.173 (0.0726) 0.076 (0.0726)		
	Schildcrout et al. (2006)	Incidence (and prevalence) among asthmatic children		One or more symptoms			0.52 (0.0567)		

[a] The following abbreviations are used to describe the national surveys conducted by the National Center for Health Statistics: HIS refers to the National Health Interview Survey; NHDS—National Hospital Discharge Survey; NHAMCS—National Hospital Ambulatory Medical Care Survey.
[b] See ftp://ftp.cdc.gov/pub/Health_Statistics/NCHS/Datasets/NHDS/
[c] See ftp://ftp.cdc.gov/pub/Health_Statistics/NCHS/Datasets/NHAMCS/

Linn et al. (2000)

Linn et al. (2000) evaluated associations between air pollution and hospital admissions for cardiopulmonary illnesses in metropolitan Los Angeles during 1992-1995. In a single-pollutant Poisson regression model, daily average of NO_2 (year-round) was found significantly associated with same-day asthma hospital admissions for both age groups (i.e., 0-29 and 30-99). The results for winter and autumn were also reported but insignificant.

Moolgavkar (2003)

Moolgavkar (2003) presented re-analyses of Moolgavkar(2000a; 2000b; 2000c) of the associations between air pollution and daily deaths and hospital admissions in Los Angeles and Cook counties in the United States.[5] The author also reported the results of generalized linear model (GLM) analyses using natural splines with the same degree of freedom as the smoothing splines he used in the generalized additive model (GAM) analyses. In single-pollutant Poisson regression models, hospital admissions for chronic obstructive disorder (COPD) (ICD-9 code 490-496) were associated with daily average of NO_2 levels at lags of 0, 1, 2, 3, 4 and 5 days for individuals 65 and older. The association was strongest at lag 0 using both GAM (stringent convergence) and GLM.

Ito et al. (2007)

Ito et al. (2007) assessed associations between air pollution and asthma emergency department visits in New York City for all ages. Specifically they examined the temporal relationships among air pollution and weather variables in the context of air pollution health effects models. The authors compiled daily data for $PM_{2.5,}$ O_3, NO_2, SO_2, CO, temperature, dew point, relative humidity, wind speed, and barometric pressure for New York City for the years 1999-2002.The authors evaluated the relationship between the various pollutants' risk estimates and their respective concurvities, and discuss the limitations that the results imply about the interpretability of multi-pollutant health effects models.

NYDOH (2006)

New York State Department of Health (NYDOH) investigated whether day-to-day variations in air pollution were associated with asthma emergency department (ED) visits in Manhattan and Bronx, NYC and compared the magnitude of the air pollution effect between the two communities. NYDOH (2006) used Poisson regression to test for effects of 14 key air contaminants on daily ED visits, with control for temporal cycles, temperature, and day-of-week effects. The core analysis utilized the average exposure for the zero- to four-day lags. Mean daily NO_2 was found significantly associated with asthma ED visits in Bronx but not Manhattan. Their findings of more significant air pollution effects in the Bronx are likely to relate in part to

[5] The principal reason for conducting these re-analyses was to assess the impact of using convergence criteria that are more stringent than the default criteria used in the S-Plus software package.

greater statistical power for identifying effects in the Bronx where baseline ED visits were greater, but they may also reflect greater sensitivity to air pollution effects in the Bronx.

Peel et al. (2005)

Peel et al. (2005) examined the associations between air pollution and respiratory emergency department visits (i.e., asthma (ICD-9 code 493, 786.09), COPD (491,492,496), upper respiratory infection (URI) (460-466, 477), pneumonia (480-486), and an all respiratory-disease group) in Atlanta, GA from 1 January 1993 to 31 August 2000. They used 3-Day Moving Average (Lags of 0, 1, and 2 Days) and unconstrained distributed lag (Lags of 0 to 13 Days) in the Poisson regression analyses. In single-pollutant models, the authors found that positive associations persisted beyond 3 days for several outcomes, and over a week for asthma. Standard deviation increases of O_3, NO_2, CO, and PM_{10} were associated with 1-3% increases in URI visits; a 2 μg/m^3 increase of $PM_{2.5}$ organic carbon was associated with a 3% increase in pneumonia visits; and standard deviation increases of NO_2 and CO were associated with 2-3% increases in chronic obstructive pulmonary disease visits.

Delfino et al. (2002)

Delfino et al. (2002) examined the association between air pollution and asthma symptoms among 22 asthmatic children (9-19 years of age) followed March through April 1996 (1,248 person-days) in Southern California. Air quality data for PM_{10}, NO_2, O_3, fungi and pollen were used in a logistic model with control for temperature, relative humidity, day-of-week trends and linear time trends. The odds ratio (95% confidence interval) for asthma episodes in relation to lag0 (i.e. immediate) 20 ppb changes in 8-hr max NO_2 is 1.49 (0.95-2.33). The authors also considered subgroups of asthmatic children who were on versus not on regularly scheduled anti-inflammatory medications and found that pollutant associations were stronger during respiratory infections in subjects not on anti-inflammatory medications.

O'Connor et al. (2008)

O'Connor et al. (2008) investigated the association between fluctuations in outdoor air pollution and asthma exacerbation among 861 inner-city children (5-12 years of age) with asthma in seven US urban communities. Asthma symptom data were collected every two months during the 2-year study period. Daily pollution measurements were obtained from the Aerometric Information Retrieval System between August 1998 and July 2001. The relationship of symptoms to fluctuations in pollutant concentrations was examined by using logistic models. In single-pollutant models, significant or nearly significant positive associations were observed between higher NO_2 concentrations and each of the health outcomes. Significant positive associations with symptoms but not school absence were observed in the single-pollutant model for CO. The O_3, $PM_{2.5,}$ and SO_2 concentrations did not appear significantly associated with symptoms or school absence except for a significant association between $PM_{2.5}$ and school absence. The authors concluded that the associations with NO_2 suggest that motor vehicle emissions may be causing excess morbidity in this population. This study is not included in the NO_2 ISA only because it was published after the cut-off date, but it met all of the other criteria for inclusion in this analysis.

Ostro et al. (2001)

Ostro et al. (2001) examined relations between several air pollutants and asthma exacerbation in African-Americans children (8 to 13 years old) in central Los Angeles from August to November 1993. Air quality data for PM_{10}, $PM_{2.5,}$ NO_2, and O_3 were used in a logistic regression model with control for age, income, time trends, and temperature-related weather effects. Asthma symptom endpoints were defined in two ways: "probability of a day with symptoms" and "onset of symptom episodes". New onset of a symptom episode was defined as a day without symptoms followed by a day with symptoms. The authors found cough prevalence associated with PM_{10} and $PM_{2.5}$ and cough incidence associated with $PM_{2.5,}$ PM_{10}, and NO_2. Ozone was not significantly associated with cough among asthmatics. The authors found that both the prevalent and incident episodes of shortness of breath were associated with $PM_{2.5}$ and PM_{10}. Neither ozone nor NO_2 were significantly associated with shortness of breath among asthmatics. The authors found both the prevalence and incidence of wheeze associated with $PM_{2.5,}$ PM_{10}, and NO_2. Ozone was not significantly associated with wheeze among asthmatics.

Schildcrout et al. (2006)

Schildcrout et al. (2006) investigated the relation between ambient concentrations of the five criteria pollutants (PM_{10}, O_3, NO_2, SO_2, and CO) and asthma exacerbations (daily symptoms and use of rescue inhalers) among 990 children in eight North American cities during the 22-month prerandomization phase (November 1993-September 1995) of the Childhood Asthma Management Program. Short-term effects of CO, NO_2, PM_{10}, SO_2, and warm-season O_3 were examined in both one-pollutant and two-pollutant models, using lags of up to 2 days in logistic and Poisson regressions. Lags in CO and NO_2 were positively associated with both measures of asthma exacerbation, and the 3-day moving sum of SO_2 levels was marginally related to asthma symptoms. PM_{10} and O_3 were unrelated to exacerbations. The strongest effects tended to be seen with 2-day lags, where a 1-parts-per-million change in CO and a 20-parts-per-billion change in NO_2 were associated with symptom odds ratios of 1.08 (95% confidence interval (CI): 1.02, 1.15) and 1.09 (95% CI: 1.03, 1.15), respectively.

Schwartz et al. (1994)

Schwartz et al. (1994) studied the association between ambient air pollution exposures and respiratory illness among 1,844 schoolchildren (7-14 years of age) in six U.S. cities during five warm season months between April and August. Daily measurements of ambient SO_2, NO_2, O_3, PM_{10}, $PM_{2.5}$, light scattering, and sulfate particles were made, along with integrated 24-h measures of aerosol strong acidity. Significant associations in single pollutant models were found between SO_2, NO_2, or $PM_{2.5}$ and incidence of cough, and between sulfur dioxide and incidence of lower respiratory symptoms. Significant associations were also found between incidence of coughing symptoms and incidence of lower respiratory symptoms and PM_{10}, and a marginally significant association between upper respiratory symptoms and PM_{10}.

Pooling Multiple Health Studies

After selecting which health endpoints to analyze and which epidemiology studies provide appropriate effect estimates, we then selected a method to combine the multiple health studies to provide a single benefits estimate for each health endpoint. The purpose of pooling multiple studies together is to generate a more robust estimate by combining the evidence across multiple studies and cities. Because we used a single study for acute respiratory symptoms and a single study for hospital admission for asthma, there was no pooling necessary for those endpoints.

For the hospital admission studies for chronic lung disease, we pooled the effect estimates reported for two counties (Los Angeles, CA, and Cook, IL) from Moolgavkar (2003) using random/fixed effects. [6] For the emergency department visit studies, we pooled the three studies (Ito et al., 2007; NYDOH, 2003; Peel et al., 2005) using random/fixed effects. For the asthma studies, we pooled the three studies (O'Conner et al, 2008; Ostro et al, 2001; Schildcrout et al, 2006) using random/fixed effects for ages 4 to 12, and then we summed this results with the Delfino study (2002) for ages 13 to 18. Because asthma represents the largest benefits category in this analysis, we tested the sensitivity of the NO_2 benefits to alternate pooling choices. In general, the estimate using the Ostro study is much lower than the estimate that combines Ostro with the new studies, and the estimate for one-or-more asthma symptoms is much higher than the estimate that combines all of the asthma endpoints.

[6] Random/fixed effects pooling allows for the possibility that the effect estimates reported among different studies may in fact be estimates of different parameters, rather than just different estimates of the same underlying parameter. For additional information regarding BenMAP pooling techniques, please consult the BenMAP technical appendices available at http://www.epa.gov/air/benmap/models/BenMAPappendicesSept08.pdf .

4a.5 Valuation of Avoided Health Effects from NO_2 Exposure

The selection of valuation functions is largely consistent with the $PM_{2.5}$ NAAQS (U.S. EPA, 2006a) with two exceptions. First, in this analysis, we only estimate chronic lung disease and asthma, two types of hospital admissions, whereas the $PM_{2.5}$ NAAQS estimated changes in all respiratory hospital admissions, which generated a larger monetized value. Second, we use the any-of-19 symptoms valuation for acute respiratory symptoms instead of the "minor-restricted activity day" (MRADs) estimated for the $PM_{2.5}$ NAAQS. The valuation for any-of-19-symptoms is approximately 50% of the valuation for MRADs. Consistent with economic theory, these valuation functions include adjustments for inflation (2006$) and income growth over time (2020 income levels). Table 4a-4 describes the valuation functions used to monetize the benefits of reduced exposure to NO_2.

Table 4a-4: Central Unit Values NO_2 Health Endpoints (2006$)*

Health Endpoint	Central Unit Value Per Statistical Incidence (2020 income level)	Derivation of Distributions of Estimates
Hospital Admissions and ER Visits		
Asthma Admissions	$10,000	No distributional information available. The cost-of-illness (COI) estimates (lost earnings plus direct medical costs) are based on ICD-9 code-level information (e.g., average hospital care costs, average length of hospital stay, and weighted share of total asthma category illnesses) reported in Agency for Healthcare Research and Quality, 2000 (www.ahrq.gov).
Chronic Lung Disease Admissions	$16,000	No distributions available. The COI point estimates (lost earnings plus direct medical costs) are based on ICD-9 code level information (e.g., average hospital care costs, average length of hospital stay, and weighted share of total COPD category illnesses) reported in Agency for Healthcare Research and Quality, 2000 (www.ahrq.gov).
Asthma Emergency Room Visits	$370	No distributional information available. Simple average of two unit COI values: (1) $400 (2006$), from Smith et al. (1997) and (2) $340 (2006$), from Stanford et al. (1999).
Respiratory Ailments Not Requiring Hospitalization		
Asthma Exacerbation	$53	Asthma exacerbations are valued at $49 (2006$) per incidence, based on the mean of average WTP estimates for the four severity definitions of a "bad asthma day," described in Rowe and Chestnut (1986). This study surveyed asthmatics to estimate WTP for avoidance of a "bad asthma day," as defined by the subjects. For purposes of valuation, an asthma exacerbation is assumed to be equivalent to a day in which asthma is moderate or worse as reported in the Rowe and Chestnut (1986) study. The value is assumed have a uniform distribution between $19 and $83 (2006$).
Acute Respiratory Symptoms	$30	The valuation estimate for "any of 19 acute respiratory symptoms" is derived from Krupnick et al. (1990) assuming that this health endpoint consists either of upper respiratory symptoms (URS) or lower respiratory symptoms (LRS), or both. We assumed the following probabilities for a day of "any of 19 acute respiratory symptoms": URS with 40 percent probability, LRS with 40 percent probability, and both with 20 percent probability. The point estimate of WTP to avoid a day of "the presence of any of 19 acute respiratory symptoms" is $28 (2006$). The value is assumed have a uniform distribution between $0 and $56 (2006$).

* All estimates rounded to two significant figures. All values have been inflated to reflect values in 2006 dollars and income levels in 2020.

4a.6 Limitations and Uncertainty

Our approach incorporates methods to assess two aspects of uncertainty quantitatively: Monte Carlo analysis and sensitivity analysis. We also provide a qualitative assessment for those aspects that we are unable to address quantitatively in this analysis. Each of these analyses is described in detail in the following sections.

This analysis includes many data sources as inputs, including emission inventories, air quality data from models (with their associated parameters and inputs), population data, health effect estimates from epidemiology studies, and economic data for monetizing benefits. Each of these inputs may be uncertain and would affect the benefits estimate. When the uncertainties from each stage of the analysis are compounded, small uncertainties can have large effects on the total quantified benefits. In this analysis, we are unable to quantify the cumulative effect of all of these uncertainties, but we provide the following analyses to characterize many of the largest sources of uncertainty.

Monte Carlo analysis

Similar to other recent RIAs, we used Monte Carlo methods for estimating characterizing random sampling error associated with the concentration response functions and economic valuation functions. Monte Carlo simulation uses random sampling from distributions of parameters to characterize the effects of uncertainty on output variables, such as incidence of morbidity. Specifically, we used Monte Carlo methods to generate confidence intervals around the estimated health impact and dollar benefits. In Table 4a-5, we present the results of this Monte Carlo analysis conducted in the area-wide analysis for the NO_2 NAAQS proposal RIA as an illustrative example of the random sampling error and 95th percentile confidence intervals.

Table 4a-5: NO_2 Benefits of Attaining 50 ppb Standard (95th percentile confidence interval) [a]

		Incidence		Valuation	
Total	Asthma Exacerbation	87,000	(250 -- 220,000)	$4,700,000	($240,000 -- $13,000,000)
	Hospital Admissions, Chronic Lung Disease	28	(23 -- 35)	$490,000	($400,000 -- $560,000)
	Hospital Admissions, Asthma	27	(11 -- 50)	$300,000	($130,000 -- $460,000)
	Emergency Room Visits, Respiratory	160	(32 -- 330)	$61,000	($14,000 -- $110,000)
	Acute Respiratory Symptoms	27,000	(-7,900 -- 75,000)	$820,000	(-$220,000 -- $2,700,000)
Grand Total				**$6,300,000**	**($570,000 -- $16,000,000)**

[a] This table shows the results of the Monte Carlo analysis conducted for the area-wide analysis in the NO_2 NAAQS proposal RIA as an illustrative example of the sensitivity of the random sampling error and 95th percentile confidence intervals.

Sensitivity analyses

We performed a variety of sensitivity analyses on the benefits results to assess the sensitivity of the primary results to various data inputs and assumptions. We then changed each default input one at a time and recalculated the total monetized benefits to assess the percent change from the default. In Table 4a-6, we present the results of this sensitivity analysis conducted in the area-wide analysis for the NO_2 NAAQS proposal RIA as an illustrative example of the sensitivity of various parameters. We indicate each input parameter, the value used as the default, and the values for the sensitivity analyses, and then we provide the total monetary benefits for each input and the percent change from the default value. Descriptions of the sensitivity analyses are provided in the relevant sections of this appendix.

Table 4a-6: Sensitivity Analyses for NO_2 Health Benefits to Fully Attain the 50 ppb Standard (Area-wide analysis) [a]

		Total NO_2 Benefits (millions of 2006$)	% Change from Default
Exposure Estimation Method	30km radius	$6.3	N/A
	12km grid cell	$1.4	-77%
	15km radius	$5.1	-19%
	CBSA	$6.3	0.6%
	Unconstrained	$8.9	42%
Location of Hospital Admission Studies	w/US-based studies only	$6.3	N/A
	w/Canada-based studies only[b]	$11	79%
Simulated Attainment	Just attainment	$6.3	N/A
	Over-control attainment	$6.8	10%
	Partial Attainment (El Paso)	$5.8	-6.2%
	Partial Attainment (El Paso and Los Angeles)	$4.6	-27%
Asthma Pooling Method	Pool all endpoints together	$6.3	N/A
	Ostro et al only	$2.1	-66%
	One or more symptoms only	$6.9	11%
Interpolation Method	Inverse Distance Squared	$6.3	N/A
	Inverse Distance	$5.8	-6.2%

[a] This table shows the results of the sensitivity analysis conducted for the area-wide analysis in the NO_2 NAAQS proposal RIA as an illustrative example of the sensitivity of various parameters of this methodology.
[b] Using Canadian studies is not a direct comparison because it includes a more complete endpoint (all respiratory hospital admissions, ages 65+), whereas the US-based studies only include hospital admissions for asthma (all ages) and chronic lung disease (ages 65+).

Qualitative assessment of uncertainty and other analysis limitations

Although we strive to incorporate as many quantitative assessments of uncertainty, there are several aspects for which we are only able to address qualitatively. These aspects are important factors to consider when evaluating the relative benefits of the attainment strategies for each of the alternative standards:

1. The gradient of ambient NO_2 concentrations is difficult to estimate due to the sparsity of the monitoring network. The 12km CMAQ grid, which is the air quality modeling resolution, may be too coarse to accurately estimate the potential near-field health benefits of reducing NO_2 emissions. These uncertainties may under- or over-estimate benefits.
2. The interpolation techniques used to estimate the full attainment benefits of the alternative standards contributed some uncertainty to the analysis. The great majority of benefits estimated for the most stringent standard alternative were derived through interpolation. As noted previously in this appendix, these benefits are likely to be more uncertain than if we had modeled the air quality scenario for both NO_2 and $PM_{2.5}$. In general, the VNA interpolation approach will under-estimate benefits because it does not account for the broader spatial distribution of air quality changes that may occur due to the implementation of a regional emission control program.
3. There are many uncertainties associated with the health impact functions used in this modeling effort. These include: within study variability (the precision with which a given study estimates the relationship between air quality changes and health effects); across study variation (different published studies of the same pollutant/health effect relationship typically do not report identical findings and in some instances the differences are substantial); the application of C-R functions nationwide (does not account for any relationship between region and health effect, to the extent that such a relationship exists); the possibility of exposure misclassification in the study due to unmeasured variability in NO_2 concentrations near roadways; extrapolation of impact functions across population (we assumed that certain health impact functions applied to age ranges broader than that considered in the original epidemiological study); and various uncertainties in the C-R function, including causality and thresholds. These uncertainties may under- or over-estimate benefits.
4. Co-pollutants present in the ambient air may have contributed to the health effects attributed to NO_2 in single pollutant models. Risks attributed to NO_2 might be overestimated where concentration-response functions are based on single pollutant models. If co-pollutants are highly correlated with NO_2, their inclusion in an NO_2 health effects model can lead to misleading conclusions in identifying a specific causal

pollutant. Because this collinearity exists, many of the studies reported statistically insignificant effect estimates for both NO_2 and the co-pollutants; this is due in part to the loss of statistical power as these models control for co-pollutants. Where available, we have selected multipollutant effect estimates to control for the potential confounding effects of co-pollutants; these include NYDOH (2006), Schwartz et al. (1994) and O'Conner et al. (2007). The remaining studies include single pollutant models.

5. This analysis is for the year 2020, and projecting key variables introduces uncertainty. Inherent in any analysis of future regulatory programs are uncertainties in projecting atmospheric conditions and source level emissions, as well as population, health baselines, incomes, technology, and other factors.
6. This analysis omits certain unquantified effects due to lack of data, time and resources. These unquantified endpoints include other health effects, ecosystem effects, and visibility. EPA will continue to evaluate new methods and models and select those most appropriate for estimating the benefits of reductions in air pollution. Enhanced collaboration between air quality modelers, epidemiologists, toxicologists, ecologists, and economists should result in a more tightly integrated analytical framework for measuring benefits of air pollution policies.

4a.7 Discussion

The benefits methodology described in this appendix suggests that reducing NO_2 emissions would produce substantial health benefits in the form of fewer respiratory hospitalizations, respiratory emergency department visits and cases of acute respiratory symptoms from reduced NO_2 exposure.

This methodology is the first time that EPA has estimated the monetized human health benefits of reducing exposure to NO_2 to support a proposed change in the NAAQS. In contrast to recent $PM_{2.5}$ and ozone-related benefits assessments, there was far less analytical precedent on which to base this assessment. For this reason, we developed entirely new components of the health impact analysis, including the identification of health endpoints to be quantified and the selection of relevant effect estimates within the epidemiology literature. As the NO_2 health literature continues to evolve, EPA will reassess the health endpoints and risk estimates used in this analysis.

While monetized NO_2 benefits may appear small when compared to recent analyses for $PM_{2.5}$ benefits or ozone benefits, readers should not necessarily infer that the total monetized benefits of NO_2 emission reductions are small. The methodology described in this appendix only captures NO_2 health benefits, not the significant monetized co-benefits from reductions in $PM_{2.5}$ or ozone. Further, the size of the benefits is related to three principle factors. As demonstrated in previous RIAs, the magnitude and geographic extent of emission reductions in the control strategy necessary to bring an area into attainment are well correlated with the size of the monetized health benefits of that standard. Second, the size of monetized benefits is correlated with both the severity of those health effects correlated of NO_2 exposure. Third, the monetized benefits are in part a function of the health endpoints quantified in the analysis. Compared to the $PM_{2.5}$ co-benefits, the benefits from reduced NO_2 exposure appear small. This is primary due to the decision not to quantify NO_2-related premature mortality and other morbidity endpoints due to the uncertainties associated with estimating this endpoint. Because premature mortality generally comprises over 90% of the total monetized benefits, this decision may underestimate the monetized health benefits of reduced NO_2 exposure. Studies have shown that there is a relationship between NO_2 exposure and premature mortality, but that relationship is generally weaker than the PM-mortality relationship and efforts to quantify that relationship have been hampered by confounding with other pollutants. For most scenarios, $PM_{2.5}$ co-benefits would represent over 95% of the total monetized benefits. This result is consistent with recent RIAs, where the $PM_{2.5}$ co-benefits represent a large proportion of total monetized benefits.

It is important to note that this analysis does not attempt to estimate the benefits in any area of the country other than those counties currently served by one of the 409 monitors in the current monitoring network. We recognize that once a network of near-roadway monitors is in place, more areas could exceed the new NO_2 NAAQS and require emission reductions. However for this analysis, we lack sufficient data to predict NO_2 exposure after implementation of a near-roadway monitoring network. Therefore, we are unable to estimate the NO_2 benefits of that scenario.

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Chapter 5: Estimates of Costs and Benefits

Synopsis

As discussed in previous chapters, under the current area-wide monitoring network, we have found no costs or benefits associated with attaining an NO_2 National Ambient Air Quality Standard (NAAQS) for the selected standard of 100 ppb, as our analysis projects no monitors in the existing network to have with maximum 1-hour design values as high as 80 ppb in 2020. Therefore, this Chapter does not include area-wide estimates.

In this RIA, we also adjusted the monitors in the existing area-wide network to approximate future near-roadway peaks in those counties. This analysis relies on current and future estimated air quality concentrations at area-wide monitors, making adjustments to future year projections using derived estimates of the relationship between future year area-wide air quality peaks and current near-roadway peaks. This additional analysis, which effectively extrapolates future year near-roadway air quality from projected area-wide concentrations, represents a screening level approximation with significant additional uncertainties. This Chapter also presents the benefits and costs of this screening level analysis to approximate future near-roadway conditions. We have found no costs or benefits associated with attaining a NAAQS for the selected standard of 100 ppb, as our analysis projects no monitors in the existing network after a near-roadway adjustment at this level in 2020.

It is important to reiterate that this analysis does not attempt to estimate attainment or nonattainment for any areas of the country other than those counties currently served by one of the 409 monitors in the current network. Chapter 2 explains that the current area-wide network is focused on community-wide ambient levels of NO_2, and not near-roadway levels, which may be significantly higher. In addition, this rule includes requirements for an NO_2 monitoring network that will include monitors near major roadways. We recognize that once a network of near-roadway monitors is put in place, more areas could find themselves exceeding the new hourly NO_2 NAAQS. However for this RIA, we lack sufficient data to predict which additional counties might exceed the new NAAQS after implementation of a near-roadway monitoring network. In our area-wide analysis, we projected current area-wide monitor values to future year monitor values directly, using future year CMAQ modeling outputs that take into account expected changes in emissions from 2006 to 2020. However regional scale models such as CMAQ do not provide a sufficient level of sub-grid detail to estimate near-road concentrations. (In addition, local-scale models such as AERMOD cannot model large regions with appropriate characterization of the near-road component of ambient air quality).

5.1 Benefits and Costs for Future Near-Roadway NO_2 Levels

Tables 5-1 and 5-2 present the counties in nonattainment, tons of NOx reduction, costs, and benefits for future near roadway levels using the near road gradient adjustment at discount rates of 3% and 7% respectively. The selected standard of 100 ppb is highlighted.

Table 5-1: Benefit Cost Comparison for Near Roadway Analysis (in millions of 2006$, 3% discount rate for Benefits only)

	Standard Level	# Counties in Nonattainment	Tons of NOx Reduction	Total Costs *			Total Benefits **			Net Benefits		
30% Gradient	80 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
	100 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
	125 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
65% Gradient	80 ppb	1	680	$5.6	to	$7.7	$3.5	to	$8.6	-$4.1	to	$3.0
	100 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
	125 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
100% Gradient	80 ppb	4	21,000	$67	to	$130	$110	to	$270	-$21	to	$200
	100 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
	125 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6

* Total Cost estimates are shown as a range from $3,000/ton to $6,000/ton. Results include monitoring costs of $3.6m. Costs estimates were only available for a 3% discount rate. All estimates have been rounded to two significant figures.

**Total Benefit estimates are actually $PM_{2.5}$ co-benefits, shown as a range from Pope et al to Laden et al, at a 3% discount rate, using no-threshold functions, assuming NOx emission reductions from the mobile sector.

Table 5-2: Benefit Cost Comparison for Near Roadway Analysis (in millions of 2006$, 7% discount rate)

	Standard Level	# Counties in Nonattainment	Tons of NOx Reduction	Total Costs *			Total Benefits **			Net Benefits		
30% Gradient	80 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
	100 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
	125 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
65% Gradient	80 ppb	1	680	$5.6	to	$7.7	$3.2	to	$7.8	-$4.5	to	$2.1
	100 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
	125 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
100% Gradient	80 ppb	4	21,000	$67	to	$130	$100	to	$240	-$31	to	$180
	100 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6
	125 ppb	0	0	$3.6	to	$3.6	$0	to	$0	-$3.6	to	-$3.6

* Total Cost estimates are shown as a range from $3,000/ton to $6,000/ton. Results include monitoring costs of $3.6m. Costs estimates were only available for a 3% discount rate. All estimates have been rounded to two significant figures.

**Total Benefit estimates are actually $PM_{2.5}$ co-benefits, shown as a range from Pope et al to Laden et al, at a 3% discount rate, using no-threshold functions, assuming NOx emission reductions from the mobile sector.

5.2 Discussion of Uncertainties and Limitations

As with other NAAQS RIAs, it should be recognized that all estimates of future costs and benefits are not intended to be forecasts of the actual costs and benefits of implementing revised standards. Ultimately, states and urban areas will be responsible for developing and implementing emissions control programs to reach attainment of the NO_2 NAAQS, with the timing of attainment being determined by future decisions by states and EPA. Our estimates are intended to provide information on the general magnitude of the costs and benefits of alternative standards, rather than precise predictions of control measures, costs, or benefits. With these caveats, we expect that this analysis can provide a reasonable picture of the types of emissions controls that are currently available, the direct costs of those controls, the levels of emissions reductions that may be achieved with these controls, the air quality impact that can be expected to result from reducing emissions, and the public health benefits of reductions in ambient NO_2 levels, as well as coincident reductions in ambient fine particulates.

In the remainder of this section we re-state the most important limitations and uncertainties in the cost and benefit estimates related to the screening level near-roadway analysis.

- Due to the absence of a near-roadway monitoring network, this is a screening level analysis with several simplifying assumptions. It is provided to give a rough projection of the costs and benefits of attaining a revised NO2 standard based on a yet to be established monitoring network.
- This analysis does not take into account a large variety of localized conditions specific to individual monitors; instead, the analysis attempts to account for some local parameters by adjusting future design values based on average localized impacts near roads from onroad emissions.
- The process of adjusting from a specific 12 km CMAQ receptor to a near-road air quality estimate represents an uncertain approximation at the specific monitor level.
- This analysis is an approximation in that it derives future year (2020) peak air quality concentrations in specific locations by relying on CMAQ estimates that are averages over a 12 km grid square.

- This analysis cannot predict air quality in locations for which there is no current NO_2 monitor, or where current monitoring data is incomplete. There are 142 CBSAs for which we are proposing to add new near-road monitors. Of these, 73 either have no existing monitor in the CBSA, or have a monitor with data not complete enough to include in the near-roadway analysis. In these CBSAs, extrapolation to near-roadway levels is not possible.

- This analysis assumes area-wide monitors remain in the same location; however concentrations are adjusted to reflect near-roadway conditions.

- Because the emission reductions in this analysis are solely reductions from mobile sources, this analysis uses an estimated cost per ton for NOx emission reductions that is different from the estimated cost per ton for NOx emission reductions used in the main body of the RIA.

- This analysis omits certain unquantified effects due to lack of data, time and resources. These unquantified endpoints include NO_2 health effects, ozone co-benefits, ecosystem effects, and visibility.

Chapter 6: Statutory and Executive Order Reviews

A. Executive Order 12866: Regulatory Planning and Review

Under section 3(f)(1) of Executive Order (EO) 12866 (58 FR 51735, October 4, 1993), this action is not an "economically significant regulatory action" because it is not likely to have an annual effect on the economy of $100 million or more. Nevertheless, EPA has submitted this action to the Office of Management and Budget (OMB) for review under EO 12866 and any changes made in response to OMB recommendations have been documented in the docket for this action. In addition, EPA prepared this Regulatory Impact Analysis (RIA) of the potential costs and benefits associated with this action. However, the CAA and judicial decisions make clear that the economic and technical feasibility of attaining ambient standards are not to be considered in setting or revising NAAQS, although such factors may be considered in the development of State plans to implement the standards. Accordingly, although an RIA has been prepared, the results of the RIA have not been considered in developing this final rule.

B. Paperwork Reduction Act

The information collection requirements in this final rule will be submitted for approval to the Office of Management and Budget (OMB) under the Paperwork Reduction Act, 44 U.S.C. 3501 et seq. The information collection requirements are not enforceable until OMB approves them.

The information collected under 40 CFR part 53 (e.g., test results, monitoring records, instruction manual, and other associated information) is needed to determine whether a candidate method intended for use in determining attainment of the National Ambient Air Quality Standards (NAAQS) in 40 CFR part 50 will meet the design, performance, and/or comparability requirements for designation as a Federal reference method (FRM) or Federal equivalent method (FEM). We do not expect the number of FRM or FEM determinations to increase over the number that is currently used to estimate burden associated with NO_2 FRM/FEM determinations provided in the current ICR for 40 CFR part 53 (EPA ICR number 2358.01). As such, no change in the burden estimate for 40 CFR part 53 has been made as part of this rulemaking.

The information collected and reported under 40 CFR part 58 is needed to determine compliance with the NAAQS, to characterize air quality and associated health and ecosystem impacts, to develop emissions control strategies, and to measure progress for the air pollution program. The amendments would revise the technical requirements for NO_2 monitoring sites, require the siting and operation of additional NO2 ambient air monitors, and the reporting of the collected ambient NO_2 monitoring data to EPA's Air Quality System (AQS). The annual average reporting burden for the collection under 40 CFR part 58 (averaged over the first 3 years of this ICR) for 142 respondents is estimated to increase by a total of 38,077 labor hours per year with an increase of $3,616,487 per year. Burden is defined at 5 CFR 1320.3(b). State, local, and tribal entities are eligible for State assistance grants provided by the Federal government under the Clean Air Act which can be used for monitors and related activities.

An agency may not conduct or sponsor, and a person is not required to respond to, a collection of information unless it displays a currently valid OMB control number. The OMB control numbers for EPA's regulations in 40 CFR are listed in 40 CFR part 9.

C. Regulatory Flexibility Act

The Regulatory Flexibility Act (RFA) generally requires an agency to prepare a regulatory flexibility analysis of any rule subject to notice and comment rulemaking requirements under the Administrative Procedure Act or any other statute unless the agency certifies that the rule will not have a significant economic impact on a substantial number of small entities. Small entities include small businesses, small organizations, and small governmental jurisdictions.

For purposes of assessing the impacts of this rule on small entities, small entity is defined as: (1) a small business that is a small industrial entity as defined by the Small Business Administration's (SBA) regulations at 13 CFR 121.201; (2) a small governmental jurisdiction that is a government of a city, county, town, school district or special district with a population of less than 50,000; and (3) a small organization that is any not-for-profit enterprise which is independently owned and operated and is not dominant in its field.

After considering the economic impacts of this final rule on small entities, the Administrator certified this action will not have a significant economic impact on a substantial number of small entities. This final rule will not impose any requirements on small entities. Rather, this rule establishes national standards for allowable concentrations of NO_2 in ambient air as required by section 109 of the CAA. *American Trucking Ass'ns v. EPA,* 175 F. 3d 1027, 1044-45 (D.C. cir. 1999) (NAAQS do not have significant impacts upon small entities because NAAQS themselves impose no regulations upon small entities). Similarly, the amendments to 40 CFR part 58 address the requirements for States to collect information and report compliance with the NAAQS and will not impose any requirements on small entities.

D. Unfunded Mandates Reform Act

Title II of the Unfunded Mandates Reform Act of 1995 (UMRA), Public Law 104-4, establishes requirements for Federal agencies to assess the effects of their regulatory actions on State, local, and tribal governments and the private sector. Unless otherwise prohibited by law, under section 202 of the UMRA, EPA generally must prepare a written statement, including a cost-benefit analysis, for proposed and final rules with "Federal mandates" that may result in expenditures to State, local, and tribal governments, in the aggregate, or to the private sector, of $100 million or more in any one year. Before promulgating an EPA rule for which a written statement is required under section 202, section 205 of the UMRA generally requires EPA to identify and consider a reasonable number of regulatory alternatives and to adopt the least costly, most cost-effective or least burdensome alternative that achieves the objectives of the rule. The provisions of section 205 do not apply when they are inconsistent with applicable law. Moreover, section 205 allows EPA to adopt an alternative other than the least costly, most cost-effective or least burdensome alternative if the Administrator publishes with the final rule an explanation why that alternative was not adopted. Before EPA establishes any regulatory requirements that may significantly or uniquely affect small governments, including tribal governments, it must have developed under section 203 of the UMRA a small government agency plan. The plan must provide for notifying potentially affected small governments, enabling officials of affected small governments to have meaningful and timely input in the development of EPA regulatory proposals with significant Federal intergovernmental mandates, and informing, educating, and advising small governments on compliance with the regulatory requirements.

This action is not subject to the requirements of sections 202 and 205 of the UMRA. EPA has determined that this final rule does not contain a Federal mandate that may result in expenditures of $100 million or more for State, local, and tribal governments, in the aggregate, or the private sector in any one year. The revisions to the NO_2 NAAQS impose no enforceable duty on any State, local or Tribal governments or the private sector. The expected costs associated with the increased monitoring requirements are described in EPA's ICR document, but those costs are not expected to exceed $100 million in the aggregate for any year. Furthermore, as indicated previously, in setting a NAAQS EPA cannot consider the economic or technological feasibility of attaining ambient air quality standards. Because the Clean Air Act prohibits EPA from considering the types of estimates and assessments described in section 202 when setting the NAAQS, the UMRA does not require EPA to prepare a written statement under section 202 for the revisions to the NO_2 NAAQS.

With regard to implementation guidance, the CAA imposes the obligation for States to submit SIPs to implement the NO_2 NAAQS. In this final rule, EPA is merely providing an interpretation of those requirements. However, even if this rule did establish an independent obligation for States to submit SIPs, it is questionable whether an obligation to submit a SIP revision would constitute a Federal mandate in any case. The obligation for a State to submit a SIP that arises out of section 110 and section 191 of the CAA is not legally enforceable by a court of law, and at most is a condition for continued receipt of highway funds. Therefore, it is possible to view an action requiring such a submittal as not creating any enforceable duty within the meaning of $_2$ U.S.C. 658 for purposes of the UMRA. Even if it did, the duty could be viewed as falling within the exception for a condition of Federal assistance under $_2$ U.S.C. 658.

EPA has determined that this final rule contains no regulatory requirements that might significantly or uniquely affect small governments because it imposes no enforceable duty on any small governments. Therefore, this rule is not subject to the requirements of section 203 of the UMRA.

E. Executive Order 13132: Federalism

Executive Order 13132, entitled "Federalism" (64 FR 43255, August 10, 1999), requires EPA to develop an accountable process to ensure "meaningful and timely input by State and local officials in the development of regulatory policies that have federalism implications." "Policies that have federalism implications" is defined in the Executive Order to include regulations that have "substantial direct effects on the States, on the relationship between the national government and the States, or on the distribution of power and responsibilities among the various levels of government."

This final rule does not have federalism implications. It will not have substantial direct effects on the States, on the relationship between the national government and the States, or on the distribution of power and responsibilities among the various levels of government, as specified in Executive Order 13132. The rule does not alter the relationship between the Federal government and the States regarding the establishment and implementation of air quality improvement programs as codified in the CAA. Under section 109 of the CAA, EPA is mandated to establish NAAQS; however, CAA section 116 preserves the rights of States to establish more stringent requirements if deemed necessary by a State. Furthermore, this rule does not impact CAA section 107 which establishes that the States have primary responsibility for implementation of the NAAQS. Finally, as noted in section E (above) on UMRA, this rule does not impose significant costs on State, local, or tribal governments or the private sector. Thus, Executive Order 13132 does not apply to this rule.

F. Executive Order 13175: Consultation and Coordination with Indian Tribal Governments

Executive Order 13175, entitled "Consultation and Coordination with Indian Tribal Governments" (65 FR 67249, November 9, 2000), requires EPA to develop an accountable process to ensure "meaningful and timely input by tribal officials in the development of regulatory policies that have tribal implications." This final rule does not have tribal implications, as specified in Executive Order 13175. It does not have a substantial direct effect on one or more Indian tribes, on the relationship between the federal government and Indian tribes, or on the distribution of power and responsibilities between the federal government and tribes. The rule does not alter the relationship between the federal government and tribes as established in the CAA and the TAR. Under section 109 of the CAA, EPA is mandated to establish NAAQS; however, this rule does not infringe existing tribal authorities to regulate air quality under their own programs or under programs submitted to EPA for approval. Furthermore, this rule does not affect the flexibility afforded to tribes in seeking to implement CAA programs consistent with the TAR, nor does it impose any new obligation on tribes to adopt or implement any NAAQS. Finally, as noted in section E (above) on UMRA, this rule does not impose significant costs on tribal governments. Thus, Executive Order 13175 does not apply to this rule.

G. Executive Order 13045: Protection of Children from Environmental Health & Safety Risks

This action is not subject to Executive Order (62 FR 19885, April 23, 1997) because it is not an economically significant regulatory action as defined by Executive Order 12866. However, we believe that the environmental health risk addressed by this action could have a disproportionate effect on children. The final rule will establish uniform national ambient air quality standards for NO_2; these standards are designed to protect public health with an adequate margin of safety, as required by CAA section 109. The protection offered by these standards may be especially important for asthmatics, including asthmatic children, because respiratory effects in asthmatics are among the most sensitive health endpoints for NO_2 exposure. Because asthmatic children are considered a sensitive population, we have evaluated the potential health effects of exposure to NO_2 pollution among asthmatic children. These effects and the size of the population affected are discussed in chapters 3 and 4 of the ISA; chapters 3, 4, and 8 of the REA, and sections II.A through II.E of the preamble.

H. Executive Order 13211: Actions that Significantly Affect Energy Supply, Distribution or Use

This rule is not a "significant energy action" as defined in Executive Order 13211, "Actions Concerning Regulations That Significantly Affect Energy Supply, Distribution, or Use" (66 FR 28355 (May 22, 2001)) because it is not likely to have a significant adverse effect on the supply, distribution, or use of energy. The purpose of this rule is to establish revised NAAQS for NO_2. The rule does not prescribe specific control strategies by which these ambient standards will be met. Such strategies will be developed by States on a case-by-case basis, and EPA cannot predict whether the control options selected by States will include regulations on energy suppliers, distributors, or users. Thus, EPA concludes that this rule is not likely to have any adverse energy effects.

I. National Technology Transfer and Advancement Act

Section 12(d) of the National Technology Transfer and Advancement Act of 1995 (NTTAA), Public Law 104-113, section 12(d) (15 U.S.C. 27_2 note) directs EPA to use voluntary consensus standards in its regulatory activities unless to do so would be inconsistent with applicable law or otherwise impractical. Voluntary consensus standards are technical standards (e.g., materials specifications, test methods, sampling procedures, and business practices) that are developed or adopted by voluntary consensus standards bodies. The NTTAA directs EPA to provide Congress, through OMB, explanations when the Agency decides not to use available and applicable voluntary consensus standards.

This final rulemaking involves technical standards with regard to ambient monitoring of NO_2. The use of this voluntary consensus standard would be impractical because the analysis method does not provide for the method detection limits necessary to adequately characterize ambient NO_2 concentrations for the purpose of determining compliance with the proposed revisions to the NO_2 NAAQS.

J. Executive Order 12898: Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations

Executive Order 12898 (59 FR 7629; Feb. 16, 1994) establishes federal executive policy on environmental justice. Its main provision directs federal agencies, to the greatest extent practicable and permitted by law, to make environmental justice part of their mission by identifying and addressing, as appropriate, disproportionately high and adverse human health or environmental effects of their programs, policies, and activities on minority populations and low-income populations in the United States.

EPA has determined that this final rule will not have disproportionately high and adverse human health or environmental effects on minority or low-income populations because it increases the level of environmental protection for all affected populations without having any disproportionately high and adverse human health effects on any population, including any minority or low-income population. The final rule will establish uniform national standards for NO_2 in ambient air.