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	Subchronic bronchitis cases
	Bronchitis: chronic and acute	Low birth weight
	Hospital admissions: respiratory and cardiovascular	Pulmonary function
	Emergency room visits for asthma	Chronic respiratory diseases other than chronic bronchitis
	Nonfatal heart attacks (myocardial infarction)	Non-asthma respiratory emergency room visits
	Lower and upper respiratory illness	Visibility
	Minor restricted-activity days	Household soiling
	Work loss days	
	Asthma exacerbations (asthmatic population)	
	Infant mortality	

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

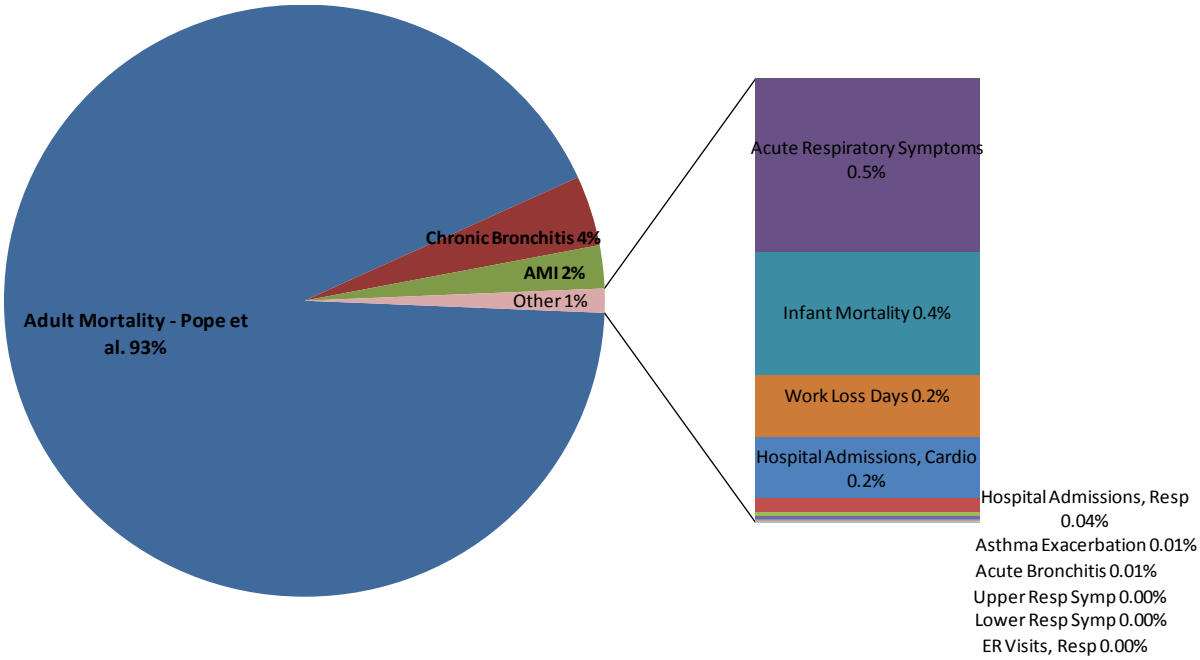
[11] In this analysis, we adjust the VSL to account for a different currency year (2006$) and to account for income growth to 2020. After applying these adjustments to the $6.3 million value, the VSL is $8.9m.

<u>3% discount rate</u> Age 0–24 Age 25–44 Age 45–54 Age 55–65 Age 66 and over	 $80,000 $90,000 $94,000 $170,000 $80,000	simple average of estimates from Russell et al. (1998) and Wittels et al. (1990). Lost earnings: Cropper and Krupnick (1990). Present discounted value of 5 years of lost earnings in (2006$): 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%)
<u>7% discount rate</u> Age 0–24 Age 25–44 Age 45–54 Age 55–65 Age 66 and over	 $80,000 $88,000 $92,000 $160,000 $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 $\mu g/m^3$, 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

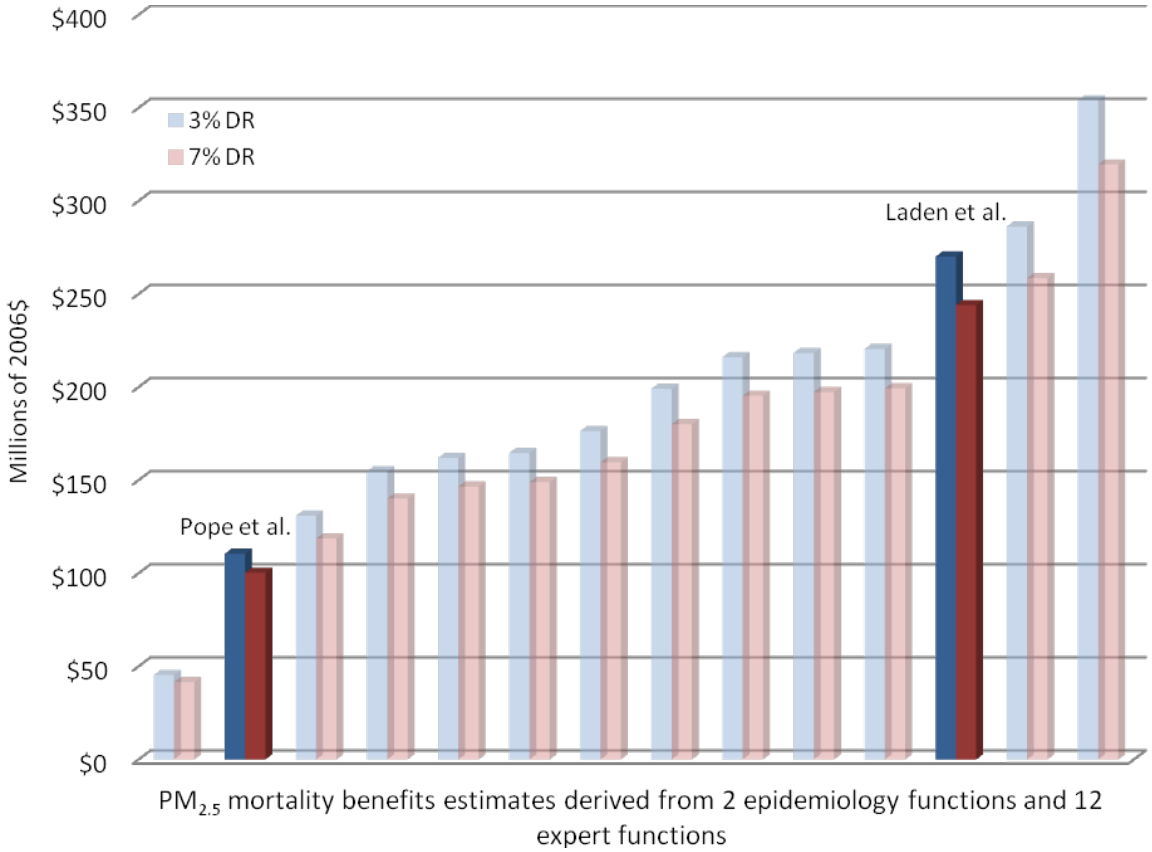
*All estimates are rounded to two significant figures. Estimates do not include confidence intervals because they were derived through the benefit-per-ton technique described above. The benefits estimates from the Expert Elicitation are provided as a reasonable characterization of the uncertainty in 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)



*

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.

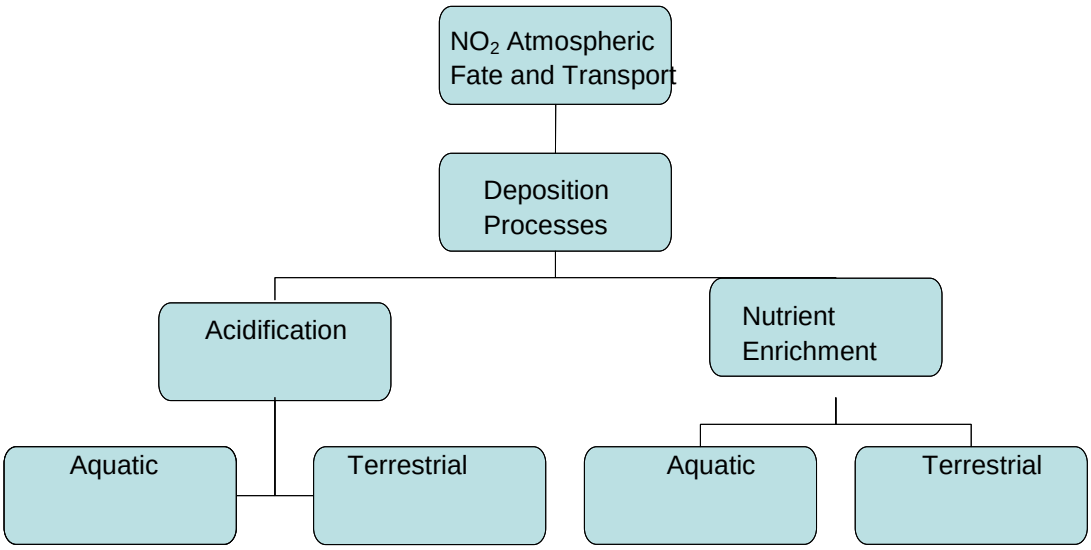
4.4 Unquantified Welfare Benefits

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



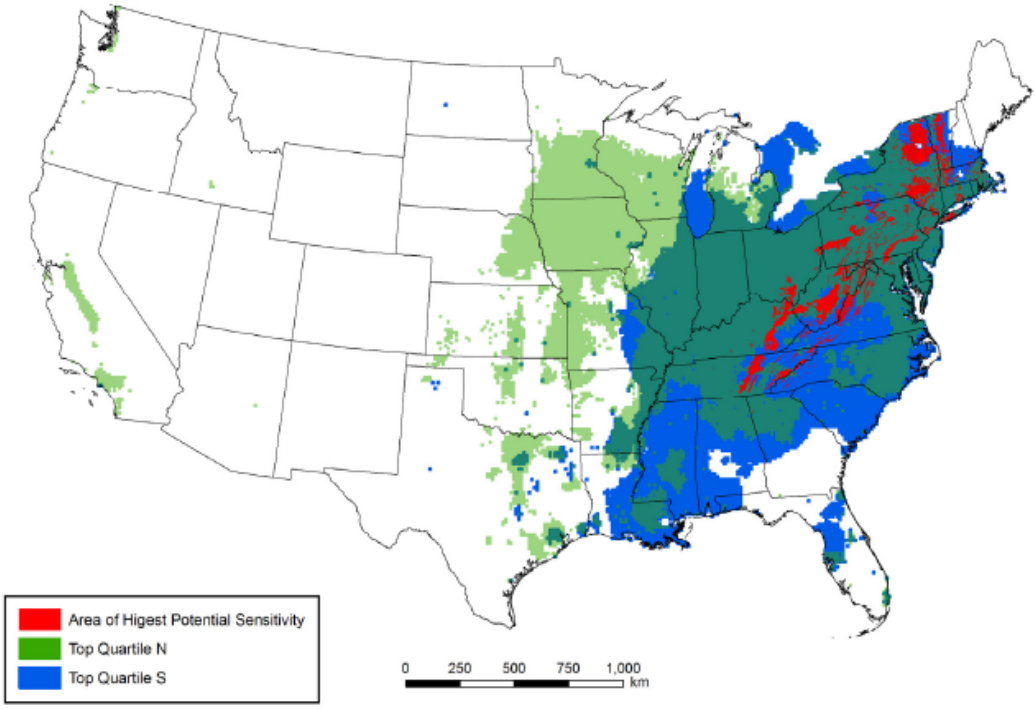
Terrestrial and Aquatic Acidification

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)



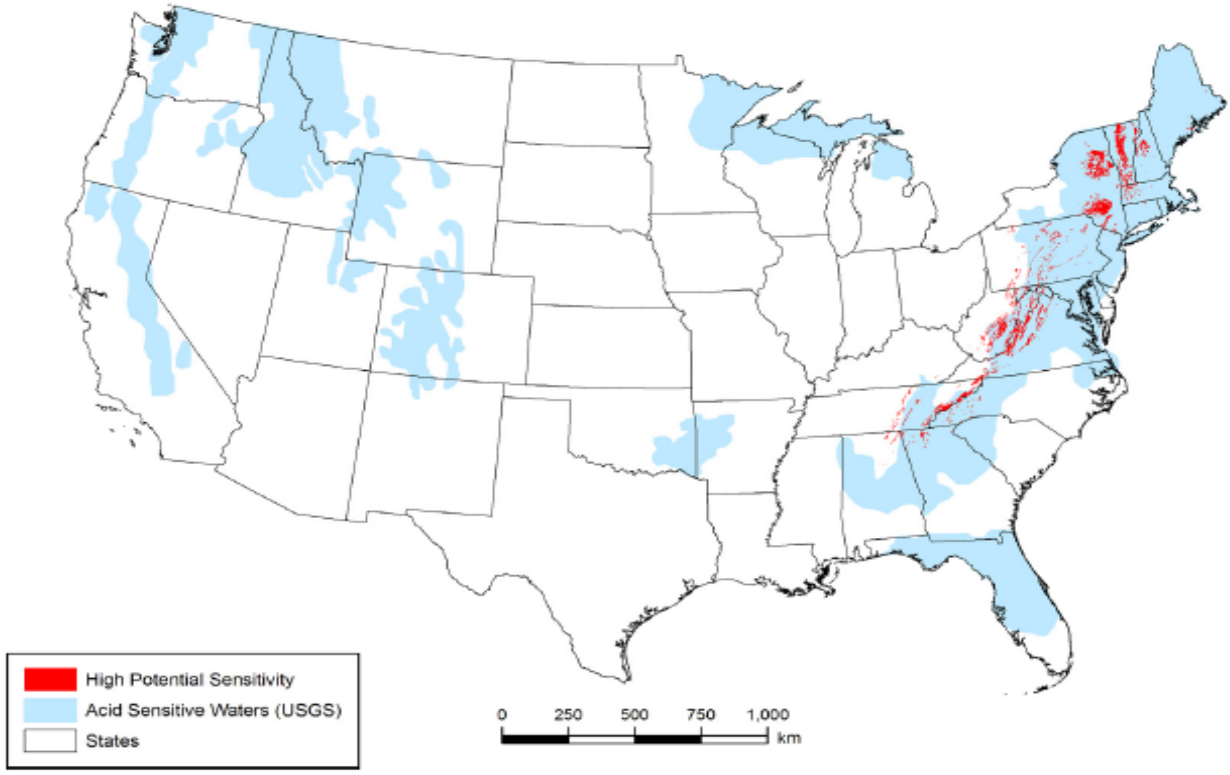
Terrestrial acidification affects several important ecological endpoints, including declines in habitat for threatened and endangered species (cultural), declines in forest

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

4.7 References

American Lung Association. 2002. Trends in Asthma Morbidity and Mortality. American Lung Association, Best Practices and Program Services, Epidemiology and Statistics Unit. Available on the Internet at <http://www.lungusa.org/data/asthma/ASTHMAdt.pdf>.

California State Parks. 2003. *Public Opinions and Attitudes on Outdoor Recreation in California 2002: An Element of the California Outdoor Recreation Plan*. Sacramento, CA: California State Parks. Available on the Internet at <http://www.parks.ca.gov/pages/795/files/poa2002final.pdf>.

Chesapeake Bay Program. “Bay Grass Abundance (Baywide)” Available on the Internet at <http://www.chesapeakebay.net/ status_baygrasses.aspx?menuitem=19652>. Last updated May 21, 2008.

Chestnut, L.G., and R.D. Rowe. 1990. A New National Park Visibility Value Estimates. In Visibility and Fine Particles, Transactions of an AWMA/EPA International Specialty Conference, C.V. Mathai, ed. Air and Waste Management Association, Pittsburgh.

Cordell, K., B. Leeworthy, G.T. Green, C. Betz, B. Stephens. n.d. *The National Survey on Recreation & the Environment*. Research Work Unit 4953. Athens, GA: Pioneering Research on Changing Forest Values in the South and Nation USDA Forest Service Southern Research Station. Available on the Internet at <http://www.srs.fs.fed.us/trends>.

Cropper, M. L. and A. J. Krupnick. 1990. The Social Costs of Chronic Heart and Lung Disease. Resources for the Future. Washington, DC. Discussion Paper QE 89-16-REV.

Fann, N., C.M. Fulcher, B.J. Hubbell. 2009. The influence of location, source, and emission type in estimates of the human health benefits of reducing a ton of air pollution. Air Qual Atmos Health (2009) 2:169–176.

Industrial Economics, Inc. 2006. Expanded Expert Judgment Assessment of the Concentration-Response Relationship Between $PM_{2.5}$ Exposure and Mortality. Prepared for the U.S. EPA, Office of Air Quality Planning and Standards, September. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/Uncertainty/pm_ee_report.pdf>.

Joslin, J.D., Kelly, J.M., van Miegroet, H. (1992). Soil chemistry and nutrition of North American spruce-fir stands: evidence for recent change. Journal of Environmental Quality, 21, 12-30.

Kahn, J.R., and W.M. Kemp. 1985. "Economic Losses Associated with the Degradation of an Ecosystem: The Case of Submerged Aquatic Vegetation in Chesapeake Bay." *Journal of Environmental Economics and Management* 12:246-263.

Kaval, P., and J. Loomis. 2003. *Updated Outdoor Recreation Use Values With Emphasis On National Park Recreation*. Final Report October 2003, under Cooperative Agreement CA 1200-99-009, Project number IMDE-02-0070.

Knowler, D. 2002. A Review of Selected Bioeconomic Models: with Environmental Influences in Fisheries. *Journal of Bioeconomics* 4:163-181.

Kochi, I., B. Hubbell, and R. Kramer. 2006. An Empirical Bayes Approach to Combining Estimates of the Value of Statistical Life for Environmental Policy Analysis. Environmental and Resource Economics. 34: 385-406.

Krewski, D., R.T. Burnett, M.S. Goldbert, K. Hoover, J. Siemiatycki, M. Jerrett, M. Abrahamowicz, and W.H. White. 2000. Reanalysis of the Harvard Six Cities Study and the American Cancer Society Study of Particulate Air Pollution and Mortality. Special Report to the Health Effects Institute. Cambridge MA. July.

Laden, F., J. Schwartz, F.E. Speizer, and D.W. Dockery. 2006. Reduction in Fine Particulate Air Pollution and Mortality. American Journal of Respiratory and Critical Care Medicine 173:667-672. Estimating the Public Health Benefits of Proposed Air Pollution Regulations. Washington, DC: The National Academies Press.

Leeworthy, V.R., and P.C. Wiley. 2001. Current Participation Patterns in Marine Recreation. Silver Spring, MD: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, Special Projects. November. Available on the Internet at < http://www.naturetourismplanning.com/pdfs/NSRE_2.pdf>.

Lipton, D. W. 1999. Pfiesteria's economic impact on seafood industry sales and recreational fishing, p. 35–38. In B. L. Gardner and L. Koch (ed.), Proceedings of the Conference, Economics of Policy Options for Nutrient Management and Pfiesteria. Center for Agricultural and Natural Resource Policy, University of Maryland, College Park.

Malm WC. 1999. Introduction to visibility. Colorado State University, Fort Collins, CO, USA, 1983, Revised edition, 1999. Available online at <vista.cira.colostate.edu/improve/Education/IntroToVisinstr.htm>.

Mrozek, J.R., and L.O. Taylor. 2002. What Determines the Value of Life? A Meta-Analysis. Journal of Policy Analysis and Management 21(2):253-270.

National Oceanic and Atmospheric Administration (NOAA). 2007. Annual Commercial Landing Statistics. August. Available on the Internet at <http://www.st.nmfs.noaa.gov/ st1/commercial/landings/annual_landings.html>.

National Research Council (NRC). 2002. Estimating the Public Health Benefits of Proposed Air Pollution Regulations. Washington, DC: The National Academies Press.

Pitchford, M. and W. Malm. 1993. Development and applications of a standard visual index. Atmospheric Environment 28(5):1049-1054.

Pope, C.A., III, R.T. Burnett, M.J. Thun, E.E. Calle, D. Krewski, K. Ito, and G.D. Thurston. 2002. "Lung Cancer, Cardiopulmonary Mortality, and Long-term Exposure to Fine Particulate Air Pollution." Journal of the American Medical Association 287:1132-1141.

Roman, Henry A., Katherine D. Walker, Tyra L. Walsh, Lisa Conner, Harvey M. Richmond, Bryan J. Hubbell, and Patrick L. Kinney. 2008. Expert Judgment Assessment of the Mortality Impact of Changes in Ambient Fine Particulate Matter in the U.S. Environ. Sci. Technol., 42(7):2268-2274.

Rowe, R. D. and L. G. Chestnut. 1986. Oxidants and Asthmatics in Los Angeles: A Benefits Analysis -- Executive Summary. Prepared for U.S. Environmental Protection Agency, Office

of Policy Analysis. Prepared by Energy and Resource Consultants, Inc. Washington, DC. EPA-230-09-86-018. March.

Sisler, J.F. (1996) Spatial and seasonal patterns and long term variability of the composition of the haze in the United States: an analysis of data from the IMPROVE network. CIRA Report, ISSN 0737-5352-32, Colorado State University.

Smith, D. H., D. C. Malone, K. A. Lawson, L. J. Okamoto, C. Battista and W. B. Saunders. 1997. A national estimate of the economic costs of asthma. Am J Respir Crit Care Med. Vol. 156 (3 Pt 1): 787-93.

Stanford, R., T. McLaughlin and L. J. Okamoto. 1999. The cost of asthma in the emergency department and hospital. Am J Respir Crit Care Med. Vol. 160 (1): 211-5.

U.S. Department of the Interior, Fish and Wildlife Service, and U.S. Department of Commerce, U.S. Census Bureau. 2007. *2006 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation*. Available on the Internet at <http://www.census.gov/prod/2008pubs/fhw06-nat.pdf >.

U.S. Environmental Protection Agency (U.S. EPA). 1999. The Benefits and Costs of the Clean Air Act 1990 to 2010: EPA Report to Congress. Office of Air and Radiation, Office of Policy, Washington, DC. November. Available on the Internet at < http://www.epa.gov/air/sect812/1990-2010/fullrept.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2000. Guidelines for Preparing Economic Analyses. EPA 240-R-00-003. National Center for Environmental Economics, Office of Policy Economics and Innovation. Washington, DC. September. Available on the Internet at <http://yosemite.epa.gov/ee/epa/eed.nsf/webpages/Guidelines.html/$file/cover.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2002. Letter to Recipients of Third Draft EPA PM Criteria Document. National Center for Environmental Assessment, Research Triangle Park, NC. June 5, 2002. Available on the Internet at <http://oaspub.epa.gov/eims/eimscomm.getfile?p_download_id=42380>.

U.S. Environmental Protection Agency (U.S. EPA). 2005. Regulatory Impact Analysis for the Final Clean Air Interstate Rule. Office of Air and Radiation. March. Available on the Internet at < http://www.epa.gov/cair/pdfs/finaltech08.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2006a. Air Quality Criteria for Ozone and Related Photochemical Oxidants. National Center for Environmental Assessment, Research Triangle Park, NC. EPA 600/R-05/004aF. February. Available on the Internet at <http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=149923>.

U.S. Environmental Protection Agency (U.S. EPA). 2006b. Willingness to Pay for Environmental Health Risk Reductions when there are Varying Degrees of Life Expectancy: A White Paper. August. Available on the Internet at <http://yosemite.epa.gov/ee/epa/eerm.nsf/vwRepNumLookup/EE-0495>.

U.S. Environmental Protection Agency (U.S. EPA). 2006c. Regulatory Impact Analysis, 2006 National Ambient Air Quality Standards for Particulate Matter, Chapter 5. Office of Air Quality Planning and Standards, Research Triangle Park, NC. October. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/RIAs/Chapter%205--Benefits.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2007. Review of the National Ambient Air Quality Standards for Ozone: Policy assessment of scientific and technical information. Staff paper. Office of Air Quality Planning and Standards. EPA-452/R-07-007a. Available on the Internet at <http://www.epa.gov/ttn/naaqs/standards/ozone/data/2007_07_ozone_staff_paper.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2008a. Regulatory Impact Analysis, 2008 National Ambient Air Quality Standards for Ground-level Ozone, Chapter 6. Office of Air Quality Planning and Standards, Research Triangle Park, NC. March. Available at <http://www.epa.gov/ttn/ecas/regdata/RIAs/6-ozoneriachapter6.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2008b. Technical Support Document: Calculating Benefit Per-Ton estimates, Ozone NAAQS Docket #EPA-HQ-OAR-2007-0225-0284. Office of Air Quality Planning and Standards, Research Triangle Park, NC. March. Available on the Internet at <http://www.regulations.gov>.

U.S. Environmental Protection Agency (U.S. EPA). 2008c. Integrated Science Assessment for Oxides of Nitrogen - Health Criteria (Final Report). National Center for Environmental Assessment, Research Triangle Park, NC. July. Available on the Internet at <http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=194645>.

U.S. Environmental Protection Agency (U.S. EPA). 2008d. Guidelines for Preparing Economic Analyses: External Review Draft. National Center for Environmental Economics, Office of

Policy Economics and Innovation. Washington, DC. September. Available on the Internet at <http://yosemite.epa.gov/ee/epa/eermfile.nsf/vwAN/EE-0516-01.pdf/$File/EE-0516-01.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2008e. Risk and Exposure Assessment to Support the Review of the NO_2 Primary National Ambient Air Quality Standard (Final Report). Office of Air Quality Planning and Standards, Research Triangle Park, NC. November. Available on the Internet at <http://www.epa.gov/ttn/naaqs/standards/nox/data/20081121_NO2_REA_final.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2008f. Integrated Science Assessment for Oxides of Nitrogen and Sulfur – Ecological Criteria National (Final Report). National Center for Environmental Assessment, Research Triangle Park, NC. EPA/600/R-08/139. December. Available on the Internet at <http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=201485>.

U.S. Environmental Protection Agency (U.S. EPA). 2009a. Regulatory Impact Analysis: National Emission Standards for Hazardous Air Pollutants from the Portland Cement Manufacturing Industry. Office of Air Quality Planning and Standards, Research Triangle Park, NC. April. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/RIAs/portlandcementria_4-20-09.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2009b. Proposed NO_2 NAAQS Regulatory Impact Analysis (RIA). Office of Air Quality Planning and Standards, Research Triangle Park, NC. November. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/RIAs/proposedno2ria.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2009c. Integrated Science Assessment for Particulate Matter (Final Report). EPA-600-R-08-139F. National Center for Environmental Assessment – RTP Division. December. Available on the Internet at <http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=216546>.

U.S. Environmental Protection Agency (U.S. EPA). 2009d. Risk and Exposure Assessment for Review of the Secondary National Ambient Air Quality Standards for Oxides of Nitrogen and Oxides of Sulfur-Main Content - Final Report. Office of Air Quality Planning and Standards, Research Triangle Park, NC. EPA-452/R-09-008a. September. Available on the Internet at <http://www.epa.gov/ttn/naaqs/standards/no2so2sec/data/NOxSOxREASep2009MainContent.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2009e. Proposed SO_2 NAAQS Regulatory Impact Analysis (RIA). Office of Air Quality Planning and Standards, Research Triangle Park, NC. November. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/RIAs/pso2full11-16-09.pdf>.

U.S. Environmental Protection Agency - Science Advisory Board (U.S. EPA-SAB). 2007. SAB Advisory on EPA's Issues in Valuing Mortality Risk Reduction. EPA-SAB-08-001. October. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/4128007E7876B8F0852573760058A978/$File/sab-08-001.pdf>.

U.S. Environmental Protection Agency - Science Advisory Board (U.S. EPA-SAB). 2008. Characterizing Uncertainty in Particulate Matter Benefits Using Expert Elicitation. EPA-COUNCIL-08-002. July. Available on the Internet at < http://yosemite.epa.gov/sab%5CSABPRODUCT.NSF/43B91173651AED9E85257487004EA6CB/$File/EPA-COUNCIL-08-002-unsigned.pdf>.

U.S. Environmental Protection Agency - Science Advisory Board (U.S. EPA-SAB). 2009a. Review of EPA's Integrated Science Assessment for Particulate Matter (First External Review Draft, December 2008). EPA-COUNCIL-09-008. May. Available on the Internet at <http://yosemite.epa.gov/sab/SABPRODUCT.NSF/81e39f4c09954fcb85256ead006be86e/73ACCA834AB44A10852575BD0064346B/$File/EPA-CASAC-09-008-unsigned.pdf>.

U.S. Environmental Protection Agency - Science Advisory Board (U.S. EPA-SAB). 2009b. Consultation on EPA's Particulate Matter National Ambient Air Quality Standards: Scope and Methods Plan for Health Risk and Exposure Assessment. EPA-COUNCIL-09-009. May. Available on the Internet at <http://yosemite.epa.gov/sab/SABPRODUCT.NSF/81e39f4c09954fcb85256ead006be86e/723FE644C5D758DF852575BD00763A32/$File/EPA-CASAC-09-009-unsigned.pdf>.

Valigura, R.A., R.B. Alexander, M.S. Castro, T.P. Meyers, H.W. Paerl, P.E. Stacy, and R.E. Turner. 2001. *Nitrogen Loading in Coastal Water Bodies: An Atmospheric Perspective*. Washington, DC: American Geophysical Union

Van Sickle, J., Baker, J.P., Simonin, H.A., Baldigo, B.P., Kretser, W.A., Sharpe, W.E. (1996). Episodic acidification of small streams in the northeastern United States: Fish mortality in field bioassays. Ecological Applications, 6, 408-421.

Viscusi, V.K., and J.E. Aldy. 2003. The Value of a Statistical Life: A Critical Review of Market Estimates throughout the World. Journal of Risk and Uncertainty 27(1):5-76.