Regulatory Impact Analysis for the Proposed Federal Transport Rule

Docket ID No. EPA-HQ-OAR-2009-0491

U.S. Environmental Protection Agency

Office of Air and Radiation

June 2010

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CHAPTER 1

EXECUTIVE SUMMARY

This Regulatory Impact Analysis (RIA) presents the health and welfare benefits, costs, and other impacts of the proposed Transport Rule focusing primarily on 2014.

1.1 Key Findings

EPA plans to lower the sulfur dioxide (SO_2) and nitrogen oxide (NO_x) emissions of the electric power industry in 32 eastern states through the proposed Transport Rule. EPA estimates in 2014 this proposed rule will have annual net benefits (in 2006$) between $120 to 290 billion using a 3% discount rate and $110 and $260 billion using a 7% discount rate. At these respective rates, the annual social costs are $2.0 billion and $2.2 billion and the annual quantified benefits are $120 to $290 billion or $110 to $270 billion. The capital costs spent for pollution controls installed for CAIR were not included in the annual social costs since the Transport Rule did not lead to their installation. Those CAIR-related capital investments are roughly estimated to have an annual social cost less than $1.15 to $ 1.29 billion (under the two discount rates.) The benefits outweigh social costs by 60 to145 to 1, or 55 to 130 to 1. The benefits are primarily from 14,000 to 36,000 fewer $PM_{2.5}$ and ozone-related premature mortalities. There are some costs and important benefits that EPA could not monetize. Upon considering these limitations and uncertainties, it remains clear that the benefits of the proposed Transport Rule are substantial and far outweigh the costs. The annualized private compliance costs to the power industry in 2014 are $2.8 billion, higher than the social costs. Consideration of the above benefit cost ratios and analysis of a greater SO_2 control suggests that, if EPA could require additional emission reductions, there could be greater net benefits. Notably, since the proposed rule expedites installation of pollution controls in 2012 that were formerly happening by 2014, the benefits of the Transport Rule in 2012 are actually even greater at the outset of the program.

The benefits and costs in 2014 of the preferred remedy (State Budgets/Limited Trading) in the proposed rule are in Table 1-1. This preferred remedy covers the electric power industry and allows intrastate emissions trading of sulfur dioxide (SO_2) and nitrogen oxides (NO_x) and limited interstate trading of these are pollutants in 32 eastern states.[1]

Table 1-1. Summary of EPA's Estimates of Benefits, Costs, and Net Benefits of the Preferred Remedy in the Proposed Transport Rule in 2014[a] (billions of 2006$)

Description	Estimate (3% Discount Rate)	Estimate (7% Discount Rate)
Social costs[b]	$2.03	$2.23
Social benefits[c,d]	$120 to $290 + B	$110 to $270 + B
Health-related benefits:	$120 to $290 + B	$110 to $260 + B
Visibility benefits[e]	$3.6	$3.6
Net benefits (benefits-costs)	$120 to $290	$110 to $260

[a] All estimates are rounded to two significant digits and represent annualized benefits and costs anticipated for the year 2014. For notational purposes, unquantified benefits are indicated with a "B" to represent the sum of additional monetary benefits and disbenefits. Data limitations prevented us from quantifying these endpoints, and as such, these benefits are inherently more uncertain than those benefits that we were able to quantify. A listing of health and welfare effects is provided in Table 1-6. Estimates here are subject to uncertainties discussed further in the body of the document.

[b] The social costs are the loss of household utility as measured in Hicksian equivalent variation.

[c] The reduction in premature mortalities account for over 90% of total monetized benefits. Benefit estimates are national except for visibility that covers Class I areas. Valuation assumes discounting over the SAB-recommended 20-year segmented lag structure described in Chapter 5. Results reflect 3 percent and 7 percent discount rates consistent with EPA and OMB guidelines for preparing economic analyses (U.S. EPA, 2000; OMB, 2003). The estimate of social benefits also includes CO2-related benefits calculated using the social cost of carbon, discussed further in chapter 5.

[d] Potential benefit categories that have not been quantified and monetized are listed in Table 1-6.

[e] Over 99% of visibility-related benefits occur within Class-1 areas located in the Eastern U.S.

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[1] The states are AL, AR, CT, DE, DC, FL, GA, IL, IN, IA, KS, KY, LA, MD, MA, MI, MN, MS, MR, NE, NJ, NY, NC, OH, OK, PA, SC, TN, TX, VA, WV, and WI.

1.1.1 Health Benefits

The proposed Transport Rule is expected to yield significant health benefits by reducing emissions of two key contributors to fine particle and ozone formation. Sulfur dioxide contributes to the formation of fine particle pollution ($PM_{2.5}$), and nitrogen oxide contributes to the formation of both $PM_{2.5}$ and ground-level ozone.

Our analyses suggest this would yield benefits in 2014 of $120 to $290 billion (based on a 3 percent discount rate) and $110 to $270 billion (based on a 7 percent discount rate) that includes the value of avoiding approximately 14,000 to 36,000 premature deaths, 22,000 nonfatal heart attacks, 11,000 hospitalizations for respiratory and cardiovascular diseases, 1.8 million lost work days, 100,000 school absences, and 10 million days when adults restrict normal activities because of respiratory symptoms exacerbated by $PM_{2.5}$ and ozone pollution.

We also estimate substantial additional health improvements for children from reductions in upper and lower respiratory illnesses, acute bronchitis, and asthma attacks. See Table 1-2 for a list of the annual reduction in health effects expected in 2014 and Table 1-3 for the estimated value of those reductions. In these tables we summarize the benefits according to whether they accrue within or beyond the Transport region (Eastern part of the US covered by the proposed rule). While not analyzed here, we expect the benefits in 2012 to be significantly larger than those modeled for 2014 because of the much greater incremental SO_2 reductions in 2012 compared to 2014 from the base case. This occurs because the proposed rule expedites the start of SO_2 emissions controls that are planned in the base case to occur after 2012 and be underway by 2014.

1.1.2 Welfare Benefits

The term *welfare benefits* covers both environmental and societal benefits of reducing pollution, such as reductions in damage to ecosystems, improved visibility and improvements in recreational and commercial fishing, agricultural yields, and forest

Table 1-2. Proposed Transport Rule: Estimated Reduction in Incidence of Adverse Health Effects in 2014 for the Proposed Remedy[a,b]

Health Effect	*Within transport region*	*Beyond transport region*	*Total*
PM-Related endpoints			
Premature Mortality			
Pope et al. (2002) (age >30)	14,000 (4,000—24,000)	130 (35—220)	14,000 (4,000—25,000)
Laden et al. (2006) (age >25)	36,000 (17,000—55,000)	320 (150—500)	36,000 (17,000—56,000)
Infant (< 1 year)	59 (-66—180)[b]	0.3 (-0.3—0.8)	59 (-66—180)
Chronic Bronchitis	9,200 (310—18,000)	89 (3—160)	9,200 (320—18,000)
Non-fatal heart attacks (age > 18)	22,000 (5,700—39,000)	250 (64—440)	22,000 (5,800—39,000)
Hospital admissions—respiratory (all ages)	3,500 (1,400—5,500)	35 (14—56)	3,500 (1,400—5,500)
Hospital admissions—cardiovascular (age > 18)	7,500 (5,200—8,800)	76 (51—93)	7,500 (5,200—8,900)
Emergency room visits for asthma (age < 18)	14,000 (7,100—21,000)	71 (36—110)	14,000 (7,200—21,000)
Acute bronchitis (age 8-12)	21,000 (-4,800—46,000)	150 (33—320)	21,000 (-4,800—46,000)
Lower respiratory symptoms (age 7-14)	250,000 (98,000—400,000)	1,700 (670—2,800)	250,000 (98,000—400,000)
Upper respiratory symptoms (asthmatics age 9-18)	190,000 (36,000—350,000)	1,300 (250—2,400)	190,000 (36,000—350,000)
Asthma exacerbation (asthmatics 6-18)	230,000 (8,300—800,000)	1,700 (11—5,700)	240,000 (8,300—800,000)
Lost work days (ages 18-65)	1,800,000 (1,500,000—2,000,000)	14,000 (12,000—17,000)	1,800,000 (1,500,000—2,000,000)
Minor restricted-activity days (ages 18-65)	10,000,000 (8,600,000—12,000,000)	86,000 (71,000—100,000)	10,000,000 (8,600,000—12,000,000)

Ozone-related endpoints

Premature mortality

Multi-city and NMMAPS-	Bell et al. (2004) (all ages)	50 (16—83)	0.6 (0.2—1)	50 (17—84)
	Schwartz et al. (2005) (all ages)	76 (23—130)	1 (0.2—2)	77 (24—130)
	Huang et al. (2005) (all ages)	83 (31—130)	1 (0.3—2)	84 (31—140)
Meta-analyses	Ito et al. (2005) (all ages)	220 (130—310)	3 (2—4)	230 (140—320)
	Bell et al. (2005) (all ages)	160 (76—250)	2 (1—3)	160 (77—250)
	Levy et al. (2005) (all ages)	230 (160—300)	3 (2—4)	230 (160—300)
Hospital admissions—respiratory causes (ages > 65)		380 (-18—730)	4 (-0.4—9)	390 (-18—740)
Hospital admissions—respiratory causes (ages <2)		290 (130—460)	4 (1—6)	300 (130—460)
Emergency room visits for asthma (all ages)		230 (-30—730)	2 (-0.4—8)	230 (-30—730)
Minor restricted-activity days (ages 18-65)		300,000 (120,000—480,000)	3,700 (1,300—6,100)	300,000 (130,000—480,000)
School absence days		110,000 (38,000—160,000)	1,300 (380—2,100)	110,000 (38,000—160,000)

[A] Estimates rounded to two significant figures; column values will not sum to total value.

[B] The negative 5th percentile estimates for certain endpoints are the result of the weak statistical power of the study used to calculate these health impacts and do not suggest imply that increases in air pollution exposure result in decreased health impacts.

Table 1-3. Estimated Monetary Value of Reductions in Incidence of Health and Welfare Effects Effects for the Proposed Remedy (in billions of 2006$)[a,b,c]

Health Effect	*Pollutant*	*Within transport region*	*Beyond transport region*[B]	*Total*
Premature Mortality (Pope et al. 2002 PM mortality and Bell et al. 2004 ozone mortality estimates)				
3% discount rate	$PM_{2.5}$ & O_3	$110 ($8.8—$330)	$0.1 ($0.08—$3)	$110 ($8.8—$340)
7% discount rate	$PM_{2.5}$ & O_3	$100 ($7.9—$300)	$0.09 ($0.07—$2.7)	$100 ($7.9—$300)
Premature Mortality (Laden et al. 2006 PM mortality and Levy et al. 2005 ozone mortality estimates)				
3% discount rate	$PM_{2.5}$ & O_3	$280 ($25—$810)	$2.5 ($0.2—$7.3)	$280 ($25—$820)
7% discount rate	$PM_{2.5}$ & O_3	$250 ($22—$300)	$2.3 ($0.2—$6.6)	$260 ($22—$310)
Chronic Bronchitis	$PM_{2.5}$	$4.3 ($0.2—$20)	$0.04 ($0.002--$0.2)	$4.3 ($0.2—$20)
Non-fatal heart attacks				
3% discount rate	$PM_{2.5}$	$2.5 ($0.4—$6)	$0.03 ($0.005—$0.07)	$2.5 ($0.4—$6)
7% discount rate	$PM_{2.5}$	$2.4 ($0.4—$5.9)	$0.03 ($0.005—$0.07)	$2.4 ($0.4—$5.9)
Hospital admissions—respiratory	$PM_{2.5}$ & O_3	$0.06 ($0.03—$0.1)	$0.00006 ($0.00003—$0.001)	$0.06 ($0.03—$0.1)
Hospital admissions—cardiovascular	$PM_{2.5}$	$0.2 ($0.1—$0.3)	$0.002 ($0.001—$0.003)	$0.2 ($0.1—$0.3)
Emergency room visits for asthma	$PM_{2.5}$ & O_3	$0.005 ($0.002—$0.008)	---	$0.005 ($0.002—$0.008)
Acute bronchitis	$PM_{2.5}$	$0.009 (-$0.0004—$0.03)	---	$0.009 (-$0.0004—$0.03)
Lower respiratory symptoms	$PM_{2.5}$	$0.005 ($0.002—$0.009)	---	$0.005 ($0.002—$0.009)
Upper respiratory symptoms	$PM_{2.5}$	$0.006 ($0.001—$0.014)	---	$0.006 ($0.001—$0.014)
Asthma exacerbation	$PM_{2.5}$	$0.012 ($0.001--$0.046)	---	$0.012 ($0.001--$0.046)
Lost work days	$PM_{2.5}$	$0.2 ($0.19—$0.24)	$0.002 ($0.0016--$0.0002)	$0.2 ($0.19—$0.24)
School loss days	O_3	$0.01 ($0.004—$0.013)	---	$0.01 ($0.004—$0.013)
Minor restricted-activity days	$PM_{2.5}$ & O_3	$0.64 ($0.34—$0.97)	$0.005 ($0.003—$0.008)	$0.64 ($0.34—$0.97)
Recreational visibility, Class I areas	$PM_{2.5}$	$3.5	$0.03	$3.6

Monetized total Benefits (Pope et al. 2002 $PM_{2.5}$ mortality and Bell et al. 2004 ozone mortality estimates)				
3% discount rate	$PM_{2.5}$, O_3	**\$120 (\$10—\$360)**	**\$1.1 (\$0.09—\$3.3)**	**\$120 (\$10—\$360)**
7% discount rate	$PM_{2.5}$, O_3	**\$110 (\$9—\$330)**	**\$0.9 (\$0.08—\$2.9)**	**\$110 (\$9—\$330)**
Monetized total Benefits (Laden et al. 2006 $PM_{2.5}$ mortality and Levy et al. 2005 ozone mortality estimates)				
3% discount rate	$PM_{2.5}$, O_3	**\$290 (\$26—\$840)**	**\$2.6 (\$0.2—\$7.5)**	**\$290 (\$26—\$840)**
7% discount rate	$PM_{2.5}$, O_3	**\$260 (\$23—\$760)**	**\$2.4 (\$0.2—\$6.8)**	**\$270 (\$24—\$760)**

[A] Estimates rounded to two significant figures.

[B] Monetary value of endpoints marked with dashes are < \$100,000. States included in transport region may be found in Chapter 2.

[C] The negative 5th percentile estimates for certain endpoints are the result of the weak statistical power of the study used to calculate these health impacts and do not suggest that increases in air pollution exposure result in decreased health impacts.

productivity. Although we are unable to monetize all welfare benefits, EPA estimates the proposed Transport Rule will yield welfare benefits of \$3.5 billion in 2014 (2006\$) for visibility improvements in southeastern Class I (national park) areas for a total of \$3.6 billion in benefits across southeastern, southwestern and California Class I areas.

Figure 1-1 summarizes an array of $PM_{2.5}$-related monetized benefits estimates based on alternative epidemiology and expert-derived PM-mortality estimate as well as the sum of ozone-related benefits using the Bell et al. (2004) mortality estimate.

Figure 1-2 summarizes the estimated net benefits for the proposed remedy by displaying all possible combinations of PM and ozone-related monetized benefits and costs. The graphic includes one estimate of ozone-related mortality and fourteen different $PM_{2.5}$ related mortality and a single 3% or 7% discounted cost estimate.[2] Each of the 14 bars in each graph represents a separate point estimate of net benefits under a certain combination of

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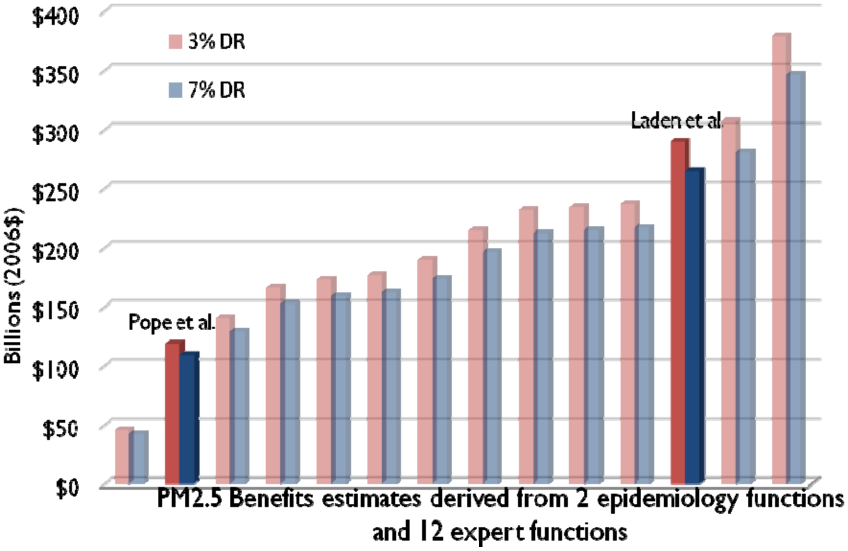
[2] Versions of this figure found in previous EPA RIA's have included the full suite of ozone mortality estimates. Because total benefits are relatively insensitive to the specification of ozone mortality estimate, for simplicity of presentation we have not included this full suite.

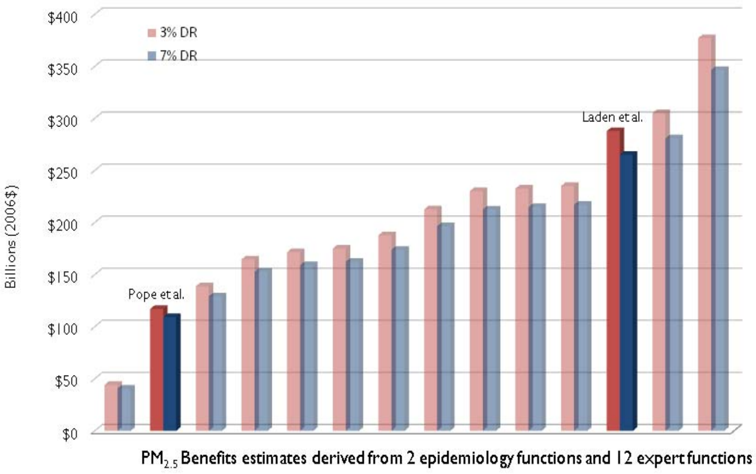
cost and benefit estimation methods. Because it is not a distribution, it is not possible to infer the likelihood of any single net benefit estimate.

Figure 1-1: Estimated Monetized Value of Estimated $PM_{2.5}$- Related Premature Mortalities Avoided According to Epidemiology or Expert-derived Derived PM Mortality Risk Estimate[A]



[A] Column total equals sum of $PM_{2.5}$-related mortality and morbidity benefits and ozone-related morbidity and mortality benefits using the Bell et al. (2004) mortality estimate.

Figure 1-2: Net Benefits of the Transport Rule According to $PM_{2.5}$ Epidemiology or Expert-derived Mortality Risk Estimate[A]



[A] Column total equals sum of $PM_{2.5}$-related mortality and morbidity benefits and ozone-related morbidity and mortality benefits using the Bell et al. (2004) mortality estimate.

1.1.3 Assessment of More and Less Stringent Scenarios

1.1.3.1 Assessment of Other Alternatives

EPA also analyzed the costs and benefits of the two alternative proposed remedies - direct control and intrastate trading programs. Finally, the Agency also considered options that were more and less stringent for the control of SO_2 emissions.

Air quality modeling was not conducted for these alternatives; thus we estimated the benefits of these alternatives by applying the same benefit per-ton approach as done for the alternative remedy options. The costs of these alternatives are estimated using IPM. Table 1-4 below presents the health-related benefits and social costs, including net social benefit, of the two scenarios alongside that of the proposed Transport Rule remedy

Table 1-4 provides the benefits of the direct control and intrastate trading remedies.

Table 1-4. Summary of Annual Benefits, Costs, and Net Benefits of Versions of the Proposed Remedy Option in 2014[a] (billions of 2006$)

Description	*Proposed Remedy- State Budgets/Limited Trading*	*Direct Control*	*Intrastate Trading*
Social costs[b]			
3 % discount rate	$2.03	$2.68	$2.49
7 % discount rate	$2.23	$2.91	$2.70
Health-related benefits[c,d]			
3 % discount rate	$118 to $288 + B	$117 to $286 + B	$113 to $276 + B
7 % discount rate	$108 to $260 + B	$108 to $262 + B	$104 to $252 + B
Net benefits (benefits-costs)[e]			
3 % discount rate	$116 to $286	$115 to $283	$110 to $273
7 % discount rate	$105 to $258	$105 to $259	$101 to $249

[a] When presenting benefits and net benefits, EPA traditionally rounds all estimates to two significant figures. In this case we have rounded to three significant digits to facilitate comparison of the benefits and costs among the proposed remedy and alternatives.

[b] The social costs are the loss of household utility as measured in Hicksian equivalent variation. More information on the social costs can be found in Chapter 8 of this RIA.

[c] Due to methodological limitations, the health benefits of the direct control and intrastate trading remedies include $PM_{2.5}$ –related benefits but omit visibility, ozone, and CO_2-related benefits. We present the $PM_{2.5}$ –related benefits of the proposed remedy, omitting these other important benefits, so that readers may compare directly the benefits of the proposed and alternate remedies. Total benefits are comprised of the value of PM-related avoided premature mortalities. The reduction in these premature mortalities in each year account for over 90 percent of total $PM_{2.5}$ –related monetized benefits. Benefits in this table are nationwide and are associated with NO_x and SO_2 reductions.

[d] Not all possible benefits or disbenefits are quantified and monetized in this analysis. Potential benefit categories that have not been quantified and monetized are listed in Table 1-6.

[e] Valuation assumes discounting over the SAB-recommended 20-year segmented lag structure. Results reflect the use of 3 percent and 7 percent discount rates consistent with EPA and OMB guidelines.

1.1.3.2 Alternatives that Are More or Less Stringent

In accordance with Circular A-4 guidance, EPA also analyzed the costs and benefits of two options that differed in their stringency from the preferred State Budgets/Limited Trading option – one less stringent, the other more stringent. Both options have the same 2012 requirements and varied in the requirements for SO_2 emissions reductions in 2014.

Unlike the preferred TR option which requires greater SO_2 reductions in 15 states (Group 1) beginning in 2014 from 2012 emissions levels, the less stringent option maintains the 2012 requirements in all subsequent years. This option allows about 1.4 million tons more SO_2 to be emitted annually than the preferred approach after 2013.

The more stringent options changes the requirement for Group 1 state SO_2 emissions reductions beginning in 2014 and moves 8 additional states to Group 1 from Group 2. Connecticut, Florida, Kansas, Maryland, Massachusetts, Minnesota, Nebraska, and New

Jersey join the 15 Group 1 states of the proposed rule, making 23 states in all and leaving 4 states and the District of Columbia in Group 2. Also, an additional 200,000 tons of SO_2 reduction is required in the 23 Group 1 states.

Table 1-5. Summary of Annual Benefits, Costs, and Net Benefits of Versions of the Proposed Remedy Option in 2014[a] (billions of 2006 dollars)

Description	*Preferred Remedy-State Budgets/Limited Trading*	*Less Stringent Scenario*	*More Stringent Scenario*
Social costs[b]			
3 % discount rate	\$2.03	\$1.12*	\$2.21*
7 % discount rate	\$2.23	\$1.23*	\$2.43*
Health-related benefits[c,d]			
3 % discount rate	\$118 to \$288	\$82 to 200	\$120 to 292
7 % discount rate	\$108 to \$262	\$76 to 184	\$110 to 267
Net benefits (benefits-costs)[e]			
3 % discount rate	\$116 to \$288	\$81 to 200	\$117 to 290
7 % discount rate	\$105 to \$260	\$74 to 182	\$107 to 264

[a] When presenting benefits and net benefits, EPA traditionally rounds all estimates to two significant figures. In this case we have rounded to three significant digits to facilitate comparison of the benefits and costs among the proposed remedy and the less and more stringent scenarios.

[b] The social costs are the loss of household utility as measured in Hicksian equivalent variation. More information on the social costs can be found in Chapter 8 of this RIA.

[c] Due to methodological limitations, the health benefits of the direct control and intrastate trading remedies include $PM_{2.5}$ –related benefits but omit visibility, ozone, and CO_2-related benefits. We present the $PM_{2.5}$ –related benefits of the proposed remedy, omitting these other important benefits, so that readers may compare directly the benefits of the proposed and alternate remedies. Total benefits are primarily of the value of PM-related avoided premature mortalities. The reduction in these premature mortalities in each year account for over 90 percent of total $PM_{2.5}$ –related monetized benefits. Benefits in this table are nationwide and are associated with NO_x and SO_2 reductions. Visibility and ozone-related benefits not calculated for the more and less stringent scenarios because these impacts were estimated using $PM_{2.5}$-related benefit per ton estimates.

[d] Not all possible benefits or disbenefits are monetized in this analysis. These are listed in Table 1-6.

[e] Valuation assumes discounting over the SAB-recommended 20-year segmented lag structure. Results reflect the use of 3 % and 7 % discount rates consistent with EPA and OMB guidelines.

1.2 Not All Benefits Quantified

EPA was unable to quantify or monetize all of the health and environmental benefits associated with the proposed Transport Rule. EPA believes these unquantified benefits are substantial, including the value of increased agricultural crop and commercial forest yields, visibility improvements, reductions in nitrogen and acid deposition and the resulting changes in ecosystem functions, and health and welfare benefits associated with reduced mercury emissions. Table 1-6 provides a list of these benefits.

Table 1-6: Human Health and Welfare Effects of Pollutants Affected by the Transport Rule

Pollutant/ Effect	*Quantified and monetized in base estimate*	*Unquantified*
PM: health[a]	Premature mortality based on cohort study estimates[b] Premature mortality based on expert elicitation estimates Hospital admissions: respiratory and cardiovascular Emergency room visits for asthma Nonfatal heart attacks (myocardial infarctions) Lower and upper respiratory illness Minor restricted activity days **Work loss days** Asthma exacerbations (among asthmatic populations Respiratory symptoms (among asthmatic populations) Infant mortality	Low birth weight Pulmonary function Chronic respiratory diseases other than chronic bronchitis Non-asthma respiratory emergency room visits UVb exposure (+/-)[c]
PM: welfare	Visibility in Class I areas	Household soiling Visibility in residential and non-class I areas UVb exposure (+/-)[c] Global climate impacts[c]
Ozone: health	Premature mortality based on short-term study estimates Hospital admissions: respiratory Emergency room visits for asthma Minor restricted activity days School loss days	Chronic respiratory damage Premature aging of the lungs Non-asthma respiratory emergency room visits UVb exposure (+/-)[c]
Ozone: welfare	Decreased outdoor worker productivity	Yields for: --Commercial forests --Fruits and vegetables, and --Other commercial and noncommercial crops Damage to urban ornamental plants Recreational demand from damaged forest aesthetics Ecosystem functions UVb exposure (+/-)[c]
NO_2: health		Respiratory hospital admissions Respiratory emergency department visits Asthma exacerbation Acute respiratory symptoms Premature mortality Pulmonary function
NO_2: welfare		Commercial fishing and forestry from acidic deposition Commercial fishing, agriculture and forestry from nutrient deposition Recreation in terrestrial and estuarine ecosystems from nutrient deposition Other ecosystem services and existence values

	for currently healthy ecosystems Coastal eutrophication from nitrogen deposition
SO_2: health	Respiratory hospital admissions Asthma emergency room visits Asthma exacerbation Acute respiratory symptoms Premature mortality Pulmonary function
SO_2: welfare	Commercial fishing and forestry from acidic deposition Recreation in terrestrial and aquatic ecosystems from acid deposition Increased mercury methylation
Mercury: health	Incidence of neurological disorders Incidence of learning disabilities Incidences in developmental delays Potential cardiovascular effects including: --Altered blood pressure regulation --Increased heart rate variability --Incidences of heart attack Potential reproductive effects
Mercury: environment	Impact on birds and mammals (e.g. reproductive effects)
Mercury: welfare	Impacts to commercial., subsistence and recreational fishing

[a] In addition to primary economic endpoints, there are a number of biological responses that have been associated with PM health effects including morphological changes and altered host defense mechanisms. The public health impact of these biological responses may be partly represented by our quantified endpoints.

[b] Cohort estimates are designed to examine the effects of long term exposures to ambient pollution, but relative risk estimates may also incorporate some effects due to shorter term exposures (see Kunzli et al., 2001 for a discussion of this issue). While some of the effects of short term exposure are likely to be captured by the cohort estimates, there may be additional premature mortality from short term PM exposure not captured in the cohort estimates included in the primary analysis.

[c] May result in benefits or disbenefits.

1.3 Costs and Economic Impacts

For the affected region, the projected annual incremental private costs of the proposed remedy option (intrastate trading with some interstate trading) to the power industry are \$3.7 billion in 2012 and \$2.8 billion in 2014. Costs are lower in 2014 than in 2012 as the rule becomes more stringent because there are larger amounts of State and Federally enforceable controls that happen between 2012 and 2014 in the baseline. There are two other remedy options that EPA examined as part of our analyses. A remedy option that relies solely on intrastate trading has projected annual incremental private costs of \$4.2 billion in 2012 and \$2.7 billion in 2014. Finally, a remedy option that applies controls directly to affected units with no trading (direct control remedy) yields projected annual incremental private costs of \$4.3 billion in 2012 and \$3.4 billion in 2014. These costs represent the total cost to the electricity-generating industry of reducing NO_x and SO_2 emissions to meet the emissions caps set out in the rule. Estimates are in 2006 dollars. These costs of the rule are estimated using the Integrated Planning Model (IPM).

In estimating the net benefits of regulation above, the appropriate cost measure is "social costs." Social costs represent the welfare costs of the rule measured as the loss of consumer utility in the macroeconomic analysis of this rule proposal.

There are several national changes in energy prices that result from the Transport Rule. Retail electricity prices are projected to increase nationally by an average of 2.5 % in 2012 and 1.5 % in 2014 with the proposed Transport Rule. The effects of the proposed rule on natural gas prices and the power-sector generation mix is also small, with a 1.7 percent or less increase in delivered gas prices projected in 2012 and 0.5 % in 2014.

There are several other types of energy impacts from the Transport Rule. A relatively small amount of coal-fired capacity, about 1.2 GW (0.3 percent of all coal-fired capacity and 0.1 % of all generating capacity), is projected to be uneconomic to maintain. In practice units projected to be uneconomic to maintain may be "mothballed," retired, or kept in service to ensure transmission reliability in certain parts of the grid. For the most part, these units are small and infrequently used generating units that are dispersed throughout the proposed Transport Rule region. Coal production for use in the power sector is projected to decrease by 0.3 % in 2012 and by 0.8 % by 2014, and we expect greater coal production in Appalachia and the West and 15 % less production in the Interior coal regions of the country with the proposed Transport Rule.

In 2014, EPA estimates that Gross Domestic Product (GDP) and consumption levels are approximately 0.01 % lower ($1.6 billion) with the proposed Transport Rule. There are declines of less than 0.05 % in GDP by region except for the Plains and West, where regional GDP increases as productive activities shift to these less regulated regions. Overall, the impacts of the proposed rule are modest, particularly in light of the large projected benefits mentioned earlier.

1.4 Small Entity and Unfunded Mandates Impacts

After preparing an analysis of small entity impacts, EPA has certified that this proposal will have no SISNOSE (significant economic impacts on a substantial number of small entities). First, of the 30 small entities (out of 81 affected) projected to have costs greater than 1 percent of revenues, around 75 percent of them operate in cost of service regions and would generally be able to pass any increased costs along to rate-payers. Furthermore, of the approximately 550 units identified by EPA as being potentially owned by small entities, approximately two-thirds of the units that have higher costs are not expected to make operational changes as a result of this rule (e.g. install control equipment or switch fuels). Their increased costs are largely due to increased cost of the fuel they would be expected to use whether or not they had to comply with the proposed rule. Further, increased fuel costs are often passed through to rate-payers as common practice in many areas of the U.S. due to fuel adder arrangements instituted by state public utility commissions. Finally, EPA's decision to exclude units smaller than 25 Megawatt capacity (MW) has already significantly reduced the burden on small entities.

EPA examined the potential economic impacts on state and municipality-owned entities associated with this rulemaking based on assumptions of how the affected states will implement control measures to meet their emissions. These impacts have been calculated to provide additional understanding of the nature of potential impacts and additional information.

According to EPA's analysis, of the 84 government entities considered in this analysis and the 482 government entities in the Transport Rule region that are included in EPA's modeling, 27 may experience compliance costs in excess of 1 percent of revenues in 2014, based on our assumptions of how the affected states implement control measures to meet their emissions budgets as set forth in this rulemaking.

Government entities projected to experience compliance costs in excess of 1 percent of revenues may have some potential for significant impact resulting from implementation of the Transport Rule. However, it is EPA's position that because these government entities can pass on their costs of compliance to rate-payers, they will not be significantly affected. Furthermore, the decision to include only units greater than 25 MW in size exempts 380 government entities that would otherwise be potentially affected by the Transport Rule.

1.5 Limitations and Uncertainties

Every analysis examining the potential benefits and costs of a change in environmental protection requirements is limited to some extent by data gaps, limitations in model capabilities (such as geographic coverage), and variability or uncertainties in the underlying scientific and economic studies used to configure the benefit and cost models. Despite the uncertainties, we believe this benefit-cost analysis provides a reasonable indication of the expected economic benefits and costs of the proposed Transport Rule.

For this analysis, such uncertainties include possible errors in measurement and projection for variables such as population growth and baseline incidence rates; uncertainties associated with estimates of future-year emissions inventories and air quality; variability in the estimated relationships between changes in pollutant concentrations and the resulting changes in health and welfare effects; and uncertainties in exposure estimation.

EPA's cost estimates assume that all states in the proposed Transport Rule region participate in the programs that reduce SO_2 and NO_x emissions from the power industry.

Below is a summary of the key uncertainties of the analysis:

Costs

- Analysis does not capture employment shifts as workers are retrained at the same company or re-employed elsewhere in the economy.
- We do not include the costs of certain relatively small permitting costs associated with Title V that new program entrants face.
- Technological innovation is not incorporated into these cost estimates.

- Economic impacts do not take into response of electric power consumers to changes in electricity prices. While this response is likely to be of small magnitude, it may have some impact on the final estimate of private compliance costs.

Benefits

- Most of the estimated PM-related benefits in this rule accrue to populations exposed to higher levels of $PM_{2.5}$. Of these estimated PM-related mortalities avoided, about 80% occur among populations initially exposed to annual mean $PM_{2.5}$ level of 10 $\mu g/m^3$ and about 97% occur among those initially exposed to annual mean $PM_{2.5}$ level of 7.5 $\mu g/m^3$; these are the lowest air quality levels considered in the Laden et al. (2006) and Pope et al. (2002) studies, respectively. This fact is important, because as we estimate PM-related mortality among populations exposed to levels of $PM_{2.5}$ that are successively lower, our confidence in the results diminishes. However, our analysis shows that the great majority of the impacts occur at higher exposures.
- There are uncertainties related to the health impact functions used in the analysis. These include: within study variability; across study variation; the application of concentration-response (C-R) functions nationwide; extrapolation of impact functions across population; and various uncertainties in the C-R function, including causality and thresholds. Therefore, benefits may be under- or over-estimates.
- Analysis is for 2014, and projecting key variables introduces uncertainty. Inherent in any analysis of future regulatory programs are uncertainties in projecting atmospheric conditions and source level emissions, as well as population, health baselines, incomes, technology, and other factors.
- This analysis omits certain unquantified effects due to lack of data, time and resources. These unquantified endpoints include other health and ecosystem effects. EPA will continue to evaluate new methods and models and select those most appropriate for estimating the benefits of reductions in air pollution. Enhanced collaboration between air quality modelers, epidemiologists, toxicologists, ecologists, and economists should result in a more tightly integrated analytical framework for measuring benefits of air pollution policies.
- $PM_{2.5}$ mortality benefits represent a substantial proportion of total monetized benefits (over 90%), and these estimates have following key assumptions and uncertainties.
    1. The $PM_{2.5}$ -related benefits of the alternative scenarios were derived through a benefit per-ton approach, which does not fully 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 SO_2.
    2. We assume that all fine particles, regardless of their chemical composition, are equally potent in causing premature mortality. This is an important assumption, because $PM_{2.5}$ produced via transported precursors emitted from EGUs may differ significantly from direct $PM_{2.5}$ released from diesel engines and other industrial

sources, but no clear scientific grounds exist for supporting differential effects estimates by particle type.

3. We assume that the health impact function for fine particles is linear within the range of ambient concentrations under consideration. Thus, the estimates include health benefits from reducing fine particles in areas with varied concentrations of $PM_{2.5,}$ including both regions that are in attainment with fine particle standard and those that do not meet the standard down to the lowest modeled concentrations.
4. To characterize the uncertainty in the relationship between $PM_{2.5}$ and premature mortality, we include a set of twelve estimates based on results of the expert elicitation study in addition to our core estimates. Even these multiple characterizations omit the uncertainty in air quality estimates, baseline incidence rates, populations exposed and transferability of the effect estimate to diverse locations. As a result, the reported confidence intervals and range of estimates give an incomplete picture about the overall uncertainty in the $PM_{2.5}$ estimates. This information should be interpreted within the context of the larger uncertainty surrounding the entire analysis.

1.6 References

Bell, M.L., et al. 2004. Ozone and short-term mortality in 95 US urban communities, 1987-2000. Journal of the American Medical Association. 292(19): p. 2372-8.

Bell, M.L., F. Dominici, and J.M. Samet. 2005. A meta-analysis of time-series studies of ozone and mortality with comparison to the national morbidity, mortality, and air pollution study. Epidemiology. 16(4): p. 436-45.

Ito, K., S.F. De Leon, and M. Lippmann. 2005. Associations between ozone and daily mortality: analysis and meta-analysis. Epidemiology. 16(4): p. 446-57.

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.

Levy JI, Baxter LK, Schwartz J. 2009. Uncertainty and variability in health-related damages from coal-fired power plants in the United States. Risk Anal. doi: 10.1111/j.1539-6924.2009.01227.x [Online 9 Apr 2009]

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.

U.S. Environmental Protection Agency (EPA). September 2000. *Guidelines for Preparing Economic Analyses.* EPA 240-R-00-003.

U.S. Office of Management and Budget (OMB). 2003. Circular A-4 Guidance to Federal Agencies on Preparation of Regulatory Analysis.

Woodruff, T.J., J. Grillo, and K.C. Schoendorf. 1997. "The Relationship Between Selected Causes of Postneonatal Infant Mortality and Particulate Infant Mortality and Particulate Air Pollution in the United States." *Environmental Health Perspectives* 105(6):608-612.

CHAPTER 2

INTRODUCTION AND BACKGROUND

2.1 Introduction

EPA is proposing actions to address the interstate transport of emissions of nitrogen oxides (NO_x) and sulfur dioxide (SO_2) that contribute significantly to nonattainment and maintenance problems with respect to the national ambient air quality standards (NAAQS) for fine particulate matter ($PM_{2.5}$) that EPA promulgated in 1997 and 2006 and for 8-hour ozone that were promulgated in 1997. In this action, EPA is proposing to both identify and eliminate emissions within states in the eastern United States that significantly contribute to nonattainment and interfere with maintenance of the ozone and $PM_{2.5}$ NAAQS in other downwind states. This document presents the health and welfare benefits of the proposed Transport Rule and compares the benefits of this rule to the estimated costs of implementing the rule in 2012 and 2014. This chapter contains background information relative to the rule and an outline of the chapters of the report.

2.2 Background

Clean Air Act (CAA) section 110(a)(2)(D)(i)(I) requires states to prohibit emissions that contribute significantly to nonattainment in, or interfere with maintenance by, any other state with respect to the National Ambient Air Quality Standards (NAAQS). In this proposed rule, the Environmental Protection Agency (EPA) is proposing actions to partially or fully address the interstate transport of emissions of sulfur dioxide (SO_2), nitrogen oxides (NO_x), and the fine particulate that they form in the atmosphere, that contribute significantly to nonattainment and interfere with maintenance with respect to the fine particulate matter ($PM_{2.5}$) NAAQS promulgated in 1997 and 2006. EPA is also proposing actions to partially or fully address the interstate transport of NO_x and the ozone that it forms in the atmosphere that contribute significantly to nonattainment and interfere with maintenance with respect to the 8-hour ozone NAAQS promulgated in 1997.

With this proposal, EPA is responding to the remand of the Clean Air Interstate Rule (CAIR) by the U.S. Court of Appeals for the D.C. Circuit in 2008. CAIR, promulgated May

12, 2005 (70 FR 25162) and the CAIR federal implementation plans (FIPs), promulgated April 26, 2006 (71 FR 25328), aimed to address the interstate transport of pollutants that contributed significantly to downwind nonattainment of the $PM_{2.5}$ and 8-hour ozone NAAQS promulgated in July 1997. In July 2008, the D.C. Circuit Court found CAIR and the CAIR FIPs unlawful. *North Carolina v. EPA*, 531 F.3d 896 (D.C. Cir. 2008). The Court's original decision vacated CAIR. Id. at 929-30. However, the Court subsequently remanded CAIR to EPA without vacatur because it found that "allowing CAIR to remain in effect until it is replaced by a rule consistent with our opinion would at least temporarily preserve the environmental values covered by CAIR." *North Carolina v. EPA*, 550 F.3d 1176, 1178 (D.C. Cir. 2008).

2.2.1 Methodology for Identifying Needed Reductions

As described in section IV of the preamble for this proposed rule, EPA is proposing a state-specific methodology to identify specific reductions that states in the eastern United States must make to satisfy the CAA section 110(a)(2)(D)(i)(I) prohibition on emissions that significantly contribute to nonattainment or interfere with maintenance in a downwind state. To facilitate implementation of the requirement that significant contribution and interference with maintenance be eliminated, EPA developed state emissions budgets. These are new emissions budgets which are based on the Agency's state-by-state analysis of each upwind state's significant contribution to nonattainment and interference with maintenance downwind. A state's emissions budget is the quantity of emissions that would remain after elimination of significant contribution and interference with maintenance in an average year, assuming no abnormal meteorology or disruptions in electricity supply. EPA proposes SO_2 and NO_x budgets for each state covered for the 24-hour and/or annual $PM_{2.5}$ NAAQS. EPA also proposes an ozone season[3] NO_x budget for each state covered for the 8-hour ozone NAAQS.

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[3] Consistent with the approach taken by the Ozone Transport Assessment Group (OTAG), the NO_X SIP call, and the CAIR, we propose to define the ozone season, for purposes of emissions reduction requirements in this rule, as May through September. We recognize that this ozone season for regulatory requirements will have differences from the official state-specific monitoring season.

2.2.2 How Reductions Will Be Achieved, and Different Options to Do So

EPA is proposing federal implementation plans (FIPs) to immediately implement the emissions reduction requirements. The FIPs would regulate electric generating units (EGUs) in the 32 covered states. EPA is proposing to regulate these sources through a program that uses state-specific budgets and allows intrastate and limited interstate trading. EPA is also taking comment on two alternative regulatory options. All three options would achieve the emission reductions necessary to address the emissions transport requirements in section 110(a)(2)(D)(i)(I) of the Clean Air Act.

2.2.3 States Covered by the Proposed Rule

In this action, EPA proposes SO_2 and NO_x emissions controls in the following 26 jurisdictions that contribute significantly to nonattainment in, or interfere with maintenance by, a downwind area with respect to the 24-hour $PM_{2.5}$ NAAQS promulgated in September 2006: Alabama, Connecticut, Delaware, District of Columbia, Georgia, Illinois, Indiana, Iowa, Kansas, Kentucky, Massachusetts, Maryland, Michigan, Minnesota, Missouri, Nebraska, New Jersey, New York, North Carolina, Ohio, Pennsylvania, Tennessee, Virginia, West Virginia, and Wisconsin.

EPA proposes SO_2 and NO_x emissions controls in the following 24 jurisdictions that contribute significantly to nonattainment in, or interfere with maintenance by, a downwind area with respect to the annual $PM_{2.5}$ NAAQS promulgated in July 1997: Alabama, Delaware, District of Columbia, Florida, Georgia, Illinois, Indiana, Iowa, Kentucky, Louisiana, Maryland, Michigan, Minnesota, Missouri, New Jersey, New York, North Carolina, Ohio, Pennsylvania, South Carolina, Tennessee, Virginia, West Virginia, and Wisconsin.

EPA also proposes ozone season NO_x emissions controls in the following 26 jurisdictions that contribute significantly to nonattainment in, or interfere with maintenance by, a downwind area with respect to the 8-hour ozone NAAQS promulgated in July 1997: Alabama, Arkansas, Connecticut, Delaware, District of Columbia, Florida, Georgia, Illinois, Indiana, Kansas, Kentucky, Louisiana, Maryland, Michigan, Mississippi, New Jersey, New York, North Carolina, Ohio, Oklahoma, Pennsylvania, South Carolina, Tennessee, Texas, Virginia, and West Virginia.

As discussed above, EPA is proposing FIPs to directly regulate EGU SO_2 and/or NO_x emissions in the 32 covered states. The proposed FIPs would require the 28 jurisdictions covered for purposes of the 24-hour and/or annual $PM_{2.5}$ NAAQS to reduce SO_2 and NO_x emissions by specified amounts. The proposed FIPs would require the 26 states covered for purposes of the 8-hour ozone NAAQS to reduce ozone season NO_x emissions by specified amounts. For the $PM_{2.5}$ NAAQS, EPA proposes two phases with an initial phase in 2012 and subsequent phase in 2012. For 8-hour ozone, EPA proposes a single phase that would start in 2012.

As discussed in detail in section IV of the preamble, the proposed approach to significant contribution and interference with maintenance would group the 28 states covered for 24-hour and/or annual $PM_{2.5}$ NAAQS in two tiers reflecting the stringency of SO_2 reductions required to eliminate that state's significant contribution and interference with maintenance. There would be a stringent SO_2 tier comprising 15 states ("group 1") and a moderate SO_2 tier comprising 13 states ("group 2") with uniform stringency within each tier.[4] For these same 28 states, there would be one annual NO_x tier with uniform stringency of NO_x reductions across all 28 states. Similarly, for the 26 states covered for the 8-hour ozone NAAQS there would be one ozone season NO_x tier with uniform stringency across all 26 states.

The proposed stringent SO_2 tier ("group 1") would include Georgia, Illinois, Indiana, Iowa, Kentucky, Michigan, Missouri, New York, North Carolina, Ohio, Pennsylvania, Tennessee, Virginia, West Virginia, and Wisconsin. The proposed moderate SO_2 tier ("group 2") would include Alabama, Connecticut, Delaware, District of Columbia, Florida, Kansas, Louisiana, Maryland, Massachusetts, Minnesota, Nebraska, New Jersey, and South Carolina.

For the 15 states in the stringent SO_2 tier ("group 1"), the 2014 phase would substantially increase the SO_2 reduction requirements (i.e., these states would have smaller SO_2 emissions budgets starting in 2014), reflecting the greater reductions needed to eliminate significant contribution and interference with maintenance from these states with respect to

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[4] With regard to interstate trading, the two SO_2 stringency tiers lead to two exclusive SO_2 trading groups.

the 24-hour $PM_{2.5}$ NAAQS. For the 13 states in the moderate SO_2 tier ("group 2") the 2014 SO_2 emissions budgets would remain the same as the 2012 SO_2 budgets for these states.

The 2014 annual NO_x emissions budgets for all 28 states covered for the 24-hour and/or annual $PM_{2.5}$ NAAQS would remain the same as the 2012 annual NO_x budgets. See Table 2-1 for proposed lists of covered states.

Table 2-1 -- Lists of Covered States for $PM_{2.5}$ and 8-Hour Ozone NAAQS

State	Covered for 24-hour and/or annual $PM_{2.5}$	Covered for 8-hour ozone
	Required to reduce SO_2 and NO_x	Required to reduce ozone season NO_x
Alabama	X	X
Arkansas		X
Connecticut	X	X
Delaware	X	X
District of Columbia	X	X
Florida	X	X
Georgia	X	X
Illinois	X	X
Indiana	X	X
Iowa	X	
Kansas	X	X
Kentucky	X	X
Louisiana	X	X
Maryland	X	X
Massachusetts	X	
Michigan	X	X

Minnesota	X	
Mississippi		X
Missouri	X	
Nebraska	X	
New Jersey	X	X
New York	X	X
North Carolina	X	X
Ohio	X	X
Oklahoma		X
Pennsylvania	X	X
South Carolina	X	X
Tennessee	X	X
Texas		X
Virginia	X	X
West Virginia	X	X
Wisconsin	X	
TOTALS	28	26

The relevant regions for $PM_{2.5}$ and ozone significant contribution are also depicted in the graphic in Figures 2-1 and 2-2, respectively. Maps are also available in Chapter 7 of this RIA.

Figure 2-1 - PM$_{2.5}$ Region (SO$_2$ and Annual NO$_x$ States) Under the Proposed Transport Rule

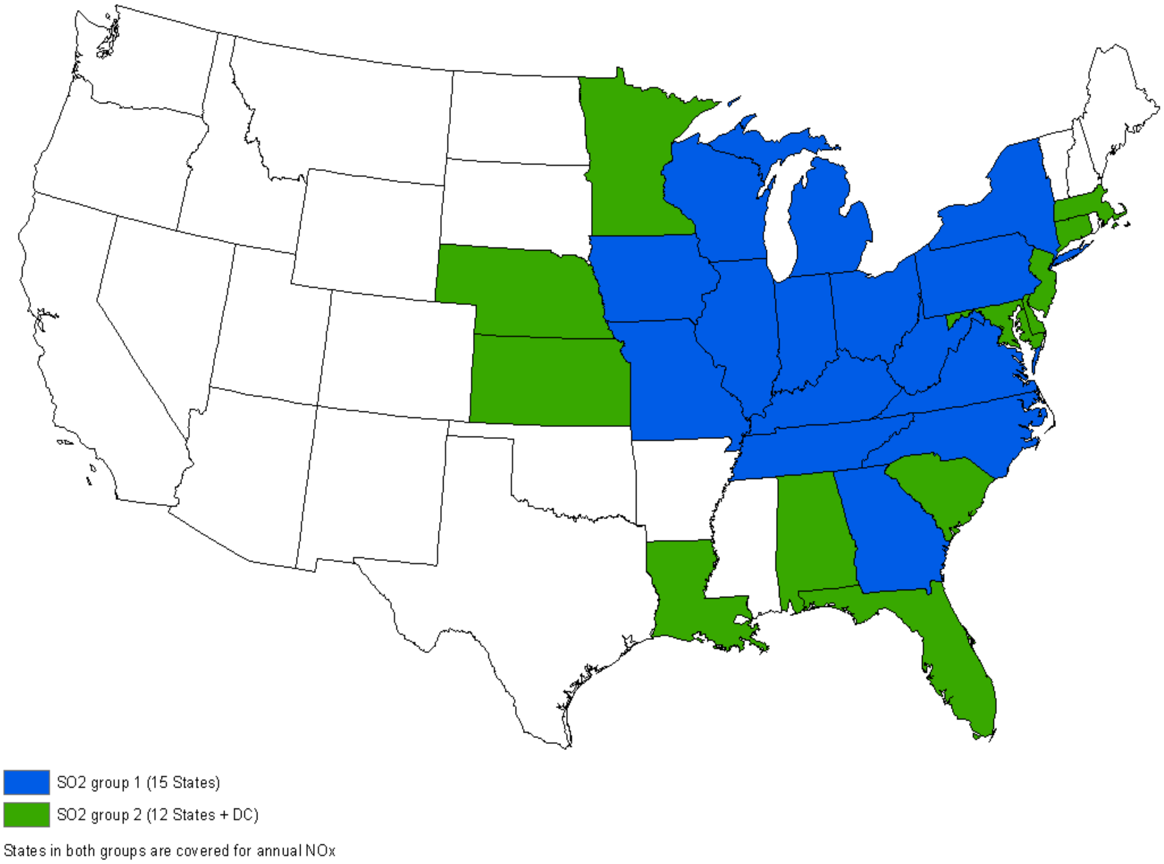
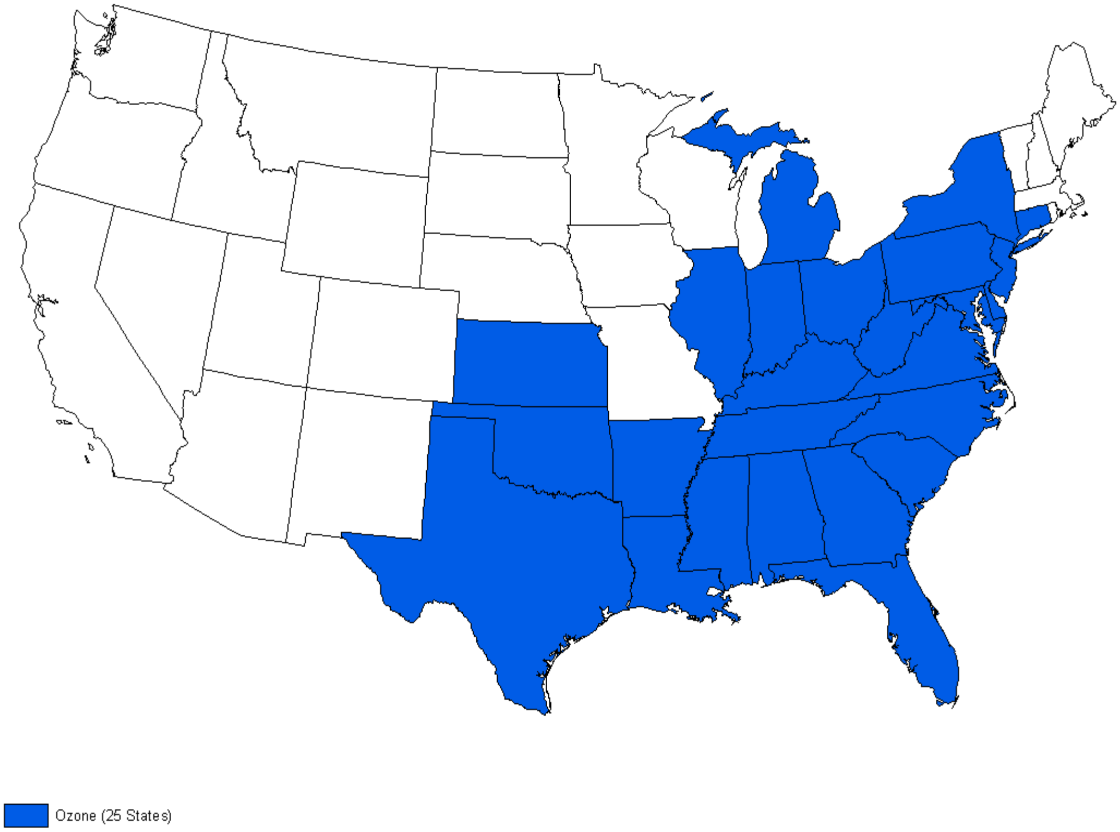


Figure 2-2. Transport Rule Ozone-season NO_x States



2.3 Regulated Entities

This action proposes to directly regulate emissions of NO_x and SO_2, from electric generating units (EGUs) with capacity greater than 25 MW in the covered states.

2.4 Baseline and Years of Analysis

The proposed rule on which this analysis is based sets forth the requirements for states to address their significant contribution to downwind nonattainment of ozone and $PM_{2.5}$ NAAQS and interference with maintenance. To address this significant contribution and interference with maintenance, EPA requires that certain states reduce their emissions of SO_2 and NO_x. The Agency considered all promulgated CAA requirements and known state actions in the baseline used to develop the estimates of benefits and costs for this rule. This baseline analysis takes into account emissions reductions associated with the implementation of all federal rules promulgated by December 2008 and assumes that the CAIR is not in effect. However, this baseline presents a unique situation. EPA has been directed to replace the CAIR; yet the CAIR remains in place and has led to significant emissions reductions in many states.

A key step in the process of developing a 110(a)(2)(D)(i)(I) rule involves analyzing existing (base case) emissions to determine which states significantly contribute to downwind nonattainment and maintenance areas. EPA cannot prejudge at this stage which states will be affected by the rule. For example, a state affected by CAIR may not be affected by the new rule and after the new rule goes into effect, the CAIR requirements will no longer apply. For a state covered by CAIR but not covered by the new rule, the CAIR requirements would not be replaced with new requirements, and therefore an increase in emissions relative to present levels could occur in that state. More fundamentally, the court has made clear that, due to legal flaws, the CAIR rule cannot remain in place and must be replaced. If EPA's base case analysis were to ignore this fact and assume that reductions from CAIR would continue indefinitely, areas that are in attainment solely due to controls required by CAIR would again face nonattainment problems, because the existing protection from upwind pollution would not be replaced. For these reasons, EPA cannot assume in its base case analysis, that the reductions required by CAIR will continue to be achieved.

Following this logic, the 2012 base case shows emissions higher than current levels in some states. Because EPA has been directed to replace CAIR, EPA believes that for many states, the absence of the CAIR NO_x program will lead to the status quo of the NO_x Budget Program, which limits ozone-season NO_X emissions and ensures the operation of NO_X controls in those states. Also, without the CAIR SO_2 program, emission requirements in many areas would revert to the comparatively less stringent requirements of the Title IV Acid Rain program. As a result, SO_2 emissions in many states would increase markedly in the 2012 base case relative to the present. Efforts to comply with ARP rules at the least-cost would occur in many cases without the operation of existing scrubbers through use of readily available, inexpensive Title IV allowances. Notably, all known controls that are required under state laws, NSPS, consent decrees, and other enforceable binding commitments through 2014 are accounted for in the base case. It is against this backdrop that the Transport Rule is analyzed and that significant contribution to nonattainment and interference with maintenance must be addressed.

The model's base case features an updated Title IV SO_2 allowance bank assumption and incorporates updates related to the Energy Independence and Security Act of 2007. Many key assumptions, notably demand for electricity, reflect the 2008 Annual Energy Outlook from the Energy Information Administration (EIA).[5] In addition, the model includes policies affecting the power sector: the Title IV of the Clean Air Act (the Acid Rain Program); the NO_x SIP Call; various New Source Review (NSR) settlements[6]; and several state rules[7] affecting emissions of SO_2 and NO_x that were finalized through February 3, 2009. IPM includes state rules that have been finalized and/or approved by a state's

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[5] For the final rule, EPA anticipates using an updated version of IPM that will reflect assumptions from AEO 2010. Key differences will include lower assumptions about future electric demand and higher capital costs accounting for ARRA.

[6] The NSR settlements include agreements between EPA and Southern Indiana Gas and Electric Company (Vectren), Public Service Enterprise Group, Tampa Electric Company, We Energies (WEPCO), Virginia Electric & Power Company (Dominion), Santee Cooper, Minnkota Power Coop, American Electric Power (AEP), East Kentucky Power Cooperative (EKPC), Nevada Power Company, Illinois Power, Mirant, Ohio Edison, and Kentucky Utilities.

[7] These include current and future state programs in Connecticut, Delaware, Georgia, Illinois, Maine, Maryland, Massachusetts, Minnesota, Missouri, New Hampshire, North Carolina, New Jersey, New York, Oregon, Texas, and Wisconsin.

legislature or environmental agency.

The years 2012 and 2014 are the compliance years for the proposed rule, though as we explain in Chapters 5 and 7 we use 2015 as a proxy for compliance in 2014 for our benefits and economic impact analysis due to availability of modeling impacts in that year. We include analyses results for each year, but we do not include benefits and economic impact estimates for 2012 due to time constraints. All estimates presented in this report represent annualized estimates of the benefits and costs of the Transport Rule in 2012 and 2015 rather than the net present value of a stream of benefits and costs in these particular years of analysis.

2.5 Control Scenarios

The option EPA is proposing for the FIPs ("State Budgets/Limited Trading") would utilize state-specific control budgets and allow for intrastate and limited interstate trading. This approach would assure environmental results while providing some limited flexibility for covered sources. The approach would also facilitate the transition from CAIR to the Transport Rule for implementing agencies and covered sources.

The first alternative remedy option for which EPA requests comment would use state-specific emissions budgets and allow intrastate trading, but prohibit interstate trading. The second alternative remedy option, for which EPA also requests comment, would use state-specific budgets and emission rate limits.

The main difference between the three remedies lies in the kinds of flexibility they provide for compliance. State Budgets/Limited Trading allows sources to trade within state lines and, as long as state emissions remain within the budgets plus variability limits, across state lines as well. State Budgets/Intrastate Trading caps each state’s emissions at its budget without variability and only allows trading within (and not between) states. Under Direct Control, each EGU must meet an emission rate limit (company-wide within state averaging allowed), and a state’s emissions must remain within its budget plus variability. Further details on each of these remedies can be found in Section V. of the Transport Rule preamble.

The proposed remedy option and the first alternative, both of which are cap-and-trade approaches, would use new allowance allocations developed on a different basis from CAIR.

Fossil-fuel electric generating units (EGUs) over 25 megawatt (MW) capacity within the proposed Transport Rule region would be covered by this action.

2.6 Benefits of Emission Controls

The benefits of the proposed Transport Rule are discussed in Chapter 5 of this report. Annual monetized benefits of \$120 to 290 billion (3 percent discount rate) or \$110 to 270 billion (7 percent discount rate) are expected for the proposed rule in 2014.

2.7 Cost of Emission Controls

EPA analyzed the costs to private industry of the proposed Transport Rule using the Integrated Planning Model (IPM). EPA has used this model in the past to analyze the impacts of regulations on the power sector and used an earlier version of this model to analyze the impacts of the CAIR rule. The social cost is measured using Hicksian equivalent variation and estimated using the EMPAX CGE model. IPM results are incorporated into the EMPAX model when calculating the social cost of the Transport Rule. EPA estimates the private industry costs of the rule to the power sector to be \$3.7 billion in 2012 and \$2.8 billion in 2014 (2006 dollars). In estimating the net benefits (benefits – costs) of the rule, EPA uses social costs of the rule that represent the costs to society of this rule. These social costs include to the impact to industries affected by changes to electricity prices resulting from implementation of the proposed Transport Rule. The social costs of the rule are estimated by the EMPAX model to be \$2.0 or \$2.2 billion in 2015 (at 3 percent and 7 percent discount rates, respectively). A description of the methodology used to model the costs and economic impacts to the power sector is discussed in Chapter 7 of this report, and a description of the methodology used to estimate the social cost of the rule and model economic impacts outside of the power sector is discussed in Chapter 8 of this report.

2.8 Organization of the Regulatory Impact Analysis

This report presents EPA’s analysis of the benefits, costs, and other economic effects of the proposed Transport Rule to fulfill the requirements of a Regulatory Impact Analysis (RIA). This RIA includes the following chapters:

- Chapter 3, Emissions Impacts, describes the emission inventories and modeling that are essential inputs into the cost and benefit assessments.
- Chapter 4, Air Quality Impacts, describes the air quality data and modeling that are important for assessing the effect on contributions to air quality from the remedy options applied in this proposed rule, and as inputs to the benefits assessment.
- Chapter 5, Benefits Analysis and Results, describes the methodology and results of the benefits analysis
- Chapter 6, Electric Power Sector Profile, describes the industry affected by the rule.
- Chapter 7, Cost, Economic, and Energy Impacts, describes the modeling conducted to estimate the cost, economic, and energy impacts to the power sector.
- Chapter 8, Macroeconomic Impacts and Social Costs, describes the modeling conducted to estimate the social cost of the rule as well as the economic impacts to industries outside of the power sector.
- Chapter 9, Statutory and Executive Order Impact Analyses, describes the small business, unfunded mandates, paperwork reduction act, environmental justice, and other analyses conducted for the rule to meet statutory and Executive Order requirements.
- Chapter 10, Comparison of Benefits and Costs, shows a comparison of the social benefits to social costs of the rule.
- Appendix A, Human Health Benefits of Direct Control and Intrastate Trading Remedies
- Appendix B, Analyses of Economic Impacts Outside of the Electric Power Sector – Intrastate Trading and Direct Control Remedies

- Appendix C, Comparison of State Level Electrical Generating Unit Emissions Under Various Regulatory Alternatives To Reduce SO_2 And NO_x Emissions Under The Transport Rule
- Appendix D, Integrated Planning Model Runs
- Appendix E, Allowance Values for Emissions Trading Programs

CHAPTER 3

EMISSIONS IMPACTS

This chapter summarizes the emissions inventories that are used to create emissions inputs to the air quality modeling that is described in Chapter 4. This chapter provides a summary of the baseline emissions inventories and the emissions reductions that were modeled for this rule. The emissions inventories are processed into a form that is required by the Comprehensive Air Quality Model with extensions (CAMx). CAMx is used to estimate base year, future baseline and post-control concentrations of ozone and PM and deposition of nitrogen and sulfur, which are combined with monitoring data to estimate population-level exposures to changes in ambient concentrations for use in estimating health and welfare effects. In the remainder of this Chapter we provide an overview of (1) the emissions components of the modeling platform, (2) the development of the 2005 base year emissions, (3) the development of 2014 future year base case emissions, and (4) the development of the future year control case emissions.

3.1 Overview of Modeling Platform and Emissions Processing Performed

The inputs to the air quality model; including emissions, meteorology, initial conditions, boundary conditions; along with the methods used to produce the inputs and the configuration of the air quality model are collectively known as a 'modeling platform'. The 2005-based air quality modeling platform used for the proposal includes 2005 base year emissions and 2005 meteorology for modeling ozone and $PM_{2.5}$ with CAMx (see http://www.camx.com/). This platform provides an update to the now more historical data in the 2001-based platform used for CAIR that included 2001 emissions, 2001 meteorology for modeling $PM_{2.5}$, and 1995 meteorology for modeling ozone. Details on the non-emissions portion of the modeling platform used for the RIA are provided in described in Chapter 4. In support of this proposal, EPA modeled the air quality in the East using a horizontal grid resolution of 12 x 12 km. This Eastern 12 km modeling domain was "nested" within a modeling domain covering the remainder of the lower 48 states using a grid resolution of 36 x 36 km[8], therefore the tables of emissions in this section cover the contiguous 48 states.

Emissions estimates were made for a 2005 base year and for the 2014 future year scenarios. All inventories include emissions from electric generating utilities (EGUs), nonEGU point sources, stationary nonpoint sources, onroad mobile sources, nonroad mobile sources and natural, biogenic emissions. These emissions were derived from the 2005 v4 emissions modeling platform, described in the 2005-based, v4 platform document (http://www.epa.gov/ttn/chief/emch/index.html#2005). The Emissions Inventories Technical Support Document for Emissions Inventories for the Transport Rule (EITSD) provides more detail on (1) the development of the 2014 base case emissions inventories for all sectors, except EGUs and (2) the procedures followed to create emissions inputs to CAMx for each scenario modeled. For details on EPA's projected emissions for the EGU sector, see Chapter 7 of this RIA.

For each of the modeling scenarios conducted: 2005 base year, 2014 base case, and 2014 control case, the emissions inventory files were processed using the Sparse Matrix Operator Kernel Emissions (SMOKE) Modeling System version 2.6 to produce the gridded model-ready emissions for input to CAMx. SMOKE was used to create the hourly, gridded emissions data for the species required by CAMx species to perform air quality modeling for all sectors, including biogenic emissions. See Chapter 4 for more details on the modeling performed with CAMx.

3.2 Development of 2005 Base Year Emissions

Emissions inventory inputs representing the year 2005 were developed to provide a base year for forecasting future air quality. The emission source sectors and the basis for current and future-year inventories are listed and defined in Table 3-1. The 2005 National Emission Inventory (NEI), version 2 from October 6, 2008 was the starting point for the U.S. inventories used for the 2005 air quality modeling. This inventory includes 2005-specific data for most point and mobile sources, while most nonpoint data were carried forward from

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[8] The air quality predictions from the 36 km Continental US (CONUS) domain were used to provide incoming "boundary" concentrations for the Eastern 12 km domain.

version 3 of the 2002 NEI. For more information on the 2005 National Emissions Inventory (NEI), upon which significant portions of the 2005 modeling platform are based, see http://www.epa.gov/ttn/chief/net/2005inventory.html. A 2006 Canadian inventory and a 1999 Mexican inventory were the latest available data from these countries and were used for the portions of Canada and Mexico within the modeling domains.

Table 3-1. Emissions Source Sectors for Current and Future-Year Inventories

Platform Sector	**2005 NEI Sector**	**Description and resolution of the data input to SMOKE**
IPM sector: *ptipm*	Point	2005v2 NEI point source EGUs mapped to the Integrated Planning Model (IPM) model using the National Electric Energy Database System (NEEDS) database. Day-specific continuous emission monitoring (CEM) emissions and non-CEM sources created for input into SMOKE.
Non-IPM sector: *ptnonipm*	Point	All 2005v2 NEI point source records not matched to the ptipm sector, annual resolution. Includes all aircraft emissions.
Average-fire sector: *avefire*	N/A	Average-year wildfire and prescribed fire emissions derived from the 2002-based Platform avefire sector, county and annual resolution.
Agricultural sector: *ag*	Nonpoint	NH_3 emissions from NEI nonpoint livestock and fertilizer application, county and annual resolution.
Area fugitive dust sector: *afdust*	Nonpoint	PM_{10} and $PM_{2.5}$ from fugitive dust sources from the NEI nonpoint inventory (e.g., building construction, road construction, paved roads, unpaved roads, agricultural dust), county and annual resolution.
Remaining nonpoint sector: *nonpt*	Nonpoint	Primarily 2002 NEI nonpoint sources not otherwise included in other SMOKE sectors, county and annual resolution. Also includes 2005 updated Residential Wood Combustion emissions and year 2005 non-California WRAP oil and gas Phase II inventory.
Nonroad sector: *nonroad*	Mobile: Nonroad	Monthly nonroad emissions from the National Mobile Inventory Model (NMIM) using NONROAD2005 version nr05c-BondBase for all states except California. Monthly emissions for California created from annual emissions submitted by the California Air Resources Board (CARB) for the 2005v2 NEI.
Locomotive, and non-C3 commercial marine: *alm_no_c3*	Mobile: Nonroad	Year 2002 non-rail maintenance locomotives, and category 1 and category 2 commercial marine vessel (CMV) emissions sources, county and annual resolution. Aircraft emissions are now included in the ptnonipm sector and category 3 emissions are now contained in the seca_c3 sector.

C3 commercial marine: *seca_c3*	Mobile : Nonroad	Annual point source formatted year 2005 category 3 (C3) CMV emissions, developed for the rule called "Control of Emissions from New Marine Compression-Ignition Engines at or Above 30 Liters per Cylinder", usually described as the Area (ECA) study, originally called SO_2 ("S") ECA (see http://www.epa.gov/otaq/oceanvessels.htm).
Onroad California, NMIM-based, and MOVES sources not subject to temperature adjustments: *on_noadj*	Mobile: onroad	Three monthly, county-level components: 1) Onroad emissions from NMIM using MOBILE6.2, other than for California. 2) California onroad, created using annual emissions submitted by CARB for the 2005v2 NEI. 3) Onroad gasoline non-motorcycle vehicle emissions from draft MOVES not subject to temperature adjustments: exhaust CO, NO_x, VOC, some VOC Hazardous Air Pollutants (HAPs), and evaporative VOC and some VOC HAPs.
Onroad cold-start gasoline exhaust mode vehicle from MOVES subject to temperature adjustments: *on_moves_startpm*	Mobile: onroad	Monthly, county-level draft MOVES-based onroad non-motorcycle gasoline emissions subject to temperature adjustments. Limited to exhaust mode only for PM species and Naphthalene. California emissions not included. This sector is limited to cold start mode emissions that contain different temperature adjustment curves from running exhaust (see on_moves_runpm sector).
Onroad running gasoline exhaust mode vehicle from MOVES subject to temperature adjustments: *on_moves_runpm*	Mobile: onroad	Monthly, county-level draft MOVES-based onroad non-motorcycle gasoline emissions subject to temperature adjustments. Limited to exhaust mode only for PM species and Naphthalene. California emissions not included. This sector is limited to running mode emissions that contain different temperature adjustment curves from cold start exhaust (see on_moves_startpm sector).
Biogenic: *biog*	N/A	Hour-specific, grid cell-specific emissions generated from the BEIS3.14 model -includes emissions in Canada and Mexico.
Other point sources not from the NEI: *othpt*	N/A	Point sources from Canada's 2006 inventory and Mexico's Phase III 1999 inventory, annual resolution. Also includes annual U.S. offshore oil 2005v2 NEI point source emissions.
Other point sources not from the NEI, Hg only: *othpt_hg*	N/A	For 2005 only, the annual year 2000 Canada speciated mercury point source emissions. Note that the '_hg' sectors were not included in the future-year modeling.
Other nonpoint and nonroad not from the NEI: *othar*	N/A	Annual year 2006 Canada (province resolution) and year 1999 Mexico Phase III (municipio resolution) nonpoint and nonroad mobile inventories, annual resolution.

Other nonpoint sources not from the NEI, Hg only: ***othar_hg***	N/A	For 2005 only, the annual year 2000 Canada speciated mercury from nonpoint sources.
Other onroad sources not from the NEI: ***othon***	N/A	Year 2006 Canada (province resolution) and year 1999 Mexico Phase III (municipio resolution) onroad mobile inventories, annual resolution.

The onroad and nonroad emissions were primarily based on the National Mobile Inventory Model (NMIM) monthly, county, process level emissions from the 2005 NEI version 2 (v2). The 2005 onroad mobile emissions were augmented for onroad gasoline emissions sources with emissions based on a draft version of the Motor Vehicle Emissions Simulator (MOVES) (http://www.epa.gov/otaq/models/moves/) for carbon monoxide (CO), NO_x, VOC, $PM_{2.5}$, particulate matter less than ten microns (PM_{10}), naphthalene, and some VOC HAPs[9]. To account for the temperature dependence of $PM_{2.5}$, MOVES-based temperature adjustment factors were applied to gridded, hourly emissions using gridded, hourly meteorology. Additional information on this approach is available in the 2005-based platform documentation.

The 2005 annual NO_x and SO_2 emissions for sources in the ptipm sector as defined in Table 3-1 are based primarily on data from EPA's Clean Air Markets Division's Continuous Emissions Monitoring (CEM) program, with other pollutants estimated using emission factors and the CEM annual heat input. As noted in Table 3-1, the 2005 EGUs include those units operating in 2005 that are matched to the units modeled by the version of IPM used for this proposal, and include records with and without data submitted to the CEM program. For EGUs without CEMs, emissions were obtained from the state-submitted data in the NEI.

For the 2005 base year, the annual EGU NEI emissions in the ptipm sector were allocated to hourly emissions values needed for modeling based on the 2004, 2005, and 2006 CEM data. The NO_x CEM data were used to create NO_x -specific profiles, the SO_2 data were used to create SO_2-specific profiles, and the heat input data were used to allocate all other pollutants. The three years of data were used to create monthly and hourly profiles by state, while the 2005 data were used to create profiles for allocating monthly emissions to daily

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[9] The final version of MOVES was not available at the time we created the emissions for this proposed rule.

and hourly values. This approach to temporal allocation was used for both the 2005 base year and 2014 base and control emissions in order provide a temporal consistency across all scenarios modeled.

The nonpoint inventory was augmented with an oil and gas exploration inventory that includes emissions within the following states: Arizona, Colorado, Montana, Nevada, New Mexico, North Dakota, Oregon, South Dakota, Utah, and Wyoming. The commercial marine category 3 (C3) vessel emissions were augmented with gridded 2005 emissions from the previous modeling efforts for the rule called "Control of Emissions from New Marine Compression-Ignition Engines at or Above 30 Liters per Cylinder". The 2005 point source daily wildfire and prescribed burning emissions were replaced with average-year county-based inventories. Additionally, the inventories were processed to provide the hourly, gridded emissions for the model-species needed by CAMx. All of these details are further described in the 2005-based platform documentation.

Tables 3-2 and 3-3 provide summaries of SO_2 and NO_x emissions by state by sector for the 2005 base year for those states within the Eastern 12 km modeling domain. Emissions for other states that are within the 36 km modeling domain are available in the EITSD. All sectors listed are defined in Table 3-1. In the tables, the EGU column summarizes all units matched to the IPM model (ptipm sector) and the nonEGU column is for other point source units (ptnonipm sector). The Nonpoint column shows emissions for all nonpoint stationary sources (nonpt, ag, and afdust sectors). The Nonroad column summarizes emissions for nonroad mobile sources, including aircraft, locomotive, and marine sources including the C3 commercial marine (nonroad, alm_no_c3, and seca_c3 sectors). The Onroad column summarizes emissions for the combined NEI and draft MOVES-based emissions, in which emissions from the draft MOVES were used when available, and NEI emissions based on MOBILE6 were used for the remainder (on_noadj, on_moves_runpm, and on_moves_startpm sectors). Finally, the Fires column represents the average-year fire (avefire sector) emissions for wildfires and prescribed burning mentioned previously.

Table 3-2. 2005 Base Year SO_2 Emissions (tons/year) for Lower 48 States by Sector

State	EGU	NonEGU	Nonpoint	Nonroad	Onroad	Fires	Total
Alabama	460,123	70,346	52,325	6,397	3,199	983	593,372
Arizona	52,733	23,966	2,571	6,154	2,909	2,888	91,221
Arkansas	66,384	13,066	27,260	5,678	1,632	728	114,749
California	622	33,097	77,672	101,270	4,935	6,735	224,330
Colorado	64,174	1,549	6,810	4,897	2,526	1,719	81,675
Connecticut	10,356	1,831	18,455	2,548	1,128	4	34,320
Delaware	32,378	34,859	5,859	11,648	422	6	85,173
District of Columbia	1,082	686	1,559	414	172	0	3,914
Florida	417,321	57,475	70,490	93,543	10,285	7,018	656,131
Georgia	616,054	56,116	56,829	13,331	5,690	2,010	750,031
Idaho	0	17,151	2,915	2,304	794	3,845	27,010
Illinois	330,382	156,154	5,395	19,302	5,716	20	516,969
Indiana	878,978	95,200	59,775	9,436	3,981	24	1,047,396
Iowa	130,264	61,241	19,832	8,838	1,702	25	221,902
Kansas	136,520	13,142	36,381	8,035	1,824	103	196,005
Kentucky	502,731	25,811	34,229	6,942	2,711	364	572,787
Louisiana	109,851	165,737	2,378	73,233	2,399	892	354,489
Maine	3,887	18,519	9,969	3,725	834	150	37,084
Maryland	283,205	34,988	40,864	17,819	2,966	32	379,874
Massachusetts	85,768	19,620	25,261	25,335	2,168	93	158,245
Michigan	349,877	76,510	42,066	14,533	7,204	91	490,280
Minnesota	101,666	25,169	14,747	10,410	2,558	631	155,181
Mississippi	74,117	29,892	6,796	6,003	2,158	1,051	120,016
Missouri	284,384	78,307	44,573	10,464	4,251	186	422,165
Montana	19,715	11,056	2,600	3,813	767	1,422	39,373
Nebraska	74,955	6,429	29,575	9,199	1,326	105	121,589
Nevada	53,363	2,253	12,477	2,877	565	1,346	72,881
New Hampshire	51,445	3,245	7,408	805	630	38	63,571
New Jersey	57,044	7,640	10,726	23,484	2,486	61	101,441
New Mexico	30,628	7,831	3,193	3,541	1,517	3,450	50,161
New York	180,847	58,562	125,158	20,908	5,628	113	391,216
North Carolina	512,231	66,150	22,020	42,743	5,341	696	649,181
North Dakota	137,371	9,458	6,455	5,986	443	66	159,779
Oklahoma	110,081	40,482	7,542	5,015	2,699	469	166,288
Ohio	1,116,084	118,468	19,810	15,615	6,293	22	1,276,292
Oregon	12,304	9,825	9,845	13,717	1,537	4,896	52,124
Pennsylvania	1,002,202	85,411	68,349	11,972	5,363	32	1,173,328
Rhode Island	176	2,743	3,365	2,494	208	1	8,987
South Carolina	218,782	31,495	30,016	20,477	2,976	646	304,393
South Dakota	12,215	1,698	10,347	3,412	511	498	28,682
Tennessee	266,148	78,206	32,714	6,288	4,834	277	388,468

State	EGU	NonEGU	Nonpoint	Nonroad	Onroad	Fires	Total
Texas	534,949	223,625	109,215	52,749	13,470	1,178	935,187
Utah	34,813	9,132	3,577	2,439	1,633	1,934	53,527
Vermont	9	902	5,385	385	305	49	7,036
Virginia	220,248	69,440	32,923	18,420	3,829	399	345,259
Washington	3,409	24,211	7,254	28,137	2,823	407	66,241
West Virginia	469,456	48,314	14,589	2,133	1,095	215	535,802
Wyoming	89,874	22,321	6,721	2,674	663	1,106	123,359
Wisconsin	180,200	66,807	6,369	7,129	3,110	70	263,685
Grand Total	**10,019,774**	**1,953,744**	**1,117,009**	**596,847**	**123,547**	**19,345**	**13,830,266**

Table 3-3. 2005 Base Year NO_x Emissions (tons/year) for Lower 48 States by Sector

State	EGU	NonEGU	Nonpoint	Nonroad	Onroad	Fires	Total
Alabama	133,051	74,830	32,024	61,623	142,221	3,814	447,562
Arizona	79,776	15975	8650	62711	159501	10532	337,145
Arkansas	35,407	37,478	21,453	63,493	81,014	2,654	241,499
California	6,992	90,687	121,882	523,800	665,225	24,563	1,433,149
Colorado	73,909	20,971	43,652	50,856	109,231	6,271	304,890
Connecticut	6,865	5,824	12,554	21,785	69,645	14	116,688
Delaware	11,917	5,567	3,259	15,567	22,569	23	58,902
District of Columbia	492	501	1,740	3,494	9,677	0	15,904
Florida	217,263	53,778	29,533	277,888	460,474	25,600	1,064,537
Georgia	111,017	53,297	38,919	95,175	279,449	7,955	585,812
Idaho	19	10,354	30,317	22,087	34,858	14,024	111,659
Illinois	127,923	97,504	47,645	223,697	276,507	71	773,347
Indiana	213,503	73,647	30,185	110,100	187,426	88	614,949
Iowa	72,806	39,299	15,150	92,965	91,795	90	312,105
Kansas	90,220	70,785	42,286	86,553	76,062	378	366,285
Kentucky	164,743	35,432	17,557	90,669	127,435	1,326	437,163
Louisiana	63,791	165,162	27,559	301,170	112,889	3,254	673,824
Maine	1,100	18,309	7,423	13,379	38,469	566	79,246
Maryland	62,574	24,621	21,715	55,812	129,796	137	294,656
Massachusetts	25,618	18,429	34,373	74,419	118,148	341	271,327
Michigan	120,005	94,139	43,499	101,087	279,816	330	638,876
Minnesota	83,836	64,438	56,700	115,873	146,138	2,300	469,286
Mississippi	45,166	53,985	12,212	79,394	98,060	3,833	292,649
Missouri	127,431	38,604	32,910	123,228	183,022	678	505,873
Montana	39,858	5,356	14,415	40,687	32,312	5,187	137,815
Nebraska	52,426	12,156	13,820	107,180	58,643	381	244,607
Nevada	47,297	17,191	5,379	27,747	40,247	4,910	142,771
New Hampshire	8,827	3,241	11,235	9,246	32,537	137	65,223
New Jersey	30,114	20,598	26,393	88,486	157,736	223	323,550
New Mexico	75,483	43,925	69,175	45,552	71,596	12,582	318,313
New York	63,465	55,122	87,608	121,363	282,072	412	610,042
North Carolina	111,576	44,502	18,869	135,936	225,756	11,424	548,064
North Dakota	76,381	7,545	10,046	59,635	21,575	240	175,422
Oklahoma	86,204	73,465	94,574	55,424	117,240	1,709	428,617
Ohio	258,687	71,715	41,466	173,988	270,383	81	816,321
Oregon	9,383	22,927	17,059	78,284	85,045	17,857	230,555
Pennsylvania	176,870	89,208	53,435	118,774	266,649	117	705,053
Rhode Island	545	2,164	2,964	7,798	13,456	4	26,930
South Carolina	53,823	29,069	20,281	68,146	128,765	2,357	302,441
South Dakota	15,650	5,035	5,766	30,324	24,850	1,817	83,442
Tennessee	102,934	60,353	18,676	82,331	207,410	1,012	472,717

State	EGU	NonEGU	Nonpoint	Nonroad	Onroad	Fires	Total
Texas	176,170	292,806	274,338	377,246	615,715	4,890	1,741,166
Utah	65,261	19,466	13,844	26,985	74,024	7,052	206,632
Vermont	297	799	3,438	3,951	13,316	179	21,980
Virginia	62,512	60,101	53,605	91,298	194,173	1,456	463,145
Washington	17,634	25,427	16,911	121,014	145,871	1,484	328,341
West Virginia	159,804	36,913	14,519	32,739	50,040	785	294,801
Wisconsin	72,170	40,688	21,994	75,981	147,952	256	359,042
Wyoming	89,315	30,516	40,480	35,482	27,084	4,035	226,912
Grand Total	**3,728,110**	**2,233,904**	**1,683,487**	**4,682,422**	**7,203,874**	**189,429**	**19,721,235**

3.3 Development of Future Year Base Case Emissions

The future base case scenarios represent predicted emissions in the absence of any further controls beyond those Federal measures already promulgated. For EGUs (ptipm sector), all state and other programs available at the time of modeling have been included. For mobile sources (on_noadj, on_moves_runpm, and on_moves_startpm sectors), all national measures available at the time of modeling have been included. The future base case scenarios do reflect projected economic changes and fuel usage for EGU and mobile sectors, as described in the EITSD. For nonEGU point (ptnonipm sector) and nonpoint stationary sources (nonpt, ag, and afdust sectors), any local control programs that might be necessary for areas to attain the 1997 $PM_{2.5}$ NAAQS annual standard, 2006 PM NAAQS (24-hour) standard, and the 1997 ozone NAAQS are not included in the future base case projections. This is because the nonattainment areas for the 1997 $PM_{2.5}$ and ozone standards were not announced until 2004 and 2005 respectively, and the corresponding state implementation plans (SIPs) were not due until 2007 and 2008, thereby preventing the inclusion of these local measures in the 2005 emissions inventory.

Table 3-4 shows a summary of the 2005 and 2014 modeled base case emissions for the lower 48 states. Tables 3-6 and 3-7 below provide summaries of SO_2 and NO_x in the 2014 base case for each sector for the 37 states included in the 12 km modeling domain. The EITSD provides summaries for carbon monoxide, volatile organic compounds, directly emitted $PM_{2.5}$, and ammonia for each state in the nationwide 36 km modeling domain. For information on the topic of the significant contribution of some states on air quality issues in other states, please see Table 2-1.

Table 3-4. Summary of Modeled Base Case Annual Emissions (tons/year) for 48 States by Sector

Source Sector NO_x Emissions	2005	2014
EGU Point	3,728,110	2,908,844
Non-EGU Point	2,233,904	2,201,601
Nonpoint	1,683,487	1,679,404
Nonroad	4,682,422	3,706,913
On-road	7,203,874	3,410,053
Average Fire	189,429	189,429
Total NO_x, All Sources	**19,721,235**	**14,096,244**
Source Sector SO_2 Emissions		
EGU Point	10,381,405	8,469,820
Non-EGU Point	2,116,137	1,923,949
Nonpoint	1,252,645	1,252,127
Nonroad	768,671	604,519
On-road	144,216	31,067
Average Fire	49,095	49,095
Total SO_2, All Sources	**14,712,170**	**12,330,575**

The 2014 base case EGU emissions were obtained from version 3.02 EISA of the Integrated Planning Model (IPM) (http://www.epa.gov/airmarkt/progsregs/epa-ipm/index.html). The IPM is a multiregional, dynamic, deterministic linear programming model of the U.S. electric power sector; version 3.02 EISA features an updated Title IV SO_2 allowance bank assumption, reflects state rules and consent decrees through February 3, 2009, and incorporates updates related to the Energy Independence and Security Act of 2007. Units with advanced controls (e.g., scrubber, SCR) that were not required to run for compliance with Title IV, New Source Review (NSR), state settlements, or state-specific rules were allowed in IPM to decide on the basis of abatement cost minimization whether to operate those controls or to use allowances for compliance. Further details on the future year

EGU emissions used for air quality modeling can be found in the IPM Documentation. Note that controls from the NO_x SIP call were assumed to have been implemented by 2005 and captured in the 2005 NEI, and reductions from the Clean Air Interstate Rule are not included in the 2014 base case emissions.

Mobile source inventories of onroad and nonroad mobile emissions were created for 2014 using a combination of the NMIM and draft MOVES models. The future onroad emissions reflect control programs including the Light-Duty Vehicle Tier 2 Rule, the Onroad Heavy-Duty Rule, and the Mobile Source Air Toxics (MSAT) final rule. Emission reductions and increases from the Renewable Fuel Standard version 2 (RFS2) are not included. The future case nonroad mobile emissions reductions for these years include reductions to locomotives, various nonroad engines including diesel engines and various marine engine types, fuel sulfur content, and evaporative emissions standards. A summary of the included mobile source control programs is shown in Table 3-5. A more comprehensive list of control programs included for mobile sources is available in the EITSD.

The 2014 onroad emissions were primarily based on the National Mobile Inventory Model (NMIM) monthly, county, process level emissions. The emissions from onroad gasoline sources were augmented with emissions based on the same preliminary version of MOVES as was used for 2005. The same MOVES-based $PM_{2.5}$ temperature adjustment factors were also applied as in 2005 for running mode emissions; however, cold start emissions used year-specific temperature adjustment factors.

Nonroad mobile emissions were created only with NMIM using a consistent approach as was used for 2005, but emissions were calculated using NMIM future-year equipment population estimates and control programs for 2014. Emissions for locomotives and category 1 and 2 (C1 and C2) commercial marine vessels were derived for 2014 based on emissions published in the Locomotive Marine Rule, Regulatory Impact Assessment, Chapter 3 (see http://www.epa.gov/otaq/locomotives.htm#2008final).

Table 3-5. Summary of Mobile Source Control Programs Included in 2014 Base Case

National Onroad Rules: Tier 2 Rule (February 28, 2000) Onroad Heavy-Duty Rule (February 24, 2009) Final Mobile Source Air Toxics Rule (MSAT2) (February 9, 2007) Renewable Fuel Standard (March 26, 2010)
Local Onroad Programs: National Low Emission Vehicle Program (NLEV) (March 2, 1998) Ozone Transport Commission (OTC) LEV Program (January, 1995)
National Nonroad Controls: Tier 1 nonroad diesel rule (June 17, 2004) Phase 1 nonroad SI rule (July 3, 1995) Marine SI rule (October 4, 1996) Nonroad diesel rule (October 23, 1998) Phase 2 nonroad nonhandheld SI rule (March 30, 1999) Phase 2 nonroad handheld SI rule (April 25, 2000) Nonroad large SI and recreational engine rule (November 8, 2002) Clean Air Nonroad Diesel Rule - Tier 4 (June 29, 2004) Locomotive and marine rule (May 6, 2008) Nonroad SI rule (October 8, 2008)
Aircraft: Itinerant (ITN) operations at airports adjusted to year 2014
Locomotives: Clean Air Nonroad Diesel Final Rule – Tier 4 (June 29, 2004) Locomotive rule (April 16, 2008) Locomotive and marine rule (May 6, 2008)
Commercial Marine: Locomotive and marine rule (May 6, 2008) Clean Air Nonroad Diesel Final Rule – Tier 4 (June 29, 2004) Commercial Marine Rule (December 29, 1999) Tier 1 Commercial Marine Rule (February 28, 2003)

In the 2014 base case, we used the 2005 base year emissions for Canada and Mexico because future year emissions for sources in these countries were not available.

For nonEGU point sources, emissions were projected by including emissions reductions and increases from a variety of source data[10]. For nonEGU point sources,

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[10] Controls from the NO_X SIP call were assumed to have been in place by 2005 and captured in the 2005 NEI v2.

emissions were not grown using economic growth projections, but, rather were held constant at the emissions levels in 2005. Emissions reductions were applied to nonEGU point source to reflect known plant closures, refinery and other consent decrees, and reductions stemming from several Maximum Achievable Control Technology (MACT) standards. Since aircraft at airports were treated as point emissions sources in the 2005 NEI v2, we applied projection factors based on activity growth projected by the Federal Aviation Administration Terminal Area Forecast (TAF) system, published December 2008 for these sources.

Emissions from stationary nonpoint sources were projected using procedures specific to individual source categories. Refueling emissions were projected using the refueling results from the NMIM runs performed for the onroad mobile sector. Portable fuel container emissions were projected using estimates from previous rulemaking inventories compiled by the Office of Transportation and Air Quality (OTAQ). Emissions of ammonia and dust from animal operations were projected based on animal population data from the Department of Agriculture and EPA. Residential wood combustion was projected by replacement of obsolete woodstoves with new woodstoves and a 1 percent annual increase in fireplaces. Landfill emissions were projected using MACT controls. All other nonpoint sources were held constant between 2005 and the 2014 future year scenarios.

Table 3-6. 2014 Base Case SO_2 Emissions (tons/year) for Lower 48 States by Sector[A]

State	EGU	NonEGU	Nonpoint	Nonroad	Onroad	Fires	Total
Alabama	322,130	69,150	52,313	1,873	605	983	447,053
Arizona	20,945	23,982	2,566	51	738	2,888	51,170
Arkansas	88,187	13,055	27,256	142	347	728	129,715
California	5,052	24,869	77,614	108,132	2,002	6,735	224,404
Colorado	72,119	1,562	6,808	47	550	1,719	82,805
Connecticut	5,512	1,834	18,440	1,294	340	4	27,424
Delaware	7,806	10,974	5,857	14,891	101	6	39,635
District of Columbia	0	686	1,559	4	42	0	2,291
Florida	192,903	57,521	70,480	108,579	2,159	7,018	438,660
Georgia	173,210	56,014	56,813	8,263	1,307	2,010	297,617
Idaho	1	17,153	2,912	21	177	3,845	24,109
Illinois	200,475	133,109	5,381	390	1,221	20	340,596
Indiana	804,294	95,037	59,764	193	810	24	960,122
Iowa	163,966	60,195	19,817	85	360	25	244,448

State	EGU	NonEGU	Nonpoint	Nonroad	Onroad	Fires	Total
Kansas	65,125	13,048	36,375	54	313	103	115,018
Kentucky	739,592	23,804	34,210	258	528	364	798,756
Louisiana	94,824	151,216	2,372	78,097	470	892	327,871
Maine	11,650	18,520	9,945	4,215	160	150	44,640
Maryland	42,635	34,994	40,851	16,966	631	32	136,109
Massachusetts	16,299	19,624	25,237	32,043	594	93	93,890
Michigan	275,637	76,437	42,066	7,536	1,107	91	402,874
Minnesota	61,447	25,112	14,728	468	618	631	103,004
Mississippi	48,149	24,427	6,785	1,280	385	1,051	82,077
Missouri	500,649	77,086	44,543	214	796	186	623,474
Montana	16,863	7,597	2,593	24	115	1,422	28,614
Nebraska	115,695	6,431	29,570	55	217	105	152,073
Nevada	20,155	2,266	12,475	25	196	1,346	36,463
New Hampshire	6,608	3,246	7,393	45	148	38	17,478
New Jersey	37,669	6,756	10,712	26,589	799	61	82,586
New Mexico	13,708	7,834	3,190	24	280	3,450	28,486
New York	141,354	58,584	125,196	10,853	1,594	113	337,694
North Carolina	140,585	66,046	21,994	52,897	961	696	283,179
North Dakota	80,320	9,458	6,450	35	78	66	96,407
Ohio	841,194	105,123	19,810	2,085	1,171	22	969,405
Oklahoma	165,773	36,924	7,534	45	524	469	211,269
Oregon	13,366	9,831	9,846	14,530	397	4,896	52,866
Pennsylvania	972,977	76,256	68,324	4,117	1,169	32	1,122,875
Rhode Island	0	2,745	3,364	3,128	85	1	9,323
South Carolina	156,096	31,453	30,002	24,380	551	646	243,128
South Dakota	13,459	1,699	10,341	22	94	498	26,113
Tennessee	600,066	77,605	32,696	173	829	277	711,646
Texas	373,950	155,720	109,194	36,109	2,511	1,178	678,662
Utah	25,414	7,157	3,574	25	310	1,934	38,414
Vermont	0	903	5,380	7	101	49	6,440
Virginia	135,741	69,177	32,899	15,624	918	399	254,758
Washington	19,155	21,136	7,229	27,880	687	407	76,494
West Virginia	496,307	41,817	14,581	96	201	215	553,217
Wisconsin	117,253	66,456	6,370	638	675	70	191,462
Wyoming	53,505	22,320	6,718	17	95	1,106	83,761
Grand Total	**8,469,820**	**1,923,949**	**1,252,127**	**604,519**	**31,067**	**49,094**	**12,330,575**

[A] Emission estimates apply to all fossil Electrical Generating Units, including those with capacity < 25 MW

Table 3-7. 2014 Base Case NO$_x$ Emissions (tons/year) for Lower 48 States by Sector[A]

State	EGU	NonEGU	Nonpoint	Nonroad	Onroad	Fires	Total
Alabama	118,420	74,622	31,939	45,932	67,011	3,814	341,738
Arizona	72,747	16,130	8,615	43,037	77,732	10,532	228,793
Arkansas	44,792	37,491	21,422	44,299	38,965	2,654	189,623
California	18,394	89,084	121,496	429,644	346,901	24,563	1,030,082
Colorado	61,641	21,139	43,556	35,480	59,980	6,271	228,067
Connecticut	2,821	5,854	12,451	14,410	31,534	14	67,084
Delaware	4,513	5,567	3,245	15,270	8,736	23	37,354
District of Columbia	1	501	1,738	2,398	3,929	0	8,567
Florida	180,801	55,343	29,457	278,920	225,478	25,600	795,599
Georgia	48,091	53,557	38,797	71,011	130,240	7,955	349,651
Idaho	398	10,367	30,294	15,832	20,727	14,024	91,642
Illinois	80,228	93,059	47,540	151,373	131,403	71	503,674
Indiana	200,899	73,523	30,107	76,024	94,217	88	474,858
Iowa	68,146	38,831	15,038	65,751	48,836	90	236,692
Kansas	78,920	70,730	42,238	61,613	35,950	378	289,829
Kentucky	148,509	34,979	17,413	65,805	57,759	1,326	325,791
Louisiana	45,457	161,766	27,515	274,697	52,360	3,254	565,049
Maine	2,535	18,316	7,257	13,169	18,061	566	59,904
Maryland	19,990	24,687	21,626	52,501	53,040	137	171,981
Massachusetts	6,619	18,527	34,207	75,654	46,748	341	182,096
Michigan	97,455	94,079	43,360	73,939	135,806	330	444,969
Minnesota	51,859	64,372	56,545	84,040	71,161	2,300	330,277
Mississippi	37,142	52,440	12,133	58,559	42,525	3,833	206,632
Missouri	82,979	38,744	32,677	88,233	90,001	678	333,312
Montana	36,800	5,368	14,359	28,367	14,161	5,187	104,242
Nebraska	52,970	12,173	13,779	75,252	27,856	381	182,411
Nevada	29,198	17,323	5,375	19,272	17,188	4,910	93,266
New Hampshire	2,515	3,255	11,129	6,587	16,260	137	39,883
New Jersey	16,268	19,089	26,298	78,875	63,254	223	204,007
New Mexico	51,340	43,953	69,146	31,864	34,564	12,582	243,449
New York	28,350	55,359	87,826	92,841	129,376	412	394,164
North Carolina	61,747	44,573	18,669	133,455	104,150	11,424	374,018
North Dakota	59,556	7,549	10,009	42,972	9,925	240	130,251
Ohio	164,945	69,157	41,352	120,900	122,426	81	518,861
Oklahoma	81,122	72,525	94,513	39,539	58,382	1,709	347,790
Oregon	13,889	22,985	17,081	68,854	51,973	17,857	192,639
Pennsylvania	196,151	84,111	53,246	83,885	118,122	117	535,632

State	EGU	NonEGU	Nonpoint	Nonroad	Onroad	Fires	Total
Rhode Island	281	2,186	2,957	7,384	6,772	4	19,584
South Carolina	47,512	28,969	20,271	62,400	62,996	2,357	224,505
South Dakota	15,514	5,039	5,722	22,021	12,254	1,817	62,367
Tennessee	68,779	59,694	18,542	59,145	104,711	1,012	311,883
Texas	166,177	282,509	274,163	289,605	241,009	4,890	1,258,353
Utah	64,088	19,285	13,824	18,576	35,500	7,052	158,325
Vermont	0	803	3,397	2,771	8,563	179	15,713
Virginia	32,115	60,216	53,464	75,461	92,291	1,456	315,003
Washington	18,374	24,825	16,728	106,915	83,318	1,484	251,644
West Virginia	100,103	35,700	14,459	23,798	22,863	785	197,708
Wisconsin	53,774	40,729	21,974	53,848	71,163	256	241,744
Wyoming	73,919	30,518	40,455	24,735	11,876	4,035	185,538
Grand Total	**2,908,844**	**2,201,601**	**1,679,404**	**3,706,913**	**3,410,053**	**189,429**	**14,096,244**

[A] Emission estimates apply to all fossil Electrical Generating Units, including those with capacity < 25 MW

3.4 Development of Future Year Control Case Emissions

For the future year control case modeling, the emissions for all sectors were unchanged from the base case modeling except for those from EGUs. The IPM model was used by CAMD to prepare the 2014 control case EGU emissions as described in Chapter 7. The changes in EGU SO_2 and NO_x emissions as a result of the control case for the lower 48 states are summarized in Table 3-8. State-specific summaries of EGU NO_x and SO_2 for the lower 48 states for the control case are shown in Tables 3-9 and 3-10, respectively. For EGU emission changes for each remedy analyzed, and for 2012 as well as 2014, please refer to Appendix C. Additional details on the changes that resulted from the control case are provided in the Transport Rule EITSD.

Table 3-8. Summary of Emissions Changes for the Transport Rule in Lower 48 States

Item	Pollutant NO_X	 SO_2
2014 EGU Emissions		
Base Case EGU Emissions (tons)	2,908,844	8,469,820
Control EGU Emissions (tons)	2,089,744	4,045,465
Reductions to Base Case in Control Case (tons)	819,101	4,424,358
Percentage Reduction of Base EGU Emissions	28.2%	52.2%
Total 2014 Man-made Emissions*		
Total Base Case Emissions (tons)	14,096,244	12,330,575
Total Control Case Emissions (tons)	13,277,143	7,906,217
Percentage Reduction of All Manmade Emissions	5.8%	35.9%

* In this table, man-made emissions includes average fires

Table 3-9. State Specific Changes in Annual EGU NO_x for the Lower 48 States[A]

State	2014 Base NO_x	2014 Controlled NO_x	EGU NO_x Reduction (tons)	EGU NO_x Reduction (%)
Alabama	118,420	61,259	57,161	48.3%
Arizona	72,747	72,705	42	0.1%
Arkansas	44,792	26,260	18,532	41.4%
California	18,394	18,429	-35	-0.2%
Colorado	61,641	62,018	-377	-0.6%
Connecticut	2,821	2,833	-12	-0.4%
Delaware	4,513	4,933	-420	-9.3%
District of Columbia	1	1	0	-3.7%
Florida	180,801	110,603	70,198	38.8%
Georgia	48,091	44,285	3,806	7.9%
Idaho	398	397	1	0.3%
Illinois	80,228	57,366	22,862	28.5%
Indiana	200,899	112,379	88,519	44.1%

State	2014 Base NO_x	2014 Controlled NO_x	EGU NO_x Reduction (tons)	EGU NO_x Reduction (%)
Iowa	68,146	52,986	15,160	22.2%
Kansas	78,920	39,958	38,963	49.4%
Kentucky	148,509	71,314	77,195	52.0%
Louisiana	45,457	37,156	8,301	18.3%
Maine	2,535	2,530	5	0.2%
Maryland	19,990	20,070	-80	-0.4%
Massachusetts	6,619	7,016	-397	-6.0%
Michigan	97,455	63,135	34,320	35.2%
Minnesota	51,859	35,426	16,433	31.7%
Mississippi	37,142	23,099	14,043	37.8%
Missouri	82,979	67,437	15,541	18.7%
Montana	36,800	36,789	10	0.0%
Nebraska	52,970	35,067	17,903	33.8%
Nevada	29,198	29,200	-2	0.0%
New Hampshire	2,515	2,456	59	2.3%
New Jersey	16,268	12,717	3,552	21.8%
New Mexico	51,340	51,358	-18	0.0%
New York	28,350	28,593	-243	-0.9%
North Carolina	61,747	59,663	2,085	3.4%
North Dakota	59,556	59,548	9	0.0%
Ohio	164,945	99,333	65,612	39.8%
Oklahoma	81,122	50,434	30,688	37.8%
Oregon	13,889	13,889	0	0.0%
Pennsylvania	196,151	114,884	81,267	41.4%
Rhode Island	281	278	4	1.4%
South Carolina	47,512	34,505	13,007	27.4%
South Dakota	15,514	15,509	5	0.0%
Tennessee	68,779	28,079	40,699	59.2%
Texas	166,177	148,002	18,175	10.9%
Utah	64,088	64,070	18	0.0%
Virginia	32,115	30,436	1,680	5.2%
Washington	18,374	18,359	15	0.1%
West Virginia	100,103	48,149	51,954	51.9%
Wisconsin	53,774	40,923	12,851	23.9%
Wyoming	73,919	73,908	10	0.0%
Total	**2,908,844**	**2,089,743**	**819,101**	**28.2%**

[A] Emission estimates apply to all fossil Electrical Generating Units, including those with capacity < 25 MW

Table 3-10. State Specific Changes in Annual EGU SO_2 for the Lower 48 States[A]

State	2014 Base SO_2	2014 Controlled SO_2	EGU SO_2 Reduction (tons)	EGU SO_2 Reduction (%)
Alabama	322130	172198	149,932	46.5%
Arizona	20945	23477	-2,532	-12.1%
Arkansas	88187	119945	-31,758	-36.0%
California	5052	5052	0	0.0%
Colorado	72119	88324	-16,204	-22.5%
Connecticut	5512	2586	2,926	53.1%
Delaware	7806	8919	-1,113	-14.3%
District of Columbia	0	0	0	N/A
Florida	192903	137985	54,918	28.5%
Georgia	173210	92329	80,882	46.7%
Idaho	1	0	1	100.0%
Illinois	200475	164733	35,742	17.8%
Indiana	804294	240599	563,695	70.1%
Iowa	163966	102419	61,547	37.5%
Kansas	65125	51248	13,878	21.3%
Kentucky	739592	123831	615,761	83.3%
Louisiana	94824	94892	-67	-0.1%
Maine	11650	11669	-19	-0.2%
Maryland	42635	42756	-120	-0.3%
Massachusetts	16299	9340	6,959	42.7%
Michigan	275637	173414	102,223	37.1%
Minnesota	61447	48819	12,628	20.6%
Mississippi	48149	62356	-14,207	-29.5%
Missouri	500649	192644	308,004	61.5%
Montana	16863	19093	-2,229	-13.2%
Nebraska	115695	75094	40,601	35.1%
Nevada	20155	20531	-376	-1.9%
New Hampshire	6608	7290	-682	-10.3%
New Jersey	37669	14555	23,114	61.4%
New Mexico	13708	13027	681	5.0%
New York	141354	57047	84,307	59.6%
North Carolina	140585	96924	43,661	31.1%
North Dakota	80320	88320	-8,000	-10.0%
Ohio	841194	232948	608,245	72.3%
Oklahoma	165773	165994	-221	-0.1%
Oregon	13366	20187	-6,821	-51.0%
Pennsylvania	972977	153204	819,773	84.3%
Rhode Island	0	0	0	N/A
South Carolina	156096	131128	24,968	16.0%

South Dakota	13459	28897	-15,438	-114.7%
Tennessee	600066	106762	493,304	82.2%
Texas	373950	467765	-93,815	-25.1%
Utah	25414	29117	-3,703	-14.6%
Virginia	135741	57496	78,245	57.6%
Washington	19155	18863	292	1.5%
West Virginia	496307	127646	368,662	74.3%
Wisconsin	117253	85788	31,464	26.8%
Wyoming	53505	58254	-4,750	-8.9%
Total	**8,469,820**	**4,045,465**	**4,424,358**	**52.2%**

[A] Emission estimates apply to all fossil Electrical Generating Units, including those with capacity < 25 MW

CHAPTER 4

AIR QUALITY MODELING AND IMPACTS

4.1 Air Quality Impacts

This section summarizes the methods for and results of estimating air quality for the 2014 base case and control scenario for the purposes of the benefit analysis. EPA has focused on the health, welfare, and ecological effects that have been linked to air quality changes. These air quality changes include the following:

1. Ambient particulate matter (PM_{10} and $PM_{2.5}$)–as estimated using a national-scale applications of the Comprehensive Air Quality Model with Extensions (CAMx; Environ, 2009); and
2. Visibility degradation (i.e., regional haze), as developed using empirical estimates of light extinction coefficients and efficiencies in combination with CAMx modeled reductions in pollutant concentrations.

The air quality estimates in this section are based on the emission changes summarized in the preceding section. These air quality results are in turn associated with human populations and ecosystems to estimate changes in health and welfare effects. In Section 4.1.1, we describe the air quality modeling platform and in Section 4.2, we cover the impacts on $PM_{2.5}$ and ozone. Lastly, in Section 4.3, we discuss the estimation of visibility degradation.

4.1.1 Air Quality Modeling Platform

We use the emissions inputs summarized above with national scale and regional scale application of the CAMx modeling system to estimate $PM_{2.5}$ and ozone air quality in the contiguous U.S. CAMx is a three-dimensional grid-based Eulerian photochemical model designed to estimate $PM_{2.5}$ and ozone concentrations over annual time periods. Consideration of the different processes that affect primary (directly emitted) and secondary (formed by atmospheric processes) $PM_{2.5}$ in different locations is fundamental to understanding and assessing the effects of pollution control measures that affect $PM_{2.5}$ and ozone concentrations at the surface.[11] Because it accounts for spatial and temporal variations as well as differences in the reactivity of emissions, CAMx is useful for evaluating the impacts of the rule on $PM_{2.5}$ and ozone concentrations. Version 5 of CAMx was employed for this Transport Rule modeling, as described in the Air Quality Modeling Technical Support Document (EPA, 2010).

For this analysis we used CAMx to simulate air quality for every hour of every day of the year. These model applications required a variety of input files that contain information pertaining to the modeling domain and simulation period. In addition to the CAMx model, our modeling system includes (1) emissions for a 2005 base year and emissions for the 2014 base case and control scenario, (2) meteorology for the year 2005, and (3) estimates of intercontinental transport (i.e., boundary concentrations) from a global photochemical model. Using these data, CAMx generates hourly predictions of ozone and $PM_{2.5}$ component species concentrations. As discussed in the Air Quality Modeling TSD, we use the relative predictions from the model by combining the 2005 base-year and each future-year scenario with speciated ambient air quality observations to determine the expected change in 2014 concentrations due to the rule. After completing this process, we then calculated annual mean $PM_{2.5}$ and seasonal mean ozone air quality metrics as inputs to the health and welfare C-R functions of the benefits analysis.

4.1.1.1 Simulation Periods

For use in this benefits analysis, the simulation period modeled by CAMx included separate full-year application for each of the three emissions scenarios (i.e., 2005 base year and the 2014 base case and 2014 control scenario).

4.1.1.2 Air Quality Modeling Domain

Although air quality estimate are provided for the entire U.S., the focus of our analysis is on the Eastern U.S. since this is the geographic area of importance for this rule.

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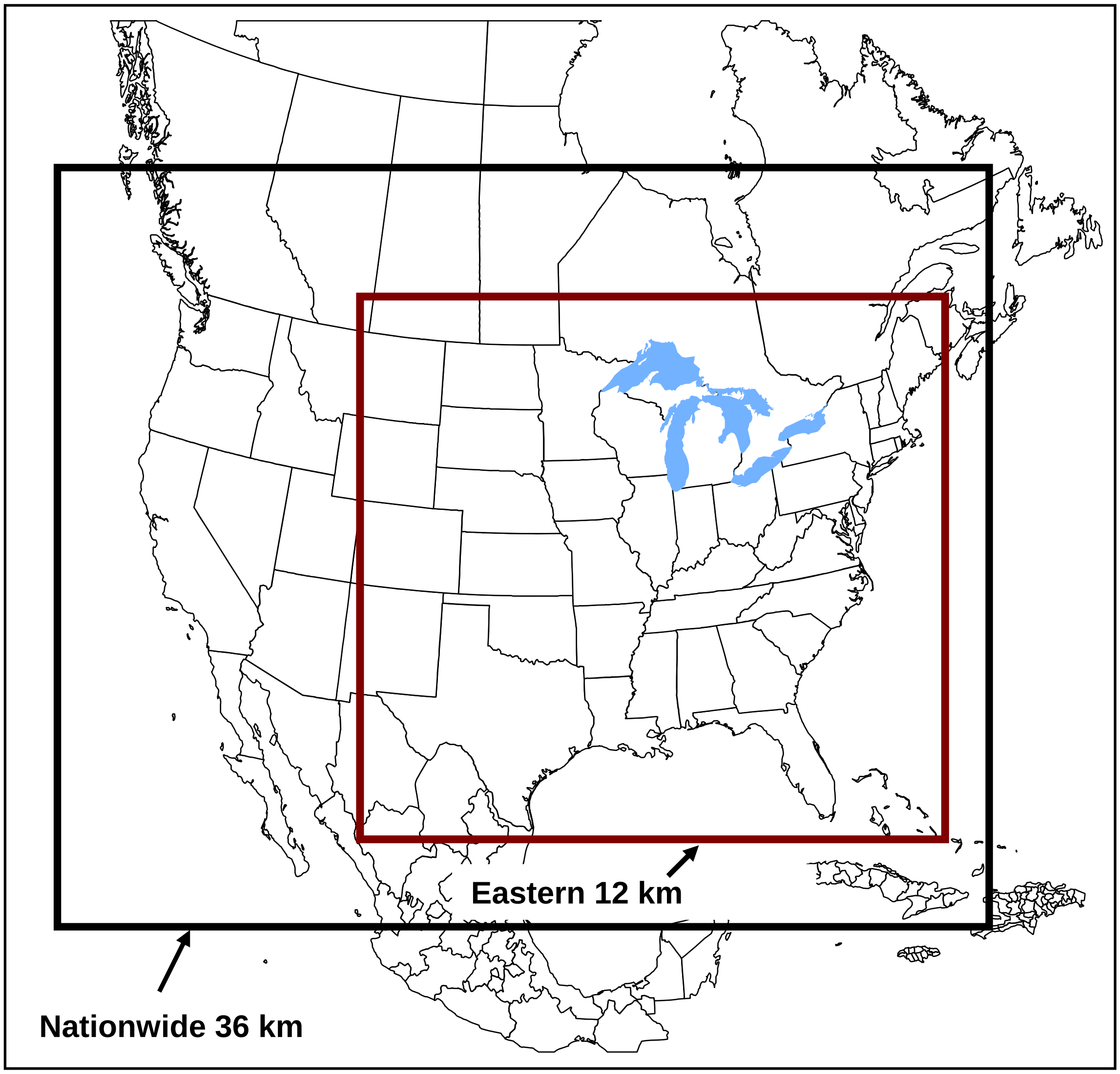
[11] Given the focus of this rule on secondarily formed particles it is important to employ a Eulerian model such as CAMx. The impact of secondarily formed pollutants typically involves primary precursor emissions from a multitude of widely dispersed sources, and chemical and physical processes of pollutants are best addressed using an air quality model that employs an Eulerian grid model design.

The areas modeled (i.e., modeling domains) are segmented into rectangular blocks referred to as grid cells. The model actually predicts pollutant concentrations for each of these grid cells. Our modeling for the East (referred to as the Eastern regional scale domain) was performed at a horizontal resolution of 12 x 12 km. Modeling for the remainder of the U.S. (referred to as the national scale domain) was performed at a resolution of 36 x 36 km. The national and regional scale modeling domains contain 14 vertical layers with the top of the modeling domain at about 16,200 meters, or approximately 100 mb. The Eastern domain is nested within the National domain, as shown in Figure 4-1.

Figure 4-1. National and Eastern U.S. air quality modeling domains.



4.1.1.3 Air Quality Model Inputs

CAMx requires a variety of input files that contain information pertaining to the modeling domain and simulation period. These include gridded, hourly emissions estimates and meteorological data, and initial and boundary conditions. Separate emissions inventories were prepared for the 2005 base year and each future-year scenario. All other inputs were specified for the 2005 base year model application and remained unchanged for each future-year modeling scenario.

CAMx requires detailed emissions inventories containing temporally allocated emissions for each grid-cell in the modeling domain for each species being simulated. The previously described annual emission inventories were preprocessed into model-ready inputs through the SMOKE emissions preprocessing system. Meteorological inputs reflecting 2005 conditions across the contiguous U.S. were derived from Version 5 of the Mesoscale Model (MM5). These inputs included horizontal wind components (i.e., speed and direction), temperature, moisture, vertical diffusion rates, and rainfall rates for each grid cell in each vertical layer. Details of the annual 2005 MM5 modeling are provided in the Air Quality Modeling TSD.

The lateral boundary and initial species concentrations are provided by a three-dimensional global atmospheric chemistry and transport model (GEOS-CHEM). The lateral boundary species concentrations varied with height and time (every 3 hours). Terrain elevations and land use information were obtained from the U.S. Geological Survey database at 10 km resolution and aggregated to the roughly 36 km horizontal resolution used for this CAMx application. The development of model inputs is discussed in greater detail in the Air Quality Modeling TSD, which is available in the docket for this rule.

4.1.1.4 Air Quality Model Evaluation

An operational model performance evaluation for ozone and $PM_{2.5}$ and its related speciated components (e.g., sulfate, nitrate, elemental carbon, organic carbon) was performed to estimate the ability of the CAMx modeling system to replicate 2005 base year concentrations. This evaluation principally comprises statistical assessments of model predictions versus observations paired in time and space on an hourly, daily, or weekly basis depending on the sampling period of measured data. Details on the evaluation methodology and the calculation of performance statistics are provided in the Air Quality Modeling TSD. Overall, the model performance statistics for ozone, sulfate, and nitrate from the CAMx 2005 simulation are within or close to the ranges found in other recent applications. The normalized mean bias for 8-hour daily maximum ozone concentrations was -2.9 percent and the normalized mean error was 13.2 percent for the months of May through September 2005, based on an aggregate of all observed-predicted pairs within the 12 km modeling domain. The two $PM_{2.5}$ species that are most relevant for today's proposal are sulfate and nitrate. For the summer months of June though August, when observed sulfate concentrations are highest in the East, the model predictions of 24-hour average sulfate were lower than the corresponding measured values by 7 percent at urban sites and by 9 to 10 percent at rural sites in the IMPROVE[12] and CASTNet[13] monitoring networks, respectively. For the winter months of December through February, when observed nitrate concentrations are highest in the East, the model predictions of 24-hour average particulate nitrate were lower than the corresponding measured values by 12 percent at urban sites and by 4 percent at rural sites in the IMPROVE monitoring network. The model performance statistics by season for ozone and $PM_{2.5}$ component species are provided in the Air Quality Modeling TSD. These model performance results give us confidence that our applications of CAMx using this 2005 modeling platform provide a scientifically credible approach for assessing ozone and $PM_{2.5}$ concentrations for the purposes of the Transport Rule.

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[12] Interagency Monitoring of PROtected Visual Environments (IMPROVE). Debell, L.J., et. al. Spatial and Seasonal Patterns and Temporal Variability of Haze and its Constituents in the United States: Report IV. November 2006.

[13] Clean Air Status and Trends Network (CASTNet) 2005 Annual Report. EPA Office of Air and Radiation, Clean Air Markets Division. Washington, DC. December 2006.

4.2 Results for $PM_{2.5}$ and Ozone

4.2.1 Converting CAMx $PM_{2.5}$ Outputs to Benefits Inputs

CAMx generates predictions of hourly $PM_{2.5}$ species concentrations for every grid. The species include a primary fraction and several secondary particles (e.g., sulfates, nitrates, and organics). $PM_{2.5}$ is calculated as the sum of the primary and the secondary formed particles. Future-year estimates of $PM_{2.5}$ were calculated using relative reduction factors (RRFs) applied to 2005 ambient $PM_{2.5}$ species concentrations. Gridded fields of species concentrations were created by interpolating ambient data from the $PM_{2.5}$ speciation network and IMPROVE data. The ambient data were interpolated to the 36 km and 12 km grid resolutions.

The procedures for determining the RRFs are similar to those in EPA guidance for modeling the $PM_{2.5}$ standard (EPA, 2007). This guidance recommends that model predictions be used in a relative sense to estimate changes expected to occur in each $PM_{2.5}$ species. The procedure for calculating future year $PM_{2.5}$ values is called the Modeled Attainment Test Software (MATS). EPA used this procedure to estimate the ambient impacts of the Transport Rule emissions controls. For the purposes of projecting future $PM_{2.5}$ concentrations for input to the benefits calculations, we applied the MATS procedure using the base year 2005 modeling results and each of the results from each of the 2014 base case and 2014 control scenario. In our application of MATS for $PM_{2.5}$ we used temporally scaled speciated $PM_{2.5}$ monitoring data from 2005 as the set of base-year measured concentrations. Temporal scaling is based on the ratios of model-predicted future case $PM_{2.5}$ species concentrations to the corresponding model-predicted 2005 concentrations. Output files from this process include both quarterly and annual mean $PM_{2.5}$ mass concentrations which are then manipulated within SAS to produce a BenMAP input file containing 364 daily values (created by replicating the quarterly mean values for each day of the appropriate season).

The MATS procedures documented in the Air Quality Modeling TSD are applicable for projecting future nonattainment and maintenance sites and downwind receptor areas for the transport analysis. Those procedures are similar as those performed for the PM benefits analysis in Chapter 5 with the following exceptions:

1) The benefits analysis uses interpolated $PM_{2.5}$ data that cover all of the grid cells in the modeling domain, whereas the nonattainment analysis is performed at each ambient monitoring site using measured $PM_{2.5}$ data (only the species data are interpolated).

2) The benefits analysis is anchored by the interpolated $PM_{2.5}$ data from the single year of 2005, whereas the nonattainment analysis uses design values from three, 3-year periods (i.e., 2003-2005, 2004-2006, and 2005-2007) at individual monitoring sites.

4.2.2 $PM_{2.5}$ Air Quality Results

Table 4-1 summaries the projected ambient $PM_{2.5}$ concentrations for the 2014 base case and 2014 impacts associated with rule. This table includes the annual mean concentrations averaged across all model grid cells in the East along with the average change between the 2014 base and control concentrations. We also provide the population-weighted average that better reflects the baseline levels and predicted changes for more populated areas of the East. This measure , therefore better reflects the potential benefits of these predicted changes through exposure changes to the affected populations. As shown, the average annual mean concentrations of $PM_{2.5}$ across populated areas of the East declines by roughly 9.7 percent (or 6.27 $\mu g/m^3$) in 2014. The population-weighted average mean concentration declined by 11 percent (or 1.21 $\mu g/m^3$) in 2014. This indicates the rule generates greater absolute air quality improvements in more populated, urban areas.

Table 4-1. Summary of Base Case $PM_{2.5}$ Air Quality and Changes Due to the Transport Rule.

		2014	
Statistic	**Base Case**	**Control Case**	**Percent Change**[a]
$PM_{2.5}$ ($\mu g/m^3$)			
Minimum Annual Mean	1.94	1.95	-0.5%
Maximum Annual Mean	31.3	31.05	0.8%
Average Annual Mean	6.95	6.27	9.7%
Pop-Weighted Average Annual Mean [b]	10.81	1.21	11%

[a] The percent change is defined as the control case value minus the base case value multiplied by 100. A negative value denotes an increase in $PM_{2.5}$ concentration.

[b] Calculated by summing the product of the projected CAMx grid-cell population and the estimated concentration, for that grid-cell and then dividing by the total population.

Table 4-2 provides information on the populations in 2014 that will experience improved PM air quality. Significant populations that live in areas with meaningful reductions in annual mean $PM_{2.5}$ concentrations resulting from the rule. As shown, in 2014, about 20 percent of the U.S. population located in the modeling domain are predicted to experience reductions of greater than 1.75 $\mu g/m^3$. Furthermore, over 43 percent of this population will benefit from reductions in annual mean $PM_{2.5}$ concentrations of greater than 1.25 $\mu g/m^3$ and almost 88 percent will live in areas with reductions of greater than 0.5 $\mu g/m^3$.

Table 4-2. Distribution of $PM_{2.5}$ Air Quality Improvements Over Population in 2014 Due to the Transport Rule for the Eastern U.S.

Change in Annual Mean $PM_{2.5}$ Concentrations (µg/m^3)	2014 Population[b]	
	Number (millions)	Percent (%)
0 > Δ $PM_{2.5}$ Conc ≤ 0.25	3,332,940	1%
0.25 > Δ $PM_{2.5}$ Conc ≤ 0.5	27,405,217	11%
0.5 > Δ $PM_{2.5}$ Conc ≤ 0.75	39,549,835	16%
0.75 > Δ $PM_{2.5}$ Conc ≤ 1.0	34,181,327	14%
1.0 > Δ $PM_{2.5}$ Conc ≤ 1.25	33,590,794	14%
1.25 > Δ $PM_{2.5}$ Conc ≤ 1.5	34,097,507	14%
1.5 > Δ $PM_{2.5}$ Conc ≤ 1.75	21,029,053	9%
Δ $PM_{2.5}$ Conc > 1.75	48,151,617	20%

[a] The change is defined as the control case value minus the base case value.

[b] Population counts and percentages are for the fraction of the national population located in the eastern 37 state modeling domain (as shown in Figure 4-1) considered in modeling health benefits for the rule.

4.2.3 Converting CAMx Outputs to Full-Season Profiles for Benefits Analysis

This study extracted hourly, surface-layer ozone concentrations for each grid-cell from the standard CAMx output file containing hourly average ozone values. These model predictions are used in conjunction with the observed concentrations obtained from the Aerometric Information Retrieval System (AIRS) to generate ozone concentrations for the entire ozone season.[14,15] The predicted changes in ozone concentrations from the future-year base case to future-year control scenario serve as inputs to the health and welfare C-R functions of the benefits analysis (i.e., BenMAP).

To estimate ozone-related health and welfare effects, full-season ozone data are required for every grid cell. Given available ozone monitoring data, we generated full-season ozone profiles for each location in the contiguous 48 States in two steps: (1) we combine monitored observations and modeled ozone predictions to interpolate hourly ozone concentrations to a grid of 8 km by 8 km population grid-cells, and (2) we converted these

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[14] The ozone season for this analysis is defined as the 5-month period from May to September; however, to estimate certain crop yield benefits, the modeling results were extended to include months outside the 5-month ozone season.

[15] Based on AIRS, there were 961 ozone monitors with sufficient data, i.e., 50 percent or more days reporting at least 9 hourly observations per day (8 am to 8 pm) during the ozone season.

full-season hourly ozone profiles to an ozone measure of interest, such as the daily average.[16,17]

4.2.4 Ozone Air Quality Results

This section provides a summary of the predicted ambient ozone concentrations from the CAMx model for the 2014 base case and changes associated with the rule. Table 4-3 provides those ozone metrics for grid cells in the Eastern U.S. that enter the C-R functions for health benefits endpoints. The population-weighted average reflects the baseline levels and predicted changes for more populated areas of the nation. This measure, therefore, will better reflect the potential benefits of these predicted changes through exposure changes to these populations.

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[16] The 12 km grid squares contain the population data used in the health benefits analysis model, BenMAP. See Chapter 5 for a discussion of this model.

[17] This approach is a generalization of planar interpolation that is technically referred to as enhanced Voronoi Neighbor Averaging (EVNA) spatial interpolation.

Table 4-3. Summary of CAMx Derived Population-Weighted Ozone Season Air Quality Metrics for Health Benefits Endpoints Due to the Transport Rule for the Eastern U.S.

	2014		
Statistic[a]	Base Case	Change [b]	Percent Change [b]
Population-Weighted Average (ppb) [c]			
Daily 8-Hour Average Concentration	45.3	0.24	0.5%

[a] This ozone metric is calculated at the CAMx grid-cell level for use in health effects estimates based on the results of spatial and temporal Voronoi Neighbor Averaging.

[b] The change is defined as the control case value minus the base case value. The percent change is the "Change" divided by the "Base Case," and then multiplied by 100 to convert the value to a percentage.

[c] Calculated by summing the product of the projected CAMx grid-cell population and the estimated CAMx grid-cell seasonal ozone concentration, and then dividing by the total population.

4.3 Visibility Degradation Estimates

Visibility degradation is often directly proportional to decreases in light transmittal in the atmosphere. Scattering and absorption by both gases and particles decrease light transmittance. To quantify changes in visibility, our analysis computes a light-extinction coefficient, based on the work of Sisler (1996), which shows the total fraction of light that is decreased per unit distance. This coefficient accounts for the scattering and absorption of light by both particles and gases, and accounts for the higher extinction efficiency of fine particles compared to coarse particles. Fine particles with significant light-extinction efficiencies include sulfates, nitrates, organic carbon, elemental carbon (soot), and soil (Sisler, 1996).

Based upon the light-extinction coefficient, we also calculated a unitless visibility index, called a "deciview," which is used in the valuation of visibility. The deciview metric provides a scale for perceived visual changes over the entire range of conditions, from clear to hazy. Under many scenic conditions, the average person can generally perceive a change of one deciview. The higher the deciview value, the worse the visibility. Thus, an improvement in visibility is a decrease in deciview value.

Table 4-4 provides the visibility improvements, measured in annual average deciviews, expected to occur in the Eastern and Western U.S. As shown, Class I visibility regions in the Eastern U.S., including such regions as the Great Smoky Mountains and Shenandoah, are expected to see significant improvements in visibility. By 2014, such regions in the Eastern U.S. are expected to see improvements of over 1 deciview (9 percent), and such regions in the Western U.S. are expected to see improvements of over 0.04 deciviews (or less than 1 percent).

Table 4-4. Summary of Basecase Recreational Visibility and Changes by Region: (annual average deciviews)

	2014		
Class I Visibility Regions	Base Case (Deciviews)	Change in Annual Average (Deciviews)	Percent Change in Annual Average (%)
Eastern US	14.35	1.36	9
Western US	9.62	0.04	< 1

4.4 References

Environ, 2009. Comprehensive Air Quality Model with Extensions Version 5 User's Guide. Environ International Corporation. Novato, CA. March 2009.

Sisler, J.F. July 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*. Fort Collins, CO: Cooperative Institute for Research in the Atmosphere, Colorado State University.

U.S. Environmental Protection Agency (EPA). 2007. Guidance on the Use of Models and Other Analyses for Demonstrating Attainment of Air Quality Goals for Ozone, $PM_{2.5}$, and Regional Haze. Office of Air Quality Planning and Standards, Research Triangle Park, NC.

U.S. Environmental Protection Agency (EPA). 2010. Air Quality Modeling Technical Support Document for the Transport Rule. Office of Air Quality Planning and Standards, Research Triangle Park, NC.

CHAPTER 5

BENEFITS ANALYSIS AND RESULTS

Synopsis

This chapter contains a subset of the estimated health and welfare benefits of the proposed Transport Rule remedy in 2014. This rule is expected to yield significant reductions in SO_2 and NO_x from EGUs, which in turn would lower overall ambient levels of $PM_{2.5}$ and ozone across much of the eastern U.S. In this chapter we quantify the health and welfare benefits resulting from these air quality improvements.

We estimate the monetized benefits of the proposed remedy to be \$120 billion to \$290 billion at a 3% discount rate and \$110 billion to \$270 billion at a 7% discount rate in 2014. The benefits of the alternative remedies may be found in the benefit-cost comparison chapter. All estimates are in 2006\$. We estimate the benefits of the proposed remedy using modeled changes in ambient pollution concentrations while the benefits of the alternate remedies are based on a benefit per ton approach described below. This benefits analysis accounts for both decreases and increases in emissions across the country resulting from aspects of the proposed provisions of the rule from reductions in NO_x and SO_2. These estimates omit the benefits from several important categories, including ecosystem benefits and the direct health benefits from reducing exposure to NO_2 and SO_2 due to time constraints. While not quantified here, because the level of SO_2 and NO_x emission reductions in 2012 exceed those in 2014, we expect the total benefits in 2012 to exceed those in 2014.

5.1 Overview

This chapter contains a subset of the estimated health and welfare benefits of the proposed and alternate rule remedies for the Transport Rule in 2014. The Transport Rule is expected to yield significant aggregate reductions in SO_2 and NO_x from EGUs, which in turn would lower overall ambient levels of $PM_{2.5}$ and ozone across much of the eastern U.S. The analysis in this chapter aims to characterize the benefits of these air quality changes by answering two key questions:

1. What are the health and welfare effects of changes in ambient particulate matter ($PM_{2.5}$) and ozone air quality resulting from reductions in precursors including NO_x and SO_2?
2. What is the economic value of these effects?

In this analysis we consider an array of health and welfare impacts attributable to changes in $PM_{2.5}$ and ozone air quality. The 2009 $PM_{2.5}$ Integrated Science Assessment (U.S. EPA, 2009d) and the 2006 ozone criteria document (U.S. EPA, 2006a) identify the human health effects associated with these ambient pollutants, which include premature mortality and a variety of morbidity effects associated with acute and chronic exposures. PM welfare effects include visibility impairment and materials damage. Ozone welfare effects include damages to agricultural and forestry sectors. NO_x welfare effects include aquatic and terrestrial acidification and nutrient enrichment (U.S. EPA, 2008f). SO_2 welfare effects include aquatic and terrestrial acidification and increased mercury methylation (U.S. EPA, 2008f). Though models exist for quantifying these ecosystem impacts, time and resource constraints precluded us from quantifying most of those effects in this analysis.

Table 5-1 summarizes the total monetized benefits of the proposed remedy in 2014. This table reflects the economic value of the change in $PM_{2.5}$ and ozone-related human health impacts occurring as a result of the proposed and alternate Transport Rule.

Table 5-2 summarizes the human health and welfare benefits categories contained within the primary benefits estimate, those categories that were unquantified due to limited data or time.

Table 5-1: Estimated monetized benefits of the proposed Transport Rule (billions of 2006$)[A]

Benefits Estimate	*Within Transport Region[B]*	*Outside Transport Region*	*Total*
Pope et al. (2002) $PM_{2.5}$ mortality and Bell et al. (2004) ozone mortality estimates			
Using a 3% discount rate	\$120 +B (\$10—\$360)	\$1.1 +B (\$0.09—\$3.3)	\$120 +B (\$10—\$360)
Using a 7% discount rate	\$110 +B (\$9—\$330)	\$0.9 +B (\$0.08—\$2.9)	\$110 +B (\$9—\$330)
Laden et al. (2006) $PM_{2.5}$ mortality and Levy et al. (2005) ozone mortality estimates			
Using a 3% discount rate	\$290 +B (\$26—\$840)	\$2.6 +B (\$0.2—\$7.5)	\$290 +B (\$26—\$840)
Using a 7% discount rate	\$260 +B (\$23—\$760)	\$2.4 +B (\$0.2—\$6.8)	\$270 +B (\$24—\$760)

[A] For notational purposes, unquantified benefits are indicated with a "B" to represent the sum of additional monetary benefits and disbenefits. Data limitations prevented us from quantifying these endpoints, and as such, these benefits are inherently more uncertain than those benefits that we were able to quantify. A detailed listing of unquantified health and welfare effects is provided in Table 5-2. Estimates here are subject to uncertainties discussed further in the body of the document

[B] Rounded to two significant figures.

The benefits analysis in this chapter relies on an array of data inputs—including air quality modeling, health impact functions and valuation estimates among others—which are themselves subject to uncertainty and may also in turn contribute to the overall uncertainty in this analysis. As a means of characterizing this uncertainty we employ two primary techniques. First, we use Monte Carlo methods for characterizing random sampling error associated with the concentration response functions from epidemiological studies and economic valuation functions. Second, because this characterization of random statistical error may omit important sources of uncertainty we also employ the results of an expert elicitation on the relationship between premature mortality and ambient $PM_{2.5}$ concentration (Roman et al., 2008); this provides additional insight into the likelihood of different outcomes and about the state of knowledge regarding the benefits estimates. Both approaches have different strengths and weaknesses, which are fully described in Chapter 5 of the PM NAAQS RIA (U.S. EPA, 2006).

Given that reductions in premature mortality dominate the size of the overall monetized benefits, more focus on uncertainty in mortality-related benefits gives us greater confidence in our uncertainty characterization surrounding total benefits. Certain EPA RIA's including the 2008 Ozone NAAQS RIA (U.S. EPA, 2008a) contained a suite of sensitivity analyses, only some of which we include here due in part to time constraints. In particular, these analyses characterized the sensitivity of the monetized benefits to the specification of alternate cessation lags and income growth adjustment factors. The estimated benefits increased or decreased in proportion to the specification of alternate income growth adjustments and cessation lags, making it possible for readers to infer the sensitivity of the results in this RIA to these parameters by referring to the PM NAAQS RIA (2006d) and Ozone NAAQS RIA (2008a).

For example, the use of an alternate lag structure would change the $PM_{2.5}$-related mortality benefits discounted at 3% discounted by between 10.4% and –27%; when discounted at 7%, these benefits change by between 31% and -49%. When applying higher and lower income growth adjustments, the monetary value of $PM_{2.5}$ and ozone-related premature changes between 30% and -10%; the value of chronic endpoints change between 5% and -2% and the value of acute endpoints change between 6% and -7%.

Below we include a new analysis (Figures 5-19 to 5-21) in which we bin the estimated number of avoided $PM_{2.5}$-related premature mortalities resulting from the implementation of the Transport Rule according to the projected 2014 baseline $PM_{2.5}$ air quality levels. This presentation is consistent with our approach to applying $PM_{2.5}$ mortality risk coefficients that have not been adjusted to incorporate an assumed threshold. The very large proportion of the avoided PM-related impacts we estimate in this analysis occur among populations exposed at or above the LML of each study, increasing our confidence in the PM mortality analysis. Approximately 80% of the avoided impacts occur at or above an annual mean $PM_{2.5}$ level of 10 $\mu g/m^3$ (the LML of the Laden et al. 2006 study); about 97% occur at or above an annual mean $PM_{2.5}$ level of 7.5 $\mu g/m^3$ (the LML of the Pope et al. 2002 study). As we model mortality impacts among populations exposed to levels of $PM_{2.5}$ that are successively lower than the LML of each study our confidence in the results diminishes. However, the analysis below confirms that the great majority of the impacts occur at or above each study's LML.

Table 5-2: Human Health and Welfare Effects of Pollutants Affected by the Transport Rule

Pollutant/ Effect	*Quantified and monetized in base estimate*	*Unquantified*
PM: health[a]	Premature mortality based on cohort study estimates[b] and expert elicitation estimates Hospital admissions: respiratory and cardiovascular Emergency room visits for asthma Nonfatal heart attacks (myocardial infarctions) Lower and upper respiratory illness Minor restricted activity days Work loss days Asthma exacerbations (among asthmatic populations) Respiratory symptoms (among asthmatic populations) Infant mortality	Low birth weight, pre-term birth and other reproductive outcomes Pulmonary function Chronic respiratory diseases other than chronic bronchitis Non-asthma respiratory emergency room visits UVb exposure (+/-)[c]
PM: welfare	Visibility in Class I areas in SE, SW, and CA regions	Household soiling Visibility in residential areas Visibility in non-class I areas and class 1 areas in NW, NE, and Central regions UVb exposure (+/-)[c] Global climate impacts[c]
Ozone: health	Premature mortality based on short-term study estimates Hospital admissions: respiratory Emergency room visits for asthma Minor restricted activity days School loss days	Chronic respiratory damage Premature aging of the lungs Non-asthma respiratory emergency room visits UVb exposure (+/-)[c]
Ozone: welfare	Decreased outdoor worker productivity	Yields for: --Commercial forests --Fruits and vegetables, and --Other commercial and noncommercial crops Damage to urban ornamental plants Recreational demand from damaged forest aesthetics Ecosystem functions UVb exposure (+/-)[c] Climate impacts
NO_2: health		Respiratory hospital admissions Respiratory emergency department visits Asthma exacerbation Acute respiratory symptoms Premature mortality Pulmonary function
NO_X: welfare		Commercial fishing and forestry from acidic deposition effects Commercial fishing, agriculture and forestry from

		nutrient deposition effects Recreation in terrestrial and estuarine ecosystems from nutrient deposition effects Other ecosystem services and existence values for currently healthy ecosystems Coastal eutrophication from nitrogen deposition effects
SO_2: health		Respiratory hospital admissions Asthma emergency room visits Asthma exacerbation Acute respiratory symptoms Premature mortality Pulmonary function
SO_X: welfare		Commercial fishing and forestry from acidic deposition effects Recreation in terrestrial and aquatic ecosystems from acid deposition effects Increased mercury methylation
Mercury: health		Incidence of neurological disorders Incidence of learning disabilities Incidences in developmental delays Potential cardiovascular effects including: --Altered blood pressure regulation --Increased heart rate variability --Incidences of Myocardial infarction Potential reproductive effects
Mercury: welfare		Impact on birds and mammals (e.g. reproductive effects) Impacts to commercial, subsistence and recreational fishing

[A] In addition to primary economic endpoints, there are a number of biological responses that have been associated with PM health effects including morphological changes and altered host defense mechanisms. The public health impact of these biological responses may be partly represented by our quantified endpoints.

[B] Cohort estimates are designed to examine the effects of long term exposures to ambient pollution, but relative risk estimates may also incorporate some effects due to shorter term exposures (see Kunzli et al., 2001 for a discussion of this issue). While some of the effects of short term exposure are likely to be captured by the cohort estimates, there may be additional premature mortality from short term PM exposure not captured in the cohort estimates included in the primary analysis.

[C] May result in benefits or disbenefits.

The benefits analysis presented in this chapter incorporates an array of policy and technical changes that the Agency has adopted since the publication of the benefits chapter accompanying the promulgated CAIR in 2005 (U.S. EPA, 2005):

1. *Incorporation of additional long-term PM mortality studies*. The 2005 CAIR analysis quantified PM-related mortality using a C-R function drawn the extended analysis of American Cancer Society (ACS) cohort, as reported in Pope et al. (2002). In this analysis we complement this estimate with a C-R function drawn from the Laden et al. (2006) reanalysis of the Harvard Six Cities (H6C) cohort. Rather than estimating PM-related

mortality using a single estimate, we now report two core estimates based on these ACS and H6C studies.

2. *Inclusion of twelve PM-mortality estimates based on EPA's expert elicitation study*. As a means of characterizing uncertainty in the PM-mortality relationship, in 2005 EPA undertook an expert elicitation (Roman et al., 2008). The 2005 CAIR analysis included the results of the pilot expert elicitation. This analysis presents PM-mortality estimates based on the 12 risk estimates derived from the final elicitation.[18]

3. *Quantification of short-term ozone mortality*. The 2005 CAIR analysis considered short-term ozone mortality in a sensitivity analysis. Consistent with recommendations from the 2008 National Academy of Sciences report (NAS), we incorporate short-term ozone mortality estimates in our primary benefits estimate (NRC, 2008).

4. *Use of a revised Value of Statistical Life (VSL)*. The Agency continues to update its guidance on valuing mortality risk reductions and until a final report is available, EPA now uses a distribution of VSL as recommended in EPA's guidance. The mean value of this distribution is $6.3 million (2006$). We discuss this issue in further depth below.

5. *Projection of baseline mortality rates*. Beginning in late 2005, after the completion of the final CAIR benefits analysis, the Agency began projecting into the future county-level mortality rates (Abt, 2005). Mortality rates are a key input to the calculation of air pollution-related premature mortality. Using projected rates generally results in a lower number of estimated excess mortality because of projected increases in life expectancy and concurrent reductions in risk of death at younger ages.

In general, for a given air quality change, the first four methodological changes increase, and the fifth decreases, the overall magnitude of the health impacts and monetized benefits compared to the approach used for the 2005 CAIR benefits analysis.

5.2 Benefits Analysis Methods

We follow a "damage-function" approach in calculating total benefits of the modeled changes in environmental quality. This approach estimates changes in individual health and

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[18] As we discuss below, the characterization of PM-related mortality using this expert elicitation responds in part to 2002 National Academy Sciences (NAS) recommendations regarding the propagation of uncertainty characterization throughout the benefits chapter.

welfare endpoints (specific effects that can be associated with changes in air quality) and assigns values to those changes assuming independence of the individual values. Total benefits are calculated simply as the sum of the values for all non-overlapping health and welfare endpoints. The "damage-function" approach is the standard method for assessing costs and benefits of environmental quality programs and has been used in several recent published analyses (Levy et al., 2009; Hubbell et al., 2009; Tagaris et al., 2009).

To assess economic value in a damage-function framework, the changes in environmental quality must be translated into effects on people or on the things that people value. In some cases, the changes in environmental quality can be directly valued, as is the case for changes in visibility. In other cases, such as for changes in ozone and PM, a health and welfare impact analysis must first be conducted to convert air quality changes into effects that can be assigned dollar values.

For the purposes of this RIA, the health impacts analysis (HIA) is limited to those health effects that are directly linked to ambient levels of air pollution and specifically to those linked to ozone and PM. There may be other, indirect health impacts associated with implementing emissions controls, such as occupational health impacts for coal miners.

The welfare impacts analysis is limited to changes in the environment that have a direct impact on human welfare. For this analysis, we are limited by the available data to examine impacts of changes in visibility in Class 1 areas. We also provide qualitative discussions of the impact of changes in other environmental and ecological effects, for example, changes in deposition of nitrogen and sulfur to terrestrial and aquatic ecosystems, but we are unable to place an economic value on these changes due to time and resource limitations.

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

5.2.1 Health Impact Assessment

The Health Impact Assessment (HIA) quantifies the changes in the incidence of adverse health impacts resulting from changes in human exposure to $PM_{2.5}$ and ozone air quality. HIAs are a well-established approach for estimating the retrospective or prospective change in adverse health impacts expected to result from population-level changes in exposure to pollutants (Levy et al. 2009). PC-based tools such as the environmental Benefits Mapping and Analysis Program (BenMAP) can systematize health impact analyses by applying a database of key input parameters, including health impact functions and population projections. Analysts have applied the HIA approach to estimate human health impacts resulting from hypothetical changes in pollutant levels (Hubbell et al. 2005; Davidson et al. 2007, Tagaris et al. 2009). EPA and others have relied upon this method to predict future changes in health impacts expected to result from the implementation of regulations affecting air quality (U.S. EPA, 2008a).

The HIA approach used in this analysis involves three basic steps: (1) utilizing CAMx-generated projections of $PM_{2.5}$ and ozone air quality and estimating the change in the spatial distribution of the ambient air quality; (2) determining the subsequent change in population-level exposure; (3) calculating health impacts by applying concentration-response relationships drawn from the epidemiological literature (Hubbell et al. 2009) to this change in population exposure.

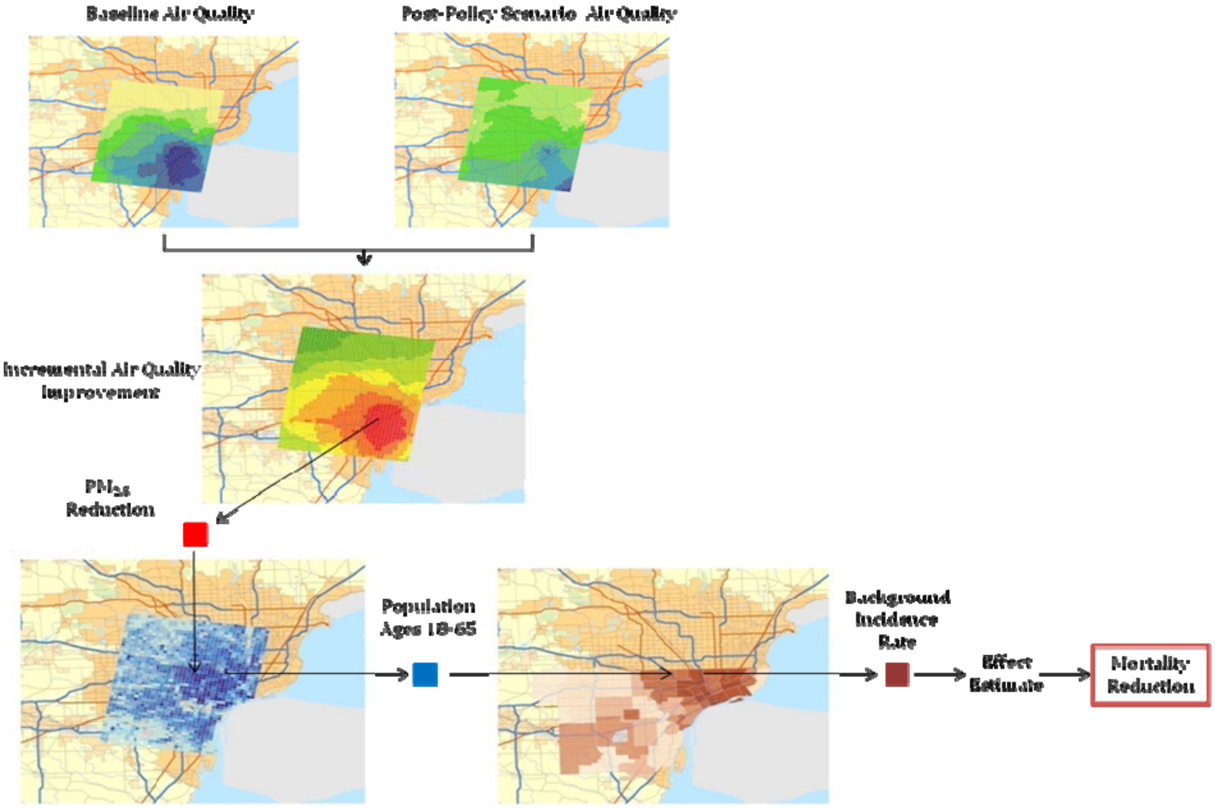
A typical health impact function might look as follows:

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

where y_0 is the baseline incidence rate for the health endpoint being quantified (for example, a health impact function quantifying changes in mortality would use the baseline, or background, mortality rate for the given population of interest); *Pop* is the population affected by the change in air quality; Δx is the change in air quality; and β is the effect coefficient drawn from the epidemiological study. Tools such as BenMAP can systematize the HIA calculation process, allowing users to draw upon a library of existing air quality monitoring data, population data and health impact functions.

Figure 5-1 provides a simplified overview of this approach.

Figure 5-1: Illustration of BenMAP Approach



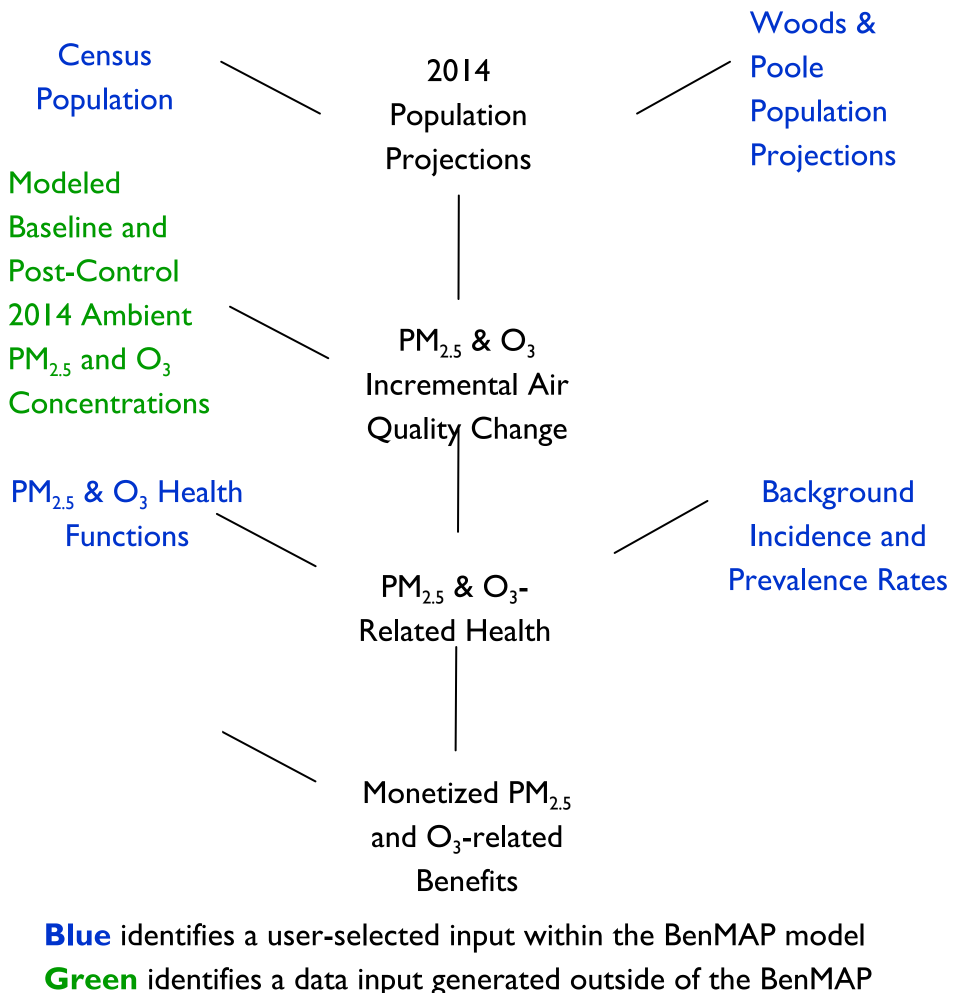
5.2.2 Economic Valuation of Health Impacts

After quantifying the change in adverse health impacts, the final step is to estimate the economic value of these avoided impacts. The appropriate economic value for a change in a health effect depends on whether the health effect is viewed *ex ante* (before the effect has occurred) or *ex post* (after the effect has occurred). Reductions in ambient concentrations of air pollution generally lower the risk of future adverse health effects by a small amount for a large population. The appropriate economic measure is therefore *ex ante* Willingness to Pay (WTP) for changes in risk. However, epidemiological studies generally provide estimates of the relative risks of a particular health effect avoided due to a reduction in air pollution. A convenient way to use this data in a consistent framework is to convert probabilities to units of avoided statistical incidences. This measure is calculated by dividing individual WTP for a risk reduction by the related observed change in risk. For example, suppose a measure is able to reduce the risk of premature mortality from 2 in 10,000 to 1 in 10,000 (a reduction of 1 in 10,000). If individual WTP for this risk reduction is $100, then the WTP for an avoided statistical premature mortality amounts to $1 million ($100/0.0001 change in risk). Using this approach, the size of the affected population is automatically taken into account by the number of incidences predicted by epidemiological studies applied to the relevant population. The same type of calculation can produce values for statistical incidences of other health endpoints.

For some health effects, such as hospital admissions, WTP estimates are generally not available. In these cases, we use the cost of treating or mitigating the effect as a primary estimate. For example, for the valuation of hospital admissions we use the avoided medical costs as an estimate of the value of avoiding the health effects causing the admission. These cost of illness (COI) estimates generally (although not in every case) understate the true value of reductions in risk of a health effect. They tend to reflect the direct expenditures related to treatment but not the value of avoided pain and suffering from the health effect.

We use the BenMAP model (Abt Associates, 2008) to estimate the health impacts and monetized health benefits for the proposed remedy. Figure 5-2 below shows the data inputs and outputs for the BenMAP model.

Figure 5-2: Data inputs and outputs for the BenMAP model



5.2.3 Benefit Per-Ton Estimates

Benefit per-ton (BPT) estimates quantify the health impacts and monetized human health benefits of an incremental change in air pollution precursor emissions. In situations when we are unable to perform air quality modeling because of resource or time constraints, this approach can provide a reliable estimate of the benefits of emission reduction scenarios. EPA has used the benefit per-ton technique in previous RIAs, including the recent Ozone NAAQS RIA (U.S. EPA, 2008) and NO_2 NAAQS RIA (U.S. EPA, 2010b). Time constraints prevented the Agency from modeling the air quality changes resulting from either the intrastate and direct control remedies or the more and less stringent SO_2 caps and so we estimate a subset of these health benefits using $PM_{2.5}$ benefit per-ton estimates. The assessment of the alternate scenarios omits ozone-related benefits for two reasons. First, the overall level of ozone-related benefits in the modeled case is relatively small compared to those associated with $PM_{2.5}$ reductions (see table 5-17 below), due in part to the fairly modest summer time NO_x emission reductions under this scenario. The level of summertime NO_x emission reductions of the alternate scenarios are very similar to the modeled scenario, suggesting that the omission of ozone-related impacts would not greatly influence the overall level of benefits. Second, the complex non-linear chemistry of ozone formation introduces uncertainty to the development and application of a benefit per ton estimate. Taken together, these factors argued against developing an ozone benefit per ton estimate for this RIA.

For this analysis, EPA applies $PM_{2.5}$ BPT estimates that are methodologically consistent with those reported in Fann et al. (2009), but have been adjusted for this analysis to better match the spatial distribution of air quality changes projected for the Transport Rule. To derive the BPT estimates for this analysis, we:

1. *Quantified the $PM_{2.5}$ -related human and monetized health benefits of the SO_2 emission reductions of the proposed remedy*. We first quantified the health impacts and monetized benefits of total $PM_{2.5}$ mass formed from the SO_2 reductions of the proposed remedy, allowing us to isolate the PM air quality impacts from SO_2 reductions alone.[19] This procedure allowed us to develop $PM_{2.5}$ BPT estimates that quantified the $PM_{2.5}$-related benefits of incremental changes in SO_2 emissions. Because reductions in NO_x emissions are relatively small in each scenario, and previous EPA modeling indicates that $PM_{2.5}$ formation is less sensitive to NO_x emission reductions on a per-µg/m^3 basis (Fann et al, 2009), we did not quantify the NO_x-related $PM_{2.5}$ changes.

2. *Divided the health impacts and monetized benefits by the emission reduction.* This calculation yields BPT estimates for PM-related SO_2.

The resulting BPT estimates were then multiplied by the projected SO_2 emission reductions for the Direct Control and Interstate Trading scenarios to produce an estimate of the PM- and

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[19] The Transport Rule includes both SO_2 and NOx emissions reductions. In general SO_2 is a precursor to particulate sulfate and NOx is a precursor to particulate nitrate. However, there are also several interactions between the $PM_{2.5}$ precursors which cannot be easily quantified. For example, under conditions in which SO_2 levels are reduced by a substantial margin, "nitrate replacement" may occur. This occurs when particulate ammonium sulfate concentrations are reduced, thereby freeing up excess gaseous ammonia. The excess ammonia is then available to react with gaseous nitric acid to form particulate nitrate. The impact of nitrate replacement is also affected by concurrent NOx reductions. NOx reductions can lead to decreases in nitrate, which competes with the process of nitrate replacement. NOx reductions can also lead to reductions in photochemical by-products which can reduce both particulate sulfate and secondary organic carbon PM concentrations. Due to the complex nature of these interactions, EPA performed a sensitivity modeling analysis in which only SO_2 emissions were reduced. We calculated benefits from this air quality modeling run to generate an SO_2-only benefit per ton estimate. The results of the SO_2-only sensitivity run may be found in the EPA Benefits TSD [Docket No. EPA-HQ-OAR-2010-0491]

ozone-related health impacts and monetized benefits. There is no analogous approach for estimating a BPT for visibility, and so the benefits of the alternative remedies omit this important monetized benefit.

5.3 Uncertainty Characterization

In any complex analysis using estimated parameters and inputs from numerous models, there are likely to be many sources of uncertainty and this analysis is no exception. As outlined both in this and preceding chapters, many inputs were used to derive the estimate of benefits for the proposed remedy, including emission inventories, air quality models (with their associated parameters and inputs), epidemiological health effect estimates, estimates of values (both from WTP and COI studies), population estimates, income estimates, and estimates of the future state of the world (i.e., regulations, technology, and human behavior). Each of these inputs may be uncertain and, depending on its role in the benefits analysis, may have a disproportionately large impact on estimates of total benefits. For example, emissions estimates are used in the first stage of the analysis. As such, any uncertainty in emissions estimates will be propagated through the entire analysis. When compounded with uncertainty in later stages, small uncertainties in emission levels can lead to large impacts on total benefits.

The National Research Council (NRC) (2002, 2008) 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 general, the NRC concluded that EPA's general methodology for calculating the benefits of reducing air pollution is reasonable and informative in spite of inherent uncertainties. Since the publication of these reports, EPA's Office of Air and Radiation (OAR) continues to make progress toward the goal of characterizing the aggregate impact of uncertainty in key modeling elements on both health incidence and benefits estimates in two key ways: Monte Carlo analysis and expert-derived concentration-response functions. In this analysis, we use both of these two methods to assess uncertainty quantitatively, as well as provide a qualitative assessment for those aspects that we are unable to address quantitatively.

First, we used Monte Carlo methods for characterizing random sampling error associated with the concentration response functions from epidemiological studies and random effects modeling to characterize both sampling error and variability across the economic valuation functions. Monte Carlo simulation uses random sampling from distributions of parameters to characterize the effects of uncertainty on output variables, such as incidence of premature mortality. Specifically, we used Monte Carlo methods to generate confidence intervals around the estimated health impact and dollar benefits. The reported standard errors in the epidemiological studies determined the distributions for individual effect estimates.

Second, because characterization of random statistical error omits important sources of uncertainty (e.g., in the functional form of the model—e.g., whether or not a threshold may exist), we also incorporate the results of an expert elicitation on the relationship between premature mortality and ambient $PM_{2.5}$ concentration (Roman et al., 2008). Use of the expert elicitation and incorporation of the standard errors approaches provide insights into the likelihood of different outcomes and about the state of knowledge regarding the benefits estimates. However, there are significant unquantified uncertainties present in upstream inputs including emission and air quality. Both approaches have different strengths and weaknesses, which are fully described in Chapter 5 of the PM NAAQS RIA (U.S. EPA, 2006).

In benefit analyses of air pollution regulations conducted to date, the estimated impact of reductions in premature mortality has accounted for 85% to 95% of total monetized benefits. Therefore, it is particularly important to attempt to characterize the uncertainties associated with reductions in premature mortality. The health impact functions used to estimate avoided premature deaths associated with reductions in ozone have associated standard errors that represent the statistical errors around the effect estimates in the underlying epidemiological studies. In our results, we report credible intervals based on these standard errors, reflecting the uncertainty in the estimated change in incidence of avoided premature deaths. We also provide multiple estimates, to reflect model uncertainty between alternative study designs.

For premature mortality associated with exposure to PM, we follow the same approach used in the RIA for 2006 PM NAAQS (U.S. EPA, 2006), presenting two empirical estimates of premature deaths avoided, and a set of twelve estimates based on results of the expert elicitation study. Even these multiple characterizations, including confidence intervals, omit the contribution to overall uncertainty of uncertainty in air quality changes, baseline incidence rates, populations exposed and transferability of the effect estimate to diverse locations. Furthermore, the approach presented here does not yet include methods for addressing correlation between input parameters and the identification of reasonable upper and lower bounds for input distributions characterizing uncertainty in additional model elements. As a result, the reported confidence intervals and range of estimates give an incomplete picture about the overall uncertainty in the estimates. This information should be interpreted within the context of the larger uncertainty surrounding the entire analysis.

In 2006 the EPA requested an NAS study to evaluate the extent to which the epidemiological literature to that point improved the understanding of ozone-related mortality. The NAS found that short-term ozone exposure was likely to contribute to ozone-related mortality (NRC, 2008) and issued a series of recommendations to EPA, including that the Agency should:

1. Present multiple short-term ozone mortality estimates, including those based on multi-city analyses such as the National Morbidity, Mortality and Air Pollution Study (NMMAPS) as well as meta-analytic studies.
2. Report additional risk metrics, including the percentage of baseline mortality attributable to short-term exposure.
3. Remove reference to a no-causal relationship between ozone exposure and premature mortality.

The quantification and presentation of ozone-related premature mortality in this chapter is responsive to these NRC recommendations and generally consistent with EPA's recent ozone reconsideration analysis (U.S. EPA, 2010a).

Some key sources of uncertainty in each stage of both the PM and ozone health impact assessment are the following:

- gaps in scientific data and inquiry;
- variability in estimated relationships, such as epidemiological effect estimates, introduced through differences in study design and statistical modeling;
- errors in measurement and projection for variables such as population growth rates;
- errors due to misspecification of model structures, including the use of surrogate variables, such as using PM_{10} when $PM_{2.5}$ is not available, excluded variables, and simplification of complex functions; and
- biases due to omissions or other research limitations.

In Table 5-3 we summarize some of the key uncertainties in the benefits analysis.

Table 5-3. Primary Sources of Uncertainty in the Benefits Analysis

1. *Uncertainties Associated with Impact Functions*
    - The value of the ozone or PM effect estimate in each impact function.
    - Application of a single impact function to pollutant changes and populations in all locations.
    - Similarity of future-year impact functions to current impact functions.
    - Correct functional form of each impact function.
    - Extrapolation of effect estimates beyond the range of ozone or PM concentrations observed in the source epidemiological study.
    - Application of impact functions only to those subpopulations matching the original study population.
2. *Uncertainties Associated with CAMx-Modeled Ozone and PM Concentrations*
    - Responsiveness of the models to changes in precursor emissions from the control policy.
    - Projections of future levels of precursor emissions, especially ammonia and crustal materials.
    - Lack of ozone and $PM_{2.5}$ monitors in all rural areas requires extrapolation of observed ozone data from urban to rural areas.
3. *Uncertainties Associated with PM Mortality Risk*
    - Limited scientific literature supporting a direct biological mechanism for observed epidemiological evidence.
    - Direct causal agents within the complex mixture of PM have not been identified.
    - The extent to which adverse health effects are associated with low-level exposures that occur many times in the year versus peak exposures.
    - The extent to which effects reported in the long-term exposure studies are associated with historically higher levels of PM rather than the levels occurring during the period of study.
    - Reliability of the $PM_{2.5}$ monitoring data in reflecting actual $PM_{2.5}$ exposures.
4. *Uncertainties Associated with Possible Lagged Effects*
    - The portion of the PM-related long-term exposure mortality effects associated with changes in annual PM levels that would occur in a single year is uncertain as well as the portion that might occur in subsequent years.
5. *Uncertainties Associated with Baseline Incidence Rates*
    - Some baseline incidence rates are not location specific (e.g., those taken from studies) and therefore may not accurately represent the actual location-specific rates.
    - Current baseline incidence rates may not approximate well baseline incidence rates in 2014.
    - Projected population and demographics may not represent well future-year population and demographics.
6. *Uncertainties Associated with Economic Valuation*
    - Unit dollar values associated with health and welfare endpoints are only estimates of mean WTP and therefore have uncertainty surrounding them.
    - Mean WTP (in constant dollars) for each type of risk reduction may differ from current estimates because of differences in income or other factors.
7. *Uncertainties Associated with Aggregation of Monetized Benefits*
    - Health and welfare benefits estimates are limited to the available impact functions. Thus, unquantified or unmonetized benefits are not included.

5.4 Benefits Analysis Data Inputs

In Figure 5-2, we summarized the key data inputs to the health impact and economic valuation estimate. Below we summarize the data sources for each of these inputs, including demographic projections, effect coefficients, incidence rates and economic valuation. We indicate where we have updated key data inputs since the 2005 CAIR benefits analysis.

5.4.1 Demographic Data

Quantified and monetized human health impacts depend on the demographic characteristics of the population, including age, location, and income. We use projections based on economic forecasting models developed by Woods and Poole, Inc (Woods and Poole, 2008). The Woods and Poole (WP) database contains county-level projections of population by age, sex, and race out to 2030. Projections in each county are determined simultaneously with every other county in the United States to take into account patterns of economic growth and migration. The sum of growth in county-level populations is constrained to equal a previously determined national population growth, based on Bureau of Census estimates (Hollman et al., 2000). According to WP, linking county-level growth projections together and constraining to a national-level total growth avoids potential errors introduced by forecasting each county independently. County projections are developed in a four-stage process:

1. First, national-level variables such as income, employment, and populations are forecasted.

2. Second, employment projections are made for 172 economic areas defined by the Bureau of Economic Analysis, using an "export-base" approach, which relies on linking industrial-sector production of non-locally consumed production items, such as outputs from mining, agriculture, and manufacturing with the national economy. The export-based approach requires estimation of demand equations or calculation of historical growth rates for output and employment by sector.

3. Third, population is projected for each economic area based on net migration rates derived from employment opportunities and following a cohort-component method based on fertility and mortality in each area.

4. Fourth, employment and population projections are repeated for counties, using the economic region totals as bounds. The age, sex, and race distributions for each region or

county are determined by aging the population by single year of age by sex and race for each year through 2014 based on historical rates of mortality, fertility, and migration.

5.4.2 Effect Coefficients

The first step in selecting effect coefficients is to identify the health endpoints to be quantified. We base our selection of health endpoints on consistency with EPA's Integrated Science Assessments (which replace the Criteria Document), with input and advice from the EPA Science Advisory Board - Health Effects Subcommittee (SAB-HES), a scientific review panel specifically established to provide advice on the use of the scientific literature in developing benefits analyses for air pollution regulations (http://www.epa.gov/sab/). In general, we follow a weight of evidence approach, based on the biological plausibility of effects, availability of concentration-response functions from well conducted peer-reviewed epidemiological studies, cohesiveness of results across studies, and a focus on endpoints reflecting public health impacts (like hospital admissions) rather than physiological responses (such as changes in clinical measures like Forced Expiratory Volume (FEV1)).

There are several types of data that can support the determination of types and magnitude of health effects associated with air pollution exposures. These sources of data include toxicological studies (including animal and cellular studies), human clinical trials, and observational epidemiology studies. All of these data sources provide important contributions to the weight of evidence surrounding a particular health impact. However, only epidemiology studies provide direct concentration-response relationships which can be used to evaluate population-level impacts of reductions in ambient pollution levels in a health impact assessment.

For the data-derived estimates, we relied on the published scientific literature to ascertain the relationship between PM and adverse human health effects. We evaluated epidemiological studies using the selection criteria summarized in Table 5-4. These criteria include consideration of whether the study was peer-reviewed, the match between the pollutant studied and the pollutant of interest, the study design and location, and characteristics of the study population, among other considerations. The selection of C-R functions for the benefits analysis is guided by the goal of achieving a balance between comprehensiveness and scientific defensibility. In general, the use of results from more than a single study can provide a more robust estimate of the relationship between a pollutant and a given health effect. However, there are often differences between studies examining the same endpoint, making it difficult to pool the results in a consistent manner. For example, studies may examine different pollutants or different age groups. For this reason, we consider very carefully the set of studies available examining each endpoint and select a consistent subset that provides a good balance of population coverage and match with the pollutant of interest. In many cases, either because of a lack of multiple studies, consistency problems, or clear superiority in the quality or comprehensiveness of one study over others, a single published study is selected as the basis of the effect estimate.

Table 5-4. Criteria Used when Selecting C-R functions

Consideration	*Comments*
Peer-Reviewed Research	Peer-reviewed research is preferred to research that has not undergone the peer-review process.
Study Type	Among studies that consider chronic exposure (e.g., over a year or longer), prospective cohort studies are preferred over ecological studies because they control for important individual-level confounding variables that cannot be controlled for in ecological studies.
Study Period	Studies examining a relatively longer period of time (and therefore having more data) are preferred, because they have greater statistical power to detect effects. More recent studies are also preferred because of possible changes in pollution mixes, medical care, and lifestyle over time. However, when there are only a few studies available, studies from all years will be included.
Population Attributes	The most technically appropriate measures of benefits would be based on impact functions that cover the entire sensitive population but allow for heterogeneity across age or other relevant demographic factors. In the absence of effect estimates specific to age, sex, preexisting condition status, or other relevant factors, it may be appropriate to select effect estimates that cover the broadest population to match with the desired outcome of the analysis, which is total national-level health impacts. When available, multi-city studies are preferred to single city studies because they provide a more generalizable representation of the C-R function.
Study Size	Studies examining a relatively large sample are preferred because they generally have more power to detect small magnitude effects. A large sample can be obtained in several ways, either through a large population or through repeated observations on a smaller population (e.g., through a symptom diary recorded for a panel of asthmatic children).
Study Location	U.S. studies are more desirable than non-U.S. studies because of potential differences in pollution characteristics, exposure patterns, medical care system, population behavior, and lifestyle.
Pollutants Included in Model	When modeling the effects of ozone and PM (or other pollutant combinations) jointly, it is important to use properly specified impact functions that include both pollutants. Using single-pollutant models in cases where both pollutants are expected to affect a health outcome can lead to double-counting when pollutants are correlated.
Measure of PM	For this analysis, impact functions based on $PM_{2.5}$ are preferred to PM_{10} because of the focus on reducing emissions of $PM_{2.5}$ precursors, and because air quality modeling was conducted for this size fraction of PM. Where $PM_{2.5}$ functions are not available, PM_{10} functions are used as surrogates, recognizing that there will be potential downward (upward) biases if the fine fraction of PM_{10} is more (less) toxic than the coarse fraction.
Economically Valuable Health Effects	Some health effects, such as forced expiratory volume and other technical measurements of lung function, are difficult to value in monetary terms. These health effects are not quantified in this analysis.
Non-overlapping Endpoints	Although the benefits associated with each individual health endpoint may be analyzed separately, care must be exercised in selecting health endpoints to include in the overall benefits analysis because of the possibility of double-counting of benefits.

When several effect estimates for a pollutant and a given health endpoint have been selected, they are quantitatively combined or pooled to derive a more robust estimate of the relationship. The BenMAP Technical Appendices provides details of the procedures used to combine multiple impact functions (Abt Associates, 2008). In general, we used fixed or random effects models to pool estimates from different studies of the same endpoint. Fixed effects pooling simply weights each study's estimate by the inverse variance, giving more weight to studies with greater statistical power (lower variance). Random effects pooling accounts for both within-study variance and between-study variability, due, for example, to differences in population susceptibility. We used the fixed effects model as our null hypothesis and then determined whether the data suggest that we should reject this null hypothesis, in which case we would use the random effects model. Pooled impact functions are used to estimate hospital admissions and asthma exacerbations. For more details on methods used to pool incidence estimates, see the BenMAP Manual Appendices (Abt Associates, 2008), which are available with the BenMAP software at http://www.epa.gov/benmap.html.

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

The specific studies from which effect estimates for the primary analysis are drawn are included in Table 5-5. We highlight in blue those studies that have been added since the 2005 CAIR benefits analysis and incorporated into the central benefits estimate. In all cases where effect estimates are drawn directly from epidemiological studies, standard errors are used as a partial representation of the uncertainty in the size of the effect estimate. Below we provide the basis for selecting these studies.

Table 5-5. Health Endpoints and Epidemiological Studies Used to Quantify Health Impacts[a]

Endpoint	Pollutant	Study	Study Population
Premature Mortality			
Premature mortality—daily time series	O3 (8-hour max)	Bell et al.(2004) (NMMAPS study) Huang et al. (2004) (multi-city) Schwartz (2005) (multi-city) Meta-analyses: Bell et al. (2005) Ito et al. (2005) Levy et al. (2005)	All ages
Premature mortality—cohort study, all-cause	$PM_{2.5}$ (annual avg)	Pope et al. (2002) Laden et al. (2006)	>29 years >25 years
Premature mortality, total exposures	$PM_{2.5}$ (annual avg)	Expert Elicitation (Roman et al., 2008)	>24 years
Premature mortality—all-cause	$PM_{2.5}$ (annual avg)	Woodruff et al. (2006)	Infant (<1 year)
Chronic Illness			
Chronic bronchitis	$PM_{2.5}$ (annual avg)	Abbey et al. (1995)	>26 years
Nonfatal heart attacks	$PM_{2.5}$ (24-hour avg)	Peters et al. (2001)	Adults (>18 years)
Hospital Admissions			
Respiratory	O_3 (8-hour max)	*Pooled estimate:* Schwartz (1995)—ICD 460–519 (all resp) Schwartz (1994a; 1994b)—ICD 480–486 (pneumonia) Moolgavkar et al. (1997)—ICD 480–487 (pneumonia) Schwartz (1994b)—ICD 491–492, 494–496 (COPD) Moolgavkar et al. (1997)—ICD 490–496 (COPD)	>64 years
		Burnett et al. (2001)	<2 years
	$PM_{2.5}$ (24-hour avg)	*Pooled estimate:* Moolgavkar (2003)—ICD 490–496 (COPD) Ito (2003)—ICD 490–496 (COPD)	>64 years
	$PM_{2.5}$ (24-hour avg)	Moolgavkar (2000)—ICD 490–496 (COPD)	20–64 years
	$PM_{2.5}$ (24-hour avg)	Ito (2003)—ICD 480–486 (pneumonia)	>64 years
	$PM_{2.5}$ (24-hour	Sheppard (2003)—ICD 493 (asthma)	<65 years

Endpoint	Pollutant	Study	Study Population
	avg)		
Cardiovascular	$PM_{2.5}$ (24-hour avg)	*Pooled estimate:* Moolgavkar (2003)—ICD 390–429 (all cardiovascular) Ito (2003)—ICD 410–414, 427–428 (ischemic heart disease, dysrhythmia, heart failure)	>64 years
	$PM_{2.5}$ (24-hour avg)	Moolgavkar (2000)—ICD 390–429 (all cardiovascular)	20–64 years
Asthma-related ER visits	O_3 (8-hour max)	*Pooled estimate:* Peel et al. (2005) Wilson et al.(2005)	 All ages All ages
	$PM_{2.5}$ (24-hour avg)	Norris et al. (1999)	0–18 years
Other Health Endpoints			
Acute bronchitis	$PM_{2.5}$ (annual avg)	Dockery et al. (1996)	8–12 years
Upper respiratory symptoms	PM_{10} (24-hour avg)	Pope et al. (1991)	Asthmatics, 9–11 years
Lower respiratory symptoms	$PM_{2.5}$ (24-hour avg)	Schwartz and Neas (2000)	7–14 years
Asthma exacerbations	$PM_{2.5}$ (24-hour avg)	Pooled estimate: Ostro et al. (2001) (cough, wheeze and shortness of breath) Vedal et al. (1998) (cough)	6–18 years[b]
Work loss days	$PM_{2.5}$ (24-hour avg)	Ostro (1987)	18–65 years
School absence days	O_3 (8-hour max)	*Pooled estimate:* Gilliland et al. (2001) Chen et al. (2000)	5–17 years[c]
Minor Restricted Activity Days (MRADs)	O3 (8-hour max)	Ostro and Rothschild (1989)	18–65 years
	$PM_{2.5}$ (24-hour avg)	Ostro and Rothschild (1989)	18–65 years

[a] Studies or air quality metrics highlighted in blue represent updates incorporated since the 2005 CAIR RIA

[b] The original study populations were 8 to 13 for the Ostro et al. (2001) study and 6 to 13 for the Vedal et al. (1998) study. Based on advice from the Science Advisory Board Health Effects Subcommittee (SAB-HES), we extended the applied population to 6 to 18, reflecting the common biological basis for the effect in children in the broader age group. See: U.S. Science Advisory Board. 2004. Advisory Plans for Health Effects Analysis in the Analytical Plan for EPA's Second Prospective Analysis –Benefits and Costs of the Clean Air Act, 1990—2020. EPA-SAB-COUNCIL-ADV-04-004. See also National Research Council (NRC). 2002. *Estimating the Public Health Benefits of Proposed Air Pollution Regulations*. Washington, DC: The National Academies Press.

[c] Gilliland et al. (2001) studied children aged 9 and 10. Chen et al. (2000) studied children 6 to 11. Based on recent advice from the National Research Council and the EPA SAB-HES, we have calculated reductions in school absences for all school-aged children based on the biological similarity between children aged 5 to 17.

5.4.2.1 $PM_{2.5}$ Premature Mortality Effect Coefficients

Both long- and short-term exposures to ambient levels of $PM_{2.5}$ air pollution have been associated with increased risk of premature mortality. The size of the mortality risk estimates from epidemiological studies, the serious nature of the effect itself, and the high monetary value ascribed to prolonging life make mortality risk reduction the most significant health endpoint quantified in this analysis.

Although a number of uncertainties remain to be addressed by continued research (NRC, 2002), a substantial body of published scientific literature documents the correlation between elevated $PM_{2.5}$ concentrations and increased mortality rates (U.S. EPA, 2009d). Time-series methods have been used to relate short-term (often day-to-day) changes in $PM_{2.5}$ concentrations and changes in daily mortality rates up to several days after a period of elevated $PM_{2.5}$ concentrations. Cohort methods have been used to examine the potential relationship between community-level PM exposures over multiple years (i.e., long-term exposures) and community-level annual mortality rates. Researchers have found statistically significant associations between $PM_{2.5}$ and premature mortality using both types of studies. In general, the risk estimates based on the cohort studies are larger than those derived from time-series studies. Cohort analyses are thought to better capture the full public health impact of exposure to air pollution over time, because they account for the effects of long-term exposures and possibly some component of short-term exposures (Kunzli et al., 2001; NRC, 2002). This section discusses some of the issues surrounding the estimation of $PM_{2.5}$-related premature mortality. To demonstrate the sensitivity of the benefits estimates to the specific sources of information regarding the impact of $PM_{2.5}$ exposures on the risk of premature death, we are providing estimates in our results tables based on studies derived from the epidemiological literature and from the EPA sponsored expert elicitation. The epidemiological studies from which these estimates are drawn are described below. The expert elicitation project and the derivation of effect estimates from the expert elicitation results are described in the 2006 $PM_{2.5}$ NAAQS RIA and Roman et al. (2008). In the interest of brevity we do not repeat those details here. However, Figure 5-18 summarizes the estimated $PM_{2.5}$-related premature mortalities avoided using risk estimates drawn from the expert elicitation.

Over a dozen epidemiological studies have found significant associations between various measures of long-term exposure to PM and elevated rates of annual mortality, beginning with Lave and Seskin (1977). Most of the published studies found positive (but not always statistically significant) associations with available PM indices such as total suspended particles

(TSP). However, exploration of alternative model specifications sometimes raised questions about causal relationships (e.g., Lipfert et al., 1989). These early "ecological cross-sectional" studies (Lave and Seskin, 1977; Ozkaynak and Thurston, 1987) were criticized for a number of methodological limitations, particularly for inadequate control at the individual level for variables that are potentially important in causing mortality, such as wealth, smoking, and diet.

Over the last 17 years, several studies using "prospective cohort" designs have been published that appear to be consistent with the earlier body of literature. These new "prospective cohort" studies reflect a significant improvement over the earlier work because they include individual level information with respect to health status and residence. The most extensive analyses have been based on data from two prospective cohort groups, often referred to as the Harvard "Six-Cities Study" (Dockery et al., 1993; Laden et al., 2006) and the "American Cancer Society or ACS study" (Pope et al., 1995; Pope et al., 2002; Pope et al., 2004, Krewski et al. 2009); these studies have found consistent relationships between fine particle indicators and premature mortality across multiple locations in the United States. A third major data set comes from the California-based 7th Day Adventist Study (e.g., Abbey et al., 1999), which reported associations between long-term PM exposure and mortality in men. Results from this cohort, however, have been inconsistent, and the air quality results are not geographically representative of most of the United States, and the lifestyle of the population is not reflective of much of the U.S. population. Analysis is also available for a cohort of adult male veterans diagnosed with hypertension has been examined (Lipfert et al., 2000; Lipfert et al., 2003, 2006). The characteristics of this group differ from the cohorts in the Six-Cities, ACS, and 7th Day Adventist studies with respect to income, race, health status, and smoking status. Unlike previous long-term analyses, this study found some associations between mortality and ozone but found inconsistent results for PM indicators. Because of the selective nature of the population in the veteran's cohort, we have chosen not to include any effect estimates from the Lipfert et al. (2000) study in our benefits assessment.

Given their consistent results and broad geographic coverage, and importance in informing the NAAQS development process, the Six-Cities and ACS data have been particularly important in benefits analyses. The credibility of these two studies is further enhanced by the fact that the initial published studies (Pope et al., 1995 and Dockery et al., 1993) were subject to extensive reexamination and reanalysis by an independent team of scientific experts commissioned by the Health Effect Institute (HEI) (Krewski et al., 2000). The final results of the reanalysis were then independently peer reviewed by a Special Panel of the HEI Health Review Committee. The results of these reanalyses confirmed and expanded the conclusions of the original investigators. While the HEI reexamination lends credibility to the original studies, it also highlights sensitivities concerning the relative impact of various pollutants, such as SO_2, the potential role of education in mediating the association between pollution and mortality, and the influence of spatial correlation modeling. Further confirmation and extension of the findings of the 1993 Six City Study and the 1995 ACS study were recently completed using more recent air quality and a longer follow-up period for the ACS cohort was published over the past several years (Pope et al., 2002, 2004; Laden et al., 2006, Krewski et al. 2009). The follow up to the Harvard Six City Study both confirmed the effect size from the first analysis and provided additional confirmation that reductions in $PM_{2.5}$ are likely to result in reductions in the risk of premature death. This additional evidence stems from the observed reductions in $PM_{2.5}$ in each city during the extended follow-up period. Laden et al. (2006) found that mortality rates consistently went down at a rate proportionate to the observed reductions in $PM_{2.5}$.

A number of additional analyses have been conducted on the ACS cohort data (Jarrett et al., 2009; Pope et al., 2009). These studies have continued to find a strong significant relationship between $PM_{2.5}$ and mortality outcomes and life expectancy. Specifically, much of the recent research has suggested a stronger relationship between cardiovascular mortality and lung cancer mortality with $PM_{2.5}$, and a less significant relationship between respiratory-related mortality and $PM_{2.5}$. The extended analyses of the ACS cohort data (Krewski et al. 2009) provides additional refinements to the analysis of PM-related mortality by (a) extend the follow-up period by 2 years to the year 2000, for a total of 18 years; (b) incorporate ecological., or neighborhood-level co-variates so as to better estimate personal exposure; (c) perform an extensive spatial analysis using land use regression modeling. These additional refinements may make this analysis well-suited for the assessment of PM-related mortality for EPA benefits analyses.

In developing and improving the methods for estimating and valuing the potential reductions in mortality risk over the years, EPA consulted with the SAB-HES. That panel recommended using long-term prospective cohort studies in estimating mortality risk reduction (U.S. EPA-SAB, 1999b). This recommendation has been confirmed by a report from the National Research Council, which stated that “it is essential to use the cohort studies in benefits analysis to capture all important effects from air pollution exposure” (NRC, 2002, p. 108). More specifically, the SAB recommended emphasis on the ACS study because it includes a much larger sample size and longer exposure interval and covers more locations (e.g., 50 cities compared to the Six Cities Study) than other studies of its kind. Because of the refinements in the extended follow-up analysis, the SAB-HES recommended using the Pope et al. (2002) study as the basis for the primary mortality estimate for adults and suggests that alternate estimates of mortality generated using other cohort and time-series studies could be included as part of the sensitivity analysis (U.S. EPA-SAB, 2004b). The PM NAAQS Risk and Exposure Assessment (U.S EPA, 2010) utilized risk coefficients drawn from the Krewski et al. (2009) study. In a December of 2009 consultation with the SAB-HES, the Agency proposed utilizing the Krewski et al. (2009) extended analysis of the ACS cohort data. The panel is scheduled to issue an advisory in early 2010.

As noted above, since 2004 SAB review, an extended follow-up of the Harvard Six cities study has been published (Laden et al., 2006) and in recent RIAs (see for example the SO_2 NAAQS, PM NAAQS, CAIR and Nonroad Diesel RIAs), we have included this estimate of mortality impacts based on application of the C-R function derived from this study. We use this specific estimate to represent the Six Cities study because it both reflects among the most up-to-date science and was cited by many of the experts in their elicitation responses. It is clear from the expert elicitation that the results published in Laden et al. (2006) are potentially influential, and in fact the expert elicitation results encompass within their range the estimates from both the Pope et al. (2002) and Laden et al. (2006) studies (see Figure 5-18 below). 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.

5.4.2.2 Ozone Premature Mortality Effect Coefficients

While particulate matter is the criteria pollutant most clearly associated with premature mortality, recent research suggests that short-term repeated ozone exposure also likely contributes to premature death. The 2006 Ozone Criteria Document found that "[c]onsistent with observed ozone-related increases in respiratory- and cardiovascular-related morbidity, several newer multi-city studies, single-city studies, and several meta-analyses of these studies have provided relatively strong epidemiologic evidence for associations between short-term ozone exposure and all-cause mortality, even after adjustment for the influence of season and PM" (U.S. EPA, 2006). The epidemiologic data are also supported by recent experimental data from both animal and human studies, which provide evidence suggestive of plausible pathways by which risk of respiratory or cardiovascular morbidity and mortality could be increased by ambient ozone. With respect to short-term exposure, the Ozone Criteria Document concluded, "This overall body of evidence is highly suggestive that ozone directly or indirectly contributes to non-accidental and cardiopulmonary-related mortality, but additional research is needed to more fully establish underlying mechanisms by which such effects occur" (U.S. EPA, 2006).

With respect to the time-series studies, the conclusion regarding the relationship between short-term ozone exposure and premature mortality is based, in part, upon recent city-specific time-series studies such as the Schwartz (2004) analysis in Houston and the Huang et al. (2004) analysis in Los Angeles.[20] This conclusion is also based on recent meta-analyses by Bell et al. (2005), Ito et al. (2005), and Levy et al. (2005), and a new analysis of the National Morbidity, Mortality, and Air Pollution Study (NMMAPS) data set by Bell et al. (2004), which specifically sought to disentangle the roles of ozone, PM, weather-related variables, and seasonality. The 2006 Criteria Document states that "the results from these meta-analyses, as well as several single- and multiple-city studies, indicate that co-pollutants generally do not appear to substantially confound the association between ozone and mortality" (p. 7-103). However, CASAC raised questions about the implications of these time-series results in a policy context. Specifically, CASAC emphasized that "…while the time-series study design is a powerful tool to detect very small effects that could not be detected using other designs, it is also a blunt tool" (U.S. EPA-SAB, 2006). They point to findings (e.g., Stieb et al., 2002, 2003) that indicated associations between premature mortality and all of the criteria pollutants, indicating that "findings of time-series studies do not seem to allow us to confidently attribute observed effects to individual pollutants" (id.). They note that "not only is the interpretation of these associations complicated by the fact that the day-to-day variation in concentrations of these pollutants is, to a varying degree, determined by meteorology, the pollutants are often part of a large and highly correlated mix of pollutants, only a very few of which are measured" (id.). Even with these uncertainties, the CASAC Ozone Panel, in its review of EPA's Staff Paper, found "…premature total non-accidental and cardiorespiratory mortality for inclusion in the quantitative risk assessment to be appropriate."

In 2006 the EPA requested an NAS study to answer four key questions regarding ozone mortality: (1) how did the epidemiological literature to that point improve our understanding of the size of the ozone mortality effect? (2) How best can EPA quantify the level of ozone mortality impacts from short-term exposure? (3) How might EPA estimate the change in life

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[20] For an exhaustive review of the city-specific time-series studies considered in the ozone staff paper, see: U.S. Environmental Protection Agency, 2007. Review of the National Ambient Air Quality Standards for Ozone: Policy Assessment of Scientific and Technical Information. Prepared by the Office of Air and Radiation. Available at http://www.epa.gov/ttn/naaqs/standards/ozone/data/2007_01_ozone_staff_paper.pdf. pp. 5-36.

expectancy? (4) What methods should EPA use to estimate the monetary value of changes in ozone-related mortality risk and life expectancy?

In 2008 the NAS (NRC, 2008) issued a series of recommendations to the EPA regarding the quantification and valuation of ozone-related short-term mortality. Chief among these was that "…short-term exposure to ambient ozone is likely to contribute to premature deaths" and the committee recommended that "ozone-related mortality be included in future estimates of the health benefits of reducing ozone exposures…" The NAS also recommended that "…the greatest emphasis be placed on the multicity and NMMAPS studies without exclusion of the meta-analyses" (NRC, 2008).

Recent evidence also suggests a relationship between long-term exposure to ozone and premature respiratory mortality in the ACS cohort (Jerrett et al. 2009). Jerrett and colleagues find that long-term exposure to ozone is linked to respiratory premature mortality in a two-pollutant model that controls for $PM_{2.5}$. This is the first long-term cohort study to have observed such a relationship. In a December of 2009 consultation with the SAB-HES, the Agency proposed utilizing the Jerrett et al. (2009) analysis of the ACS cohort data. The panel is scheduled to issue an advisory in early 2010.

In view of the findings of the Criteria document and the NAS panel, we include used estimates of short-term ozone mortality from the Bell et al. (2004) NMMAPS analysis, the Schwartz (2005) multi-city study, the Huang et al. (2005) multi-city study as well as effect estimates from the three meta-analyses (Bell et al. 2005, Levy et al. 2005 and Ito et al. 2005).

5.4.2.3 Chronic Bronchitis

CB is characterized by mucus in the lungs and a persistent wet cough for at least 3 months a year for several years in a row. CB affects an estimated 5 percent of the U.S. population (American Lung Association, 1999). A limited number of studies have estimated the impact of air pollution on new incidences of CB. Schwartz (1993) and Abbey et al. (1995) provide evidence that long-term PM exposure gives rise to the development of CB in the United States. Because the Transport Rule is expected to reduce primarily $PM_{2.5}$, this analysis uses only the Abbey et al. (1995) study, because it is the only study focusing on the relationship between $PM_{2.5}$ and new incidences of CB.

5.4.2.4 Nonfatal Myocardial Infarctions (Heart Attacks)

Nonfatal heart attacks have been linked with short-term exposures to $PM_{2.5}$ in the United States (Peters et al., 2001) and other countries (Poloniecki et al., 1997). We used a recent study by Peters et al. (2001) as the basis for the impact function estimating the relationship between $PM_{2.5}$ and nonfatal heart attacks. Peters et al. is the only available U.S. study to provide a specific estimate for heart attacks. Other studies, such as Samet et al. (2000) and Moolgavkar (2000), show a consistent relationship between all cardiovascular hospital admissions, including those for nonfatal heart attacks, and PM. Given the lasting impact of a heart attack on long-term health costs and earnings, we provide a separate estimate for nonfatal heart attacks. The estimate used in the Transport Rule analysis is based on the single available U.S. effect estimate. The finding of a specific impact on heart attacks is consistent with hospital admission and other studies showing relationships between fine particles and cardiovascular effects both within and outside the United States. Several epidemiologic studies (Liao et al., 1999; Gold et al., 2000; Magari et al., 2001) have shown that heart rate variability (an indicator of how much the heart is able to speed up or slow down in response to momentary stresses) is negatively related to PM levels. Heart rate variability is a risk factor for heart attacks and other coronary heart diseases (Carthenon et al., 2002; Dekker et al., 2000; Liao et al., 1997; Tsuji et al., 1996). As such, significant impacts of PM on heart rate variability are consistent with an increased risk of heart attacks.

5.4.2.5 Hospital and Emergency Room Admissions

Because of the availability of detailed hospital admission and discharge records, there is an extensive body of literature examining the relationship between hospital admissions and air pollution. Because of this, many of the hospital admission endpoints use pooled impact functions based on the results of a number of studies. In addition, some studies have examined the relationship between air pollution and emergency room visits. Since most emergency room visits do not result in an admission to the hospital (the majority of people going to the emergency room are treated and return home), we treat hospital admissions and emergency room visits separately, taking account of the fraction of emergency room visits that are admitted to the hospital.

The two main groups of hospital admissions estimated in this analysis are respiratory admissions and cardiovascular admissions. There is not much evidence linking ozone or PM with other types of hospital admissions. The only type of emergency room visits that have been consistently linked to ozone and PM in the United States are asthma-related visits.

To estimate avoided incidences of cardiovascular hospital admissions associated with $PM_{2.5}$, we used studies by Moolgavkar (2003) and Ito (2003). Additional published studies show a statistically significant relationship between PM_{10} and cardiovascular hospital admissions. However, given that the control options we are analyzing are expected to reduce primarily $PM_{2.5}$, we focus on the two studies that examine $PM_{2.5}$. Both of these studies provide an effect estimate for populations over 65, allowing us to pool the impact functions for this age group. Only Moolgavkar (2000) provided a separate effect estimate for populations 20 to 64.[21] Total cardiovascular hospital admissions are thus the sum of the pooled estimate for populations over 65 and the single study estimate for populations 20 to 64. Cardiovascular hospital admissions include admissions for myocardial infarctions. To avoid double-counting benefits from reductions in myocardial infarctions when applying the impact function for cardiovascular hospital admissions, we first adjusted the baseline cardiovascular hospital admissions to remove admissions for myocardial infarctions.

To estimate total avoided incidences of respiratory hospital admissions, we used impact functions for several respiratory causes, including chronic obstructive pulmonary disease (COPD), pneumonia, and asthma. As with cardiovascular admissions, additional published studies show a statistically significant relationship between PM_{10} and respiratory hospital admissions. We used only those focusing on $PM_{2.5}$. Both Moolgavkar (2000) and Ito (2003) provide effect estimates for COPD in populations over 65, allowing us to pool the impact functions for this group. Only Moolgavkar (2000) provides a separate effect estimate for populations 20 to 64. Total COPD hospital admissions are thus the sum of the pooled estimate for populations over 65 and the single study estimate for populations 20 to 64. Only Ito (2003) estimated pneumonia and only for the population 65 and older. In addition, Sheppard (2003) provided an effect estimate for asthma hospital admissions for populations under age 65. Total avoided incidences of PM-related respiratory-related hospital admissions are the sum of COPD, pneumonia, and asthma admissions.

To estimate the effects of PM air pollution reductions on asthma-related ER visits, we use the effect estimate from a study of children 18 and under by Norris et al. (1999). As noted earlier, there is another study by Schwartz examining a broader age group (less than 65), but the Schwartz study focused on PM_{10} rather than $PM_{2.5}$. We selected the Norris et al. (1999) effect

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[21] Note that the Moolgavkar (2000) study has not been updated to reflect the more stringent GAM convergence criteria. However, given that no other estimates are available for this age group, we chose to use the existing study. Given the very small (<5 percent) difference in the effect estimates for people 65 and older with cardiovascular hospital admissions between the original and reanalyzed results, we do not expect this choice to introduce much bias.

estimate because it better matched the pollutant of interest. Because children tend to have higher rates of hospitalization for asthma relative to adults under 65, we will likely capture the majority of the impact of $PM_{2.5}$ on asthma emergency room visits in populations under 65, although there may still be significant impacts in the adult population under 65.

To estimate avoided incidences of respiratory hospital admissions associated with ozone, we used a number of studies examining hospital admissions for a range of respiratory illnesses, including pneumonia and COPD. Two age groups, adults over 65 and children under 2, were examined. For adults over 65, Schwartz (1995) provides effect estimates for two different cities relating ozone and hospital admissions for all respiratory causes (defined as ICD codes 460–519). Impact functions based on these studies were pooled first before being pooled with other studies. Two studies (Moolgavkar et al., 1997; Schwartz, 1994a) examine ozone and pneumonia hospital admissions in Minneapolis. One additional study (Schwartz, 1994b) examines ozone and pneumonia hospital admissions in Detroit. The impact functions for Minneapolis were pooled together first, and the resulting impact function was then pooled with the impact function for Detroit. This avoids assigning too much weight to the information coming from one city. For COPD hospital admissions, two studies are available: Moolgavkar et al. (1997), conducted in Minneapolis, and Schwartz (1994b), conducted in Detroit. These two studies were pooled together. To estimate total respiratory hospital admissions for adults over 65, COPD admissions were added to pneumonia admissions, and the result was pooled with the Schwartz (1995) estimate of total respiratory admissions. Burnett et al. (2001) is the only study providing an effect estimate for respiratory hospital admissions in children under 2.

We used two studies as the source of the concentration-response functions we used to estimate the effects of ozone exposure on asthma-related emergency room (ER) visits: Peel et al. (2005) and Wilson et al. (2005). We estimated the change in ER visits using the effect estimate(s) from each study and then pooled the results using the random effects pooling technique (see Abt, 2005). The Peel et al. study (2005) estimated asthma-related ER visits for all ages in Atlanta, using air quality data from 1993 to 2000. Using Poisson generalized estimating equations, the authors found a marginal association between the maximum daily 8-hour average ozone level and ER visits for asthma over a 3-day moving average (lags of 0, 1, and 2 days) in a single pollutant model. Wilson et al. (2005) examined the relationship between ER visits for respiratory illnesses and asthma and air pollution for all people residing in Portland, Maine from 1998–2000 and Manchester, New Hampshire from 1996–2000. For all models used in the analysis, the authors restricted the ozone data incorporated into the model to the months ozone levels are usually measured, the spring-summer months (April through September). Using the generalized additive model, Wilson et al. (2005) found a significant association between the maximum daily 8-hour average ozone level and ER visits for asthma in Portland, but found no significant association for Manchester. Similar to the approach used to generate effect estimates for hospital admissions, we used random effects pooling to combine the results across the individual study estimates for ER visits for asthma. The Peel et al. (2005) and Wilson et al. (2005) Manchester estimates were not significant at the 95 percent level, and thus, the confidence interval for the pooled incidence estimate based on these studies includes negative values. This is an artifact of the statistical power of the studies, and the negative values in the tails of the estimated effect distributions do not represent improvements in health as ozone concentrations are increased. Instead, these should be viewed as a measure of uncertainty due to limitations in the statistical power of the study. We included both hospital admissions and ER visits as separate endpoints associated with ozone exposure because our estimates of hospital admission costs do not include the costs of ER visits and most asthma ER visits do not result in a hospital admission.

5.4.2.6 Acute Health Events and School/Work Loss Days

In addition to mortality, chronic illness, and hospital admissions, a number of acute health effects not requiring hospitalization are associated with exposure to ambient levels of ozone and PM. The sources for the effect estimates used to quantify these effects are described below.

Around 4 percent of U.S. children between the ages of 5 and 17 experience episodes of acute bronchitis annually (American Lung Association, 2002c). Acute bronchitis is characterized by coughing, chest discomfort, slight fever, and extreme tiredness, lasting for a number of days. According to the MedlinePlus medical encyclopedia,[22] with the exception of cough, most acute bronchitis symptoms abate within 7 to 10 days. Incidence of episodes of acute bronchitis in children between the ages of 5 and 17 were estimated using an effect estimate developed from Dockery et al. (1996).

Incidences of lower respiratory symptoms (e.g., wheezing, deep cough) in children aged 7 to 14 were estimated using an effect estimate from Schwartz and Neas (2000).

Because asthmatics have greater sensitivity to stimuli (including air pollution), children with asthma can be more susceptible to a variety of upper respiratory symptoms (e.g., runny or stuffy nose; wet cough; and burning, aching, or red eyes). Research on the effects of air

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[22] See http://www.nlm.nih.gov/medlineplus/ency/article/000124.htm, accessed January 2002.

pollution on upper respiratory symptoms has thus focused on effects in asthmatics. Incidences of upper respiratory symptoms in asthmatic children aged 9 to 11 are estimated using an effect estimate developed from Pope et al. (1991).

Health effects from air pollution can also result in missed days of work (either from personal symptoms or from caring for a sick family member). Days of work lost due to $PM_{2.5}$ were estimated using an effect estimate developed from Ostro (1987). Children may also be absent from school because of respiratory or other diseases caused by exposure to air pollution. Most studies examining school absence rates have found little or no association with $PM_{2.5}$, but several studies have found a significant association between ozone levels and school absence rates. We used two recent studies, Gilliland et al. (2001) and Chen et al. (2000), to estimate changes in absences (school loss days) due to changes in ozone levels. The Gilliland et al. study estimated the incidence of new periods of absence, while the Chen et al. study examined absence on a given day. We converted the Gilliland estimate to days of absence by multiplying the absence periods by the average duration of an absence. We estimated an average duration of school absence of 1.6 days by dividing the average daily school absence rate from Chen et al. (2000) and Ransom and Pope (1992) by the episodic absence rate from Gilliland et al. (2001). This provides estimates from Chen et al. (2000) and Gilliland et al. (2001), which can be pooled to provide an overall estimate.

MRAD result when individuals reduce most usual daily activities and replace them with less strenuous activities or rest, yet not to the point of missing work or school. For example, a mechanic who would usually be doing physical work most of the day will instead spend the day at a desk doing paper and phone work because of difficulty breathing or chest pain. The effect of $PM_{2.5}$ and ozone on MRAD was estimated using an effect estimate derived from Ostro and Rothschild (1989).

For the Transport Rule, we have followed the SAB-HES recommendations regarding asthma exacerbations in developing the primary estimate. To prevent double-counting, we focused the estimation on asthma exacerbations occurring in children and excluded adults from the calculation.[23] Asthma exacerbations occurring in adults are assumed to be captured in the general population endpoints such as work loss days and MRADs. Consequently, if we had included an adult-specific asthma exacerbation estimate, we would likely double-count incidence for this endpoint. However, because the general population endpoints do not cover children (with regard to asthmatic effects), an analysis focused specifically on asthma exacerbations for children (6 to 18 years of age) could be conducted without concern for double-counting.

To characterize asthma exacerbations in children, we selected two studies (Ostro et al., 2001; Vedal et al., 1998) that followed panels of asthmatic children. Ostro et al. (2001) followed a group of 138 African-American children in Los Angeles for 13 weeks, recording daily occurrences of respiratory symptoms associated with asthma exacerbations (e.g., shortness of breath, wheeze, and cough). This study found a statistically significant association between $PM_{2.5}$, measured as a 12-hour average, and the daily prevalence of shortness of breath and wheeze endpoints. Although the association was not statistically significant for cough, the results were still positive and close to significance; consequently, we decided to include this endpoint, along with shortness of breath and wheeze, in generating incidence estimates (see below). Vedal et al. (1998) followed a group of elementary school children, including 74 asthmatics, located on the west coast of Vancouver Island for 18 months including measurements of daily peak expiratory flow (PEF) and the tracking of respiratory symptoms (e.g., cough, phlegm, wheeze, chest tightness) through the use of daily diaries. Association between PM_{10} and respiratory symptoms for the asthmatic population was only reported for two

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[23] Estimating asthma exacerbations associated with air pollution exposures is difficult, due to concerns about double counting of benefits. Concerns over double counting stem from the fact that studies of the general population also include asthmatics, so estimates based solely on the asthmatic population cannot be directly added to the general population numbers without double counting. In one specific case (upper respiratory symptoms in children), the only study available is limited to asthmatic children, so this endpoint can be readily included in the calculation of total benefits. However, other endpoints, such as lower respiratory symptoms and MRADs, are estimated for the total population that includes asthmatics. Therefore, to simply add predictions of asthma-related symptoms generated for the population of asthmatics to these total population-based estimates could result in double counting, especially if they evaluate similar endpoints. The SAB-HES, in commenting on the analytical blueprint for 812, acknowledged these challenges in evaluating asthmatic symptoms and appropriately adding them into the primary analysis (SAB-HES, 2004). However, despite these challenges, the SAB-HES recommends the addition of asthma-related symptoms (i.e., asthma exacerbations) to the primary analysis, provided that the studies use the panel study approach and that they have comparable design and baseline frequencies in both asthma prevalence and exacerbation rates. Note also, that the SAB-HES, while supporting the incorporation of asthma exacerbation estimates, does not believe that the association between ambient air pollution, including ozone and PM, and the new onset of asthma is sufficiently strong to support inclusion of this asthma-related endpoint in the primary estimate.

endpoints: cough and PEF. Because it is difficult to translate PEF measures into clearly defined health endpoints that can be monetized, we only included the cough-related effect estimate from this study in quantifying asthma exacerbations. We employed the following pooling approach in combining estimates generated using effect estimates from the two studies to produce a single asthma exacerbation incidence estimate. First, we pooled the separate incidence estimates for shortness of breath, wheeze, and cough generated using effect estimates from the Ostro et al. study, because each of these endpoints is aimed at capturing the same overall endpoint (asthma exacerbations) and there could be overlap in their predictions. The pooled estimate from the Ostro et al. study is then pooled with the cough-related estimate generated using the Vedal study. The rationale for this second pooling step is similar to the first; both studies are attempting to quantify the same overall endpoint (asthma exacerbations).

5.4.2.7 School Absences

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

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

5.4.2.8 Outdoor Worker Productivity

To monetize benefits associated with increased worker productivity resulting from improved ozone air quality, we used information reported in Crocker and Horst (1981). Crocker and Horst examined the impacts of ozone exposure on the productivity of outdoor citrus workers. The study measured productivity impacts. Worker productivity is measuring the value of the loss in productivity for a worker who is at work on a particular day, but due to ozone, cannot work as hard. It only applies to outdoor workers, like fruit and vegetable pickers, or construction workers. Here, productivity impacts are measured as the change in income associated with a change in ozone exposure, given as the elasticity of income with respect to ozone concentration. The reported elasticity translates a ten percent reduction in ozone to a 1.4 percent increase in income. Given the national median daily income for outdoor workers engaged in strenuous activity reported by the U.S. Census Bureau (2002), $81 per day (2006$), a ten percent reduction in ozone yields about $0.97 in increased daily wages. We adjust the national median daily income estimate to reflect regional variations in income using a factor based on the ratio of county median household income to national median household income. No information was available for quantifying the uncertainty associated with the central valuation estimate. Therefore, no uncertainty analysis was conducted for this endpoint.

5.4.3 Baseline Incidence Estimates

Epidemiological studies of the association between pollution levels and adverse health effects generally provide a direct estimate of the relationship of air quality changes to the *relative risk* of a health effect, rather than estimating the absolute number of avoided cases. For example, a typical result might be that a 10 ppb decrease in daily ozone levels might, in turn, decrease hospital admissions by 3 percent. The baseline incidence of the health effect is necessary to convert this relative change into a number of cases. A baseline incidence rate is the estimate of the number of cases of the health effect per year in the assessment location, as it corresponds to baseline pollutant levels in that location. To derive the total baseline incidence per year, this rate must be multiplied by the corresponding population number. For example, if the baseline incidence rate is the number of cases per year per million people, that number must be multiplied by the millions of people in the total population.

Table 5-6 summarizes the sources of baseline incidence rates and provides average incidence rates for the endpoints included in the analysis. For both baseline incidence and prevalence data, we used age-specific rates where available. We applied concentration-response functions to individual age groups and then summed over the relevant age range to provide an estimate of total population benefits. In most cases, we used a single national incidence rate, due to a lack of more spatially disaggregated data. Whenever possible, the national rates used are national averages, because these data are most applicable to a national assessment of benefits. For some studies, however, the only available incidence information comes from the studies themselves; in these cases, incidence in the study population is assumed to represent typical incidence at the national level. Regional incidence rates are available for hospital admissions, and county-level data are available for premature mortality. We have projected mortality rates such that future mortality rates are consistent with our projections of population growth (Abt Associates, 2008); this represents a change from the 2005 CAIR analysis, which used static rates.

For the set of endpoints affecting the asthmatic population, in addition to baseline incidence rates, prevalence rates of asthma in the population are needed to define the applicable population. Table 5-7 lists the prevalence rates used to determine the applicable population for asthma symptom endpoints. Note that these reflect current asthma prevalence and assume no change in prevalence rates in future years. We again highlight in blue those rates that have been updated since the publication of the 2005 CAIR RIA.

Table 5-6: Baseline Incidence Rates and Population Prevalence Rates for Use in Impact Functions, General Population

		Rates	
Endpoint	*Parameter*	*Value*	*Source*[a]
Mortality	Daily or annual mortality rate projected to 2015	Age-, cause-, and county-specific rate	CDC Wonder (1996–1998) U.S. Census bureau
Hospitalizations	Daily hospitalization rate	Age-, region-, and cause-specific rate	1999 NHDS public use data files[b]
Asthma ER Visits	Daily asthma ER visit rate	Age- and region- specific visit rate	2000 NHAMCS public use data files[c]; 1999 NHDS public use data files[b]
Chronic Bronchitis	Annual prevalence rate per person		1999 NHIS (American Lung Association, 2002b, Table 4)
	• Aged 18–44	0.0367	
	• Aged 45–64	0.0505	
	• Aged 65 and older	0.0587	
	Annual incidence rate per person	0.00378	Abbey et al. (1993, Table 3)
Nonfatal Myocardial Infarction (heart attacks)	Daily nonfatal myocardial infarction incidence rate per person, 18+		1999 NHDS public use data files[b]; adjusted by 0.93 for probability of surviving after 28 days (Rosamond et al., 1999)
	• Northeast	0.0000159	
	• Midwest	0.0000135	
	• South	0.0000111	
	• West	0.0000100	
Asthma Exacerbations	Incidence (and prevalence) among asthmatic African-American children		Ostro et al. (2001)
	• daily wheeze	0.076 (0.173)	
	• daily cough	0.067 (0.145)	
	• daily dyspnea	0.037 (0.074)	
	Prevalence among asthmatic children		Vedal et al. (1998)
	• daily wheeze	0.038	
	• daily cough	0.086	
	• daily dyspnea	0.045	
Acute Bronchitis	Annual bronchitis incidence rate, children	0.043	American Lung Association (2002c, Table 11)
Lower Respiratory Symptoms	Daily lower respiratory symptom incidence among children[d]	0.0012	Schwartz et al. (1994, Table 2)
Upper Respiratory Symptoms	Daily upper respiratory symptom incidence among asthmatic children	0.3419	Pope et al. (1991, Table 2)
Work Loss Days	Daily WLD incidence rate per person (18–65)		1996 HIS (Adams, Hendershot, and Marano, 1999, Table 41); U.S. Bureau of the Census (2000)
	• Aged 18–24	0.00540	
	• Aged 25–44	0.00678	
	• Aged 45–64	0.00492	

School Loss Days	Rate per person per year, assuming 180 school days per year	9.9	National Center for Education Statistics (1996) and 1996 HIS (Adams et al., 1999, Table 47);
Minor Restricted-Activity Days	Daily MRAD incidence rate per person	0.02137	Ostro and Rothschild (1989, p. 243)

[a] The following abbreviations are used to describe the national surveys conducted by the National Center for Health Statistics: HIS refers to the National Health Interview Survey; NHDS—National Hospital Discharge Survey; NHAMCS—National Hospital Ambulatory Medical Care Survey.

[b] See ftp://ftp.cdc.gov/pub/Health_Statistics/NCHS/Datasets/NHDS/.

[c] See ftp://ftp.cdc.gov/pub/Health_Statistics/NCHS/Datasets/NHAMCS/.

[d] Lower respiratory symptoms are defined as two or more of the following: cough, chest pain, phlegm, and wheeze.

Table 5-7. Asthma Prevalence Rates Used for this Analysis

	Asthma Prevalence Rates	
Population Group	*Value*	*Source*
All Ages	0.0386	American Lung Association (2002a, Table 7)—based on 1999 HIS
< 18	0.0527	American Lung Association (2002a, Table 7)—based on 1999 HIS
5–17	0.0567	American Lung Association (2002a, Table 7)—based on 1999 HIS
18–44	0.0371	American Lung Association (2002a, Table 7)—based on 1999 HIS
45–64	0.0333	American Lung Association (2002a, Table 7)—based on 1999 HIS
65+	0.0221	American Lung Association (2002a, Table 7)—based on 1999 HIS
Male, 27+	0.021	2000 HIS public use data files[a]
African American, 5 to 17	0.0726	American Lung Association (2002a, Table 9)—based on 1999 HIS
African American, <18	0.0735	American Lung Association (2002a, Table 9)—based on 1999 HIS

[a] See ftp://ftp.cdc.gov/pub/Health_Statistics/NCHS/Datasets/NHIS/2000/.

5.4.4 Economic Valuation Estimates

Reductions in ambient concentrations of air pollution generally lower the risk of future adverse health effects for a large population. Therefore, the appropriate economic measure is WTP for changes in risk of a health effect rather than WTP for a health effect that would occur with certainty (Freeman, 1993). Epidemiological studies generally provide estimates of the relative risks of a particular health effect that is avoided because of a reduction in air pollution. We converted those to units of avoided statistical incidence for ease of presentation. We calculated the value of avoided statistical incidences by dividing individual WTP for a risk reduction by the related observed change in risk.

WTP estimates generally are not available for some health effects, such as hospital admissions. In these cases, we used the cost of treating or mitigating the effect as a primary estimate. These cost-of-illness (COI) estimates generally understate the true value of reducing the risk of a health effect, because they reflect the direct expenditures related to treatment, but not the value of avoided pain and suffering (Harrington and Portney, 1987; Berger, 1987). We provide unit values for health endpoints (along with information on the distribution of the unit value) in Table 5-8. All values are in constant year 2006 dollars, adjusted for growth in real income out to 2014 using projections provided by Standard and Poor's. Economic theory argues that WTP for most goods (such as environmental protection) will increase if real income increases. Many of the valuation studies used in this analysis were conducted in the late 1980s and early 1990s. Because real income has grown since the studies were conducted, people's willingness to pay for reductions in the risk of premature death and disease likely has grown as well. We did not adjust cost of illness-based values because they are based on current costs. Similarly, we did not adjust the value of school absences, because that value is based on current wage rates. For these two reasons, these cost of illness estimates may underestimate the economic value of avoided health impacts in 2014. The discussion below provides additional details on ozone and $PM_{2.5}$-related related endpoints.

5.4.4.1 Mortality Valuation

Following the advice of the EEAC of the SAB, EPA currently uses the VSL approach in calculating the primary estimate of mortality benefits, because we believe this calculation provides the most reasonable single estimate of an individual's willingness to trade off money for reductions in mortality risk (U.S. EPA-SAB, 2000). The VSL approach is a summary measure for the value of small changes in mortality risk experienced by a large number of people. For a period of time (2004-2008), the Office of Air and Radiation (OAR) valued mortality risk reductions using a 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 $5.5 million (2000$) 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)[24] while the Agency continues its efforts to update its guidance on this issue. This approach calculates a mean value across VSL estimates derived from 26 labor market and contingent valuation studies published between 1974 and 1991. The mean VSL across these studies is \$6.3 million (2000\$).[25] 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.

As indicated in the previous section on quantification of premature mortality benefits, we assumed for this analysis that some of the incidences of premature mortality related to PM exposures occur in a distributed fashion over the 20 years following exposure. To take this into

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[24] 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.

[25] In this analysis, we adjust the VSL to account for a different currency year (2006\$) and to account for income growth to 2014. After applying these adjustments to the \$6.3 million value, the VSL is \$7.8M.

account in the valuation of reductions in premature mortality, we applied an annual 3% discount rate to the value of premature mortality occurring in future years.[26]

The economics literature concerning the appropriate method for valuing reductions in premature mortality risk is still developing. The adoption of a value for the projected reduction in the risk of premature mortality is the subject of continuing discussion within the economics and public policy analysis community. EPA strives to use the best economic science in its analyses. Given the mixed theoretical finding and empirical evidence regarding adjustments to VSL for risk and population characteristics, we use a single VSL for all reductions in mortality risk.

Although there are several differences between the labor market studies EPA uses to derive a VSL estimate and the PM air pollution context addressed here, those differences in the affected populations and the nature of the risks imply both upward and downward adjustments. Table 5-11 lists some of these differences and the expected effect on the VSL estimate for air pollution-related mortality. In the absence of a comprehensive and balanced set of adjustment factors, EPA believes it is reasonable to continue to use the $6.3 million value while acknowledging the significant limitations and uncertainties in the available literature.

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[26] The choice of a discount rate, and its associated conceptual basis, is a topic of ongoing discussion within the federal government. EPA adopted a 3% discount rate for its base estimate in this case to reflect reliance on a "social rate of time preference" discounting concept. We have also calculated benefits and costs using a 7% rate consistent with an "opportunity cost of capital" concept to reflect the time value of resources directed to meet regulatory requirements. In this case, the benefit and cost estimates were not significantly affected by the choice of discount rate. Further discussion of this topic appears in EPA's *Guidelines for Preparing Economic Analyses* (EPA, 2000b).

Table 5-8: Expected Impact on Estimated Benefits of Premature Mortality Reductions of Differences Between Factors Used in Developing Applied VSL and Theoretically Appropriate VSL

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

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

Uncertainties Specific to Premature Mortality Valuation**.** The economic benefits associated with reductions in the risk of premature mortality are the largest category of monetized benefits of the Transport Rule. In addition, in prior analyses, EPA has identified valuation of mortality-related benefits as the largest contributor to the range of uncertainty in monetized benefits (U.S. EPA, 1999b).[27] Because of the uncertainty in estimates of the value of reducing premature mortality risk, it is important to adequately characterize and understand the various types of economic approaches available for valuing reductions in mortality risk. Such an assessment also requires an understanding of how alternative valuation approaches reflect that some individuals may be more susceptible to air pollution-induced mortality or reflect differences in the nature of the risk presented by air pollution relative to the risks studied in the relevant economics literature.

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[27] This conclusion was based on an assessment of uncertainty based on statistical error in epidemiological effect estimates and economic valuation estimates. Additional sources of model error such as those examined in the PM mortality expert elicitation may result in different conclusions about the relative contribution of sources of uncertainty.

The health science literature on air pollution indicates that several human characteristics affect the degree to which mortality risk affects an individual. For example, some age groups appear to be more susceptible to air pollution than others (e.g., the elderly and children). Health status prior to exposure also affects susceptibility. An ideal benefits estimate of mortality risk reduction would reflect these human characteristics, in addition to an individual's WTP to improve one's own chances of survival plus WTP to improve other individuals' survival rates. The ideal measure would also take into account the specific nature of the risk reduction commodity that is provided to individuals, as well as the context in which risk is reduced. To measure this value, it is important to assess how reductions in air pollution reduce the risk of dying from the time that reductions take effect onward and how individuals value these changes. Each individual's survival curve, or the probability of surviving beyond a given age, should shift as a result of an environmental quality improvement. For example, changing the current probability of survival for an individual also shifts future probabilities of that individual's survival. This probability shift will differ across individuals because survival curves depend on such characteristics as age, health state, and the current age to which the individual is likely to survive.

Although a survival curve approach provides a theoretically preferred method for valuing the benefits of reduced risk of premature mortality associated with reducing air pollution, the approach requires a great deal of data to implement. The economic valuation literature does not yet include good estimates of the value of this risk reduction commodity. As a result, in this study we value reductions in premature mortality risk using the VSL approach.

Other uncertainties specific to premature mortality valuation include the following:

- *Across-study variation:* There is considerable uncertainty as to whether the available literature on VSL provides adequate estimates of the VSL for risk reductions from air pollution reduction. Although there is considerable variation in the analytical designs and data used in the existing literature, the majority of the studies involve the value of risks to a middle-aged working population. Most of the studies examine differences in wages of risky occupations, using a hedonic wage approach. Certain characteristics of both the population affected and the mortality risk facing that population are believed to affect the average WTP to reduce the risk. The appropriateness of a distribution of WTP based on the current VSL literature for valuing the mortality-related benefits of reductions in air pollution concentrations therefore depends not only on the quality of the studies (i.e., how well they measure what they are trying to measure), but also on the extent to which the risks being valued are similar and the extent to which the subjects in the studies are similar to the population affected by changes in pollution concentrations.

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

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

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

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

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

5.4.4.2 Chronic Bronchitis Valuation

The best available estimate of WTP to avoid a case of CB comes from Viscusi, Magat, and Huber (1991). The Viscusi, Magat, and Huber study, however, describes a severe case of CB to the survey respondents. We therefore employ an estimate of WTP to avoid a pollution-related case of CB, based on adjusting the Viscusi, Magat, and Huber (1991) estimate of the WTP to avoid a severe case. This is done to account for the likelihood that an average case of pollution-related CB is not as severe. The adjustment is made by applying the elasticity of WTP with respect to severity reported in the Krupnick and Cropper (1992) study. Details of this adjustment procedure are provided in the Benefits Technical Support Document (TSD) for the Nonroad Diesel rulemaking (Abt Associates, 2003).

We use the mean of a distribution of WTP estimates as the central tendency estimate of WTP to avoid a pollution-related case of CB in this analysis. The distribution incorporates uncertainty from three sources: the WTP to avoid a case of severe CB, as described by Viscusi, Magat, and Huber; the severity level of an average pollution-related case of CB (relative to that of the case described by Viscusi, Magat, and Huber); and the elasticity of WTP with respect to severity of the illness. Based on assumptions about the distributions of each of these three uncertain components, we derive a distribution of WTP to avoid a pollution-related case of CB by statistical uncertainty analysis techniques. The expected value (i.e., mean) of this distribution, which is about $340,000 (2006$), is taken as the central tendency estimate of WTP to avoid a PM-related case of CB.

5.4.4.3 Nonfatal Myocardial Infarctions Valuation

The Agency has recently incorporated into its analyses the impact of air pollution on the expected number of nonfatal heart attacks, although it has examined the impact of reductions in other related cardiovascular endpoints. We were not able to identify a suitable WTP value for reductions in the risk of nonfatal heart attacks. Instead, we use a COI unit value with two components: the direct medical costs and the opportunity cost (lost earnings) associated with the illness event. Because the costs associated with a myocardial infarction extend beyond the initial event itself, we consider costs incurred over several years. Using age-specific annual lost earnings estimated by Cropper and Krupnick (1990) and a 3% discount rate, we estimated a present discounted value in lost earnings (in 2006$) over 5 years due to a myocardial infarction of $8,774 for someone between the ages of 25 and 44, $12,932 for someone between the ages of 45 and 54, and $74,746 for someone between the ages of 55 and 65. The corresponding age-specific estimates of lost earnings (in 2006$) using a 7% discount rate are $7,855, $11,578, and $66,920, respectively. Cropper and Krupnick (1990) do not provide lost earnings estimates for populations under 25 or over 65. As such, we do not include lost earnings in the cost estimates for these age groups.

We found three possible sources in the literature of estimates of the direct medical costs of myocardial infarction:

- Wittels et al. (1990) estimated expected total medical costs of myocardial infarction over 5 years to be $51,211 (in 1986$) for people who were admitted to the hospital and survived hospitalization. (There does not appear to be any discounting used.) Wittels et al. was used to value coronary heart disease in the 812 Retrospective Analysis of the Clean Air Act. Using the CPI-U for medical care, the Wittels estimate is $144,111 in year 2006$. This estimated cost is based on a medical cost model, which incorporated therapeutic options, projected outcomes, and prices (using "knowledgeable cardiologists" as consultants). The model used medical data and medical decision algorithms to estimate the probabilities of certain events and/or medical procedures being used. The authors note that the average length of hospitalization for acute myocardial infarction has decreased over time (from an average of 12.9 days in 1980 to an average of 11 days in 1983). Wittels et al. used 10 days as the average in their study. It is unclear how much further the length of stay for myocardial infarction may have decreased from 1983 to the present. The average length of stay for ICD code 410 (myocardial infarction) in the year-2000 Agency for Healthcare Research and Quality (AHRQ) HCUP database is 5.5 days. However, this may include patients who died in the hospital (not included among our nonfatal myocardial infarction cases), whose length of stay was therefore substantially shorter than it would be if they had not died.

- Eisenstein et al. (2001) estimated 10-year costs of $44,663 in 1997$, or $64,003 in 2006$ for myocardial infarction patients, using statistical prediction (regression) models to estimate inpatient costs. Only inpatient costs (physician fees and hospital costs) were included.

- Russell et al. (1998) estimated first-year direct medical costs of treating nonfatal myocardial infarction of $15,540 (in 1995$) and $1,051 annually thereafter. Converting to year 2006$, that would be $30,102 for a 5-year period (without discounting) or $38,113 for a 10-year period.

  In summary, the three different studies provided significantly different values (see Table 5-9).

Table 5-9: Alternative Direct Medical Cost of Illness Estimates for Nonfatal Heart Attacks

Study	*Direct Medical Costs (2006$)*	*Over an x-Year Period, for x =*
Wittels et al. (1990)	$144,111[a]	5
Russell et al. (1998)	$30,102[b]	5
Eisenstein et al. (2001)	$64,003[b]	10
Russell et al. (1998)	$38,113[b]	10

[a] Wittels et al. (1990) did not appear to discount costs incurred in future years.

[b] Using a 3% discount rate. Discounted values as reported in the study.

As noted above, the estimates from these three studies are substantially different, and we have not adequately resolved the sources of differences in the estimates. Because the wage-related opportunity cost estimates from Cropper and Krupnick (1990) cover a 5-year period, we used estimates for medical costs that similarly cover a 5-year period (i.e., estimates from Wittels et al. (1990) and Russell et al. (1998). We used a simple average of the two 5-year estimates, or $65,902, and added it to the 5-year opportunity cost estimate. The resulting estimates are given in Table 5-10.

Table 5-10: Estimated Costs Over a 5-Year Period (in 2006$) of a Nonfatal Myocardial Infarction

Age Group	*Opportunity Cost*	*Medical Cost*[a]	*Total Cost*
0–24	$0	$84,955	$84,955
25–44	$10,757[b]	$84,955	$95,713
45–54	$15,855[b]	$84,955	$100,811
55–65	$91,647[b]	$84,955	$176,602
> 65	$0	$84,955	$84,955

[a] An average of the 5-year costs estimated by Wittels et al. (1990) and Russell et al. (1998).

[b] From Cropper and Krupnick (1990), using a 3% discount rate.

5.4.4.4 Hospital Admissions Valuation

In the absence of estimates of societal WTP to avoid hospital visits/admissions for specific illnesses, estimates of total cost of illness (total medical costs plus the value of lost productivity) typically are used as conservative, or lower bound, estimates. These estimates are biased downward, because they do not include the willingness-to-pay value of avoiding pain and suffering.

The International Classification of Diseases (ICD-9, 1979) code-specific COI estimates used in this analysis consist of estimated hospital charges and the estimated opportunity cost of time spent in the hospital (based on the average length of a hospital stay for the illness). We based all estimates of hospital charges and length of stays on statistics provided by the Agency for Healthcare Research and Quality (AHRQ 2000). We estimated the opportunity cost of a day spent in the hospital as the value of the lost daily wage, regardless of whether the hospitalized individual is in the workforce. To estimate the lost daily wage, we divided the 1990 median weekly wage by five and inflated the result to year 2006$ using the CPI-U "all items." The resulting estimate is $127.93. The total cost-of-illness estimate for an ICD code-specific hospital stay lasting *n* days, then, was the mean hospital charge plus $127.93 multiplied by *n*.

Table 5-11: Unit Values for Economic Valuation of Health Endpoints (2006$)

Health Endpoint	**Central Estimate of Value Per Statistical Incidence**		
	2000 Income Level	**2014 Income Level**	**Derivation of Distributions of Estimates**
Premature Mortality (Value of a Statistical Life)	$6,300,000	$7,800,000	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)	$340,000	$460,000	The WTP to avoid a case of pollution-related CB is calculated as where x is the severity of an average CB case, WTP_{13} 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, 1999b).
Nonfatal Myocardial Infarction (heart attack) 3% discount rate Age 0–24 Age 25–44 Age 45–54 Age 55–65 Age 66 and over 7% discount rate Age 0–24 Age 25–44 Age 45–54 Age 55–65 Age 66 and over	$79,685 $88,975 $93,897 $167,532 $79,685 $77,769 $87,126 $91,559 $157,477 $77,769	$79,685 $88,975 $93,897 $167,532 $79,685 $77,769 $87,126 $91,559 $157,477 $77,769	No distributional information available. Age-specific cost-of-illness values reflect lost earnings and direct medical costs over a 5-year period following a nonfatal MI. Lost earnings estimates are based on Cropper and Krupnick (1990). Direct medical costs are based on simple average of estimates from Russell et al. (1998) and Wittels et al. (1990). Lost earnings: Cropper and Krupnick (1990). Present discounted value of 5 years of lost earnings: age of onset: at 3 25–44 $8,774 $7,8 45–54 $12,932 11,5 55–65 $74,746 66,9 Direct medical expenses: An average of: 1. Wittels et al. (1990) ($102,658—no discounting) 2. Russell et al. (1998), 5-year period ($22,331 at 3% discount rate; $21,113 at 7% discount rate)
Hospital Admissions			
Chronic Obstructive Pulmonary Disease (COPD)	$16,606	$16,606	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	$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	$24,668	$24,668	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+)	$24,622	$24,622	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)	$10,385	$10,385	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	$384	$384	No distributional information available. Simple average of two unit COI values: (1) $311.55, from Smith et al. (1997) and (2) $260.67, from Stanford et al. (1999).
Respiratory Ailments Not Requiring Hospitalization			
Upper Respiratory Symptoms (URS)	$30	$30	Combinations of the three symptoms for which WTP estimates are available that closely match those listed by Pope et al. result in seven

			different "symptom clusters," each describing a "type" of URS. A dollar value was derived for each type of URS, using mid-range estimates of WTP (IEc, 1994) to avoid each symptom in the cluster and assuming additivity of WTPs. In the absence of information surrounding the frequency with which each of the seven types of URS occurs within the URS symptom complex, we assumed a uniform distribution between \$9.2 and \$43.1.
Lower Respiratory Symptoms (LRS)	\$16	\$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 $6.9 and $24.46.
Asthma Exacerbations	$43	$53	Asthma exacerbations are valued at $45 per incidence, based on the mean of average WTP estimates for the four severity definitions of a "bad asthma day," described in Rowe and Chestnut (1986). This study surveyed asthmatics to estimate WTP for avoidance of a "bad asthma day," as defined by the subjects. For purposes of valuation, an asthma exacerbation is assumed to be equivalent to a day in which asthma is moderate or worse as reported in the Rowe and Chestnut (1986) study. The value is assumed have a uniform distribution between $15.6 and $70.8.
Acute Bronchitis	$360	$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 $10 is the sum of the mid-range values recommended by IEc (1994) for two

			symptoms believed to be associated with acute bronchitis: coughing and chest tightness. The high daily estimate was taken to be twice the value of a minor respiratory restricted-activity day, or $110.
Work Loss Days (WLDs)	Variable (U.S. median = $130)	Variable (U.S. median = $130)	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)	$51	$62	Median WTP estimate to avoid one MRAD from Tolley et al. (1986). Distribution is assumed to be triangular with a minimum of $22 and a maximum of $83, with a most likely value of $52. Range is based on assumption that value should exceed WTP for a single mild symptom (the highest estimate for a single symptom—for eye irritation—is $16.00) 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.

School Absence Days	$89	$89	No distribution available

5.4.4.5 *Asthma-Related Emergency Room Visits Valuation*

To value asthma emergency room visits, we used a simple average of two estimates from the health economics literature. The first estimate comes from Smith et al. (1997), who reported approximately 1.2 million asthma-related emergency room visits in 1987, at a total cost of $186.5 million (1987$). The average cost per visit that year was $155; in 2006$, that cost was $400.88 (using the CPI-U for medical care to adjust to 2006$). The second estimate comes from Stanford et al. (1999), who reported the cost of an average asthma-related emergency room visit at $335.14, based on 1996–1997 data. A simple average of the two estimates yields a (rounded) unit value of $368.

5.4.4.6 *Minor Restricted Activity Days Valuation*

No studies are reported to have estimated WTP to avoid a minor restricted activity day. However, one of EPA's contractors, IEc (1994) has derived an estimate of willingness to pay to avoid a minor *respiratory* restricted activity day, using estimates from Tolley et al. (1986) of WTP for avoiding a combination of coughing, throat congestion and sinusitis. The IEc estimate of WTP to avoid a minor respiratory restricted activity day is $38.37 (1990$), or about $62.04 (2006$).

Although Ostro and Rothschild (1989) statistically linked ozone and minor restricted activity days, it is likely that most MRADs associated with ozone exposure are, in fact, minor *respiratory* restricted activity days. For the purpose of valuing this health endpoint, we used the estimate of mean WTP to avoid a minor respiratory restricted activity day.

5.4.4.7 *School Absences Valuation*

To value a school absence, we: (1) estimated the probability that if a school child stays home from school, a parent will have to stay home from work to care for the child; and (2) valued the lost productivity at the parent's wage. To do this, we estimated the number of families with school-age children in which both parents work, and we valued a school-loss day as the probability that such a day also would result in a work-loss day. We calculated this value by multiplying the proportion of households with school-age children by a measure of lost wages.

We used this method in the absence of a preferable WTP method. However, this approach suffers from several uncertainties. First, it omits willingness to pay to avoid the symptoms/illness that resulted in the school absence; second, it effectively gives zero value to school absences that do not result in work-loss days; and third, it uses conservative assumptions about the wages of the parent staying home with the child. Finally, this method assumes that parents are unable to work from home. If this is not a valid assumption, then there would be no lost wages.

For this valuation approach, we assumed that in a household with two working parents, the female parent will stay home with a sick child. From the Statistical Abstract of the United States (U.S. Census Bureau, 2001), we obtained: (1) the numbers of single, married and “other” (widowed, divorced or separated) working women with children; and (2) the rates of participation in the workforce of single, married and “other” women with children. From these two sets of statistics, we calculated a weighted average participation rate of 72.85 percent.

Our estimate of daily lost wage (wages lost if a mother must stay at home with a sick child) is based on the year 2006 median weekly wage among women ages 25 and older (U.S. Census Bureau, 2001). This median weekly wage is $655. Dividing by five gives an estimated median daily wage of $131. To estimate the expected lost wages on a day when a mother has to stay home with a school-age child, we first estimated the probability that the mother is in the workforce then multiplied that estimate by the daily wage she would lose by missing a workday: 72.85 percent times $131, for a total loss of $95.43. This valuation approach is similar to that used by Hall et al. (2003).

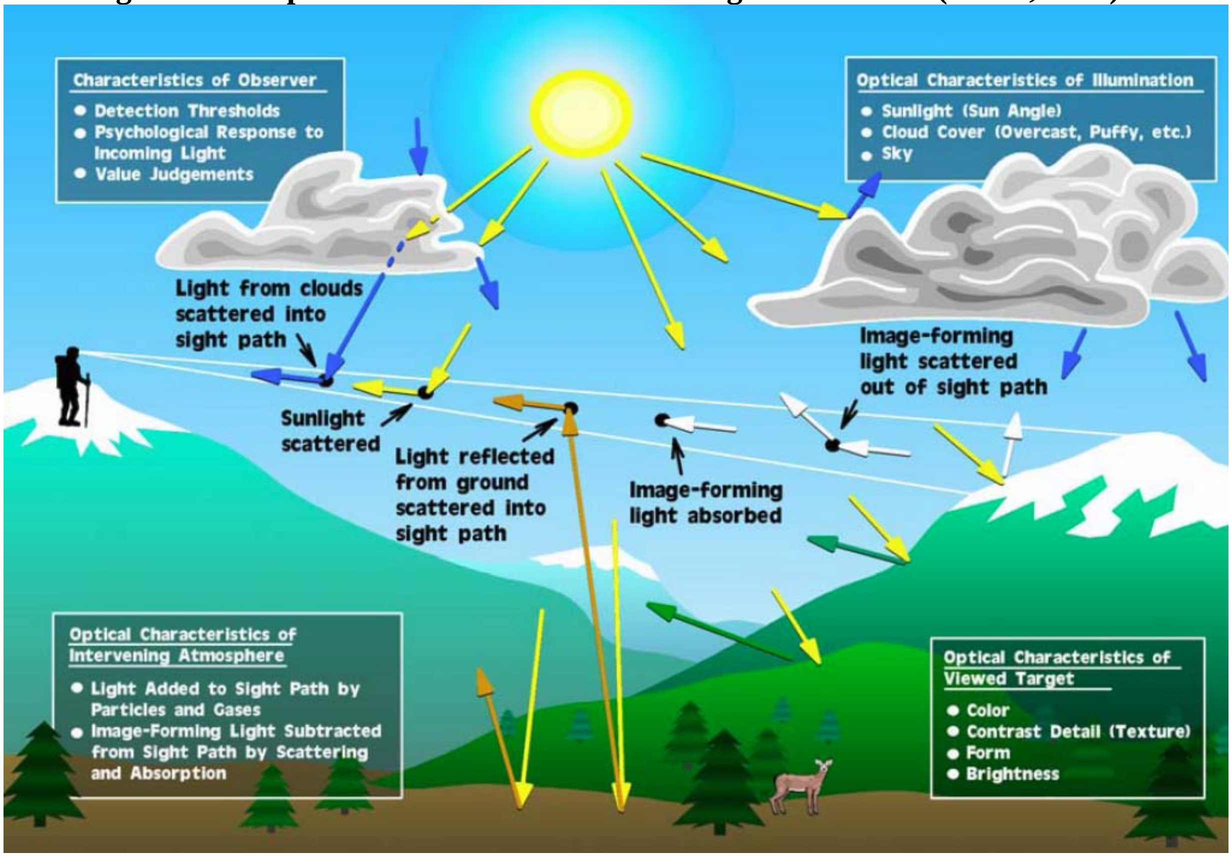
5.4.4.8 Visibility Valuation

Reductions in NO_2 and SO_2 emissions along with the secondary formation of $PM_{2.5}$ 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, 2009d).

Visibility has direct significance to people's enjoyment of daily activities and their overall sense of wellbeing (U.S. EPA, 2009d). 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. This section discusses the measurement of the economic benefits of improved visibility.

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. Pitchford and Malm characterize a change of one deciview as "a small but perceptible scenic change under many circumstances." 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. See Figure 5-3 for an illustration of the important factors affecting visibility.

Figure 5-3: Important factors involved in seeing a scenic vista (Malm, 1999)



In conjunction with the U.S. National Park Service, the U.S. Forest Service, other Federal land managers, and State organizations in the U.S., the U.S. EPA has supported visibility monitoring in national parks and wilderness areas since 1988. The monitoring network known as IMPROVE (Interagency Monitoring of Protected Visual Environments) now includes 150 sites that represent almost all of the Class I areas across the country (see Figure 5-4) (U.S. EPA, 2009d).

Figure 5-4: Mandatory Class I Areas in the U.S.

* Rainbow Lake, WI and Bradwell Bay, FL are Class 1 Areas where visibility is not an important air quality related value

Produced by NPS Air Resources Division

Annual average visibility conditions (reflecting light extinction due to both anthropogenic and non-anthropogenic sources) vary regionally across the U.S. (U.S. EPA, 2009d). 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, 1999b).

EPA distinguishes benefits from two categories of visibility changes: residential visibility and recreational visibility. 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 birdwatching. 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.

Residential visibility benefits are those that occur from visibility changes in urban, suburban, and rural areas. In previous assessments, EPA used a study on residential visibility valuation conducted in 1990 (McClelland et al., 1993). Subsequently, EPA designated the McClelland et al. study as significantly less reliable for regulatory benefit-cost analysis consistent with SAB advice (U.S. EPA-SAB, 1999). Although a wide range of published, peer-review literature supports a non-zero value for residential visibility (Brookshire et al., 1982; Rae, 1983; Tolley et al., 1986; Chestnut and Rowe, 1990c; McClleand et al., 1993; Loehman et al., 1994), the residential visibility benefits have not been calculated in this analysis.

For recreational visibility, only one existing study provides defensible monetary estimates of the value of visibility changes in a 1988 survey on recreational visibility value (Chestnut and Rowe, 1990a; 1990b). Although there are a number of other studies in the literature, they were conducted in the early 1980s and did not use methods that are considered defensible by current standards. The Chestnut and Rowe study uses the CV method. There has been a great deal of controversy and significant development of both theoretical and empirical knowledge about how to conduct CV surveys in the past decade. In EPA's judgment, the Chestnut and Rowe study contains many of the elements of a valid CV study and is sufficiently reliable to serve as the basis for monetary estimates of the benefits of visibility changes in recreational areas.[28] This study serves as an essential input to our

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[28] An SAB advisory letter indicates that "many members of the Council believe that the Chestnut and Rowe study is the best available" (EPA-SAB-COUNCIL-ADV-00-002, 1999, p. 13). However, the committee did not formally approve use of these estimates because of concerns about the peer-reviewed status of the study. EPA believes the study has received adequate review and has been cited in numerous peer-reviewed publications (Chestnut and Dennis, 1997).

estimates of the benefits of recreational visibility improvements in the primary benefits estimates.

For the purposes of this analysis, recreational visibility improvements are defined as those that occur specifically in federal Class I areas.[29] A key distinction between recreational and residential benefits is that only those people living in residential areas are assumed to receive benefits from residential visibility, while all households in the United States are assumed to derive some benefit from improvements in Class I areas. Values are assumed to be higher if the Class I area is located close to their home.[30] The Chestnut and Rowe study measured the demand for visibility in Class I areas managed by the National Park Service (NPS) in three broad regions of the country: California, the Southwest, and the Southeast. Respondents in five states were asked about their WTP to protect national parks or NPS-managed wilderness areas within a particular region. The survey used photographs reflecting different visibility levels in the specified recreational areas. The visibility levels in these photographs were later converted to deciviews for the current analysis. The survey data collected were used to estimate a WTP equation for improved visibility. In addition to the visibility change variable, the estimating equation also included household income as an explanatory variable.

The Chestnut and Rowe study did not measure values for visibility improvement in Class I areas outside the three regions. Their study covered 86 of the 156 Class I areas in the United States. We can infer the value of visibility changes in the other Class I areas by transferring values of visibility changes at Class I areas in the study regions. A complete description of the benefits transfer method used to infer values for visibility changes in Class I areas outside the study regions is provided in the Benefits TSD for the Nonroad Diesel rulemaking (Abt Associates, 2003).

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[29] The Clean Air Act designates 156 national parks and wilderness areas as Class I areas for visibility protection.

[30] For details of the visibility estimates discussed in this chapter, please refer to the Benefits TSD for the Nonroad Diesel rulemaking (Abt Associates, 2003).

The Chestnut and Rowe study (Chestnut and Rowe, 1990a; 1990b), although representing the best available estimates, has a number of limitations. These include the following:

- The age of the study (late 1980s) will increase the uncertainty about the correspondence of the estimated values to those that might be provided by current or future populations.
- The survey focused only on populations in five states, so the application of the estimated values to populations outside those states requires that preferences of populations in the five surveyed states be similar to those of non-surveyed states.
- There is an inherent difficulty in separating values expressed for visibility improvements from an overall value for improved air quality. The Chestnut and Rowe study attempted to control for this by informing respondents that "other households are being asked about visibility, human health, and vegetation protections in urban areas and at national parks in other regions." However, most of the respondents did not feel that they were able to segregate visibility at national parks entirely from residential visibility and health effects.
- It is not clear exactly what visibility improvements the respondents to the Chestnut and Rowe survey were valuing. The WTP question asked about changes in average visibility, but the survey respondents were shown photographs of only summertime conditions, when visibility is generally at its worst. It is possible that the respondents believed those visibility conditions held year-round, in which case they would have been valuing much larger overall improvements in visibility than what otherwise would be the case. For the purpose of the benefits analysis for this rule, EPA assumed that respondents provided values for changes in annual average visibility. Because most policies will result in a shift in the distribution of visibility (usually affecting the worst days more than the best days), the annual average may not be the most relevant metric for policy analysis.
- The survey did not include reminders of possible substitutes (e.g., visibility at other parks) or budget constraints. These reminders are considered to be best practice for stated preference surveys.
- The Chestnut and Rowe survey focused on visibility improvements in and around national parks and wilderness areas. The survey also focused on visibility improvements of national parks in the southwest United States. Given that national parks and wilderness areas exhibit unique characteristics, it is not clear whether the

WTP estimate obtained from Chestnut and Rowe can be transferred to other national parks and wilderness areas, without introducing additional uncertainty.

In general, the survey design and implementation reflect the period in which the survey was conducted. Since that time, many improvements to the stated preference methodology have been developed. As future survey efforts are completed, EPA will incorporate values for visibility improvements reflecting the improved survey designs.

The estimated relationship from the Chestnut and Rowe study is only directly applicable to the populations represented by survey respondents. EPA used benefits transfer methodology to extrapolate these results to the population affected by the reductions in precursor emissions associated with this rule. A general WTP equation for improved visibility (measured in deciviews) was developed as a function of the baseline level of visibility, the magnitude of the visibility improvement, and household income. The behavioral parameters of this equation were taken from analysis of the Chestnut and Rowe data. These parameters were used to calibrate WTP for the visibility changes resulting from this rule. The method for developing calibrated WTP functions is based on the approach developed by Smith et al. (2002). Available evidence indicates that households are willing to pay more for a given visibility improvement as their income increases (Chestnut, 1997). The benefits estimates here incorporate Chestnut's estimate that a 1% increase in income is associated with a 0.9% increase in WTP for a given change in visibility. A more detailed explanation of the visibility benefits methodology is provided in Appendix I of the PM NAAQS RIA (U.S. EPA, 2006).

One major source of uncertainty for the visibility benefits estimate is the benefits transfer process used. Judgments used to choose the functional form and key parameters of the estimating equation for WTP for the affected population could have significant effects on the size of the estimates. Assumptions about how individuals respond to changes in visibility that are either very small or outside the range covered in the Chestnut and Rowe study could also affect the results.

In addition, our estimate of visibility benefits is incomplete. For example, we anticipate improvement in visibility in residential areas within the Transport Rule region for which we are currently unable to monetize benefits, such as the Northeastern and Central regions of the U.S. The value of visibility benefits in areas where we were unable to monetize benefits could also be substantial. EPA requests public comment on the approach taken here to quantify the monetary value of changes in visibility in Class I areas.

5.4.4.9 Growth in WTP Reflecting National Income Growth Over Time

Our analysis accounts for expected growth in real income over time. Economic theory argues that WTP for most goods (such as environmental protection) will increase if real incomes increase. There is substantial empirical evidence that the income elasticity[31] of WTP for health risk reductions is positive, although there is uncertainty about its exact value. Thus, as real income increases, the WTP for environmental improvements also increases. Although many analyses assume that the income elasticity of WTP is unit elastic (i.e., a 10% higher real income level implies a 10% higher WTP to reduce risk changes), empirical evidence suggests that income elasticity is substantially less than one and thus relatively inelastic. As real income rises, the WTP value also rises but at a slower rate than real income.

The effects of real income changes on WTP estimates can influence benefits estimates in two different ways: through real income growth between the year a WTP study was conducted and the year for which benefits are estimated, and through differences in income between study populations and the affected populations at a particular time. Empirical evidence of the effect of real income on WTP gathered to date is based on studies examining the former. The Environmental Economics Advisory Committee (EEAC) of the Science Advisory Board (SAB) advised EPA to adjust WTP for increases in real income over time but not to adjust WTP to account for cross-sectional income differences "because of the sensitivity of making such distinctions, and because of insufficient evidence available at present" (U.S. EPA-SAB, 2000a). A recent advisory by another committee associated with the SAB, the Advisory Council on Clean Air Compliance Analysis, has provided conflicting advice. While agreeing with "the general principle that the willingness to pay to reduce mortality risks is likely to increase with growth in real income (U.S. EPA-SAB, 2004a, p. 52)" and that "The same increase should be assumed for the WTP for serious nonfatal health

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[31] Income elasticity is a common economic measure equal to the percentage change in WTP for a 1% change in income.

effects (U.S. EPA-SAB, 2004a, p. 52)," they note that "given the limitations and uncertainties in the available empirical evidence, the Council does not support the use of the proposed adjustments for aggregate income growth as part of the primary analysis (U.S. EPA-SAB, 2004a, p. 53)." Until these conflicting advisories have been reconciled, EPA will continue to adjust valuation estimates to reflect income growth using the methods described below, while providing sensitivity analyses for alternative income growth adjustment factors.

Based on a review of the available income elasticity literature, we adjusted the valuation of human health benefits upward to account for projected growth in real U.S. income. Faced with a dearth of estimates of income elasticities derived from time-series studies, we applied estimates derived from cross-sectional studies in our analysis. Details of the procedure can be found in Kleckner and Neumann (1999). An abbreviated description of the procedure we used to account for WTP for real income growth between 1990 and 2014 is presented below.

Reported income elasticities suggest that the severity of a health effect is a primary determinant of the strength of the relationship between changes in real income and WTP. As such, we use different elasticity estimates to adjust the WTP for minor health effects, severe and chronic health effects, and premature mortality. Note that because of the variety of empirical sources used in deriving the income elasticities, there may appear to be inconsistencies in the magnitudes of the income elasticities relative to the severity of the effects (*a priori* one might expect that more severe outcomes would show less income elasticity of WTP). We have not imposed any additional restrictions on the empirical estimates of income elasticity. One explanation for the seeming inconsistency is the difference in timing of conditions. WTP for minor illnesses is often expressed as a short term payment to avoid a single episode. WTP for major illnesses and mortality risk reductions are based on longer term measures of payment (such as wages or annual income). Economic theory suggests that relationships become more elastic as the length of time grows, reflecting the ability to adjust spending over a longer time period. Based on this theory, it would be expected that WTP for reducing long term risks would be more elastic than WTP for reducing short term risks. We also expect that the WTP for improved visibility in Class I areas would increase with growth in real income. The relative magnitude of the income elasticity of WTP for visibility compared with those for health effects suggests that visibility is not as much of a necessity as health, thus, WTP is more elastic with respect to income. The elasticity values used to adjust estimates of benefits in 2014 are presented in Table 5-12.

Table 5-12: Elasticity Values Used to Account for Projected Real Income Growth[a]

Benefit Category	*Central Elasticity Estimate*
Minor Health Effect	0.14
Severe and Chronic Health Effects	0.45
Premature Mortality	0.40
Visibility	0.90

[a] Derivation of estimates can be found in Kleckner and Neumann (1999) and Chestnut (1997). COI estimates are assigned an adjustment factor of 1.0.

In addition to elasticity estimates, projections of real gross domestic product (GDP) and populations from 1990 to 2020 are needed to adjust benefits to reflect real per capita income growth. For consistency with the emissions and benefits modeling, we used national population estimates for the years 1990 to 1999 based on U.S. Census Bureau estimates (Hollman, Mulder, and Kallan, 2000). These population estimates are based on application of a cohort-component model applied to 1990 U.S. Census data projections (U.S. Bureau of Census, 2000). For the years between 2000 and 2014, we applied growth rates based on the U.S. Census Bureau projections to the U.S. Census estimate of national population in 2000. We used projections of real GDP provided in Kleckner and Neumann (1999) for the years 1990 to 2010.[32] We used projections of real GDP (in chained 1996 dollars) provided by Standard and Poor's (2000) for the years 2010 to 2014.[33]

Using the method outlined in Kleckner and Neumann (1999) and the population and income data described above, we calculated WTP adjustment factors for each of the elasticity estimates listed in Table 5-13. Benefits for each of the categories (minor health effects,

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[32] U.S. Bureau of Economic Analysis, Table 2A (1992$) (available at http://www.bea.doc.gov/bea/dn/0897nip2/tab2a.htm.) and U.S. Bureau of Economic Analysis, Economics and Budget Outlook. Note that projections for 2007 to 2010 are based on average GDP growth rates between 1999 and 2007.

[33] In previous analyses, we used the Standard and Poor's projections of GDP directly. This led to an apparent discontinuity in the adjustment factors between 2010 and 2011. We refined the method by applying the relative growth rates for GDP derived from the Standard and Poor's projections to the 2010 projected GDP based on the Bureau of Economic Analysis projections.

severe and chronic health effects, premature mortality, and visibility) are adjusted by multiplying the unadjusted benefits by the appropriate adjustment factor. Note that, for premature mortality, we applied the income adjustment factor to the present discounted value of the stream of avoided mortalities occurring over the lag period. Also note that because of a lack of data on the dependence of COI and income, and a lack of data on projected growth in average wages, no adjustments are made to benefits based on the COI approach or to work loss days and worker productivity. This assumption leads us to underpredict benefits in future years because it is likely that increases in real U.S. income would also result in increased COI (due, for example, to increases in wages paid to medical workers) and increased cost of work loss days and lost worker productivity (reflecting that if worker incomes are higher, the losses resulting from reduced worker production would also be higher).

Table 5-13: Adjustment Factors Used to Account for Projected Real Income Growth[a]

Benefit Category	*2014*
Minor Health Effect	1.04
Severe and Chronic Health Effects	1.16
Premature Mortality	1.14
Visibility	1.35

[a] Based on elasticity values reported in Table 5-3, U.S. Census population projections, and projections of real GDP per capita.

5.5 Unquantified Health and Welfare Benefits

This analysis is limited by the available data and resources. As such, we are not able to quantify several welfare benefit categories, as shown in Table 5-2. In this section, we provide a qualitative assessment of some of the primary welfare benefit categories from reducing NO_2 and SO_2 emissions: health and ecosystem benefits of reducing nitrogen and sulfur emissions and deposition and vegetation benefits from reducing ozone.[34] While we were unable to quantify how large these benefits might be as a result of the emission

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[34] Some quantitative estimates of the total value of certain recreational and environmental goods given current and historic emission levels are provided below. They do not reflect benefits that would accrue as a result of this result. However, these values would be expected to increase as emissions are decreased a result of this rule.

reductions achieved by this rule , previous EPA assessments show that these benefits could be substantial (U.S. EPA, 2008f; U.S. EPA, 2009c; U.S. EPA, 2007b; U.S. EPA, 1999b). The omission of these endpoints from the monetized results should not imply that the impacts are small or unimportant.

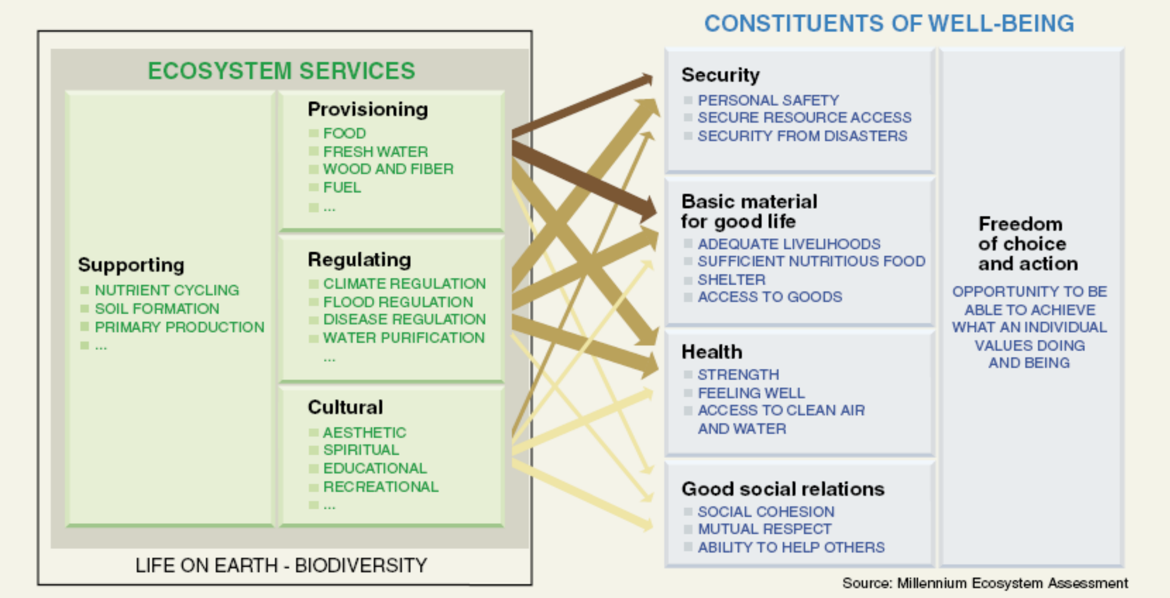
5.5.1 Ecosystem Services

Ecosystem services can be generally defined as the benefits that individuals and organizations obtain from ecosystems. EPA has defined ecological goods and services as the “outputs of ecological functions or processes that directly or indirectly contribute to social welfare or have the potential to do so in the future. Some outputs may be bought and sold, but most are not marketed” (U.S. EPA, 2006b). Figure 5-5 provides the Millennium Ecosystem Assessment’s schematic demonstrating the connections between the categories of ecosystem services and human well-being. The interrelatedness of these categories means that any one ecosystem may provide multiple services. Changes in these services can affect human well-being by affecting security, health, social relationships, and access to basic material goods (MEA, 2005).

In the Millennium Ecosystem Assessment (MEA, 2005), ecosystem services are classified into four main categories:

1. Provisioning: Products obtained from ecosystems, such as the production of food and water
2. Regulating: Benefits obtained from the regulation of ecosystem processes, such as the control of climate and disease
3. Cultural: Nonmaterial benefits that people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences
4. Supporting: Services necessary for the production of all other ecosystem services, such as nutrient cycles and crop pollination

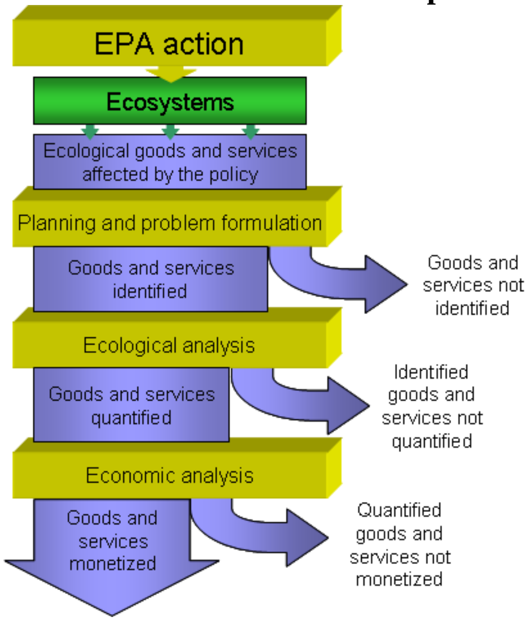
Figure 5-5: Linkages between categories of ecosystem services and components of human well-being from Millennium Ecosystem Assessment (MEA, 2005)



Source: Millennium Ecosystem Assessment

The monetization of ecosystem services generally involves estimating the value of ecological goods and services based on what people are willing to pay (WTP) to increase ecological services or by what people are willing to accept (WTA) in compensation for reductions in them (U.S. EPA, 2006b). There are three primary approaches for estimating the monetary value of ecosystem services: market-based approaches, revealed preference methods, and stated preference methods (U.S. EPA, 2006b). Because economic valuation of ecosystem services can be difficult, nonmonetary valuation using biophysical measurements and concepts also can be used. An example of a nonmonetary valuation method is the use of relative-value indicators (e.g., a flow chart indicating uses of a water body, such as boatable, fishable, swimmable, etc.). It is necessary to recognize that in the analysis of the environmental responses associated with any particular policy or environmental management action, only a subset of the ecosystem services likely to be affected are readily identified. Of those ecosystem services that are identified, only a subset of the changes can be quantified. Within those services whose changes can be quantified, only a few will likely be monetized, and many will remain nonmonetized. The stepwise concept leading up to the valuation of ecosystems services is graphically depicted in Figure 5-6.

Figure 5-6: Schematic of the benefits assessment process (U.S. EPA, 2006b)

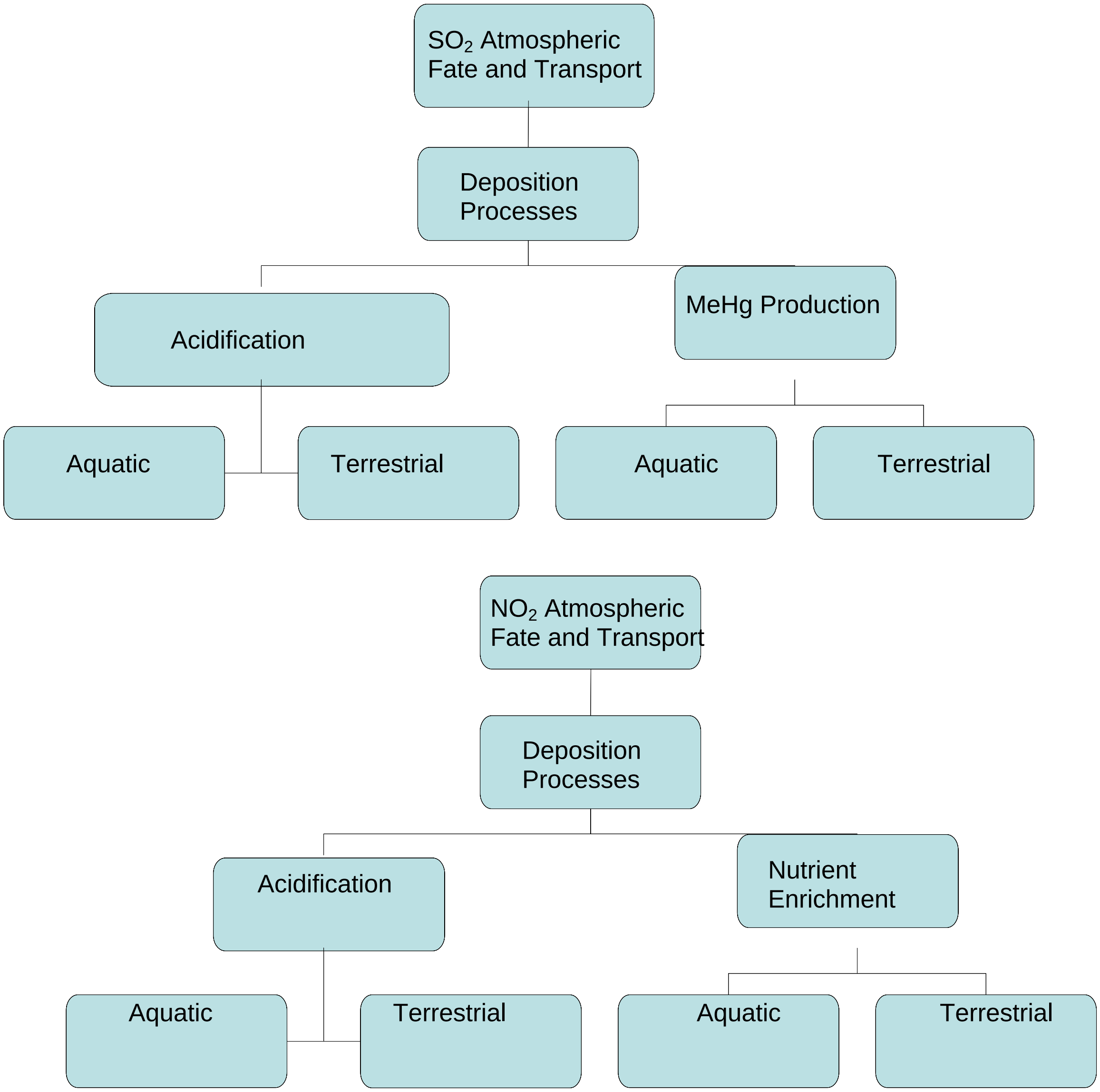


5.5.2 Ecosystem Benefits of Reduced Nitrogen and Sulfur Deposition

5.5.2.1 Science of Deposition

Nitrogen and sulfur emissions occur over large regions of North America. Once these pollutants are lofted to the middle and upper troposphere, they typically have a much longer lifetime and, with the generally stronger winds at these altitudes, can be transported long distances from their source regions. The length scale of this transport is highly variable owing to differing chemical and meteorological conditions encountered along the transport path (U.S. EPA, 2008f).. Sulfur is primarily emitted as SO_2, and nitrogen can be emitted as NO, NO_2, or NH_3. Secondary particles are formed from NO_x and SO_X gaseous emissions and associated chemical reactions in the atmosphere. Deposition can occur in either a wet (i.e., rain, snow, sleet, hail, clouds, or fog) or dry form (i.e., gases or particles). Together these emissions are deposited onto terrestrial and aquatic ecosystems across the U.S., contributing to the problems of acidification, nutrient enrichment, and methylmercury production as represented in Figure 5-7. Although there is some evidence that nitrogen deposition may have positive effects on agricultural and forest output through passive fertilization, it is likely that the overall value is very small relative to other health and welfare effects.

Figure 5-7: Schematics of Ecological Effects of Nitrogen and Sulfur Deposition



The lifetimes of particles vary with particle size. Accumulation-mode particles such as sulfates are kept in suspension by normal air motions and have a lower deposition velocity than coarse-mode particles; they can be transported thousands of kilometers and remain in the atmosphere for a number of days. They are removed from the atmosphere primarily by cloud processes. Particulates affect acid deposition by serving as cloud condensation nuclei and contribute directly to the acidification of rain. In addition, the gas-phase species that lead to the dry deposition of acidity are also precursors of particles. Therefore, reductions in NO_2 and SO_2 emissions will decrease both acid deposition and PM concentrations, but not necessarily in a linear fashion. (U.S. EPA, 2008f). Sulfuric acid is also deposited on surfaces by dry deposition and can contribute to environmental effects (U.S. EPA, 2008f).

5.5.2.2 Ecological Effects of Acidification

Deposition of nitrogen and sulfur can cause acidification, which alters biogeochemistry and affects animal and plant life in terrestrial and aquatic ecosystems across the U.S. Soil acidification is a natural process, but is often accelerated by acidifying deposition, which can decrease concentrations of exchangeable base cations in soils (U.S. EPA, 2008f). Major terrestrial effects include a decline in sensitive tree species, such as red spruce (*Picea rubens*) and sugar maple (*Acer saccharum*) (U.S. EPA, 2008f). Biological effects of acidification in terrestrial ecosystems are generally linked to aluminum toxicity and decreased ability of plant roots to take up base cations (U.S. EPA, 2008f). 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 (U.S. EPA, 2008f).

Geology (particularly surficial geology) is the principal factor governing the sensitivity of terrestrial and aquatic ecosystems to acidification from nitrogen and sulfur deposition (U.S. EPA, 2008f). Geologic formations having low base cation supply generally underlie the watersheds of acid-sensitive lakes and streams. Other factors contribute to the sensitivity of soils and surface waters to acidifying deposition, including topography, soil chemistry, land use, and hydrologic flow path (U.S. EPA, 2008f).

5.5.2.3 Aquatic Ecosystems

Aquatic effects of acidification 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[35], 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). Biological effects in aquatic ecosystems can be divided into two major categories: effects on health, vigor, and reproductive success; and effects on biodiversity. Surface water with ANC values greater than 50 μeq/L generally provides moderate protection for most fish (i.e., brook trout, others) and other aquatic organisms (U.S. EPA, 2009c). Table 5-14 provides a summary of the biological effects experienced at various ANC levels.

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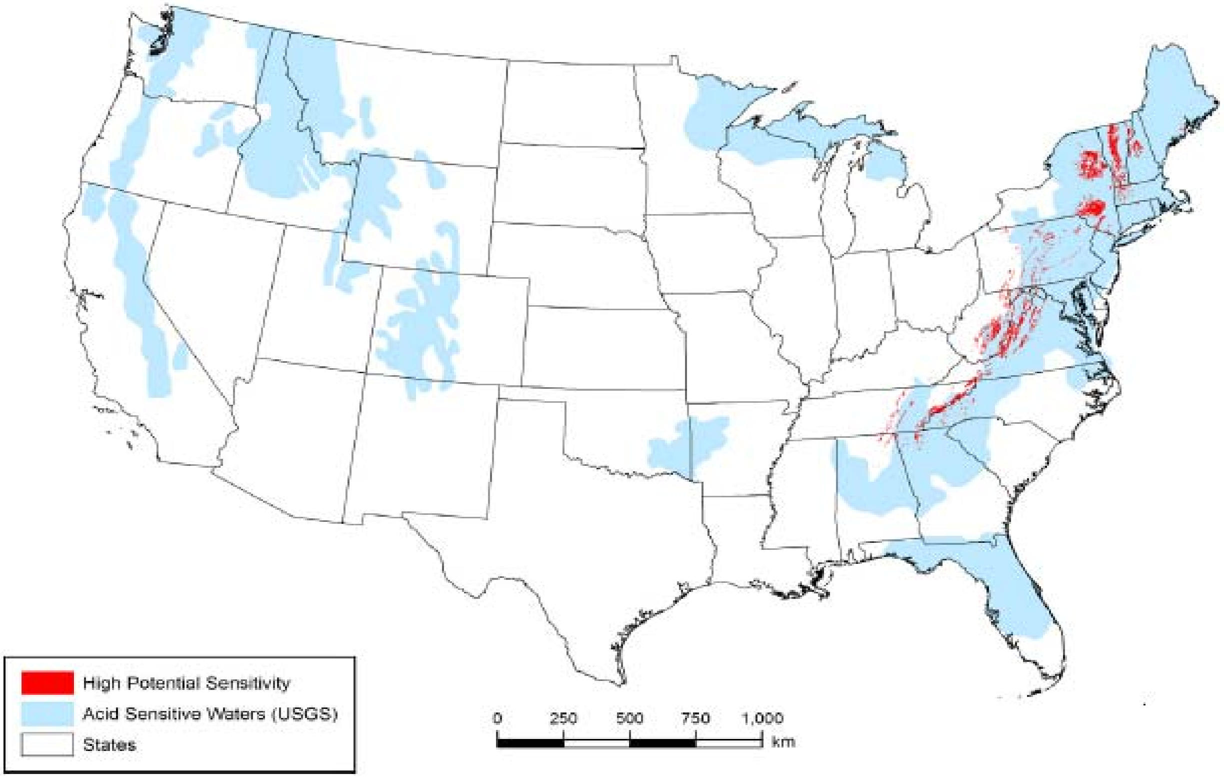
[35] Condition factor is an index that describes the relationship between fish weight and length, and is one measure of sublethal acidification stress that has been used to quantify effects of acidification on an individual fish (U.S.EPA, 2008f).

Table 5-14: Aquatic Status Categories

Category Label	ANC Levels	Expected Ecological Effects
Acute Concern	<0 micro equivalent per Liter (μeq/L)	Near complete loss of fish populations is expected. Planktonic communities have extremely low diversity and are dominated by acidophilic forms. The number of individuals in plankton species that are present is greatly reduced.
Severe Concern	0–20 μeq/L	Highly sensitive to episodic acidification. During episodes of high acidifying deposition, brook trout populations may experience lethal effects. Diversity and distribution of zooplankton communities decline sharply.
Elevated Concern	20–50 μeq/L	Fish species richness is greatly reduced (i.e., more than half of expected species can be missing). On average, brook trout populations experience sublethal effects, including loss of health, reproduction capacity, and fitness. Diversity and distribution of zooplankton communities decline.
Moderate Concern	50–100 μeq/L	Fish species richness begins to decline (i.e., sensitive species are lost from lakes). Brook trout populations are sensitive and variable, with possible sublethal effects. Diversity and distribution of zooplankton communities also begin to decline as species that are sensitive to acidifying deposition are affected.
Low Concern	>100 μeq/L	Fish species richness may be unaffected. Reproducing brook trout populations are expected where habitat is suitable. Zooplankton communities are unaffected and exhibit expected diversity and distribution.

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 5-8 illustrates those areas of the U.S. where aquatic ecosystems are at risk from acidification.

Figure 5-8: Areas Potentially Sensitive to Aquatic Acidification (U.S. EPA, 2008f)



Because 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.

While acidification is unlikely to have serious negative effects on, for example, 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. For example, there is evidence that certain population subgroups in the northeastern United States, such as the Hmong and Chippewa ethnic groups, have particularly high rates of self-caught fish consumption (Hutchison and Kraft, 1994; Peterson et al., 1994). However, it is not known if and how their consumption patterns are affected by the reductions in available fish populations caused by surface water acidification.

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 United States. Based on studies conducted in the northeastern United States, Kaval and Loomis (2003) estimated average consumer surplus values per day of $36 for recreational fishing (in 2007 dollars); therefore, the implied total annual value of freshwater fishing in the northeastern United States was $5.1 billion in 2006.[36] For recreation days, consumer surplus value is most commonly measured using recreation demand, travel cost models.

Another estimate of the overarching ecological benefits associated with reducing lake acidification levels in Adirondacks National Park can be derived from the contingent valuation (CV) survey (Banzhaf et al., 2006), which elicited values for specific improvements in acidification-related water quality and ecological conditions in Adirondack lakes. The survey described a base version with minor improvements said to result from the program, and a scope version with large improvements due to the program and a gradually worsening status quo. After adapting and transferring the results of this study and converting the 10-year annual payments to permanent annual payments using discount rates of 3% and 5%, the WTP estimates ranged from $48 to $107 per year per household (in 2004 dollars) for the base version and $54 to $154 for the scope version. Using these estimates, the aggregate annual benefits of eliminating all anthropogenic sources of NO_x and SO_x emissions were estimated to range from $291 million to $829 million (U.S. EPA, 2009c).[37]

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. These services are disrupted by the toxic effects of acidification on fish and other aquatic life. Although it is difficult to quantify these services and how they are affected by acidification, some of these services may be captured through measures of provisioning and cultural services.

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[36] These estimates reflect the total value of the service, not the marginal change in the value of the service as a result of the emission reductions achieved by this rule.

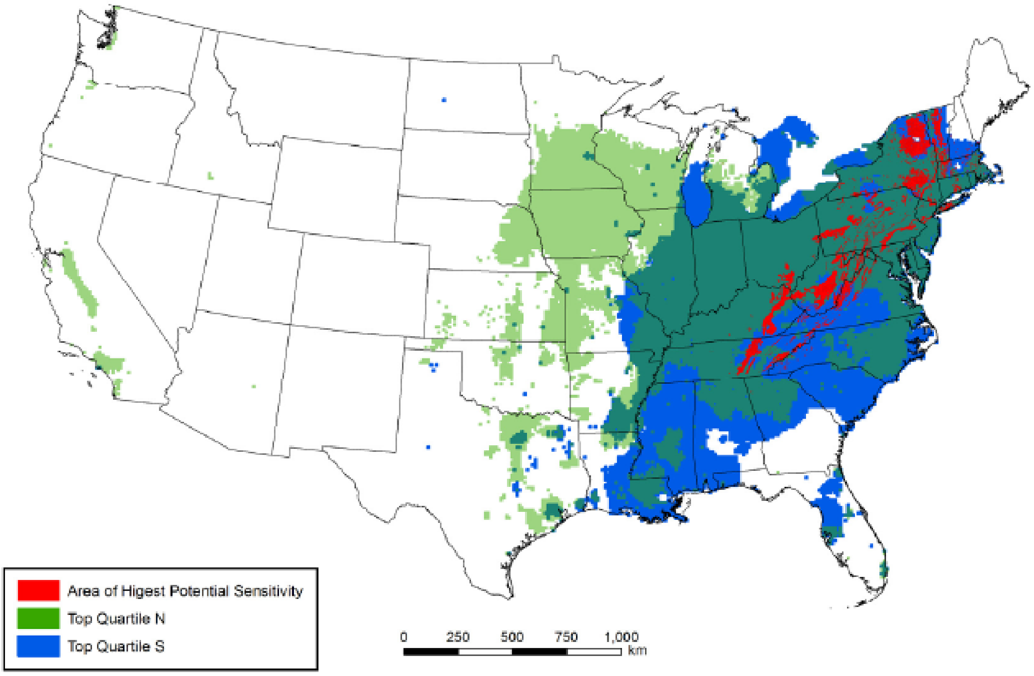
[37] These estimates reflect the total value of the service, not the marginal change in the value of the service as a result of the emission reductions achieved by this rule.

5.5.2.4 Terrestrial 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. Inorganic aluminum is toxic to some tree roots. 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. In the U.S., terrestrial effects of acidification are best described for forested ecosystems (especially red spruce and sugar maple ecosystems) with additional information on other plant communities, including shrubs and lichen (U.S. EPA, 2008f).

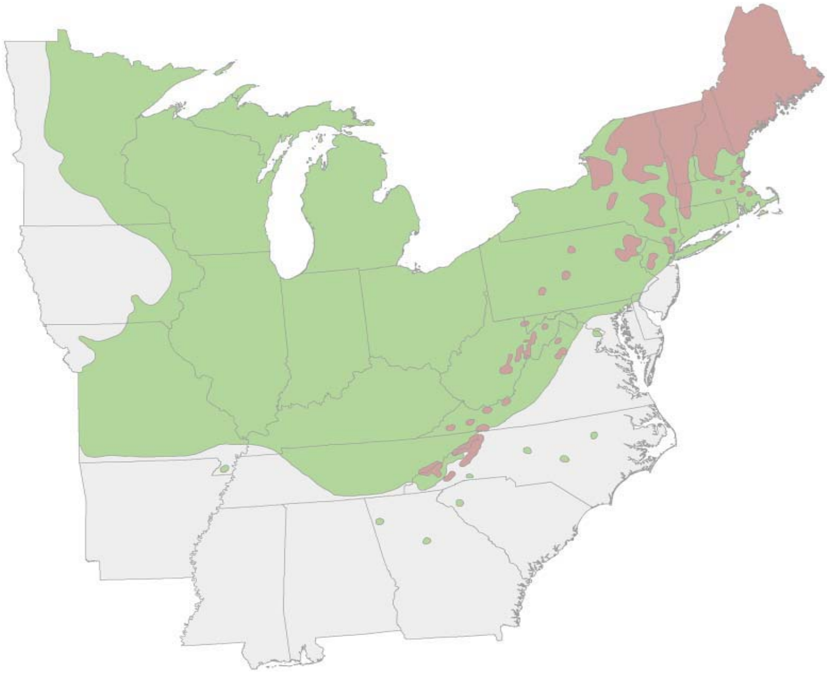
Certain ecosystems in the continental U.S. are potentially sensitive to terrestrial acidification, which is the greatest concern regarding nitrogen and sulfur deposition U.S. EPA (2008f). Figure 5-9 depicts the areas across the U.S. that are potentially sensitive to terrestrial acidification.

Figure 5-9: Areas Potentially Sensitive to Terrestrial Acidification (U.S. EPA, 2008f)



Both coniferous and deciduous forests throughout the eastern U.S. are experiencing gradual losses of base cation nutrients from the soil due to accelerated leaching from acidifying deposition. This change in nutrient availability may reduce the quality of forest nutrition over the long term. Evidence suggests that red spruce and sugar maple in some areas in the eastern U.S. have experienced declining health because of this deposition. For red spruce, (*Picea rubens*) dieback or decline has been observed across high elevation landscapes of the northeastern U.S., and to a lesser extent, the southeastern U.S., and acidifying deposition has been implicated as a causal factor (DeHayes et al., 1999). Figure 5-10 shows the distribution of red spruce (brown) and sugar maple (green) in the eastern U.S.

Figure 5-10: Distribution of Red Spruce (pink) and Sugar Maple (green) in the Eastern U.S. (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).

Forests in the northeastern United States provide several important and valuable provisioning services in the form of tree products. Sugar maples are a particularly important commercial hardwood tree species, providing timber and maple syrup. In the United States, sugar maple saw timber was nearly 900 million board feet in 2006 (USFS, 2006), and annual production of maple syrup was nearly 1.4 million gallons, accounting for approximately 19% of worldwide production. The total annual value of U.S. production in these years was approximately $160 million (NASS, 2008). Red spruce is also used in a variety of products including lumber, pulpwood, poles, plywood, and musical instruments. The total removal of red spruce saw timber from timberland in the United States was over 300 million board feet in 2006 (USFS, 2006).

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. Red spruce forests are home to two federally listed species and one delisted species:

1. Spruce-fir moss spider (*Microhexura montivaga*)—endangered
2. Rock gnome lichen (*Gymnoderma lineare*)—endangered
3. Virginia northern flying squirrel (*Glaucomys sabrinus fuscus*)—delisted, but important

Forestlands support a wide variety of outdoor recreational activities, including fishing, hiking, camping, off-road driving, hunting, and wildlife viewing. Regional statistics on recreational activities that are specifically forest based are not available; however, more general data on outdoor recreation provide some insights into the overall level of recreational services provided by forests. More than 30% of the U.S. adult population visited a wilderness or primitive area during the previous year and engaged in day hiking (Cordell et al., 2008). From 1999 to 2004, 16% of adults in the northeastern United States participated in off-road vehicle recreation, for an average of 27 days per year (Cordell et al., 2005). The average consumer surplus value per day of off-road driving in the United States was $25 (in 2007 dollars), and the implied total annual value of off-road driving recreation in the northeastern United States was more than $9 billion (Kaval and Loomis, 2003). More than 5% of adults in the northeastern United States participated in nearly 84 million hunting days (U.S. FWS and U.S. Census Bureau, 2007). Ten percent of adults in northeastern states participated in wildlife viewing away from home on 122 million days in 2006. For these recreational activities in the northeastern United States, Kaval and Loomis (2003) estimated average consumer surplus values per day of $52 for hunting and $34 for wildlife viewing (in 2007 dollars). The implied total annual value of hunting and wildlife viewing in the northeastern United States was, therefore, $4.4 billion and $4.2 billion, respectively, in 2006.

As previously mentioned, 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. However, one recreational activity that is directly dependent on forest conditions is fall color viewing. Sugar maple trees, in particular, are known for their bright colors and are, therefore, an essential aesthetic component of most fall color landscapes. A survey of residents in the Great Lakes area found that roughly 30% of residents reported at least one trip in the previous year involving fall color viewing (Spencer and Holecek, 2007). In a separate study conducted in Vermont, Brown (2002) reported that more than 22% of households visiting Vermont in 2001 made the trip primarily for viewing fall colors.

Two studies estimated values for protecting high-elevation spruce forests in the southern Appalachian Mountains. Kramer et al. (2003) conducted a contingent valuation study estimating households' WTP for programs to protect remaining high-elevation spruce forests from damages associated with air pollution and insect infestation. Median household WTP was estimated to be roughly $29 (in 2007 dollars) for a smaller program, and $44 for the more extensive program. Jenkins et al. (2002) conducted a very similar study in seven Southern Appalachian states on a potential program to maintain forest conditions at status quo levels. The overall mean annual WTP for the forest protection programs was $208 (in 2007 dollars). Multiplying the average WTP estimate from these studies by the total number of households in the seven-state Appalachian region results in an aggregate annual range of $470 million to $3.4 billion for avoiding a significant decline in the health of high-elevation spruce forests in the Southern Appalachian region.[38]

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.

5.5.3 Ecological Effects Associated with the Role of Sulfate in Mercury Methylation

Mercury is a highly neurotoxic contaminant that enters the food web as a methylated compound, methylmercury (U.S. EPA, 2008f). The contaminant is concentrated in higher

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[38] These estimates reflect the marginal value of the service for the hypothetical program described in the survey, not the marginal change in the value of the service as a result of the emission reductions achieved by this rule.

trophic levels, including fish eaten by humans. Experimental evidence has established that only inconsequential amounts of methylmercury can be produced in the absence of sulfate (U.S. EPA, 2008f). Many variables influence how much mercury accumulates in fish, but elevated mercury levels in fish can only occur where substantial amounts of methylmercury are present (U.S. EPA, 2008f). Current evidence indicates that in watersheds where mercury is present, increased sulfate deposition very likely results in methylmercury accumulation in fish (Drevnick et al., 2007; Munthe et al., 2007). The ISA for Oxides of Nitrogen and Sulfur: Ecological Criteria ISA concluded that evidence is sufficient to infer a casual relationship between sulfur deposition and increased mercury methylation in wetlands and aquatic environments (U.S. EPA, 2008f).

Establishing the quantitative relationship between sulfate and mercury methylation in natural settings is difficult because of the presence of multiple interacting factors in aquatic and terrestrial environments, including wetlands, aquatic environments where sulfate, sulfur-reducing bacteria (SRB), and inorganic mercury are present (U.S. EPA, 2008f). These are the three primary requirements for bacterially-mediated conversion to methylmercury. Additional factors affecting conversion include the presence of anoxic conditions, temperature, the presence and types of organic matter, the presence and types of mercury-binding species, and watershed effects (e.g., watershed type, land cover, water body limnology, and runoff loading). With regard to methylmercury, the highest concentrations in the environment generally occur at or near the sedimentary surface, below the oxic–anoxic boundary. Although mercury methylation can occur within the water column, there is generally a far greater contribution of mercury methylation from sediments because of anoxia and of greater concentrations of SRB, substrate, and sulfate. Figure 5-15 depicts the mercury cycle.

Figure 5-15: The mercury cycle in an ecosystem (USGS, 2006)

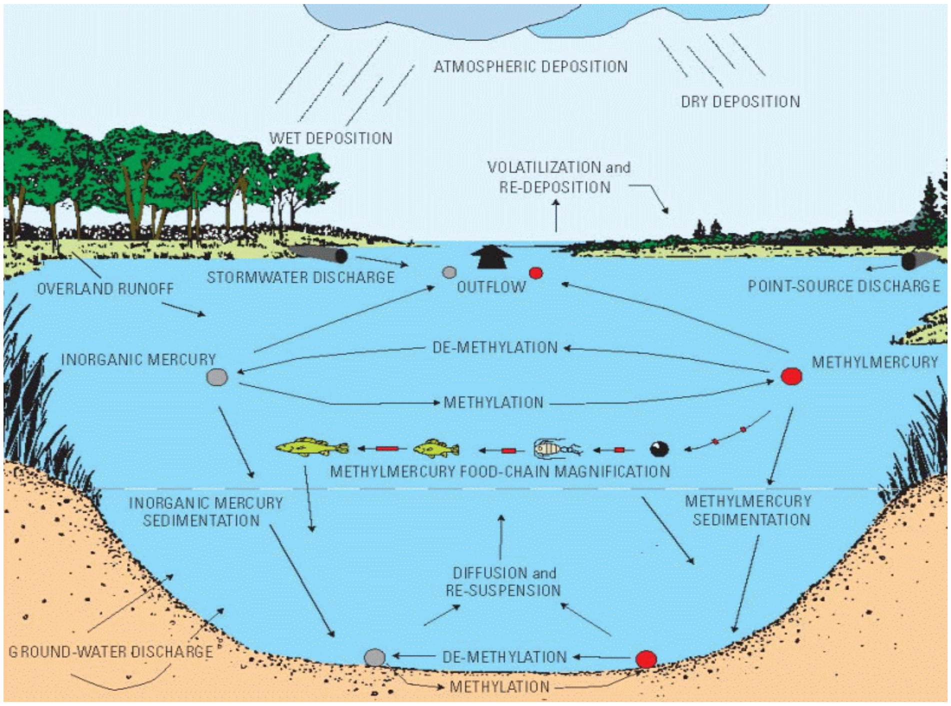
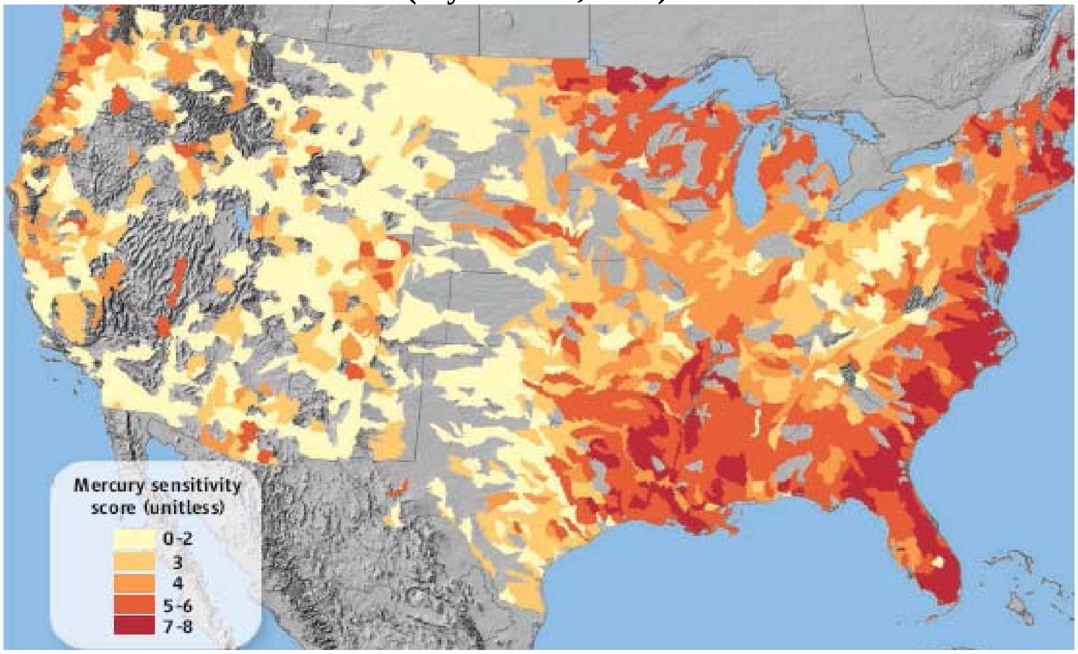


Figure 5-16 illustrates a map of mercury-sensitive watersheds based on sulfate concentrations, ANC, levels of dissolved organic carbon and pH, mercury species concentrations, and soil types to gauge the methylation sensitivity (Myers et al., 2007).

Figure 5.16: Preliminary USGS map of mercury methylation–sensitive watersheds (Myers et al., 2007)



Decreases in sulfate deposition/emissions have already shown reductions in methylmercury (U.S. EPA, 2008f). Observed decreases in methylmercury fish tissue concentrations have been linked to decreased acidification and declining sulfate and mercury deposition (Hrabik and Watras, 2002; Drevnick et al., 2007).

In the U.S., consumption of fish and shellfish are the main sources of methylmercury exposure to humans. Methylmercury builds up more in some types of fish and shellfish than in others. The levels of methylmercury in high and shellfish vary widely depending on what they eat, how long they live, and how high they are in the food chain. Most fish, including ocean species and local freshwater fish, contain some methylmercury. For example, in recent studies by EPA and the U.S. Geological Survey (USGS) of fish tissues, every fish samples contained some methylmercury.

State-level fish consumption advisories for mercury are based on state criteria, many of which are based on EPA's fish tissue criterion for methylmercury (U.S. EPA, 2001) or on U.S. Food and Drug Administration's action levels (U.S. FDA, 2001). In 2008, there were 3,361 fish advisories issued at least in part for mercury contamination (80% of all fish advisories), covering 16.8 million lake acres (40% of total lake acreage) and 1.3 million river miles (35% of total river miles) over all 50 states, one U.S. territory, and 3 tribes (U.S. EPA, 2009f). Recently, the U.S. Geological Survey (USGS) examined mercury levels in top-predator fish, bed sediment, and water from 291 streams across the U.S. (Scudder et al., 2009). USGS detected mercury contamination in every fish sampled, and the concentration of mercury in fish exceeded EPA's criterion in 27% of the sites sampled.

The ecosystem service most directly affected by sulfate-mediated mercury methylation is the provision of fish for consumption as a food source. This service is of particular importance to groups engaged in subsistence fishing, pregnant women and young children. While it is not possible to quantify the reduction in fish consumption due to the presence of methylmercury in fish from sulfur deposition, it is likely, given the number of state advisories and the EPA/FDA guidelines (U.S. EPA/FDA, 2004) on consumption for pregnant women and young children, that this service is negatively affected.

Research shows that most people's fish consumption does not cause a mercury-related health concern. However, certain people may be at higher risk because of their routinely high consumption of fish (e.g., tribal and other subsistence fishers and their families who rely heavily on fish for a substantial part of their diet). It has been demonstrated that high levels of methylmercury in the bloodstream of unborn babies and young children may harm the developing nervous system, making the child less able to think and learn. Moreover, mercury exposure at high levels can harm the brain, heart, kidneys, lungs, and immune system of people of all ages. The majority of fish consumed in the U.S. are ocean species. The methylmercury concentrations in ocean fish species are primarily influences by the global mercury pool. However, the methylmercury found in local fish can be due, at least partly, to mercury emissions from local sources.

Several studies suggest that the methylmercury content of fish may reduce these cardio-protective effects of fish consumption. Some of these studies also suggest that methylmercury may cause adverse effects to the cardiovascular system. For example, the NRC (2000) review of the literature concerning methylmercury health effects took note of two epidemiological studies that found an association between dietary exposure to methylmercury and adverse cardiovascular effects.[39] Moreover, in a study of 1,833 males in Finland aged 42 to 60 years, Solonen et al. (1995) observed a relationship between methylmercury exposure via fish consumption and acute myocardial infarction (AMI or heart attacks), coronary heart disease, cardiovascular disease, and all-cause mortality.[40] The NRC also noted a study of 917 seven year old children in the Faroe Islands, whose initial exposure to methylmercury was *in utero* although post natal exposures may have occurred as well. At seven years of age, these children exhibited an increase in blood pressure and a decrease in heart rate variability.[41] Based on these and other studies, NRC concluded in 2000 that, while "the data base is not as extensive for cardiovascular effects as it is for other end points (i.e. neurologic effects) the cardiovascular system appears to be a target for methylmercury toxicity."[42]

Since publication of the NRC report there have been some 30 published papers

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[39] National Research Council (NRC). 2000. Toxicological Effects of Methylmercury. Committee on the Toxicological Effects of Methylmercury, Board on Environmental Studies and Toxicology. National Academies Press. Washington, DC. pp.168-173.

[40] Salonen, J.T., Seppanen, K. Nyyssonen et al. 1995. "Intake of mercury from fish lipid peroxidation, and the risk of myocardial infarction and coronary, cardiovascular and any death in Eastern Finnish men." Circulation, 91 (3):645-655.

[41] Sorensen, N, K. Murata, E. Budtz-Jorgensen, P. Weihe, and Grandjean, P., 1999. "Prenatal Methylmercury Exposure As A Cardiovascular Risk Factor At Seven Years of Age", Epidemiology, pp370-375.

[42] National Research Council (NRC). 2000. Toxicological Effects of Methylmercury. Committee on the Toxicological Effects of Methylmercury, Board on Environmental Studies and Toxicology. National Academies Press. Washington, DC. p. 229.

presenting the findings of studies that have examined the possible cardiovascular effects of methylmercury exposure. These studies include epidemiological, toxicological, and toxicokinetic investigations. Over a dozen review papers have also been published. If there is a causal relationship between methylmercury exposure and adverse cardiovascular effects, then reducing exposure to methylmercury would result in public health benefits from reduced cardiovascular effects.

In early 2010, EPA sponsored a workshop in which a group of experts were asked to assess the plausibility of a causal relationship between methylmercury exposure and cardiovascular health effects and to advise EPA on methodologies for estimating population level cardiovascular health impacts of reduced methylmercury exposure. The report from that workshop is in preparation.

Because establishing the quantitative relationship between sulfate and mercury methylation in natural settings is difficult, we were unable to model the changes in the methylation process, bioaccumulation in fish tissue, and human consumption of mercury-contaminated fish that would be needed in order to estimate the human health benefits from reducing sulfate emissions in this rule.

5.5.4 Ecological Effects Associated with Gaseous Sulfur Dioxide

Uptake of gaseous sulfur dioxide in a plant canopy is a complex process involving adsorption to surfaces (leaves, stems, and soil) and absorption into leaves. SO_2 penetrates into leaves through to the stomata, although there is evidence for limited pathways via the cuticle. Pollutants must be transported from the bulk air to the leaf boundary layer in order to get to the stomata. When the stomata are closed, as occurs under dark or drought conditions, resistance to gas uptake is very high and the plant has a very low degree of susceptibility to injury. In contrast, mosses and lichens do not have a protective cuticle barrier to gaseous pollutants or stomates and are generally more sensitive to gaseous sulfur and nitrogen than vascular plants (U.S. EPA, 2008f). Acute foliar injury usually happens within hours of exposure, involves a rapid absorption of a toxic dose, and involves collapse or necrosis of plant tissues. Another type of visible injury is termed chronic injury and is usually a result of variable SO_2 exposures over the growing season. Besides foliar injury, chronic exposure to low SO_2 concentrations can result in reduced photosynthesis, growth, and yield of plants (U.S. EPA, 2008f). These effects are cumulative over the season and are often not associated with visible foliar injury. As with foliar injury, these effects vary among species and growing environment. SO_2 is also considered the primary factor causing the death of lichens in many urban and industrial areas (Hutchinson et al., 1996).

In addition to the role of sulfate deposition on methylation, the technologies installed to reduce emissions of NO_x and SO_2 associated with this proposed rule would also reduce mercury emissions. EPA recently commissioned an information collection request that will soon provide greatly improved power industry mercury emissions estimates that will enable the Agency to better estimate mercury emissions changes from its air emissions control actions. For this reason, the Agency did not estimate Hg changes in this rule and will instead wait for these new data which will be available in the near future. Due to time and resource limitations, we were unable in any event to model mercury dispersion, deposition, methylation, bioaccumulation in fish tissue, and human consumption of mercury-contaminated fish that would be needed in order to estimate the human health benefits from reducing these mercury emissions.

5.5.5 Nitrogen Enrichment

5.5.5.1 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.

In addition to affecting provisioning services through commercial fish harvests, eutrophication in estuaries may also affect 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, there were 26 million days of saltwater fishing coastal states from North Carolina to Massachusetts in 2006 (FWA and Census, 2007). Assuming an average consumer surplus value for a fishing day at \$36 (in 2007 dollars) in the Northeast and \$87 in the Southeast (Kaval and Loomis, 2003), the aggregate value was approximately \$1.3 billion (in 2007 dollars).[43] In addition, almost 6 million adults participated in motorboating in coastal states from North Carolina to Massachusetts, for a total of nearly 63 million days annually during 1999–2000 (Leeworthy and Wiley, 2001). Using a national daily value estimate of \$32 (in 2007 dollars) for motorboating (Kaval and Loomis (2003), the aggregate value of these coastal motorboating outings was \$2 billion per year.[44] Almost 7 million participated in birdwatching for 175 million days per year, and more than 3 million participated in visits to non-beach coastal waterside areas.

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 from declines in SAV (MEA, 2005). SAV can help reduce wave energy levels and thus protect shorelines against excessive erosion, which increases the risks of episodic flooding and associated damages to near-shore properties or public infrastructure or even contribute to shoreline retreat.

5.5.5.2 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, 2008f) 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

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[43] These estimates reflect the total value of the service, not the marginal change in the value of the service as a result of the emission reductions achieved by this rule.

[44] These estimates reflect the total value of the service, not the marginal change in the value of the service as a result of the emission reductions achieved by this rule.

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.

One of the main provisioning services potentially affected by N deposition is grazing opportunities offered by grasslands for livestock production in the Central U.S. Although N deposition on these grasslands can offer supplementary nutritive value and promote overall grass production, there are concerns that fertilization may favor invasive grasses and shift the species composition away from native grasses. This process may ultimately reduce the productivity of grasslands for livestock production. Losses due to invasive grasses can be significant; for example, based on a bioeconomic model of cattle grazing in the upper Great Plains, Leitch, Leistritz, and Bangsund (1996) and Leistritz, Bangsund, and Hodur (2004) estimated $130 million in losses due to a leafy spurge infestation in the Dakotas, Montana, and Wyoming. [45] However, the contribution of N deposition to these losses is still uncertain.

5.5.6 Benefits of Reducing Ozone Effects on Vegetation and Ecosystems

Ozone causes discernible injury to a wide array of vegetation (U.S. EPA, 2006a; Fox and Mickler, 1996). In terms of forest productivity and ecosystem diversity, ozone may be the pollutant with the greatest potential for regional-scale forest impacts (U.S. EPA, 2006a). Studies have demonstrated repeatedly that ozone concentrations commonly observed in polluted areas can have substantial impacts on plant function (De Steiguer et al., 1990; Pye, 1988).

When ozone is present in the air, it can enter the leaves of plants, where it can cause significant cellular damage. Like carbon dioxide (CO_2) and other gaseous substances, ozone enters plant tissues primarily through the stomata in leaves in a process called "uptake" (Winner and Atkinson, 1986). Once sufficient levels of ozone (a highly reactive substance), or its reaction products, reaches the interior of plant cells, it can inhibit or damage essential cellular components and functions, including enzyme activities, lipids, and cellular membranes, disrupting the plant's osmotic (i.e., water) balance and energy utilization patterns

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[45] These estimates reflect the total value of the service, not the marginal change in the value of the service as a result of the emission reductions achieved by this rule.

(U.S. EPA, 2006a; Tingey and Taylor, 1982). With fewer resources available, the plant reallocates existing resources away from root growth and storage, above ground growth or yield, and reproductive processes, toward leaf repair and maintenance, leading to reduced growth and/or reproduction. Studies have shown that plants stressed in these ways may exhibit a general loss of vigor, which can lead to secondary impacts that modify plants' responses to other environmental factors. Specifically, plants may become more sensitive to other air pollutants, or more susceptible to disease, pest infestation, harsh weather (e.g., drought, frost) and other environmental stresses, which can all produce a loss in plant vigor in ozone-sensitive species that over time may lead to premature plant death. Furthermore, there is evidence that ozone can interfere with the formation of mycorrhiza, essential symbiotic fungi associated with the roots of most terrestrial plants, by reducing the amount of carbon available for transfer from the host to the symbiont (U.S. EPA, 2006a).

This ozone damage may or may not be accompanied by visible injury on leaves, and likewise, visible foliar injury may or may not be a symptom of the other types of plant damage described above. Foliar injury is usually the first visible sign of injury to plants from ozone exposure and indicates impaired physiological processes in the leaves (Grulke, 2003). When visible injury is present, it is commonly manifested as chlorotic or necrotic spots, and/or increased leaf senescence (accelerated leaf aging). Because ozone damage can consist of visible injury to leaves, it can also reduce the aesthetic value of ornamental vegetation and trees in urban landscapes, and negatively affects scenic vistas in protected natural areas.

Ozone can produce both acute and chronic injury in sensitive species depending on the concentration level and the duration of the exposure. Ozone effects also tend to accumulate over the growing season of the plant, so that even lower concentrations experienced for a longer duration have the potential to create chronic stress on sensitive vegetation. Not all plants, however, are equally sensitive to ozone. Much of the variation in sensitivity between individual plants or whole species is related to the plant’s ability to regulate the extent of gas exchange via leaf stomata (e.g., avoidance of ozone uptake through closure of stomata) (U.S. EPA, 2006a; Winner, 1994). After injuries have occurred, plants may be capable of repairing the damage to a limited extent (U.S. EPA, 2006a). Because of the differing sensitivities among plants to ozone, ozone pollution can also exert a selective pressure that leads to changes in plant community composition. Given the range of plant sensitivities and the fact that numerous other environmental factors modify plant uptake and response to ozone, it is not possible to identify threshold values above which ozone is consistently toxic for all plants.

Because plants are at the base of the food web in many ecosystems, changes to the plant community can affect associated organisms and ecosystems (including the suitability of habitats that support threatened or endangered species and below ground organisms living in the root zone). Ozone impacts at the community and ecosystem level vary widely depending upon numerous factors, including concentration and temporal variation of tropospheric ozone, species composition, soil properties and climatic factors (U.S. EPA, 2006a). In most instances, responses to chronic or recurrent exposure in forested ecosystems are subtle and not observable for many years. These injuries can cause stand-level forest decline in sensitive ecosystems (U.S. EPA, 2006a, McBride et al., 1985; Miller et al., 1982). It is not yet possible to predict ecosystem responses to ozone with much certainty; however, considerable knowledge of potential ecosystem responses has been acquired through long-term observations in highly damaged forests in the United States (U.S EPA, 2006a).

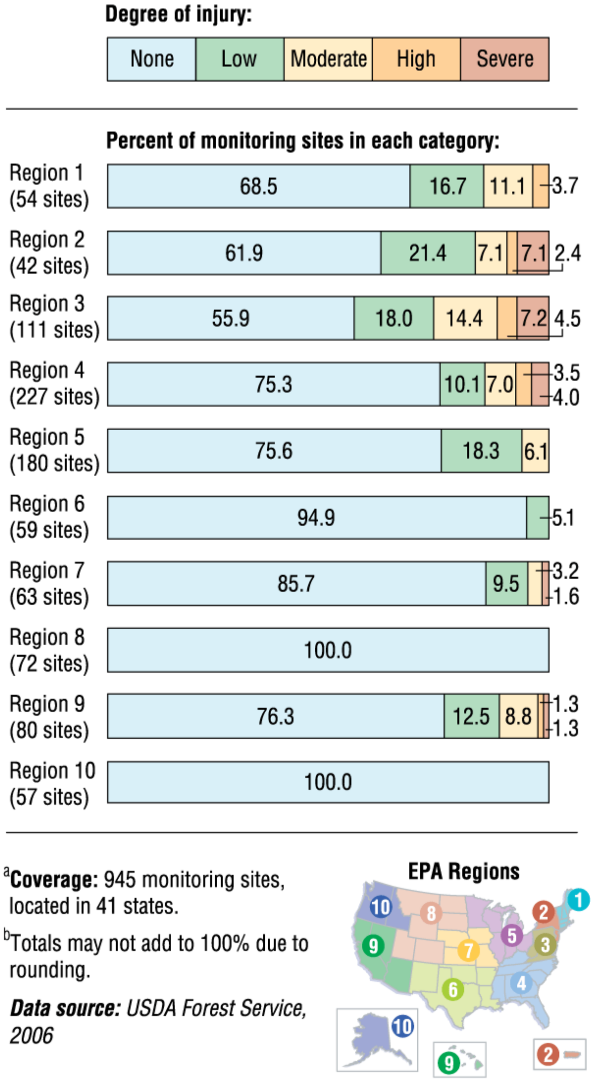
5.5.6.1 Ozone Effects on Forests

Air pollution can affect the environment and affect ecological systems, leading to changes in the ecological community and influencing the diversity, health, and vigor of individual species (U.S. EPA, 2006a). Ozone has been shown in numerous studies to have a strong effect on the health of many plants, including a variety of commercial and ecologically important forest tree species throughout the United States (U.S. EPA, 2007b).

In the U.S., this data comes from the U.S. Department of Agriculture (USDA) Forest Service Forest Inventory and Analysis (FIA) program. As part of its Phase 3 program, formerly known as Forest Health Monitoring, FIA examines ozone injury to ozone-sensitive plant species at ground monitoring sites in forestland across the country (excluding woodlots and urban trees). FIA looks for damage on the foliage of ozone-sensitive forest plant species at each site that meets certain minimum criteria. Because ozone injury is cumulative over the course of the growing season, examinations are conducted in July and August, when ozone injury is typically highest.

Monitoring of ozone injury to plants by the USDA Forest Service has expanded over the last 10 years from monitoring sites in 10 states in 1994 to nearly 1,000 monitoring sites in 41 states in 2002. The data underlying the indictor in Figure 5-13 are based on averages of all observations collected in 2002, the latest year for which data are publicly available at the time the study was conducted, and are broken down by U.S. EPA Regions. Ozone damage to forest plants is classified using a subjective five-category biosite index based on expert opinion, but designed to be equivalent from site to site. Ranges of biosite values translate to no injury, low or moderate foliar injury (visible foliar injury to highly sensitive or moderately sensitive plants, respectively), and high or severe foliar injury, which would be expected to result in tree-level or ecosystem-level responses, respectively (U.S. EPA, 2006a; Coulston, 2004). The highest percentages of observed high and severe foliar injury, which are most likely to be associated with tree or ecosystem-level responses, are primarily found in the Mid-Atlantic and Southeast regions.

Figure 5-13: Ozone Injury to Forest Plants in U.S. by EPA Regions, 2002[a, b]



[a] **Coverage:** 945 monitoring sites, located in 41 states.

[b] Totals may not add to 100% due to rounding.

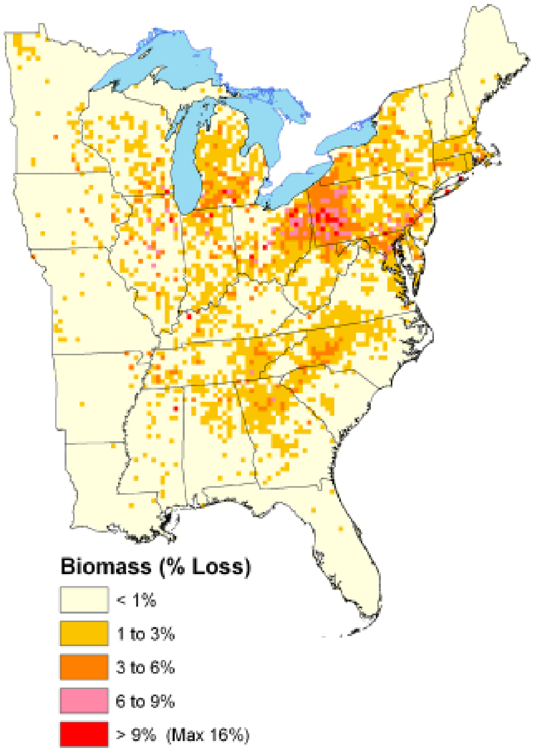
***Data source:** USDA Forest Service, 2006*

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

Some of the common tree species in the United States that are sensitive to ozone are black cherry (*Prunus serotina*), tulip-poplar (*Liriodendron tulipifera)*, and eastern white pine (*Pinus strobus)*. Ozone-exposure/tree-response functions have been developed for each of these tree species, as well as for aspen (*Populus tremuliodes*), and ponderosa pine (*Pinus ponderosa*) (U.S. EPA, 2007b). Other common tree species, such as oak (*Quercus* spp*.)* and hickory (*Carya* spp.), are not as sensitive to ozone. Consequently, with knowledge of the distribution of sensitive species and the level of ozone at particular locations, it is possible to estimate a "biomass loss" for each species across their range. As shown in Figure 5-14, current ambient levels of ozone are associated with significant biomass loss across large geographic areas (U.S. EPA, 2009b). However, this information is unavailable this rule.

To estimate the biomass loss for forest ecosystems across the eastern United States, the biomass loss for each of the seven tree species was calculated using the three-month, 12-hour W126 exposure metric at each location, along with each tree's individual C-R functions. The W126 exposure metric was calculated using monitored ozone data from CASTNET and AQS sites, and a three-year average was used to mitigate the effect of variations in meteorological and soil moisture conditions. The biomass loss estimate for each species was then multiplied by its prevalence in the forest community using the U.S. Department of Agriculture (USDA) Forest Service IV index of tree abundance calculated from Forest Inventory and Analysis (FIA) measurements (Prasad, 2003). Sources of uncertainty include the ozone-exposure/plant-response functions, the tree abundance index, and other factors (e.g., soil moisture). Although these factors were not considered, they can affect ozone damage (Chappelka, 1998).

Figure 5-14: Estimated Black Cherry, Yellow Poplar, Sugar Maple, Eastern White Pine, Virginia Pine, Red Maple, and Quaking Aspen Biomass Loss due to Current Ozone Exposure, 2006-2008 (U.S. EPA, 2009b)



Ozone damage to the plants including the trees and understory in a forest can affect the ability of the forest to sustain suitable habitat for associated species particularly threatened and endangered species that have existence value – a nonuse ecosystem service - for the public. Similarly, damage to trees and the loss of biomass can affect the forest's provisioning services in the form of timber for various commercial uses. In addition, ozone can cause discoloration of leaves and more rapid senescence (early shedding of leaves), which could negatively affect fall-color tourism because the fall foliage would be less available or less attractive. Beyond the aesthetic damage to fall color vistas, forests provide the public with many other recreational and educational services that may be impacted by reduced forest health including hiking, wildlife viewing (including bird watching), camping, picnicking, and hunting. Another potential effect of biomass loss in forests is the subsequent loss of climate regulation service in the form of reduced ability to sequester carbon (Felzer et al., 2005).

5.5.6.2 Ozone Effects on Crops and Urban Ornamentals

Laboratory and field experiments have also shown reductions in yields for agronomic crops exposed to ozone, including vegetables (e.g., lettuce) and field crops (e.g., cotton and wheat). Damage to crops from ozone exposures includes yield losses (i.e., in terms of weight, number, or size of the plant part that is harvested), as well as changes in crop quality (i.e., physical appearance, chemical composition, or the ability to withstand storage) (U.S. EPA, 2007b). The most extensive field experiments, conducted under the National Crop Loss Assessment Network (NCLAN) examined 15 species and numerous cultivars. The NCLAN results show that "several economically important crop species are sensitive to ozone levels typical of those found in the United States" (U.S. EPA, 2006a). In addition, economic studies have shown reduced economic benefits as a result of predicted reductions in crop yields, directly affecting the amount and quality of the provisioning service provided by the crops in question, associated with observed ozone levels (Kopp et al., 1985; Adams et al., 1986; Adams et al., 1989). According to the Ozone Staff Paper, there has been no evidence that crops are becoming more tolerant of ozone (U.S. EPA, 2007b). Using the Agriculture Simulation Model (AGSIM) (Taylor, 1994) to calculate the agricultural benefits of reductions in ozone exposure, U.S. EPA estimated that meeting a W126 standard of 21 ppm-hr would produce monetized benefits of approximately $160 million to $300 million (inflated to 2006 dollars) (U.S. EPA, 2007b). [46]

Urban ornamentals are an additional vegetation category likely to experience some degree of negative effects associated with exposure to ambient ozone levels. Because ozone causes visible foliar injury, the aesthetic value of ornamentals (such as petunia, geranium, and poinsettia) in urban landscapes would be reduced (U.S. EPA, 2007b). Sensitive ornamental species would require more frequent replacement and/or increased maintenance (fertilizer or pesticide application) to maintain the desired appearance because of exposure to ambient ozone (U.S. EPA, 2007b). In addition, many businesses rely on healthy-looking vegetation for their livelihoods (e.g., horticulturalists, landscapers, Christmas tree growers, farmers of leafy crops, etc.) and a variety of ornamental species have been listed as sensitive

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[46] These estimates illustrate the value of vegetation effects from a substantial reduction of ozone concentrations, not the marginal change in ozone concentrations anticipated a result of the emission reductions achieved by this rule.

to ozone (Abt Associates, 1995). The ornamental landscaping industry is valued at more than $30 billion (inflated to 2006 dollars) annually, by both private property owners/tenants and by governmental units responsible for public areas (Abt Associates, 1995). Therefore, urban ornamentals represent a potentially large unquantified benefit category. This aesthetic damage may affect the enjoyment of urban parks by the public and homeowners' enjoyment of their landscaping and gardening activities. In the absence of adequate exposure-response functions and economic damage functions for the potential range of effects relevant to these types of vegetation, we cannot conduct a quantitative analysis to estimate these effects.

5.5.7 Unquantified SO_2 and NO_2 -Related Human Health Benefits

Following an extensive evaluation of health evidence from epidemiologic and laboratory studies, the Integrated Science Assessment for Sulfur Dioxide concluded that there is a causal relationship between respiratory health effects and short-term exposure to SO_2 (U.S. EPA, 2008). The immediate effect of SO_2 on the respiratory system in humans is bronchoconstriction. Asthmatics are more sensitive to the effects of SO_2 likely resulting from preexisting inflammation associated with this disease. A clear concentration-response relationship has been demonstrated in laboratory studies following exposures to SO_2 at concentrations between 20 and 100 ppb, both in terms of increasing severity of effect and percentage of asthmatics adversely affected. Based on our review of this information, we identified four short-term morbidity endpoints that the SO_2 ISA identified as a "causal relationship": asthma exacerbation, respiratory-related emergency department visits, and respiratory-related hospitalizations. The differing evidence and associated strength of the evidence for these different effects is described in detail in the SO_2 ISA. The SO_2 ISA also concluded that the relationship between short-term SO_2 exposure and premature mortality was "suggestive of a causal relationship" because it is difficult to attribute the mortality risk effects to SO_2 alone. Although the SO_2 ISA stated that studies are generally consistent in reporting a relationship between SO_2 exposure and mortality, there was a lack of robustness of the observed associations to adjustment for pollutants. We did not quantify these benefits due to time constraints.

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). The NO_2 ISA provides a comprehensive review of the current evidence of health and environmental effects of NO_2. The NO_2 ISA concluded that the evidence "is sufficient to infer a likely causal relationship between short-term NO_2 exposure and adverse effects on the respiratory system" (ISA, section 5.3.2.1). 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). The NO_2 ISA concluded that the relationship between short-term NO_2 exposure and premature mortality was "suggestive but not sufficient to infer a causal relationship" because it is difficult to attribute the mortality risk effects to NO_2 alone. Although the NO_2 ISA stated that studies consistently reported a relationship between NO_2 exposure and mortality, the effect was generally smaller than that for other pollutants such as PM. We did not quantify these benefits due to time constraints.

5.6 Social Cost of Carbon and Greenhouse Gas Benefits

EPA has assigned a dollar value to reductions in carbon dioxide (CO_2) emissions using recent estimates of the "social cost of carbon" (SCC). The SCC is an estimate of the monetized damages associated with an incremental increase in carbon emissions in a given year. It is intended to include (but is not limited to) changes in net agricultural productivity, human health, property damages from increased flood risk, and the value of ecosystem services due to climate change. The SCC estimates used in this analysis were developed through an interagency process that included EPA and other executive branch entities, and concluded in February 2010. EPA first used these SCC estimates in the benefits analysis for the final joint EPA/DOT Rulemaking to establish Light-Duty Vehicle Greenhouse Gas Emission Standards and Corporate Average Fuel Economy Standards; see the rule's preamble for discussion about application of SCC (75 FR 25324; 5/7/10). The SCC Technical Support Document (SCC TSD) provides a complete discussion of the methods used to develop these SCC estimates.[47]

The interagency group selected four SCC values for use in regulatory analyses, which we have applied in this analysis: $5, $21, $35, and $65 per metric ton of CO_2 emissions[48] in 2010, in 2007 dollars. The first three values are based on the average SCC from three integrated assessment models, at discount rates of 2.5, 3, and 5 percent, respectively. SCCs at several discount rates are included because the literature shows that the SCC is quite sensitive to assumptions about the discount rate, and because no consensus exists on the appropriate rate to use in an intergenerational context. The fourth value is the 95th percentile of the SCC from all three models at a 3 percent discount rate. It is included to represent higher-than-expected impacts from temperature change further out in the tails of the SCC distribution. Low probability, high impact events are incorporated into all of the SCC values through explicit consideration of their effects in two of the three models as well as the use of a probability density function for equilibrium climate sensitivity. Treating climate sensitivity probabilistically results in more high temperature outcomes, which in turn lead to higher projections of damages.

The SCC increases over time because future emissions are expected to produce larger incremental damages as physical and economic systems become more stressed in response to greater climatic change. Note that the interagency group estimated the growth rate of the SCC directly using the three integrated assessment models rather than assuming a constant annual growth rate. This helps to ensure that the estimates are internally consistent with other modeling assumptions. The SCC estimates for the analysis years of 2014, in 2006 dollars are provided in Table 5-15.

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[47] Docket ID EPA-HQ-OAR-2009-0472-114577, *Technical Support Document: Social Cost of Carbon for Regulatory Impact Analysis Under Executive Order 12866,* Interagency Working Group on Social Cost of Carbon, with participation by Council of Economic Advisers, Council on Environmental Quality, Department of Agriculture, Department of Commerce, Department of Energy, Department of Transportation, Environmental Protection Agency, National Economic Council, Office of Energy and Climate Change, Office of Management and Budget, Office of Science and Technology Policy, and Department of Treasury (February 2010). Also available at http://www.epa.gov/otaq/climate/regulations.htm

[48] The interagency group decided that these estimates apply only to CO_2 emissions. Given that warming profiles and impacts other than temperature change (e.g. ocean acidification) vary across GHGs, the group concluded "transforming gases into CO_2-equivalents using GWP, and then multiplying the carbon-equivalents by the SCC, would not result in accurate estimates of the social costs of non-CO_2 gases" (SCC TSD, pg 13).

When attempting to assess the incremental economic impacts of carbon dioxide emissions, the analyst faces a number of serious challenges. A recent report from the National Academies of Science (NRC 2009) points out that any assessment will suffer from uncertainty, speculation, and lack of information about (1) future emissions of greenhouse gases, (2) the effects of past and future emissions on the climate system, (3) the impact of changes in climate on the physical and biological environment, and (4) the translation of these environmental impacts into economic damages. As a result, any effort to quantify and monetize the harms associated with climate change will raise serious questions of science, economics, and ethics and should be viewed as provisional.

The interagency group noted a number of limitations to the SCC analysis, including the incomplete way in which the integrated assessment models capture catastrophic and non-catastrophic impacts, their incomplete treatment of adaptation and technological change, uncertainty in the extrapolation of damages to high temperatures, and assumptions regarding risk aversion. The limited amount of research linking climate impacts to economic damages makes the interagency modeling exercise even more difficult. The interagency group hopes that over time researchers and modelers will work to fill these gaps and that the SCC estimates used for regulatory analysis by the Federal government will continue to evolve with improvements in modeling. Additional details on these limitations are discussed in the SCC TSD.

In light of these limitations, the interagency group has committed to updating the current estimates as the science and economic understanding of climate change and its impacts on society improves over time. Specifically, the interagency group has set a preliminary goal of revisiting the SCC values within two years or at such time as substantially updated models become available, and to continue to support research in this area.

Applying the global SCC estimates to the estimated reductions in CO_2 emissions for the range of policy scenarios, we estimate the dollar value of the climate related benefits captured by the models for each analysis year. For internal consistency, the annual benefits are discounted back to NPV terms using the same discount rate as each SCC estimate (i.e.

5%, 3%, and 2.5%) rather than 3% and 7%.[49] These estimates are provided in Table 5-16.

Table 5-15. Social Cost of Carbon (SCC) Estimates (per tonne of CO_2) for 2014 (in 2006$)[a]

Discount Rate and Statistic		*SCC estimate*
5%	Average	$5.4
3%	Average	$22.7
2.5%	Average	$36.7
3%	95%ile	$69.2

[a] The SCC values are dollar-year and emissions-year specific. SCC values represent only a partial accounting of climate impacts.

Table 5-16. Monetized Benefits of CO_2 Emissions Reductions in 2014 (in millions of 2006$)[a]

Discount Rate and Statistic		*SCC estimate*
5%	Average	$82
3%	Average	$350
2.5%	Average	$560
3%	95%ile	$1,100

[a] The SCC values are dollar-year and emissions-year specific. SCC values represent only a partial accounting of climate impacts.

5.7 Benefits Results

Applying the impact and valuation functions described previously in this chapter to the estimated changes in ozone and PM yields estimates of the changes in physical damages (e.g., premature mortalities, cases, admissions, and change in light extinction) and the associated monetary values for those changes. Estimates of physical health impacts among those states in either the ozone or $PM_{2.5}$ trading region, or outside the trading region, are presented in Table 5-15. Monetized values for both health and welfare endpoints within the trading region are presented in Table 5-16, along with total aggregate monetized benefits. All of the monetary benefits are in constant-year 2006 dollars. The $PM_{2.5}$-related benefits of the Direct Control and Intrastate Trading scenarios were within about 5% of the preferred

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[49] It is possible that other benefits or costs of proposed regulations unrelated to CO_2 emissions will be discounted at rates that differ from those used to develop the SCC estimates.

remedy. The results of this analysis may be found in Appendix A. The benefits of the more and less stringent SO_2 sensitivity analyses may be found in the Chapter 10 cost-benefit comparison chapter.

Table 5-17: Estimated Reduction in Incidence of Adverse Health Effects of the Proposed remedy (95% confidence intervals)[A]

Health Effect	*Within transport region*	*Beyond transport region*	*Total*
PM-Related endpoints			
Premature Mortality			
Pope et al. (2002) (age >30)	14,000 (4,000—24,000)	130 (35—220)	14,000 (4,000—25,000)
Laden et al. (2006) (age >25)	36,000 (17,000—55,000)	320 (150—500)	36,000 (17,000—56,000)
Infant (< 1 year)	59 (-66—180)	0.3 (-0.3—0.8)	59 (-66—180)
Chronic Bronchitis	9,200 (310—18,000)	89 (3—160)	9,200 (320—18,000)
Non-fatal heart attacks (age > 18)	22,000 (5,700—39,000)	250 (64—440)	23,000 (5,800—39,000)
Hospital admissions—respiratory (all ages)	3,500 (1,400—5,500)	35 (14—56)	3,500 (1,400—5,500)
Hospital admissions—cardiovascular (age > 18)	7,500 (5,200—8,800)	76 (51—93)	7,500 (5,200—8,900)
Emergency room visits for asthma (age < 18)	14,000 (7,100—21,000)	71 (36—110)	14,000 (7,200—21,000)
Acute bronchitis (age 8-12)	21,000 (-4,800—46,000)	150 (33—320)	21,000 (-4,800—46,000)
Lower respiratory symptoms (age 7-14)	250,000 (98,000—400,000)	1,700 (670—2,800)	250,000 (98,000—400,000)
Upper respiratory symptoms (asthmatics age 9-18)	190,000 (36,000—350,000)	1,300 (250—2,400)	190,000 (36,000—350,000)
Asthma exacerbation (asthmatics 6-18)	230,000 (8,300—800,000)	1,700 (11—5,700)	240,000 (8,300—800,000)
Lost work days (ages 18-65)	1,800,000 (1,500,000—2,000,000)	14,000 (12,000—17,000)	1,800,000 (1,500,000—2,000,000)
Minor restricted-activity days (ages 18-65)	10,000,000 (8,600,000—12,000,000)	86,000 (71,000—100,000)	11,000,000 (8,600,000—12,000,000)

Ozone-related endpoints				
Premature mortality				
Multi-city and NMMAPS-	Bell et al. (2004) (all ages)	50 (16—83)	0.6 (0.2—1)	50 (17—84)
	Schwartz et al. (2005) (all ages)	76 (23—130)	1 (0.2—2)	77 (24—130)
	Huang et al. (2005) (all ages)	83 (31—130)	1 (0.3—2)	84 (31—140)
Meta-analyses	Ito et al. (2005) (all ages)	220 (130—310)	3 (2—4)	230 (140—320)
	Bell et al. (2005) (all ages)	160 (76—250)	2 (1—3)	160 (77—250)
	Levy et al. (2005) (all ages)	230 (160—300)	3 (2—4)	230 (160—300)
Hospital admissions—respiratory causes (ages > 65)		380 (-18—730)	4 (-0.4—9)	390 (-18—740)
Hospital admissions—respiratory causes (ages <2)		290 (130—460)	4 (1—6)	300 (130—460)
Emergency room visits for asthma (all ages)		230 (-30—730)	2 (-0.4—8)	230 (-30—730)
Minor restricted-activity days (ages 18-65)		300,000 (120,000—480,000)	3,700 (1,300—6,100)	300,000 (130,000—480,000)
School absence days		110,000 (38,000—160,000)	1,300 (380—2,100)	110,000 (38,000—160,000)

[A] Estimates rounded to two significant figures; column values will not sum to total value.

[B] The negative estimates for certain endpoints are the result of the weak statistical power of the study used to calculate these health impacts and do not suggest that increases in air pollution exposure result in decreased health impacts.

Table 5-18: Estimated Economic Value of Health and Welfare Benefits (95% confidence intervals, billions of 2006$)[A]

Health Effect	*Pollutant*	*Within transport region*	*Beyond transport region*[B]	*Total*
Premature Mortality (Pope et al. 2002 PM mortality and Bell et al. 2004 ozone mortality estimates)				
3% discount rate	$PM_{2.5}$ & O_3	$110 ($8.8—$330)	$0.1 ($0.08—$3)	$110 ($8.8—$340)
7% discount rate	$PM_{2.5}$ & O_3	$100 ($7.9—$300)	$0.09 ($0.07—$2.7)	$100 ($7.9—$300)
Premature Mortality (Laden et al. 2006 PM mortality and Levy et al. 2005 ozone mortality estimates)				
3% discount rate	$PM_{2.5}$ & O_3	$280 ($25—$810)	$2.5 ($0.2—$7.3)	$280 ($25—$820)
7% discount rate	$PM_{2.5}$ & O_3	$250 ($22—$300)	$2.3 ($0.2—$6.6)	$260 ($22—$310)
Chronic Bronchitis	$PM_{2.5}$	$4.3 ($0.2—$20)	$0.04 ($0.002--$0.2)	$4.3 ($0.2—$20)
Non-fatal heart attacks				
3% discount rate	$PM_{2.5}$	$2.5 ($0.4—$6)	$0.03 ($0.005—$0.07)	$2.5 ($0.4—$6)
7% discount rate	$PM_{2.5}$	$2.4 ($0.4—$5.9)	$0.03 ($0.005—$0.07)	$2.4 ($0.4—$5.9)
Hospital admissions—respiratory	$PM_{2.5}$ & O_3	$0.06 ($0.03—$0.1)	$0.00006 ($0.00003—$0.001)	$0.06 ($0.03—$0.1)
Hospital admissions—cardiovascular	$PM_{2.5}$	$0.2 ($0.1—$0.3)	$0.002 ($0.001—$0.003)	$0.2 ($0.1—$0.3)
Emergency room visits for asthma	$PM_{2.5}$ & O_3	$0.005 ($0.002—$0.008)	---	$0.005 ($0.002—$0.008)
Acute bronchitis	$PM_{2.5}$	$0.009 (-$0.0004—$0.03)[c]	---	$0.009 (-$0.0004—$0.03)
Lower respiratory symptoms	$PM_{2.5}$	$0.005 ($0.002—$0.009)	---	$0.005 ($0.002—$0.009)
Upper respiratory symptoms	$PM_{2.5}$	$0.006 ($0.001—$0.014)	---	$0.006 ($0.001—$0.014)
Asthma exacerbation	$PM_{2.5}$	$0.012 ($0.001--$0.046)	---	$0.012 ($0.001--$0.046)
Lost work days	$PM_{2.5}$	$0.2 ($0.19—$0.24)	$0.002 ($0.0002--$0.002)	$0.2 ($0.19—$0.24)
School loss days	O_3	$0.01 ($0.004—$0.013)	---	$0.01 ($0.004—$0.013)
Minor restricted-activity days	$PM_{2.5}$ & O_3	$0.64 ($0.34—$0.97)	$0.005 ($0.003—$0.008)	$0.64 ($0.34—$0.97)
Recreational visibility, Class I areas	$PM_{2.5}$	$3.5	$0.03	$3.6
Social cost of carbon (3% discount rate, 2014 value)	CO_2			$0.35

Monetized total Benefits **(Pope et al. 2002 $PM_{2.5}$ mortality and Bell et al. 2004 ozone mortality estimates)**				
3% discount rate	$PM_{2.5}$, O_3	**$120** **($10—$360)**	**$1.1** **($0.09—$3.3)**	**$120** **($10—$360)**
7% discount rate	$PM_{2.5}$, O_3	**$110** **($9—$330)**	**$0.9** **($0.08—$2.9)**	**$110** **($9—$330)**
Monetized total Benefits **(Laden et al. 2006 $PM_{2.5}$ mortality and Levy et al. 2005 ozone mortality estimates)**				
3% discount rate	$PM_{2.5}$, O_3	**$290** **($26—$840)**	**$2.6** **($0.2—$7.5)**	**$290** **($26—$840)**
7% discount rate	$PM_{2.5}$, O_3	**$260** **($23—$760)**	**$2.4** **($0.2—$6.8)**	**$270** **($24—$760)**

[A] Estimates rounded to two significant figures.

[B] Monetary value of endpoints marked with dashes are < $100,000. States included in transport region may be found in chapter 2.

[C] The negative estimates for certain endpoints are the result of the weak statistical power of the study used to calculate these health impacts and do not suggest that increases in air pollution exposure result in decreased health impacts.

Not all known PM- and ozone-related health and welfare effects could be quantified or monetized. The monetized value of these unquantified effects is represented by adding an unknown “B” to the aggregate total. The estimate of total monetized health benefits is thus equal to the subset of monetized PM- and ozone-related health and welfare benefits plus B, the sum of the nonmonetized health and welfare benefits; this B represents both uncertainty and a bias in this analysis, as it reflects those benefits categories that we are unable quantify in this analysis.

Total monetized benefits are dominated by benefits of mortality risk reductions. The primary analysis projects that the proposed remedy will result in between 14,000 and 36,000 $PM_{2.5}$ and ozone-related avoided premature deaths annually in 2014. Our estimate of total monetized benefits in 2014 proposed remedy is between $120 billion and $290 billion using a 3 percent discount rate and between $110 billion and $270 using a 7 percent discount rate. Health benefits account for between 97 and 99 percent of total benefits depending on the $PM_{2.5}$ and ozone mortality estimates used, in part because we are unable to quantify most of the non-health benefits. The monetized benefit associated with reductions in the risk of premature mortality, which accounts for between $110 and $280 billion in 2014, depending again on the PM and ozone mortality risk estimates used, is between 90 and 96 percent of total monetized health benefits. The next largest benefit is for reductions in chronic illness (CB and nonfatal heart attacks), although this value is more than an order of magnitude lower than for premature mortality. Hospital admissions for respiratory and cardiovascular causes, visibility, MRADs, work loss days, school absence days, and worker productivity account for the majority of the remaining benefits. The remaining categories each account for a small percentage of total benefit; however, they represent a large number of avoided incidences affecting many individuals. A comparison of the incidence table to the monetary benefits table reveals that there is not always a close correspondence between the number of incidences avoided for a given endpoint and the monetary value associated with that endpoint. For example, there are almost 100 times more work loss days than premature mortalities, yet work loss days account for only a very small fraction of total monetized benefits. This reflects the fact that many of the less severe health effects, while more common, are valued at a lower level than the more severe health effects. Also, some effects, such as hospital admissions, are valued using a proxy measure of WTP. As such, the true value of these effects may be higher than that reported in Table 5-16.

Figures 5-15 and 5-16 illustrates the geographic distribution of avoided $PM_{2.5}$ and ozone-related mortalities estimated to result from the proposed remedy. Figure 5-17 plots the cumulative distribution of reductions in all-cause premature mortality attributable to reductions in $PM_{2.5}$ and ozone resulting from the proposed remedy. Among the 10 counties containing the most populous cities in the U.S., the three experiencing the largest reduction in the percentage of $PM_{2.5}$ and ozone-related premature mortality are located within the Transport Rule region: New York, Chicago and Philadelphia. While not quantified in this RIA, we expect the Transport Rule to produce important public health benefits for populations living in Canada. Approximately 90% of the Canadian population lives within 100 miles of the U.S. border, suggesting that some of the air quality improvements projected in areas near the U.S.-Canada border would be enjoyed by Canadian populations as well. A recent analysis (Chestnut and Mills, 2005) of the U.S. Acid Rain Program estimates annual benefits of the program in 2010 to both Canada and the United States at $122 billion and costs for that year at $3 billion (2000$)—a 40-to-1 benefit/cost ratio. These quantified benefits in the United States and Canada are the result of improved air quality prolonging lives, reducing heart attacks and other cardiovascular and respiratory problems, and improving visibility. The complete report is available in volume 77, issue 3, of the *Journal of Environmental Management*.

These figures show that while there are very large health benefits throughout most of the East, there could be several areas where a very small disbenefit could result if further governmental actions do not occur to address them in the future. There are several upcoming planned federal actions that could lead to further large reductions throughout the US of ambient levels of fine particles and ozone. Additionally, state actions to address regional haze in the near future and the existing NAAQS for fine particles and ozone could address these situations. There are also other state actions, such as the recent Colorado Clean Air – Clean Jobs Act of April 2010 that is likely to convert much of the front range coal-fired generation in Colorado to natural gas in the near future.

Figure 5-15: Estimated reduction in excess $PM_{2.5}$-related premature mortalities estimated to occur in each county in 2014 as a result of the proposed remedy.

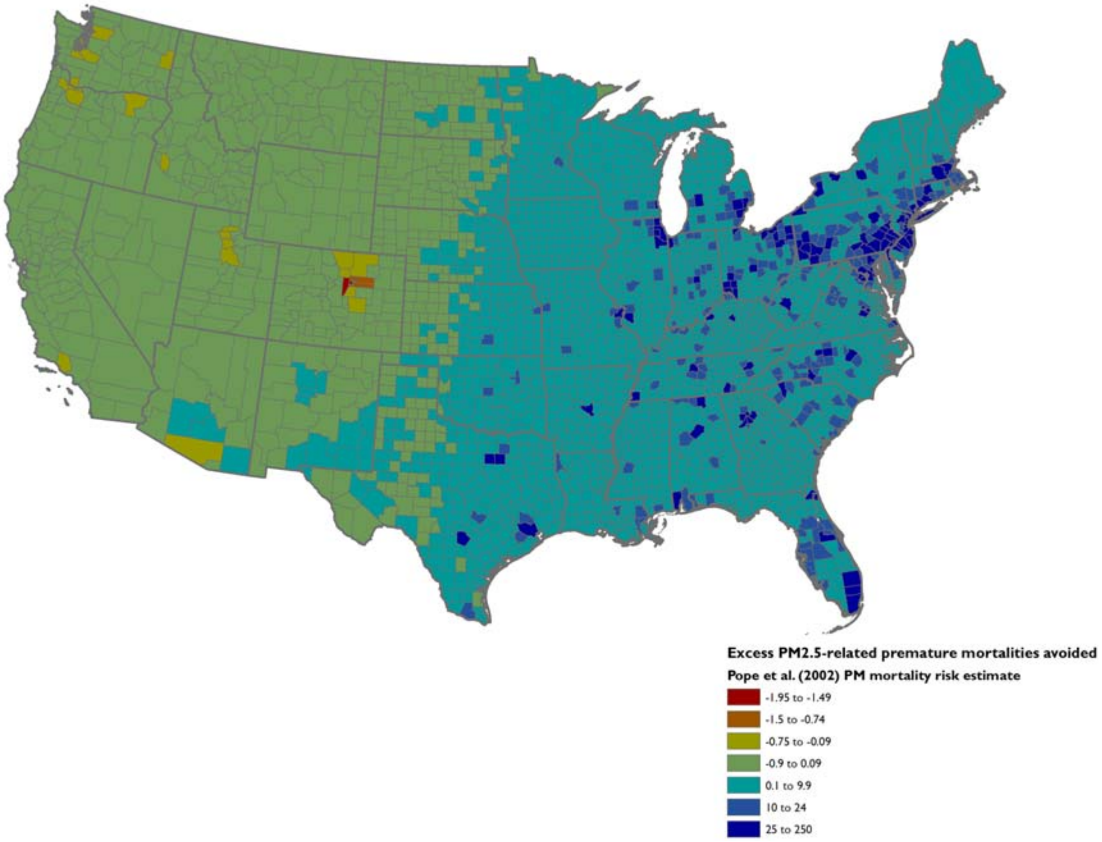


Figure 5-16: Estimated reduction in excess ozone-related premature mortalities estimated to occur in each county in 2014 as a result of the proposed remedy

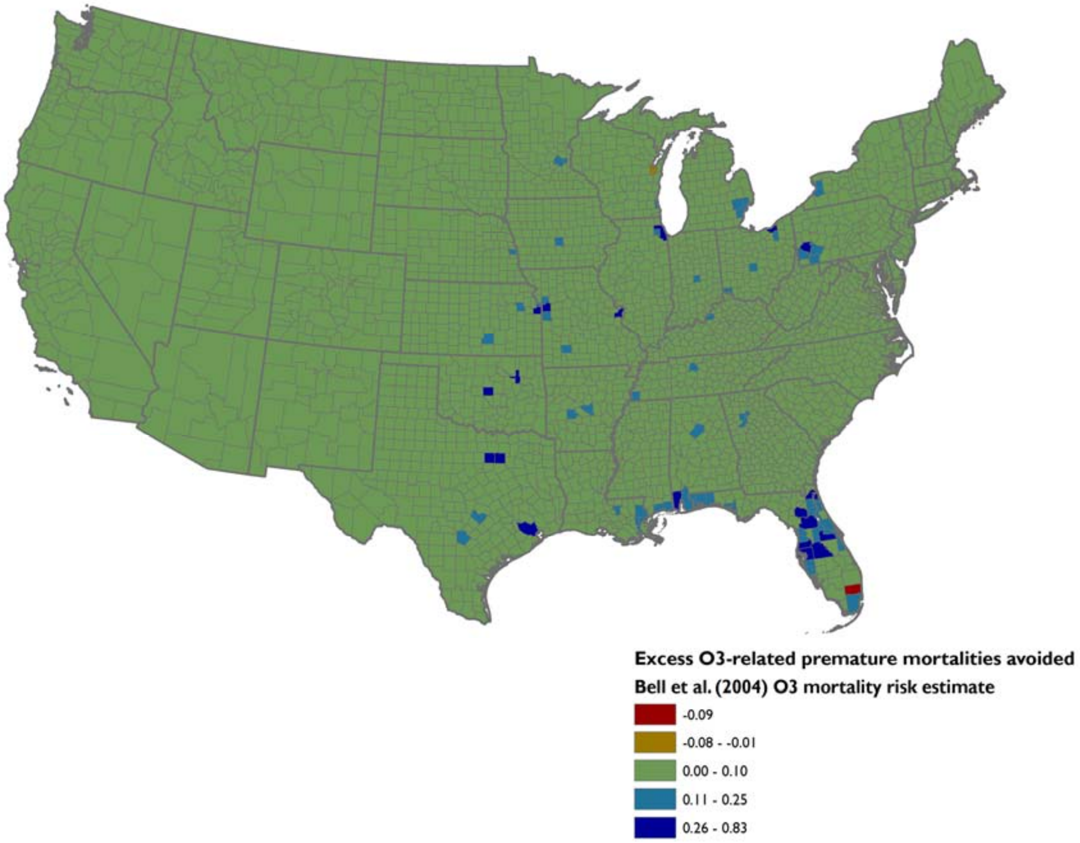
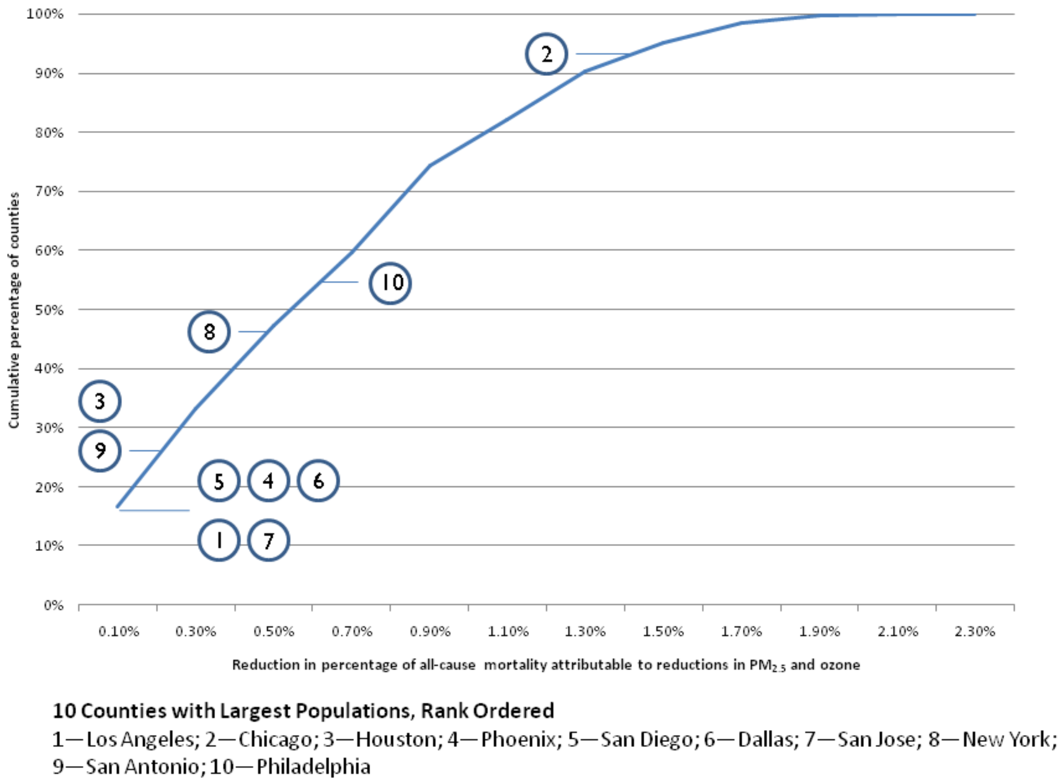


Figure 5-17: Cumulative percentage of the reduction in all-cause mortality attributable to reductions in $PM_{2.5}$ and Ozone resulting from the proposed remedy by county in 2014[A]



[A] Bell et al. 2005 ozone mortality estimate and Pope et al. 2002 $PM_{2.5}$ mortality estimates.

Figure 5-18 summarizes an array of $PM_{2.5}$-related monetized benefits estimates based on alternative epidemiology and expert-derived PM-mortality estimate as well as the sum of ozone-related benefits using the Bell et al. (2004) mortality estimate.

Based on our review of the current body of scientific literature, EPA estimated PM-related mortality without applying an assumed concentration threshold. EPA's Integrated Science Assessment for Particulate Matter (U.S. EPA, 2009b), which was recently 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. Consistent with this finding, we have conformed the threshold sensitivity analysis to the current state of the PM science improved upon our previous approach for estimating the sensitivity of the benefits estimates to the presence of an assumed threshold by incorporating a new "Lowest Measured Level" (LML) assessment.

This approach summarizes the distribution of avoided PM mortality impacts according to the baseline (i.e. pre-Transport Rule) $PM_{2.5}$ levels experienced by the population receiving the $PM_{2.5}$ mortality benefit (Figure 5-19). We identify on this figure the lowest air quality levels measured in each of the two primary epidemiological studies EPA uses to quantify PM-related mortality. This information allows readers to determine the portion of PM-related mortality benefits occurring above or below the LML of each study; in general, our confidence in the estimated PM mortality decreases as we consider air quality levels further below the LML in the two epidemiological studies. While the LML analysis provides some insight into the level of uncertainty in the estimated PM mortality benefits, EPA does not view the LML as a threshold and continues to quantify PM-related mortality impacts using a full range of modeled air quality concentrations.

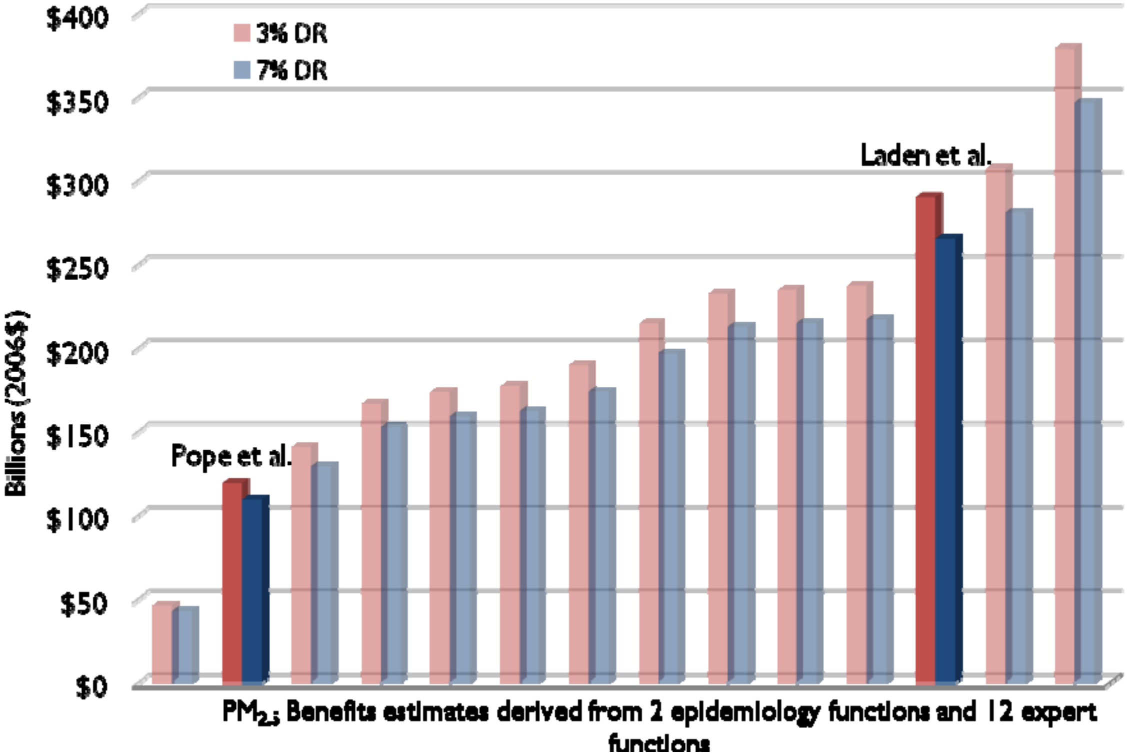
The very large proportion of the avoided PM-related impacts we estimate in this analysis occur among populations exposed at or above the LML of each study (Figures 5-20 and 5-21), increasing our confidence in the PM mortality analysis. Approximately 80% of the avoided impacts occur at or above an annual mean $PM_{2.5}$ level of 10 $\mu g/m^3$ (the LML of the Laden et al. 2006 study); about 97% occur at or above an annual mean $PM_{2.5}$ level of 7.5 $\mu g/m^3$ (the LML of the Pope et al. 2002 study). As we model mortality impacts among populations exposed to levels of $PM_{2.5}$ that are successively lower than the LML of each study our confidence in the results diminishes. However, the analysis above confirms that the great majority of the impacts occur at or above each study's LML.

As an example, when considering mortality impacts among populations living in areas with an annual mean PM level of 8 ug/m3, we would place greater confidence in estimates drawn from the Pope et al. 2002 study, as this air quality level is above the LML of this study. Conversely, we would place equal confidence when estimating mortality impacts among populations living in locations where the annual mean PM levels are above 10 ug/m3 because this value is at or above the LML of each study.

Finally, Figure 5-22 illustrates the percentage of population exposed to different levels of annual mean PM2.5 levels in the baseline and after the implementation of the Transport Rule in 2014. The Transport Rule reduces overall PM2.5 levels substantially, particularly among highly exposed populations located within the states covered by the rule. Locations of the U.S. where annual mean PM levels are below the lowest measured level of the Pope study--western states in particular--are generally unaffected by the rule. However, for populations in the far western portion of the Transport Rule region, where annual mean PM2.5 concentrations are below 7.5 ug/m3, there are benefits of the rule, although the relative magnitude of those benefits compared to benefits in the majority of the areas covered by the Transport Rule is small. In these areas there is lower confidence in the magnitude of the benefits associated with reductions in long-term PM2.5. In addition, we note that prior to the implementation of the Transport Rule, 89% of the population live in areas where PM2.5 levels are projected to be above the lowest measured levels of the Pope study. Taken together, this information increases our confidence in the estimated mortality reductions for this rule.

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

Figure 5-18: Estimated $PM_{2.5}$- related premature mortalities avoided according to epidemiology or expert-derived PM mortality risk estimate[A]



[A] Column total equals sum of $PM_{2.5}$-related mortality and morbidity benefits and ozone-related morbidity and mortality benefits using the Bell et al. (2004) mortality estimate.

Figure 5-19: Distribution of $PM_{2.5}$-related mortality impacts by baseline $PM_{2.5}$ levels, $PM_{2.5}$ epidemiology study and lowest measured level (LML) of each study

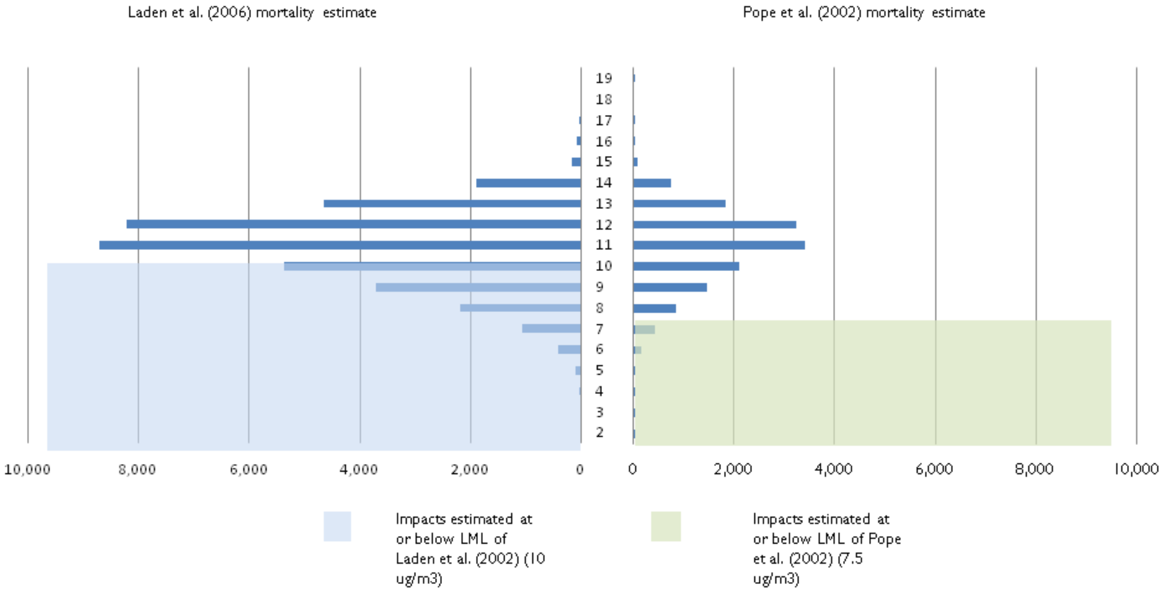
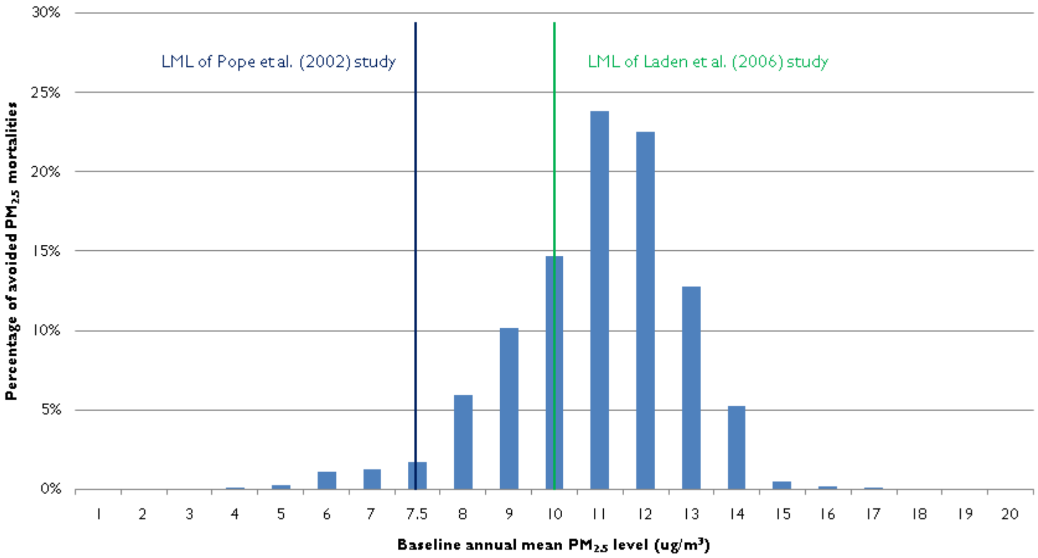


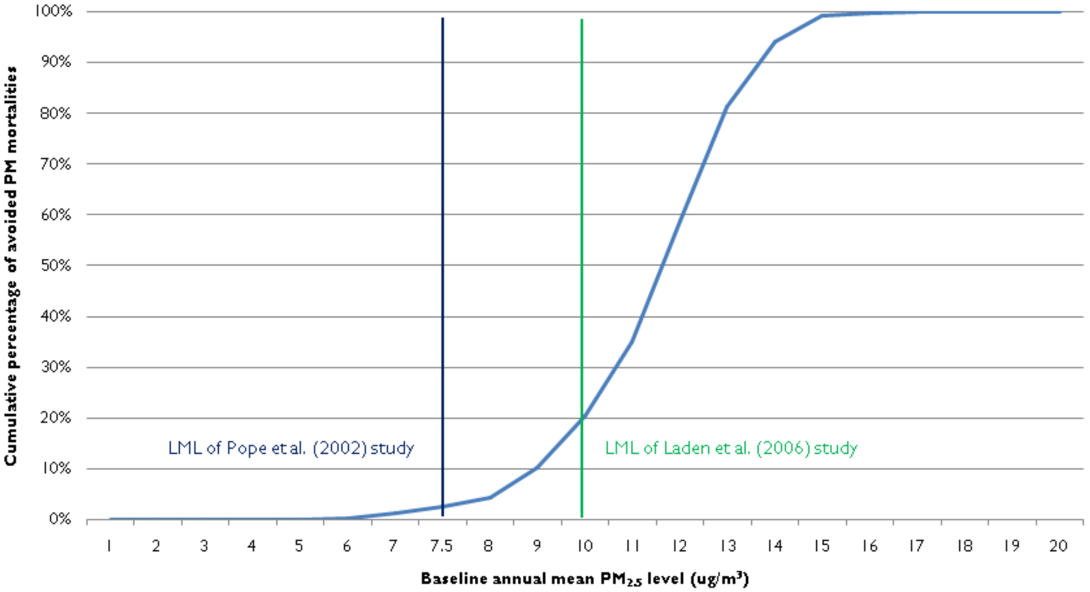
Figure 5-20: Percentage of total PM-related mortalities avoided by baseline air quality level



Of the total mortalities avoided:

97% occur among populations exposed to PM levels at or above the LML of the **Pope et al.** study.
80% occur among populations exposed to PM levels at or above the LML of the **Laden et al.** study.

Figure 5-21: Cumulative percentage of total PM-related mortalities avoided by baseline air quality level

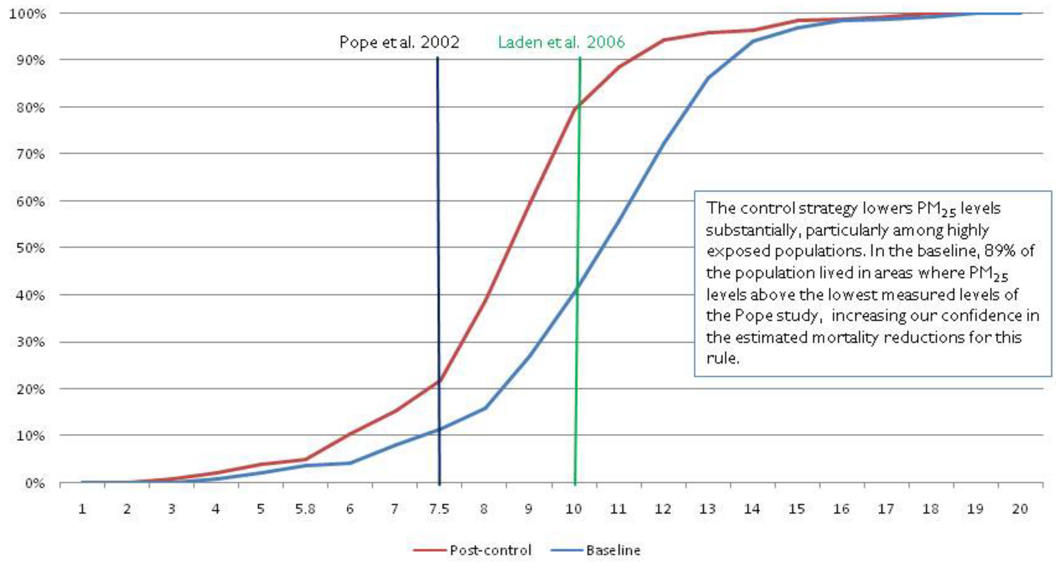


Of the total mortalities avoided:

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

80% occur among populations exposed to PM levels at or above the LML of the **Laden et al.** study.

Figure 5-22: Cumulative percentage of adult population at annual mean $PM_{2.5}$ levels (pre- and post-2014 Transport Rule)



The control strategy lowers PM_{25} levels substantially, particularly among highly exposed populations. In the baseline, 89% of the population lived in areas where PM_{25} levels above the lowest measured levels of the Pope study, increasing our confidence in the estimated mortality reductions for this rule.

5.8 Discussion

This analysis demonstrates the significant health and welfare benefits of the Transport Rule. We estimate that by 2014 the rule will have reduced the number of $PM_{2.5}$ and ozone-related premature mortalities by between 14,000 and 36,000, produce substantial non-mortality benefits and significantly improve visibility in Class 1 areas. This rule promises to yield significant welfare impacts as well, though the quantification of those endpoints in this RIA is incomplete. These significant health and welfare benefits suggest the important role that pollution from the EGU sector plays in the public health impacts of air pollution.

Inherent in any complex RIA such as this one are multiple sources of uncertainty. Some of these we characterized through our quantification of statistical error in the concentration response relationships and our use of the expert elicitation-derived PM mortality functions. Others, including the projection of atmospheric conditions and source-level emissions, the projection of baseline morbidity rates, incomes and technological development are unquantified. When evaluated within the context of these uncertainties, the health impact and monetized benefits estimates in this RIA can provide useful information regarding the public health impacts attributable to EGUs.

5.9 References

Abbey, D.E., N. Nishino, W.F. McDonnell, R.J. Burchette, S.F. Knutsen, W. Lawrence Beeson, and J.X. Yang. 1999. Long-term inhalable particles and other air pollutants related to mortality in nonsmokers [see comments]. American Journal of Respiratory and Critical Care Medicine 159(2):373-382.

Abbey, D.E., B.L. Hwang, R.J. Burchette, T. Vancuren, and P.K. Mills. 1995. Estimated Long-Term Ambient Concentrations of PM(10) and Development of Respiratory Symptoms in a Nonsmoking Population. Archives of Environmental Health 50(2): 139-152.

Abt Associates, Inc. 2005. U.S. EPA. Urban ornamental plants: sensitivity to ozone and potential economic losses. Memorandum to Bryan Hubbell and Zachary Pekar.

Abt Associates, Inc. April 2003. Proposed Nonroad Land-based Diesel Engine Rule: Air Quality Estimation, Selected Health and Welfare Benefits Methods, and Benefit Analysis Results. Prepared for Office of Air Quality Planning and Standards, U.S. EPA.

Abt Associates, Inc. 2008. Environmental Benefits and Mapping Program (Version 3.0). Bethesda, MD. Prepared for U.S. Environmental Protection Agency Office of Air Quality Planning and Standards. Research Triangle Park, NC. Available on the Internet at <http://www.epa.gov/air/benmap>.

Adams PF, Hendershot GE, Marano MA. 1999. Current Estimates from the National Health Interview Survey, 1996. Vital Health Stat 10(200):1-212.

Adams, R. M., Glyer, J. D., Johnson, S. L., McCarl, B. A. 1989. A reassessment of the economic effects of ozone on U.S. agriculture. Journal of the Air Pollution Control Association, 39, 960-968.

Adams, R. M., Hamilton, S. A., McCarl, B. A. 1986. The benefits of pollution control: the case of ozone and U.S. agriculture. American Journal of Agricultural Economics, 34, 3-19.

Agency for Healthcare Research and Quality (AHRQ). 2000. HCUPnet, Healthcare Cost and Utilization Project.

American Lung Association. 1999. Chronic Bronchitis. Available on the Internet at <http://www.lungusa.org/diseases/lungchronic.html>.

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>.

Banzhaf, S., D. Burtraw, D. Evans, and A. Krupnick. 2006. "Valuation of Natural Resource Improvements in the Adirondacks." Land Economics 82:445-464.

Bell, M.L., et al. 2004. Ozone and short-term mortality in 95 US urban communities, 1987-2000. Journal of the American Medical Association. 292(19): p. 2372-8.

Bell, M.L., F. Dominici, and J.M. Samet. 2005. A meta-analysis of time-series studies of ozone and mortality with comparison to the national morbidity, mortality, and air pollution study. Epidemiology. 16(4): p. 436-45.

Berger, M.C., G.C. Blomquist, D. Kenkel, and G.S. Tolley. 1987. Valuing Changes in Health Risks: A Comparison of Alternative Measures. The Southern Economic Journal 53:977-984.

Brookshire, D.S., Thayer, M.A., Schulze, W.D. & D'Arge, R.C. 1982. "Valuing Public Goods: A Comparison of Survey and Hedonic Approaches." The American Economic Review. 72(1): 165-177.

Brown, L.H. 2002. Profile of the Annual Fall Foliage Tourist in Vermont: Travel Year 2001. Report prepared for the Vermont Department of Tourism and Marketing and the Vermont Tourism Data Center in association with the University of Vermont, Burlington, VT.

Burnett RT, Smith-Doiron M, Stieb D, Raizenne ME, Brook JR, Dales RE, et al. 2001. Association between ozone and hospitalization for acute respiratory diseases in children less than 2 years of age. Am J Epidemiol 153(5):444-452.

Carnethon, M.R., D. Liao, G.W. Evans, W.E. Cascio, L.E. Chambless, W.D. Rosamond, and G. Heiss. 2002. Does the Cardiac Autonomic Response to Postural Change Predict Incident Coronary Heart Disease and Mortality? The Atherosclerosis Risk in Communities Study. American Journal of Epidemiology 155(1):48-56.

Centers for Disease Control: Wide-ranging OnLine Data for Epidemiologic Research (CDC Wonder) (data from years 1996-1998), Centers for Disease Control and Prevention (CDC), U.S. Department of Health and Human Services, Available on the Internet at <http://wonder.cdc.gov>.

Chappelka, A.H., Samuelson, L.J. 1998. Ambient ozone effects on forest trees of the eastern United States: a review. New Phytologist, 139, 91-108.

Chen L, Jennison BL, Yang W, Omaye ST. 2000. Elementary school absenteeism and air pollution. Inhal Toxicol 12(11):997-1016.

Chesapeake Bay Program. 2008. Underwater Bay Grasses. Available on the Internet at < http://www.chesapeakebay.net/baygrasses.aspx>. Last accessed March 24, 2010.

Chestnut, L.G and Mills, D.M. (2005) A fresh look at the benefits and costs of the US Acid Rain Program. Journal of Environmental Management. Vol 77, No 3, pp 252—266.

Chestnut, L.G., and R.D. Rowe. 1990a. 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.

Chestnut, L.G., and R.D. Rowe. 1990b. Preservation Values for Visibility Protection at the National Parks: Draft Final Report. Prepared for Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency, Research Triangle Park, NC, and Air Quality Management Division, National Park Service, Denver, CO.

Chestnut, L.G. and R.D. Rowe. 1990c. Economic Valuation of Changes in Visibility: A State of the Science Assessment for NAPAP. Section B5 in NAPAP State of Science and Technology Report 27.

Chestnut, L.G. April 15, 1997. Draft Memorandum: Methodology for Estimating Values for Changes in Visibility at National Parks.

Cordell, H.K., C.J. Betz, G. Green, and M. Owens. 2005. Off-Highway Vehicle Recreation in the United States, Regions and States: A National Report from the National Survey on Recreation and the Environment (NSRE). Prepared for the U.S. Department of Agriculture Forest Service, Southern Research Station, National OHV Policy and Implementation Teams, Athens, GA. Available on the Internet at <http://www.fs.fed.us/recreation/programs/ohv/OHV_final_report.pdf>.

Cordell, K., B. Leeworthy, G.T. Green, C. Betz, B. Stephens. 2008. The National Survey on Recreation & the Environment. Research Work Unit 4953. Pioneering Research on Changing Forest Values in the South and Nation, U.S. Department of Agriculture Forest Service, Southern Research Station, Athens, GA. Available on the Internet at <http://www.srs.fs.fed.us/trends>.

Coulston, J.W., Riitters, K.H., Smith, G.C. 2004. A preliminary assessment of the Montreal process indicators of air pollution for the United States. Environmental Monitoring and Assessment, 95, 57-74.

Crocker, T.D. and R.L. Horst, Jr. Hours of Work, Labor Productivity, and Environmental Conditions: A Case Study. The Review of Economics and Statistics, 1981. 63: p. 361-368.

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.

Davidson K, Hallberg A, McCubbin D, Hubbell BJ. 2007. Analysis of $PM_{2.5}$ Using the Environmental Benefits Mapping and Analysis Program (BenMAP). J Toxicol Environ Health 70: 332—346.

Dekker, J.M., R.S. Crow, A.R. Folsom, P.J. Hannan, D. Liao, C.A. Swenne, and E.G. Schouten. 2000. Low Heart Rate Variability in a 2-Minute Rhythm Strip Predicts Risk of Coronary Heart Disease and Mortality From Several Causes: The ARIC Study. Circulation 2000 102:1239-1244.

De Steiguer, J., Pye, J., Love, C. 1990. Air Pollution Damage to U.S. Forests. Journal of Forestry, 88(8), 17-22.

DeHayes, D.H., P.G. Schaberg, G.J. Hawley, and G.R. Strimbeck. 1999. Acid rain impacts on calcium nutrition and forest health. Bioscience 49(10):789–800.

Dockery, D.W., J. Cunningham, A.I. Damokosh, L.M. Neas, J.D. Spengler, P. Koutrakis, J.H. Ware, M. Raizenne, and F.E. Speizer. 1996. Health Effects of Acid Aerosols On North American Children-Respiratory Symptoms. Environmental Health Perspectives 104(5):500-505.

Dockery, D.W., C.A. Pope, X.P. Xu, J.D. Spengler, J.H. Ware, M.E. Fay, B.G. Ferris, and F.E. Speizer. 1993. An Association between Air Pollution and Mortality in Six U.S. Cities. New England Journal of Medicine 329(24):1753-1759.

Drevnick, P.E., D.E. Canfield, P.R. Gorski, A.L.C. Shinneman, D.R. Engstrom, D.C.G. Muir, G.R. Smith, P.J. Garrison, L.B. Cleckner, J.P. Hurley, R.B. Noble, R.R. Otter, and J.T.Oris. 2007. Deposition and cycling of sulfur controls mercury accumulation in Isle Royale fish. Environmental Science and Technology 41(21):7266–7272.

Eisenstein, E.L., L.K. Shaw, K.J. Anstrom, C.L. Nelson, Z. Hakim, V. Hasselblad and D.B.

Mark. 2001. Assessing the Clinical and Economic Burden of Coronary Artery Disease: 1986-1998. Medical Care 39(8):824-35.

Estimating the Public Health Benefits of Proposed Air Pollution Regulations. Washington, DC: The National Academies Press.

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 2:169–176.

Felzer, B., J. Reilly, J. Melillo, D Kicklighter, M. Sarogim, C Wang, R. Prinn, Q. Zhuang. 2005. Future effects of Ozone on Carbon Sequestration and Climate Change Policy using a Global Biogeochemical Model. Climatic Change (2005) 73: 345–373

Fox, S., Mickler, R. A. (Eds.). 1996. Impact of Air Pollutants on Southern Pine Forests, Ecological Studies. (Vol. 118, 513 pp.) New York: Springer-Verlag.

Freeman(III), AM. 1993. The Measurement of Environmental and Resource Values: Theory and Methods. Washington, DC: Resources for the Future.

Gilliland FD, Berhane K, Rappaport EB, Thomas DC, Avol E, Gauderman WJ, et al. 2001. The effects of ambient air pollution on school absenteeism due to respiratory illnesses. Epidemiology 12(1):43-54.

Gold, D.R., A. Litonjua, J. Schwartz, E. Lovett, A. Larson, B. Nearing, G. Allen, M. Verrier, R. Cherry., and R. Verrier. 2000. Ambient Pollution and Heart Rate Variability. Circulation 101(11):1267-73.

Grulke, N.E. 2003. The physiological basis of ozone injury assessment attributes in Sierran conifers. In A. Bytnerowicz, M.J. Arbaugh, & R. Alonso (Eds.), Ozone air pollution in the Sierra Nevada: Distribution and effects on forests. (pp. 55-81). New York, NY: Elsevier Science, Ltd.

Guallar, et. al., 2002. "Mercury, Fish Oils, and the Risk of Myocardial Infarction." New England Journal of Medicine, Vol. 374, No. 22, November.

Hall JV, Brajer V, Lurmann FW. 2003. Economic Valuation of Ozone-related School Absences in the South Coast Air Basin of California. Contemporary Economic Policy 21(4):407-417.

Hallgren et al. 2001. "Markers of high fish intake are associated with decreased risk of a first myocardial infarction," British Journal of Nutrition, 86, 397-404.

Harrington, W., and P.R. Portney. 1987. Valuing the Benefits of Health and Safety Regulation. Journal of Urban Economics 22:101-112.

Heck, W.W. &Cowling E.B. 1997. The need for a long term cumulative secondary ozone standard – an ecological perspective. Environmental Management, January, 23-33.

Hollman, F.W., T.J. Mulder, and J.E. Kallan. January 2000. Methodology and Assumptions for the Population Projections of the United States: 1999 to 2100. Population Division Working Paper No. 38, Population Projections Branch, Population Division, U.S. Census Bureau, Department of Commerce.

Hrabik, T.R., and C.J. Watras. 2002. Recent declines in mercury concentration in a freshwater fishery: isolating the effects of de-acidification and decreased atmospheric mercury deposition in Little Rock Lake. Science of the Total Environment 297:229–237.

Huang Y. Dominici F. Bell M. 2005. Bayesian Hierarchical Distributed Lag Models for Summer Ozone Exposure and Cardio-Respiratory Mortality Environmetrics, 16, 547-562.

Hubbell BJ, Fann N, Levy JI. 2009. Methodological Considerations in Developing Local-Scale Health Impact Assessments: Balancing National, Regional and Local Data. Air Qual Atmos Health doi: 10.1007/s11869-009-0037-z [online 31 March 2009].

Hubbell BJ, Hallberg A, McCubbin D, Post, E. 2005. Health-Related Benefits of Attaining the 8-Hr Ozone Standard. Environ Health Perspect 113: 73—82.

Hutchison, R., and C.E. Kraft. 1994. Hmong Fishing Activity and Fish Consumption. Journal of Great Lakes Research 20(2):471–487.

Hutchinson, J., D. Maynard, and L. Geiser. 1996. Air Quality and Lichens —A Literature Review Emphasizing the Pacific Northwest, USA. USDA Forest Service, Pacific Northwest Region Air Resource Management Program, U.S. Forest Service, U.S. Department of Agriculture.

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>.

Industrial Economics, Incorporated (IEc). 1994. Memorandum to Jim DeMocker, Office of Air and Radiation, Office of Policy Analysis and Review, U.S. Environmental Protection Agency. March 31.

Intergovernmental Panel on Climate change (IPCC). 2007. Climate Change 2007 - Synthesis Report Contribution of Working Groups I, II and III to the Fourth Assessment Report of the IPCC.

Ito, K. 2003. Associations of Particulate Matter Components with Daily Mortality and Morbidity in Detroit, Michigan. In Revised Analyses of Time-Series Studies of Air Pollution and Health. Special Report. Health Effects Institute, Boston, MA.

Ito, K., S.F. De Leon, and M. Lippmann. 2005. Associations between ozone and daily mortality: analysis and meta-analysis. Epidemiology. 16(4): p. 446-57.

Jaffe DH, Singer ME, Rimm AA. 2003. Air pollution and emergency department visits for asthma among Ohio Medicaid recipients, 1991-1996. Environ Res 91(1):21-28.

Jenkins, D.H., J. Sullivan, and G.S. Amacher. 2002. Valuing high altitude spruce-fir forest improvements: Importance of forest condition and recreation activity. Journal of Forest Economics 8:77–99.

Jerrett M, Burnett RT, Pope CA, III, et al. 2009. Long-Term Ozone Exposure and Mortality. N Engl J Med 360:1085-95.

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.

Kleckner, N., and J. Neumann. June 3, 1999. Recommended Approach to Adjusting WTP Estimates to Reflect Changes in Real Income. Memorandum to Jim Democker, U.S. EPA/OPAR.

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.

Kopp, R. J., Vaughn, W. J., Hazilla, M., Carson, R. 1985. Implications of environmental policy for U.S. agriculture: the case of ambient ozone standards. Journal of Environmental Management, 20, 321-331.

Kramer, A., T. Holmes, and M. Haefel. 2003. Contingent valuation of forest ecosystem protection. Pp. 303–320 in Forests in a Market Economy. Edited by E.O. Sills and K.L. Abt. Dordrecht, The Netherlands: Kluwer Academic Publishers.

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.

Krewski D, Jerrett M, Burnett RT, Ma R, Hughes E, Shi, Y, et al. 2009. Extended follow-up and spatial analysis of the American Cancer Society study linking particulate air pollution and mortality. HEI Research Report, 140, Health Effects Institute, Boston, MA.

Krupnick, A.J., and M.L. Cropper. 1992. The Effect of Information on Health Risk Valuations. Journal of Risk and Uncertainty 5(2):29-48.

Kunzli, N., S. Medina, R. Kaiser, P. Quenel, F. Horak Jr, and M. Studnicka. 2001. Assessment of Deaths Attributable to Air Pollution: Should We Use Risk Estimates Based on Time Series or on Cohort Studies? American Journal of Epidemiology 153(11):1050-55.

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.

Lave, L.B., and E.P. Seskin. 1977. Air Pollution and Human Health. Baltimore: Johns Hopkins University Press for Resources for the Future.

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>.

Levy JI, Baxter LK, Schwartz J. 2009. Uncertainty and variability in health-related damages from coal-fired power plants in the United States. Risk Anal. doi: 10.1111/j.1539-6924.2009.01227.x [Online 9 Apr 2009]

Levy, J.I., S.M. Chemerynski, and J.A. Sarnat. 2005. Ozone exposure and mortality: an empiric bayes metaregression analysis. Epidemiology. 16(4): p. 458-68.

Leistritz, F. L., D. A. Bangsund, and N. M. Hodur. 2004. Assessing the Economic Impact of Invasive Weeds: The Case of Leafy Spurge (*Euphorbia esula*). Weed Technology 18:1392-1395.

Leitch, J. A, F. L Leistritz, and D. A Bangsund. 1996. Economic effect of leafy spurge in the upper Great Plains: methods, models, and results. Impact Assessment 14: 419–434.

Liao, D., J. Creason, C. Shy, R. Williams, R. Watts, and R. Zweidinger. 1999. Daily Variation of Particulate Air Pollution and Poor Cardiac Autonomic Control in the Elderly. Environ Health Perspect 107:521-5.

Liao, D., J. Cai, W.D. Rosamond, R.W. Barnes, R.G. Hutchinson, E.A. Whitsel, P. Rautaharju, and G. Heiss. 1997. Cardiac Autonomic Function and Incident Coronary Heart Disease: A Population-Based Case-Cohort Study. The ARIC Study. Atherosclerosis Risk in Communities Study. American Journal of Epidemiology 145(8):696-706.

Lipfert, F.W., S.C. Morris, and R.E. Wyzga. 1989. Acid Aerosols—the Next Criteria Air Pollutant. Environmental Science & Technology 23(11):1316-1322.

Lipfert, F.W., H. Mitchell Perry Jr., J. Philip Miller, Jack D. Baty, Ronald E. Wyzg, and Sharon E. Carmody. 2000. The Washington University-EPRI Veterans' Cohort Mortality Study: Preliminary Results. Inhalation Toxicology 12:41-74.

Lipfert, F.W.; Perry, H.M., Jr.; Miller, J.P.; Baty, J.D.; Wyzga, R.E.; Carmody, S.E. 2003. Air Pollution, Blood Pressure, and Their Long-Term Associations with Mortality. Inhalation Toxicology. 15, 493-512.

Lipfert, F.W.; Wyzga, R.E.; Baty, J.D.; Miller, J.P. 2006. Traffic Density as a Surrogate Measure of Environmental Exposures in Studies of Air Pollution Health Effects: Long-Term Mortality in a Cohort of US Veterans. Atmospheric Environment 40: 154-169.

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.

Loehman, E.T., S. Park, and D. Boldt. 1994. "Willingness to Pay for Gains and Losses in Visibility and Health." Land Economics 70(4): 478-498.

Magari, S.R., R. Hauser, J. Schwartz, P.L. Williams, T.J. Smith, and D.C. Christiani. 2001. Association of Heart rate Variability with Occupational and Environmental Exposure to Particulate Air Pollution. Circulation 104(9):986-91.

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

McBride, J.R., Miller, P.R., Laven, R.D. 1985. Effects of oxidant air pollutants on forest succession in the mixed conifer forest type of southern California. In: Air Pollutants Effects on Forest Ecosystems, Symposium Proceedings, St. P, 1985, p. 157-167.

McClelland, G., W. Schulze, D. Waldman, J. Irwin, D. Schenk, T. Stewart, L. Deck, and M. Thayer. September 1993. Valuing Eastern Visibility: A Field Test of the Contingent Valuation Method. Prepared for Office of Policy, Planning and Evaluation, U.S. Environmental Protection Agency. Available on the Internet at < http://yosemite.epa.gov/ee/epa/eerm.nsf/vwAN/EE-0008-1.pdf/$file/EE-0008-1.pdf>.

Millennium Ecosystem Assessment Board (MEA). 2005. Ecosystems and Human Well-being: Synthesis. Washington, DC: World Resources Institute.

Miller, P.R., O.C. Taylor, R.G. Wilhour. 1982. Oxidant air pollution effects on a western coniferous forest ecosystem. Corvallis, OR: U.S. Environmental Protection Agency, Environmental Research Laboratory (EPA600-D-82-276).

Moolgavkar SH, Luebeck EG, Anderson EL. 1997. Air pollution and hospital admissions for respiratory causes in Minneapolis St. Paul and Birmingham. Epidemiology. 8(4):364-370.

Moolgavkar, S.H. 2003. Air Pollution and Daily Deaths and Hospital Admissions in Los Angeles and Cook Counties. In Revised Analyses of Time-Series Studies of Air Pollution and Health. Special Report. Boston, MA: Health Effects Institute.

Moolgavkar, S.H. 2000. Air Pollution and Hospital Admissions for Diseases of the Circulatory System in Three U.S. Metropolitan Areas. Journal of the Air and Waste Management Association 50:1199-1206.

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.

Munthe, J., R.A. Bodaly, B.A. Branfireun, C.T. Driscoll, C.C. Gilmour, R. Harris, M. Horvat, M. Lucotte, and O. Malm. 2007. Recovery of mercury-contaminated fisheries. AMBIO:A Journal of the Human Environment 36:33–44.

Myers, M.D., M.A. Ayers, J.S. Baron, P.R. Beauchemin, K.T. Gallagher, M.B. Goldhaber, D.R. Hutchinson, J.W. LaBaugh, R.G. Sayre, and S.E. Schwarzbach. 2007. USGS goals for the coming decade. Science 318:200–201.

National Agricultural Statistics Service (NASS). 2008. Maple Syrup – June 12, 2008: Maple Syrup Production Up 30 Percent Nationwide. U.S. Department of Agriculture, National Agricultural Statistics Service, New England Agricultural Statistics, Concord, NH.

National Center for Education Statistics (NCHS). 1996. The Condition of Education 1996, Indicator 42: Student Absenteeism and Tardiness. U.S. Department of Education. Washington, DC.

National Center for Health Statistics (NCHS). 1999a. Health Interview Survey (HIS). National Vital Statistics Reports. U.S. Department of Health and Human Services, Centers for Disease Control and Prevention, National Center for Health Statistics. Washington, DC. Volume 47, Number 19. June 30.

National Center for Health Statistics (NCHS). 1999b. National Hospital Discharge Survey (NHDS). Available on the internet at <ftp://ftp.cdc.gov/pub/Health_Statistics/NCHS/Datasets/NHDS>.

National Center for Health Statistics (NCHS). 2000. National Hospital Ambulatory Medical Care Survey (NHAMCS). Available on the internet at <ftp://ftp.cdc.gov/pub/Health_Statistics/NCHS/Datasets/NHAMCS>.

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). 2000. Toxicological Effects of Methylmercury. Committee on the Toxicological Effects of Methylmercury, Board on Environmental Studies and Toxicology. National Academies Press. Washington, DC. p. 229.

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

National Research Council (NRC). 2008. Estimating Mortality Risk Reduction and Economic Benefits from Controlling Ozone Air Pollution. National Academies Press. Washington, DC.

Neumann, J.E., M.T. Dickie, and R.E. Unsworth. March 31, 1994. Linkage Between Health Effects Estimation and Morbidity Valuation in the Section 812 Analysis—Draft Valuation Document. Industrial Economics Incorporated (IEc) Memorandum to Jim DeMocker, U.S. Environmental Protection Agency, Office of Air and Radiation, Office of Policy Analysis and Review.

Norris, G., S.N. YoungPong, J.Q. Koenig, T.V. Larson, L. Sheppard, and J.W. Stout. 1999. An Association between Fine Particles and Asthma Emergency Department Visits for Children in Seattle. Environmental Health Perspectives 107(6):489-493.

Ostro, B., M. Lipsett, J. Mann, H. Braxton-Owens, and M. White. 2001. Air Pollution and Exacerbation of Asthma in African-American Children in Los Angeles. Epidemiology 12(2):200-208.

Ostro, B.D. 1987. Air Pollution and Morbidity Revisited: A Specification Test. Journal of Environmental Economics Management 14:87-98.

Ostro, B.D. and S. Rothschild. 1989. Air Pollution and Acute Respiratory Morbidity: An Observational Study of Multiple Pollutants. Environmental Research 50:238-247.

Ozkaynak, H., and G.D. Thurston. 1987. Associations between 1980 U.S. Mortality Rates and Alternative Measures of Airborne Particle Concentration. Risk Analysis 7(4):449-461.

Peel, J. L., P. E. Tolbert, M. Klein, et al. 2005. Ambient air pollution and respiratory emergency department visits. Epidemiology. Vol. 16 (2): 164-74.

Peters, A., D.W. Dockery, J.E. Muller, and M.A. Mittleman. 2001. Increased Particulate Air Pollution and the Triggering of Myocardial Infarction. Circulation 103:2810-2815.

Peterson, D.E., M.S. Kanarek, M.A. Kuykendall, J.M. Diedrich, H.A. Anderson, P.L. Remington, and T.B. Sheffy. 1994. Fish Consumption Patterns and Blood Mercury Levels in Wisconsin Chippewa Indians. Archives of Environmental Health 49(1):53–58.

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

Poloniecki, J.D., R.W. Atkinson., A.P. de Leon., and H.R. Anderson. 1997. Daily Time Series for Cardiovascular Hospital Admissions and Previous Day's Air Pollution in London, UK. Occupational and Environmental Medicine 54(8):535-540.

Pope, C.A., III, D.W. Dockery, J.D. Spengler, and M.E. Raizenne. 1991. Respiratory Health and PM_{10} Pollution: A Daily Time Series Analysis. American Review of Respiratory Diseases 144:668-674.

Pope, C.A., III, M.J. Thun, M.M. Namboodiri, D.W. Dockery, J.S. Evans, F.E. Speizer, and C.W. Heath, Jr. 1995. Particulate Air Pollution as a Predictor of Mortality in a Prospective Study of U.S. Adults. American Journal of Respiratory Critical Care Medicine 151:669-674.

Pope, CA III, E Majid, D Dockery. 2009. Fine Particle Air Pollution and Life Expectancy in the United States. New England Journal of Medicine 360: 376—386.

Pope, C.A., III, R.T. Burnett, G.D. Thurston, M.J. Thun, E.E. Calle, D. Krewski, and J.J. Godleski. 2004. Cardiovascular Mortality and Long-term Exposure to Particulate Air Pollution. Circulation 109: 71-77.

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.

Prasad, A.M. and Iverson, L.R. 2003. Little's range and FIA importance value database for 135 eastern U.S. tree species. Northeastern Research Station, USDA Forest Service. Available on the Internet at http://www.fs.fed.us/ne/delaware/4153/global/littlefia/index.html

Pye, J.M. 1988. Impact of ozone on the growth and yield of trees: A review. Journal of

Environmental Quality, 17, 347-360.

Rae, D.A. 1983. Benefits of Visual Air Quality in Cincinnati Results of a Contingent Ranking Survey. Prepared for Electric Power Research Institute. May.

Ransom, Michael, and C. Arden Pope. 1992. M.R. Ransom and C.A. Pope, III, Elementary school absences and PM_{10} pollution in Utah Valley. Environ. Res. 58, pp. 204–219.

Rissanen, T., S Voutilainen, T.A. Lakka, and J.T. Salonen. 2000. "Fish oil-derived fatty acids, docasahexaenoic acid and docosapentainoic acid, and the risk of acute coronary events: The Kuopio ischaemic heart disease risk factor study", Circulation. 102(22), pp2677-2679.

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 by Energy and Resource Consultants, Inc. Report to the U.S. Environmental Protection Agency, Office of Policy Analysis. EPA-230-09-86-018. Washington, DC.

Rowlatt, P., Spackman, M., Jones, S., Jones-Lee, M., Loomes, G. 1998. Valuation of deaths from air pollution. A Report for the Department of Environment, Transport and the Regions and the Department of Trade and Industry. National Economic Research Associates (NERA), London.

Russell, M.W., D.M. Huse, S. Drowns, E.C. Hamel, and S.C. Hartz. 1998. Direct Medical Costs of Coronary Artery Disease in the United States. American Journal of Cardiology 81(9):1110-1115.

Samet, J.M., S.L. Zeger, F. Dominici, F. Curriero, I. Coursac, D.W. Dockery, J. Schwartz, and A. Zanobetti. 2000. The National Morbidity, Mortality and Air Pollution Study: Part II: Morbidity, Mortality and Air Pollution in the United States. Research Report No. 94, Part II. Health Effects Institute, Cambridge MA. June.

Salonen, J.T., Seppanen, K. Nyyssonen et al. 1995. "Intake of mercury from fish lipid peroxidation, and the risk of myocardial infarction and coronary, cardiovascular and any death in Eastern Finnish men." Circulation, 91 (3):645-655.

Salonen, J.T., Seppanen,T.A. Lakka, R. Salonen and G.A. Kaplan. 2000. “Mercury accumulation and accelerated progression of carotid atherosclerosis: A population-based prospective 4-year follow-up study in men in Eastern Finland. Atherosclerosis. 148: pp265-273.

Schwartz J. 1995. Short term fluctuations in air pollution and hospital admissions of the elderly for respiratory disease. Thorax. 50(5):531-538.

Schwartz J. 1994b. Air Pollution and Hospital Admissions For the Elderly in Detroit, Michigan. Am J Respir Crit Care Med. 150(3):648-655.

Schwartz J. 1994a. PM(10) Ozone, and Hospital Admissions For the Elderly in Minneapolis St Paul, Minnesota. Arch Environ Health. 49(5):366-374.

Schwartz, J., and L.M. Neas. 2000. Fine Particles are More Strongly Associated than Coarse Particles with Acute Respiratory Health Effects in Schoolchildren. Epidemiology 11:6-10.

Schwartz, J. 2005. How sensitive is the association between ozone and daily deaths to control for temperature? Am J Respir Crit Care Med. Vol. 171 (6): 627-31.

Schwartz, J. 1993. Particulate Air Pollution and Chronic Respiratory Disease. EnvironmentResearch 62:7-13.

Scudder, B.C., Chasar, L.C., Wentz, D.A., Bauch, N.J., Brigham, M.E., Moran, P.W., and Krabbenhoft, D.P. 2009. Mercury in fish, bed sediment, and water from streams across the United States, 1998–2005: U.S. Geological Survey Scientific Investigations Report 2009–5109, 74 p.

Sheppard, L. 2003. Ambient Air Pollution and Nonelderly Asthma Hospital Admissions in Seattle, Washington, 1987-1994. In Revised Analyses of Time-Series Studies of Air Pollution and Health. Special Report. Boston, MA: Health Effects Institute.

Shogren, J., and T. Stamland. 2002. Skill and the Value of Life. Journal of Political Economy 110:1168-1197.

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. American Journal of Respiratory and Critical Care Medicine 156(3 Pt 1):787-793.

Smith, V.K., G. Van Houtven, and S.K. Pattanayak. 2002. Benefit Transfer via Preference Calibration. Land Economics 78:132-152.

Smith, G., Coulston, J., Jepsen, E., Prichard, T. 2003. A national ozone biomonitoring program—results from field surveys of ozone sensitive plants in Northeastern forests (1994-2000). Environmental Monitoring and Assessment, 87, 271-291.

Smith, V.K., G. Van Houtven, and S.K. Pattanayak. 2002. Benefit Transfer via Preference Calibration. Land Economics 78:132-152.

N. Sorensen, K. Murata, E. Budtz-Jorgensen, P. Weihe, and Grandjean, P., 1999. "Prenatal Methylmercury Exposure As A Cardiovascular Risk Factor At Seven Years of Age", Epidemiology, pp370-375.

Spencer, D.M., and D.F. Holecek. 2007. Basic characteristics of the fall tourism market. Tourism Management 28:491–504.

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.

Stern, A. 2005, "A review of the studies of the cardiovascular health effects of methylmercury with consideration of their suitability for risk assessment," Environmental Research, 98 133-142.

Stieb, D. M.; Judek, S.; Burnett, R. T. 2002. Meta-analysis of time-series studies of air pollution and mortality:effects of gases and particles and the influence of cause of death, age, and season. J. Air Waste Manage. Assoc. 52: 470-484.

Stieb, D. M.; Judek, S.; Burnett, R. T. 2003. Meta-analysis of time-series studies of air pollution and mortality: update in relation to the use of generalized additive models. J. Air Waste Manage. 53: 258-261.

Tagaris E, Liao KJ, Delucia AJ, et al. 2009. Potential impact of climate change on air-pollution related human health effects. Environ. Sci. Technol. 43: 4979—4988.

Taylor R. 1994. Deterministic versus stochastic evaluation of the aggregate economic effects of price support programs. Agricultural Systems 44: 461-473.

Tingey, D.T., and Taylor, G.E. 1982 Variation in plant response to ozone: a conceptual model of physiological events. In M.H. Unsworth & D.P. Omrod (Eds.), Effects of Gaseous Air Pollution in Agriculture and Horticulture. (pp.113-138). London, UK: Butterworth Scientific.

Tolley, G.S. et al. 1986. Valuation of Reductions in Human Health Symptoms and Risks. University of Chicago. Final Report for the U.S. Environmental Protection Agency. January

Tolley, G., A. Randall, G. Blomquist, M. Brien, R. Fabian, G. Fishelson, A. Frankel, M. Grenchik, J. Hoehn, A. Kelly, R. Krumm, E. Mensah, and T. Smith. 1986. Establishing and Valuing the Effects of Improved Visibility in Eastern United States. Prepared for U.S. Environmental Protection Agency, Office of Policy, Planning and Evaluation. U.S.Environmental Protection Agency Grant #807768-01-0.

Tsuji, H., M.G. Larson, F.J. Venditti, Jr., E.S. Manders, J.C. Evans, C.L. Feldman, D. Levy. 1996. Impact of Reduced Heart Rate Variability on Risk for Cardiac Events. The Framingham Heart Study. Circulation 94(11):2850-2855.

U.S. Bureau of Census. 2000. Population Projections of the United States by Age, Sex, Race, Hispanic Origin and Nativity: 1999 to 2100. Population Projections Program, Population Division, U.S. Census Bureau, Washington, DC. Available on the Internet at <http://www.census.gov/population/projections/nation/summary/np-t.txt>.

U.S. Bureau of the Census. 2001. Statistical Abstract of the United States, 2001, Section 12: Labor Force, Employment, and Earnings, Table No. 521. Washington, DC.

U.S. Department of Agriculture - Forest Service (USDA Forest Service). 2006. Ozone bioindicator data. Accessed 2006. Available on the Internet at <http://nrs.fs.fed.us/fia/topics/ozone/data/>.

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 (NSRE). Available on the Internet at <http://www.census.gov/prod/2008pubs/fhw06-nat.pdf >.

U.S. Environmental Protection Agency (U.S. EPA). 1999a. Regional Haze Regulations, July 1, 1999. (64 FR 35714, July 1, 1999).

U.S. Environmental Protection Agency (U.S. EPA). 1999b. 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. EPA report no. EPA410-R-99-001. 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). 2001. Water Quality Criterion for the Protection of Human Health: Methylmercury (Final). Office of Science and Technology. Office of Water. January. Washington, DC. Available on the Internet at <http://www.epa.gov/waterscience/criteria/methylmercury/document.html>.

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). 2004. Air Quality Criteria for Particulate Matter Volume II of II. National Center for Environmental Assessment, Office of Research and Development, U.S. Environmental Protection Agency, Research Triangle Park, NC EPA/600/P-99/002bF. Available on the Internet at <http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=87903>.

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 (Final). EPA/600/R-05/004aF-cF. Washington, DC: U.S. EPA. 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. Ecological Benefits Assessment Strategic Plan. EPA-240-R-06-001. Office of the Administrator. Washington, DC. October. Available on the Internet at <http://yosemite.epa.gov/ee/epa/eed.nsf/webpages/EcologBenefitsPlan.html>.

U.S. Environmental Protection Agency (U.S. EPA). 2006c. 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). 2006d. 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). 2007a. Guidance on the Use of Models and Other Analyses for Demonstrating Attainment of Air Quality Goals for Ozone, $PM_{2.5}$, and Regional Haze. Office of Air Quality Planning and Standards. EPA-454/B-07-002. April. Available on the Internet at http://www.epa.gov/scram001/guidance/guide/final-03-pm-rh-guidance.pdf

U.S. Environmental Protection Agency (U.S. EPA). 2007b. 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. July. 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. Integrated Science Assessment for Sulfur Oxides - Health Criteria (Final Report). National Center for Environmental Assessment, Research Triangle Park, NC. September. Available on the Internet at < http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=198843>.

U.S. Environmental Protection Agency (U.S. EPA). 2008e. 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). 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. The NO_x Budget Trading Program: 2008 Environmental Results. Clean Air Markets Division. September. Available on the Internet at http://www.epa.gov/airmarkt/progress/NBP_3/NBP_2008_Environmental_Results.pdf

U.S. Environmental Protection Agency (U.S. EPA). 2009c. Risk and Exposure Assessment for Review of the Secondary National Ambient Air Quality Standards for Oxides of Nitrogen and Oxides of Sulfur (Final). EPA-452/R-09-008a. Office of Air Quality Planning and Standards, Research Triangle Park, NC. 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). 2009d. 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). 2009e. 2008 National Listing of Fish Advisories. Fish Advisory Program. EPA-823-F-09-007. September. Available on the Internet at <http://www.epa.gov/waterscience/fish/advisories/tech2008.pdf>.

U.S. Environmental Protection Agency (U.S. EPA) 2010a. Regulatory Impact Analysis, National Ambient Air Quality Standards for Ozone, 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/6-ozoneriachapter6.pdf>.

U.S. Environmental Protection Agency (U.S. EPA). 2010b. Final Regulatory Impact Analysis (RIA) for the NO_2 National Ambient Air Quality Standards (NAAQS). Office of Air Quality Planning and Standards, Research Triangle Park, NC. January. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/RIAs/FinalNO2RIAfulldocument.pdf>.

U.S. Environmental Protection Agency Science Advisory Board (U.S. EPA-SAB). 1999. An SAB Advisory: The Clean Air Act Section 812 Prospective Study Health and Ecological Initial Studies. Prepared by the Health and Ecological Effects Subcommittee (HEES) of the Advisory Council on the Clean Air Compliance Analysis, Science Advisory Board, U.S. Environmental Protection Agency. Washington DC. EPA-SAB-COUNCIL-ADV-99-005.

U.S. Environmental Protection Agency – Science Advisory Board (U.S. EPA-SAB). 2000. An SAB Report on EPA's White Paper Valuing the Benefits of Fatal Cancer Risk Reduction. EPA-SAB-EEAC-00-013.

U.S. Environmental Protection Agency Science Advisory Board (U.S. EPA-SAB). 2004. Advisory on Plans for Health Effects Analysis in the Analytical Plan for EPA's Second Prospective Analysis – Benefits and Costs of the Clean Air Act, 1990-2020. EPA-SAB-COUNCIL-ADV-04- 002 March.

U.S. Environmental Protection Agency Science Advisory Board (U.S. EPA-SAB). 2006. Clean Air Scientific Advisory Committee's (CASAC) Peer Review of the Agency's 2nd Draft Ozone Staff Paper. EPA-CASAC-07-001. October. Available on the Internet at <http://yosemite.epa.gov/sab/sabproduct.nsf/AB290E0DB8B72A33852572120055858F/$File/casac-07-001.pdf>.

U.S. Environmental Protection Agency - Science Advisory Board (U.S. EPA-SAB). 2007a. 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). 2007b. Clean Air Scientific Advisory Committee's (CASAC) Review of the Agency's Final Ozone Staff Paper. EPA-CASAC-07-002. Available on the Internet at http://yosemite.epa.gov/sab/sabproduct.nsf/FE915E916333D776852572AC007397B5/$File/casac-07-002.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>.

U.S. Environmental Protection Agency and U.S. Food and Drug Administration (U.S. EPA/FDA). 2004. What You Need to Know About Mercury in Fish and Shellfish: Advice for Women Who Might Become Pregnant Women, Who are Pregnant, Nursing Mothers, Young Children. EPA-823-F-04-009. Available on the Internet at <http://www.epa.gov/waterscience/fish/files/MethylmercuryBrochure.pdf>.

U.S. Fish and Wildlife Service and U.S. Census Bureau (FWS and Census). 2007. 2006 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (FHWAR). FHW/06-NAT. U.S. Department of the Interior, U.S. Fish and Wildlife Service, Washington, DC, and U.S. Department of Commerce, U.S. Census Bureau, Washington, DC.

U.S. Food and Drug Administration (U.S. FDA). 2001. Fish and Fisheries Products Hazards and Controls Guidance: Chapter 10: Methyl Mercury. Third Edition. June. Available on the Internet at < http://www.fda.gov/Food/GuidanceComplianceRegulatoryInformation/GuidanceDocuments/Seafood/ucm092041.htm>.

U.S. Forest Service (USFS). 2006. Forest Inventory and Analysis National Program: Forest Inventory Data Online. Online database. U.S. Department of Agriculture Forest Service, Forest Inventory and Analysis, Arlington, VA. Available on the Internet at

<http://fia.fs.fed.us/tools-data>.

U.S. Geological Survey (USGS). 2006. Investigations and Monitoring of Mercury in Indiana by the U.S. Geological Survey. U.S. Department of the Interior, U.S. Geological Survey, Indiana Water Science Center, Indianapolis, IN. Available athttp://in.water.usgs.gov/mercury.

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.

Vedal, S., J. Petkau, R. White, and J. Blair. 1998. Acute Effects of Ambient Inhalable Particles in Asthmatic and Nonasthmatic Children. American Journal of Respiratory and Critical Care Medicine 157(4):1034-1043.

Virtanen JK, S Voutilainen, T.H. Rissanen, J Mursu, T.P.Tuomainen, M.J. Korhonen, V.P.Walkonen, K. Seppanen,J.A. Laukkanen, J.T.Salonen.Grandjean, P., K. Murata, E. Budtz-Jorgensen, and P.Weihe. 2004. “Autonomic Activity in Methylmercury Neurotoxicity:14-Year Follow-Up of a Faroese Birth Cohort”, The Journal of Pediatrics, February, pp169-176.

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.

Viscusi, W.K., W.A. Magat, and J. Huber. 1991. Pricing Environmental Health Risks: Survey Assessments of Risk-Risk and Risk-Dollar Trade-Offs for Chronic Bronchitis. Journal of Environmental Economics and Management 21:32-51.

Wilson, A. M., C. P. Wake, T. Kelly, et al. 2005. Air pollution, weather, and respiratory emergency room visits in two northern New England cities: an ecological time-series study. Environ Res. Vol. 97 (3): 312-21.

Winner, W.E. 1994. Mechanistic analysis of plant responses to air pollution. Ecological Applications, 4(4), 651-661.

Winner, W.E., and C.J. Atkinson. 1986. Absorption of air pollution by plants, and consequences for growth. Trends in Ecology and Evolution 1:15-18.

Wittels, E.H., J.W. Hay, and A.M. Gotto, Jr. 1990. Medical Costs of Coronary Artery Disease in the United States. American Journal of Cardiology 65(7):432-440.

Woodruff, T.J., J. Grillo, and K.C. Schoendorf. 1997. The Relationship Between Selected Causes of Postneonatal Infant Mortality and Particulate Air Pollution in the United States. Environmental Health Perspectives 105(6):608-612.

Woodruff TJ, Parker JD, Schoendorf KC. 2006. Fine particulate matter ($PM_{2.5}$) air pollution and selected causes of postneonatal infant mortality in California. Environmental Health Perspectives 114(5):786-90.

Woods & Poole Economics Inc. 2008. Population by Single Year of Age CD. CD-ROM. Woods & Poole Economics, Inc. Washington, D.C.

Yoshizawa, K et. al. 2002. "Mercury and the Risk of Coronary Heart Disease in Men," New England Journal of Medicine, Vol. 374, No. 22, November.

CHAPTER 6

ELECTRIC POWER SECTOR PROFILE

This chapter discusses important aspects of the power sector that relate to the Transport Rule, including the types of power-sector sources affected by the Transport Rule, and provides background on the power sector and electric generating units (EGUs). In addition, this chapter provides some historical background on EPA regulation of and future projections for the power sector.

6.1 Power Sector Overview

The production and delivery of electricity to customers consists of three distinct segments: generation, transmission, and distribution.

6.1.1 Generation

Electricity generation is the first process in the delivery of electricity to consumers. Most of the capacity for generating electricity involves creating heat to rotate turbines which, in turn, create electricity. The power sector consists of over 17,000 generating units, comprising fossil-fuel-fired units, nuclear units, and hydroelectric and other renewable sources dispersed throughout the country (see Table 6-1).

Table 6-1. Existing Electricity Generating Capacity by Energy Source, 2008

Energy Source	Number of Generators	Generator Nameplate Capacity (MW)	Generator Net Summer Capacity (MW)
Coal	1,445	337,300	313,322
Petroleum	3,768	63,655	57,445
Natural Gas	5,467	454,611	397,432
Other Gases	102	2,262	1,995
Nuclear	104	106,147	100,755
Hydroelectric Conventional	3,996	77,731	77,930
Wind Farms	494	24,980	24,651
Solar Thermal and Photovoltaic Projects	89	539	536
Wood and Wood Derived Fuels	353	7,730	6,864
Geothermal	228	3,281	2,256
Other Biomass	1,412	4,854	4,186
Pumped Storage	151	20,355	21,858
Other	49	1,042	942
Total	17,658	1,104,486	1,010,171

Source: EIA Electric Power Annual 2008, Table 1.2

These electric generating sources provide electricity for commercial, industrial, and residential uses, each of which consumes roughly a quarter to a third of the total electricity produced (see Table 6-2). Some of these uses are highly variable, such as heating and air conditioning in residential and commercial buildings, while others are relatively constant, such as industrial processes that operate 24 hours a day.

Table 6-2. Total U.S. Electric Power Industry Retail Sales in 2008 (Billion kWh)

		Sales/Direct Use (Billion kWh)	Share of Total End Use
Retail Sales	Residential	1,380	35%
	Commercial	1,336	34%
	Industrial	1,009	26%
	Transportation	8	0.2%
Direct Use		173	4%
Total End Use		**3,906**	**100%**

Source: EIA Electric Power Annual 2008, Table 7.2

In 2008, electric generating sources produced 4,157 billion kWh to meet electricity demand. Roughly 70 percent of this electricity was produced through the combustion of fossil fuels, primarily coal and natural gas, with coal accounting for approximately half of the total (see Table 6-3).

Table 6-3. Electricity Net Generation in 2008 (Billion kWh)

	Net Generation (Billion kWh)	**Fuel Source Share**
Coal	1,986	48%
Petroleum	32	1%
Natural Gas	883	21%
Other Gases	12	0.3%
Nuclear	806	20%
Hydroelectric	255	6%
Other	146	4%
Total	**4,119**	100%

Source: EIA Electric Power Annual 2008, Table 1.1
Note: Retail sales and net generation are not equal because net generation includes net exported electricity and loss of electricity that occurs through transmission and distribution.

Coal-fired generating units typically supply "base-load" electricity, the portion of electricity loads which are continually present, and typically operate throughout the day. Along with nuclear generation, these coal units meet the part of demand that is relatively constant. Gas-fired generation, on the other hand, is more often used to meet the variable portion of the electricity load and typically supplies "peak" power, when there is increased demand for electricity (for example, when businesses operate throughout the day or when people return home from work and run appliances and heating/air-conditioning), versus late at night or very early morning, when demand for electricity is reduced.

6.1.2 Transmission

Transmission is the term used to describe the movement of electricity over a network of high voltage lines, from electric generators to substations where power is stepped down for local distribution. In the US and Canada, there are three separate interconnected networks of high voltage transmission lines[50], each operating at a common frequency. Within each of these transmission networks, there are multiple areas where the operation of power plants is monitored and controlled to ensure that electricity generation and load are kept in balance. In some areas, the operation of the transmission system is under the control of a single regional operator; in others, individual utilities coordinate the operations of their generation, transmission, and distribution systems to balance their common generation and load needs.

6.1.3 Distribution

Distribution of electricity involves networks of lower voltage lines and substations that take the higher voltage power from the transmission system and step it down to lower voltage levels to match the needs of customers. The transmission and distribution system is the classic example of a natural monopoly, in part because it is not practical to have more than one set of lines running from the electricity generating sources to substations or from substations to residences and business, and in part because both transmission and generation are characterized by very high capital costs and very low variable operating costs.

Transmission has generally been developed by the larger vertically integrated utilities that typically operate generation and distribution networks. Distribution is handled by a large number of utilities that often purchase and sell electricity, but do not generate it. Transmission and distribution have been considered differently from generation in efforts to restructure the industry. As discussed below, electricity restructuring has focused primarily on efforts to reorganize the industry to encourage competition in the generation segment of the industry, including ensuring open access of generation to the transmission and distribution services needed to deliver power to consumers. In many state efforts, this has also included separating generation assets from transmission and distribution assets into

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[50] These three network interconnections are the western US and Canada, corresponding approximately to the area west of the Rocky Mountains; eastern US and Canada, not including most of Texas; and a third network operating in most of Texas.

separate economic entities. Transmission and distribution remain price-regulated throughout the country based on the cost of service.

6.2 Deregulation and Restructuring

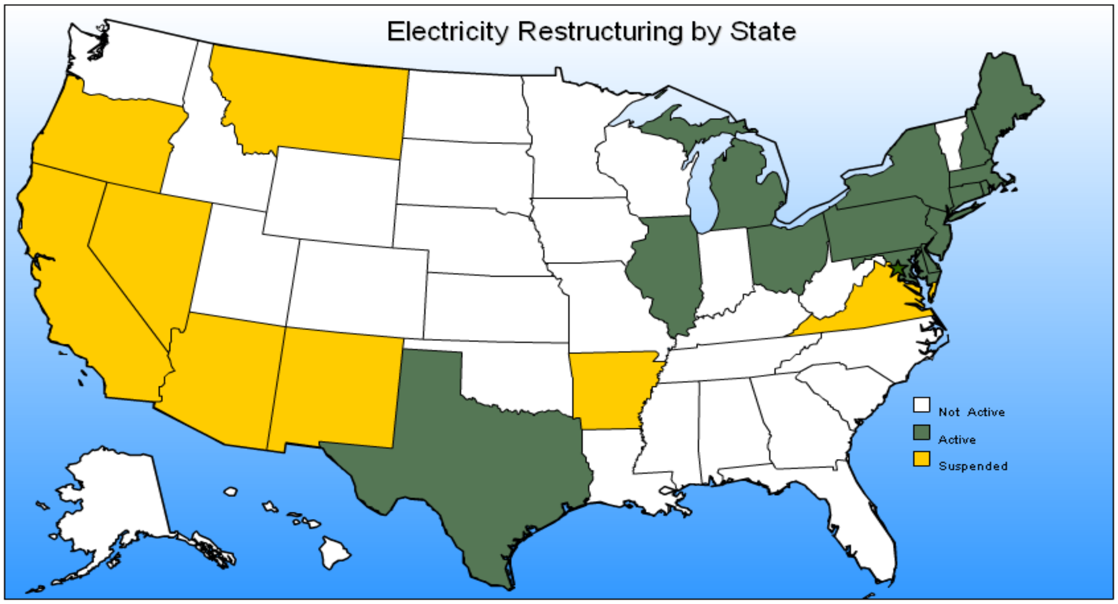
The process of restructuring and deregulation of wholesale and retail electric markets has changed the structure of the electric power industry. In addition to reorganizing asset management between companies, restructuring sought a functional unbundling of the generation, transmission, distribution, and ancillary services the power sector has historically provided, with the aim of enhancing competition in the generation segment of the industry.

Beginning in the 1970s, government policy shifted against traditional regulatory approaches and in favor of deregulation for many important industries, including transportation (notably commercial airlines), communications, and energy, which were all thought to be natural monopolies (prior to 1970) that warranted governmental control of pricing. However, deregulation efforts in the power sector were most active during the 1990s. Some of the primary drivers for deregulation of electric power included the desire for more efficient investment choices, the possibility of lower electric rates, reduced costs of combustion turbine technology that opened the door for more companies to sell power, and complexity of monitoring utilities' cost of service and establishing cost-based rates for various customer classes (see Figure 6-1).

The pace of restructuring in the electric power industry slowed significantly in response to market volatility and financial turmoil associated with bankruptcy filings of key energy companies in California. By the end of 2001, restructuring had either been delayed or suspended in eight states that previously enacted legislation or issued regulatory orders for its implementation (shown as "Suspended" in Figure 6-1 below). Another 18 other states that had seriously explored the possibility of deregulation in 2000 reported no legislative or regulatory activity in 2001 (DOE, EIA, 2003a) ("Not Active" in Figure 6-1 below). Currently, there are 15 states where price deregulation of generation (restructuring) has occurred ("Active" in Figure 6-1 below). Thirteen of these states are in the Transport Rule region. The effort is more or less at a standstill; there have been no recent proposals to the

Federal Energy Regulatory Commission (FERC) for actions aimed at wider restructuring, and no new states have begun retail deregulation activity.

Figure 6-1. Status of State Electricity Industry Restructuring Activities

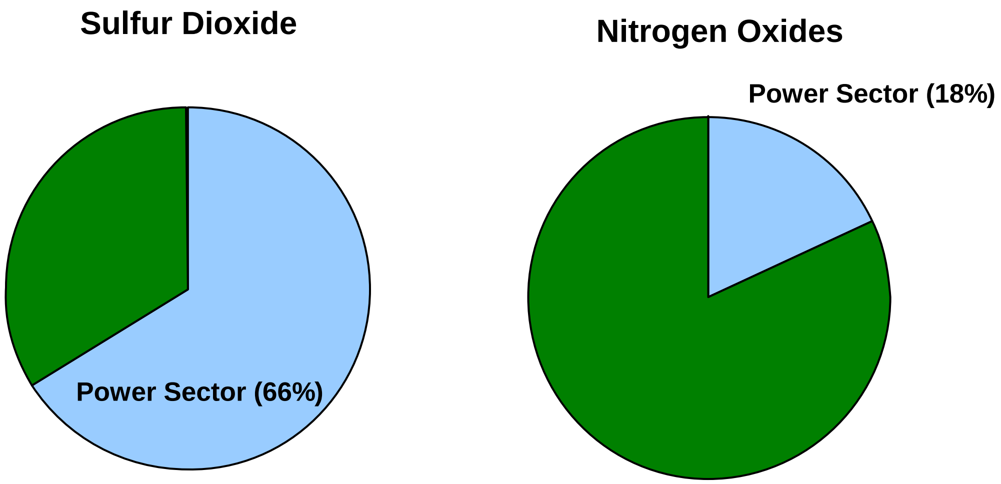


Source: EIA http://www.eia.doe.gov/cneaf/electricity/page/restructuring/restructure_elect.html (January 2010).

6.3 Pollution and EPA Regulation of Emissions

The burning of fossil fuels, which generates about 70 percent of our electricity nationwide, results in air emissions of SO_2 and NO_x, important precursors in the formation of fine particles and ozone (NO_x only). The power sector is a major contributor of both these pollutants, and reductions of SO_2 and NO_x emissions are critical to EPA's efforts to bring about attainment with the fine particle and ozone NAAQS through programs like the Transport Rule. In 2008, the power sector accounted for 66 percent of total nationwide SO_2 emissions and 18 percent of total nationwide NO_x emissions (see Figure 6-2).

Figure 6-2. Emissions of SO_2 and NO_x from the Power Sector (2008)



Source: EPA http://www.epa.gov/ttn/chief/trends/

Different types of fossil fuel-fired units vary widely in their air emissions levels for SO_2 and NO_x, particularly when uncontrolled. For coal-fired units, NO_x emissions rates can vary from under 0.05 lbs/mmBtu (for a unit with selective catalytic reduction (SCR) for NO_x removal) to over 1 lb/mmBtu for an uncontrolled cyclone boiler. NO_x emissions from coal-fired power plants are formed during combustion and are a result of both nitrogen in coal and nitrogen in the air. SO_2 emissions rates can vary from under 0.1 lbs/mmBtu (for some units with flue gas desulfurization (FGD) for SO_2 removal) to over 5 lbs/mmBtu for units burning higher sulfur coal without any pollution controls. For an uncontrolled coal plant, SO_2 emissions are directly related to the amount of sulfur in the coal.

Oil- and gas-fired units also have a wide range of NO_x emissions depending on both the plant type and the controls installed. Gas-fired units with SCR can have emissions rates under 0.01 lbs/mmBtu, while completely uncontrolled units can have emissions rates in excess of 0.5 lbs/mmBtu. Gas-fired units have very little SO_2 emissions. NO_x emissions rates on oil-fired units can range from under 0.1 lbs/mmBtu (for units with new combustion controls) to over 0.6 lbs/mmBtu for units without combustion controls. SO_2 emissions for oil-fired units can range from under 0.1 lbs/mmBtu for units burning low sulfur distillate oil to over 2 lbs/mmBtu for units burning high sulfur residual oil.

6.4 Pollution Control Technologies

There are three options for reducing SO_2 emissions from coal-burning power plants. Units may switch from higher to lower sulfur coal, blend higher sulfur coal with lower sulfur coal, or use FGD, commonly referred to as scrubbers. According to data submitted to EPA for compliance with the Title IV Acid Rain Program, the SO_2 emissions rates for coal-fired units without controls varied from under 0.4 lbs/mmBtu to over 5 lbs/mmBtu depending on the type of coal combusted. With controls, rates range from as low as 0.03 lbs/mmBtu to close to 1 lb/mmBtu.

It is generally easier to switch to a coal within the same rank (e.g., bituminous or sub-bituminous) because these coals will have similar heat contents and other characteristics. Switching completely to sub-bituminous coal (which typically has lower sulfur content) from bituminous coal is likely to require some modifications to the unit. Limited blending of sub-bituminous coal with bituminous coal can often be done with fewer modifications.

The two most commonly used scrubber types include wet scrubbers and spray dryers, also known as dry scrubbers. Wet scrubbers can use a variety of sorbents to capture SO_2, including limestone and magnesium-enhanced lime. The choice of sorbent can affect the performance, size, and capital and operating costs of the scrubber. New wet scrubbers typically achieve at least 95 percent SO_2 removal. Spray dryers use lime-based slurry and can achieve over 90 percent removal.

One method of reducing NO_x emissions is through the use of combustion controls (such as low NO_x burners and over-fire air). Combustion controls adjust the coal combustion conditions to those where less formation of NO_x occurs. Post-combustion controls remove the NO_x after it has been formed. The most common post-combustion control is SCR. In SCR systems ammonia (NH_3) is injected, which combines with the NO_x in the flue gas, to form nitrogen and water and uses a catalyst to enhance the reaction. These systems can reduce NO_x by 90 percent and achieve emissions rates of around 0.06 lbs/mmBtu. Another post-combustion control is Selective Non-catalytic Reduction (SNCR). In this technology

NO_x also is removed by injecting ammonia, but no catalyst is used. SNCR systems can reduce NO_x by up to 40 percent.

Some of the same control options available to coal-fired units are also applicable to units fueled by oil or gas. Combustion controls, SCR, and SNCR can also be applied to oil- and gas-fired boilers for NO_x control. Combustion controls and SCR are also routinely used for NO_x control on gas turbines.

For more detail on the cost and performance assumptions of pollution controls, see the documentation for the Integrated Planning Model (IPM), a dynamic linear programming model that EPA uses to examine air pollution control policies for SO_2 and NO_x throughout the contiguous United States for the entire power system. Documentation for IPM can be found at www.epa.gov/airmarkets/epa-ipm and in the TSD "Updates to EPA Base Case v3.02 EISA Using the Integrated Planning Model."

6.5 Regulation of the Power Sector

At the federal level, efforts to reduce emissions of SO_2 have been occurring since 1970. Policy makers have recognized the need to address these harmful emissions, and incremental steps have been taken to ensure that the country meets air quality standards. The Transport Rule is the next step towards realizing attainment of the national standards for $PM_{2.5}$ and ozone.

Even before widespread regulation of SO_2 and NO_x for the power sector, total suspended particulate matter (TSP) was a related target of state and federal action. Because larger particulates are visible as dark smoke from smokestacks, most states had regulations by 1970 limiting the opacity of emissions. Requirements for taller smokestacks also mitigated local impacts of TSP. Notably, such regulations effectively addressed large-diameter, filterable particulate matter rather than condensable particulate matter (such as $PM_{2.5}$) associated with SO_2 and NO_x emissions, which are not visible at the smokestack and have impacts far from their sources.

Federal regulation of SO_2 and NO_x emissions at power plants began with the 1970 Clean Air Act. The Act required the Agency to develop New Source Performance Standards (NSPS) for a number of source categories including coal-fired power plants. The first NSPS for power plants (subpart D) required new units to limit SO_2 emissions either by using scrubbers or by using low sulfur coal. NO_x was required to be limited through the use of low NO_x burners. A new NSPS (subpart Da), promulgated in 1978, tightened the standards for SO_2, requiring scrubbers on all new units.

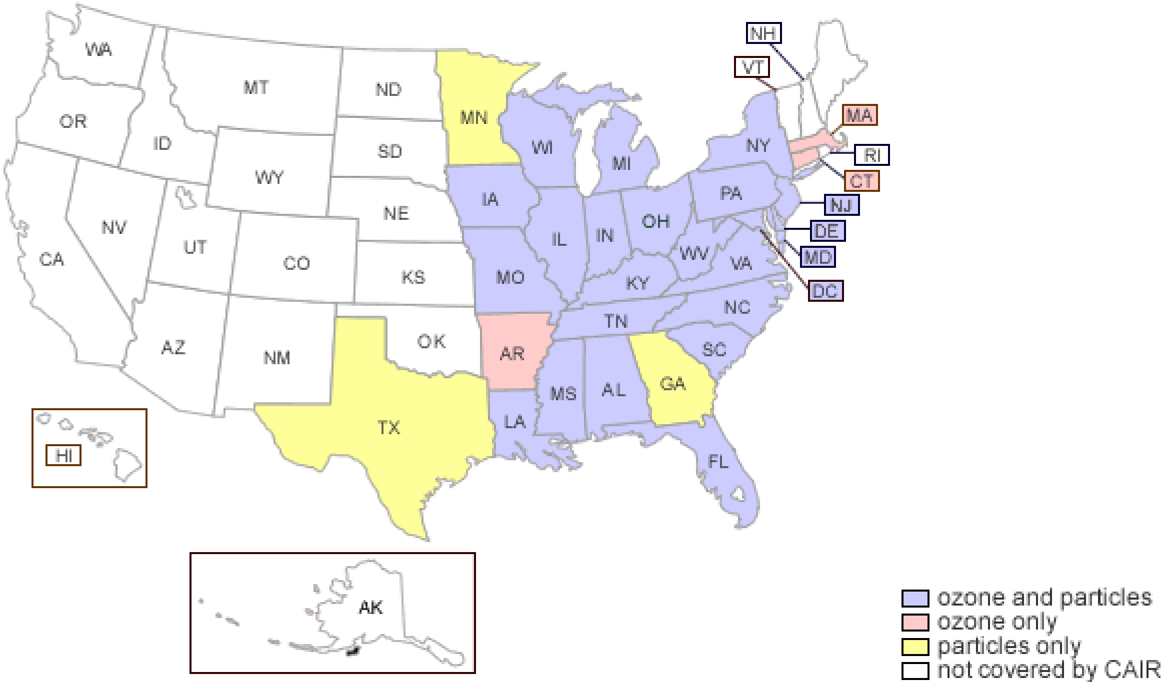
The 1990 Clean Air Act Amendments (CAAA) placed a number of new requirements on power plants. The Acid Rain Program, established under Title IV of the 1990 CAAA, requires major reductions of SO_2 and NO_x emissions. The SO_2 program sets a permanent cap on the total amount of SO_2 that can be emitted by electric power plants in the contiguous United States at about one-half of the amount of SO_2 these sources emitted in 1980. Using a market-based cap and trade mechanism allows flexibility for individual combustion units to select their own methods of compliance with the SO_2 reduction requirements. The program uses a more traditional approach to NO_x emissions limitations for certain coal-fired electric utility boilers, with the objective of achieving a 2 million ton reduction from projected NO_x emission levels that would have been emitted in 2000 without implementation of Title IV.

The Acid Rain Program comprises two phases for SO_2 and NO_x. Phase I applied primarily to the largest coal-fired electric generating sources from 1995 through 1999 for SO_2 and from 1996 through 1999 for NO_x. Phase II for both pollutants began in 2000. For SO_2, it applies to thousands of combustion units generating electricity nationwide; for NO_x it generally applies to affected units that burned coal during 1990 through 1995. The Acid Rain Program has led to the installation of a number of scrubbers on existing coal-fired units as well as significant fuel switching to lower sulfur coals. Under the NO_x provisions of Title IV, most existing coal-fired units installed low NO_x burners.

The CAAA also placed much greater emphasis on control of NO_x to reduce ozone nonattainment. This led to the formation of several regional NO_x trading programs as well as intrastate NO_x trading programs in states such as Texas. The northeastern states of the Ozone Transport Commission (OTC) required existing sources to meet Reasonably Available Control Technology (RACT) limits on NO_x in 1995 and in 1999 began an ozone-season cap and trade program to achieve deeper reductions. In 1998, EPA promulgated regulations (the NO_x SIP Call) that required 21 states in the eastern United States and the District of Columbia to reduce NO_x emissions that contributed to nonattainment in downwind states using the cap and trade approach. This program began in May of 2003 and has resulted in the installation of significant amounts of selective catalytic reduction.

The Clean Air Interstate Rule (CAIR) built on EPA's efforts in the NO_x SIP call to address specifically interstate pollution transport for ozone, and was EPA's first attempt to address interstate pollution transport for $PM_{2.5}$. It required significant reductions in emissions of SO_2 and NO_x in 28 states and the District of Columbia (see Figure 6-3 below). EGUs were found to be a major source of the SO_2 and NO_x emissions which contributed to fine particle concentrations and ozone problems downwind. Although the D.C. Circuit remanded the rule to EPA in 2008, it did so without vacatur, allowing the rule to remain in effect while EPA addresses the remand. Thus, CAIR is continuing to help states address ozone and $PM_{2.5}$ nonattainment and improve visibility by reducing transported precursors of SO_2 and NO_x through the implementation of three separate cap and trade compliance programs for annual NO_x, ozone season NO_x, and annual SO_2 emissions from power plants.

Figure 6-3. States Covered under the Clean Air Interstate Rule



Perhaps in anticipation of complying with CAIR, especially the more stringent second phase that was set to begin in 2015, several sources have recently been installing or planning to install advanced controls for SO_2 and NO_x to begin operating in the 2010 to 2015 timeframe. Many EPA New Source Review (NSR) settlements also require controls in those years, as do state rules in Georgia, Illinois, and Maryland. States like North Carolina, New York, Connecticut, Massachusetts, and Delaware have also moved to control these emissions to address nonattainment. Thus both federal and state efforts are continuing to bring about sizeable reductions in SO_2 and NO_x from the power sector. Section 7-1 below discusses how these recent activities are reflected in the Transport Rule base case. Details of the NSR settlements and state controls can be found in the IPM documentation referenced earlier.

6.6 Price Elasticity of Electricity

Electricity performs a vital and high-value function in the economy; as a result, electricity consumers are generally unable or unwilling to alter consumption as the price increases. Demand for electricity, especially in the short run, is not very sensitive to changes in prices and is considered relatively price inelastic, although some demand reduction does occur. With that in mind, EPA modeling does not incorporate a “demand response” in its electric generation modeling (Chapter 7) to any increases in electricity prices because of the reasons mentioned. Electricity demand is considered to be constant in EPA modeling applications and the reduction in production costs that would result from lower demand is not considered. This leads to some overstatement in the private compliance costs that EPA estimates. Notably, the “compliance costs” are the changes in the electric power generation costs in the base case and pollution control options that are evaluated in Chapter 7. In simple terms, it is the resource costs of what the power industry will directly expend to comply with EPA’s requirements. This is not the “social cost” of the rule which has been separately explained and estimated in EMPAX modeling in Chapter 8.

6.7 Reference

EIA Electric Power Annual 2008. DOE/EIA-0348 (2008). Available at: http://www.eia.doe.gov/cneaf/electricity/epa/epa_sum.htm

CHAPTER 7

COST, ECONOMIC, AND ENERGY IMPACTS

This chapter reports the cost, economic, and energy impact analysis performed for the Transport Rule. EPA used the Integrated Planning Model (IPM), developed by ICF Consulting, to conduct its analysis. IPM is a dynamic linear programming model that can be used to examine air pollution control policies for SO_2 and NO_x (as well as other air pollutants) throughout the contiguous United States for the entire power system. Documentation for IPM can be found at http://www.epa.gov/airmarkets/progsregs/epa-ipm, and updates specific to Transport Rule modeling are in the TSD “Updates to EPA Base Case v3.02 EISA Using the Integrated Planning Model.”

7.1 Background

Over the last decade, EPA has on several occasions used IPM to consider control options for reducing power-sector SO_2 and NO_x for regional transport. These, along with the alternative remedies analyzed alongside the proposed remedy below, both provide context and suggest alternative approaches to the Transport Rule proposed remedy (see Keohane 2009, 34–35 and Wagner 2009, 59).

Many EPA analyses with IPM have focused on legislative changes with national programs, such as EPA’s IPM analyses of the Clean Air Planning Act (S.843 in 108th Congress), the Clean Power Act (S.150 in 109th Congress), the Clear Skies Act of 2005 (S.131 in 109th Congress), the Clear Skies Act of 2003 (S.485 in 108th Congress), and the Clear Skies Manager's Mark (of S.131). These analyses are available at EPA’s website: (www.epa.gov/airmarkets/progsregs/cair/multi.html). EPA’s IPM analysis for CAIR is another example, in this case dealing with a regulatory approach focusing on the eastern US: (www.epa.gov/airmarkets/progsregs/epa-ipm/cair/index.html).

In addition, EPA conducted extensive state-by-state analysis of control levels and associated emissions projections related to identifying significant contribution to nonattainment and interference with maintenance for the Transport Rule. More details on this analysis can be found in the Significant Contribution Approach TSD.

As discussed in section 6.5, this proposed rule comes during a period when many new SO_2 and NO_x controls are being installed. Many are needed for compliance with NSR settlements and state rules, while others may have been planned in expectation of CAIR. Because CAIR remains in effect until it is replaced, emission reductions continue in the eastern US.

The base case in this RIA assumes that CAIR is not in effect, but does take into account emissions reductions associated with the implementation of all federal rules, state rules and statutes, and other binding, enforceable commitments finalized by February 3, 2009, that the power industry has for installing and operating SO_2 and NO_x emissions controls in the timeframe covered in the analysis.

EPA has made these base case assumptions recognizing that a key step in the process of developing a 110(a)(2)(D)(i)(I) rule, such as the Transport Rule, involves analyzing existing (base case) emissions to determine which states significantly contribute to downwind nonattainment and maintenance areas. EPA cannot prejudge at this stage which states will be affected by the rule. For example, a state affected by CAIR may not be affected by the new rule and after the new rule goes into effect, the CAIR requirements will no longer apply. For a state covered by CAIR but not covered by the new rule, the CAIR requirements would not be replaced with new requirements, and therefore an increase in emissions relative to present levels could occur in that state. More fundamentally, the court has made clear that, due to legal flaws, the CAIR rule cannot remain in place and must be replaced. If EPA's base case analysis were to ignore this fact and assume that reductions from CAIR would continue indefinitely, areas that are in attainment solely due to controls required by CAIR would again face nonattainment problems, because the existing protection from upwind pollution would not be replaced. For these reasons, EPA cannot assume in its base case analysis that the reductions required by CAIR will continue to be achieved.

Following this logic, the 2012 base case shows emissions higher than current levels in some states. Because EPA has been directed to replace CAIR, EPA believes that for many states, the absence of the CAIR NO_x program will lead to reversion to the NO_x Budget Trading Program (NBP), which substantially limits ozone-season NO_x emissions from electric power generation in a major part of the Eastern US and ensures the operation of NO_x controls in 20 covered states and the District of Columbia. The base case contains the NBP. Also, without the CAIR SO_2 program, there would remain the broad federal SO_2 emissions requirements for electric generation from fossil fuels in the lower 48 states for the comparatively less stringent CAA Title IV Acid Rain program. As a result, SO_2 emissions in many states would increase markedly in the 2012 base case relative to the present. Efforts to comply with ARP rules at the least-cost would occur in many cases solely through use of currently readily available, inexpensive Title IV allowances and without the operation of some existing scrubbers that do not have other binding enforceable requirements. Notably, all known controls for both SO_2 and NO_x that are required under state laws, NSPS, consent decrees, and other enforceable, binding commitments through 2014 are accounted for in the base case. These requirements are quite substantial in maintaining the operation of much of the existing advanced controls in place. It is against this backdrop that the Transport Rule is analyzed and that significant contribution to nonattainment and interference with maintenance must be addressed.

The model's base case features an updated Title IV SO_2 allowance bank assumption and incorporates updates related to the Energy Independence and Security Act of 2007. Many key assumptions, notably demand for electricity, reflect the 2008 Annual Energy Outlook from the Energy Information Administration (EIA).[51] In addition, the model includes policies affecting the power sector: the Title IV of the Clean Air Act (the Acid Rain Program); the NO_x SIP Call; various New Source Review (NSR) settlements[52]; and several

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[51] For the final rule, EPA anticipates using an updated version of IPM that will reflect assumptions from AEO 2010.

[52] The NSR settlements include agreements between EPA and Southern Indiana Gas and Electric Company (Vectren), Public Service Enterprise Group, Tampa Electric Company, We Energies (WEPCO), Virginia Electric & Power Company (Dominion), Santee Cooper, Minnkota Power Coop, American Electric Power (AEP), East Kentucky Power Cooperative (EKPC), Nevada Power Company, Illinois Power, Mirant, Ohio Edison, and Kentucky Utilities. These agreements lay out specific NO_x, SO_2, and other emissions controls for the fleets of these major Eastern companies by specified dates. Many of the pollution controls (e.g FGDs and SCRs) are required between 2010 and 2014.

state rules[53] affecting emissions of SO_2 and NO_x that were finalized through February 3, 2009. IPM includes state rules that have been finalized and/or approved by a state's legislature or environmental agency. The IPM documentation TSD contains details on all of these other binding enforceable commitments for installation and operation of advanced NO_x and SO_2 pollution controls across much of the Eastern US. In providing the results of the EPA's analysis with IPM, the Agency is using the results for the model's 2015 run year (which covers years 2014, 2015, 2016, and 2017) as a proxy for 2014 results. IPM provides actual results for 2012, 2015, and 2020.

To address air quality problems and improve public health and the environment, EPA is proposing the Transport Rule. The proposed Transport Rule requires annual SO_2 and NO_x reductions in 27 states and the District of Columbia, and also requires ozone season NO_x reductions in 25 States and the District of Columbia. Many of the Transport Rule States are affected by both the annual SO_2 and NO_x reduction requirements and the ozone season NO_x requirements.

The rule would affect roughly 5,000 fossil fuel-fired units with a nameplate capacity greater than 25 MW. These sources accounted for roughly 84 percent of nationwide SO_2 emissions and 73 percent of nationwide NO_x emissions in 2008 (see Table 7-1).

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[53] These include current and future state programs in Connecticut, Delaware, Georgia, Illinois, Maine, Maryland, Massachusetts, Minnesota, Missouri, New Hampshire, North Carolina, New Jersey, New York, Oregon, Texas, and Wisconsin the cover NO_x and SO_2 emissions controls.

Table 7-1. Annual Emissions of SO_2 and NO_x in 2008 and Percentage of Emissions in the Transport Rule Affected Region (tons)

	SO_2	NO_x
Transport Rule Annual NO_x and SO_2 States	6,439,067	2,267,008
Nationwide (Contiguous 48 States)	7,620,588	3,098,267
Emissions of Transport Rule States as Percentage of Nationwide Emissions	84%	73%

Source: EPA emissions data from all reporting units.

Note: Transport Rule annual NO_X and SO_2 states include Alabama, Connecticut, Delaware, Florida, Georgia, Illinois, Indiana, Iowa, Kansas, Kentucky, Louisiana, Maryland, Massachusetts, Michigan, Minnesota, Missouri, Nebraska, New Jersey, New York, North Carolina, Ohio, Pennsylvania, South Carolina, Tennessee, Virginia, West Virginia, and Wisconsin.

For SO_2 and annual NO_x, EPA modeled control requirements beginning in 2012 for the 27 eastern states shown in blue and green in Figure 7-1 below. In 15 of those states (shown in blue), more stringent SO_2 requirements begin in 2014. For ozone-season NO_x, separate ozone-season requirements were applied to the 25 states shown in blue in Figure 7-2. Many of the Transport Rule states are affected by both the annual SO_2 and NO_x reduction requirements and the ozone-season (May–September) NO_x requirements. Tables 7-2 and 7-3 show the emission budgets allotted to each state. For further discussion about the scope and requirements of the Transport Rule, see the Transport Rule preamble or Chapter 2 of this RIA.

Figure 7-1. Transport Rule Annual NO_x and SO_2 States

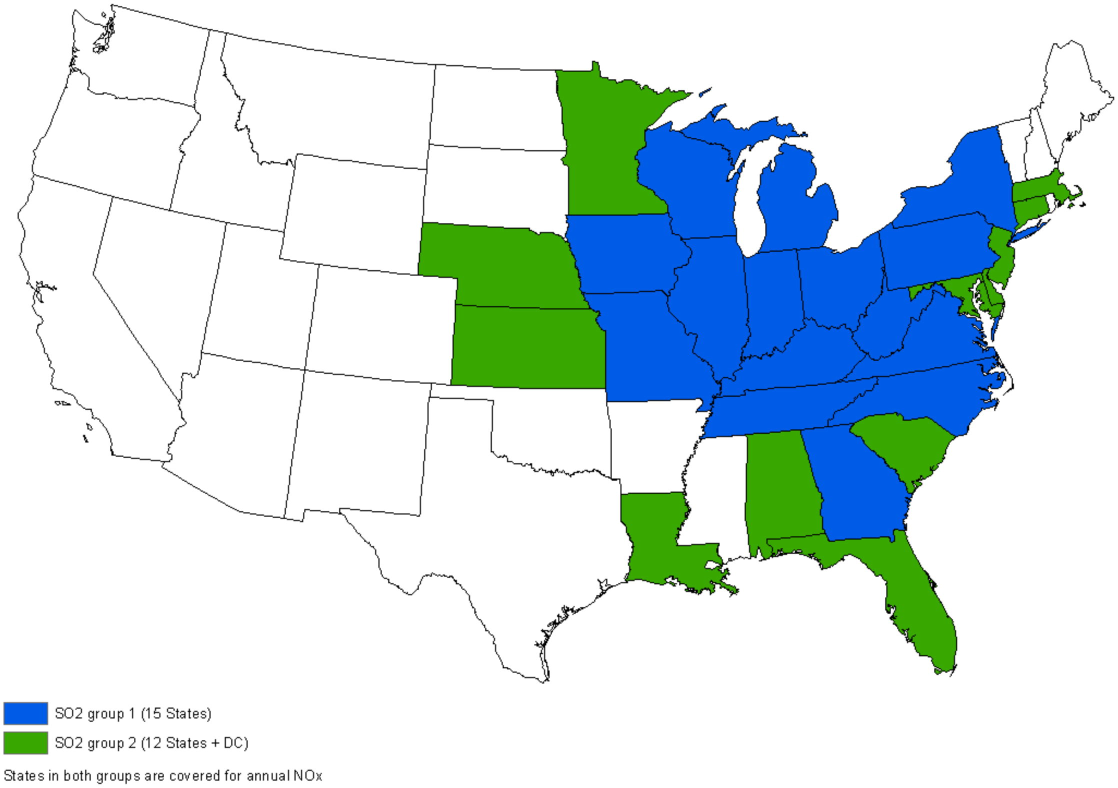

Figure 7-2. Transport Rule Ozone-season NO_x States

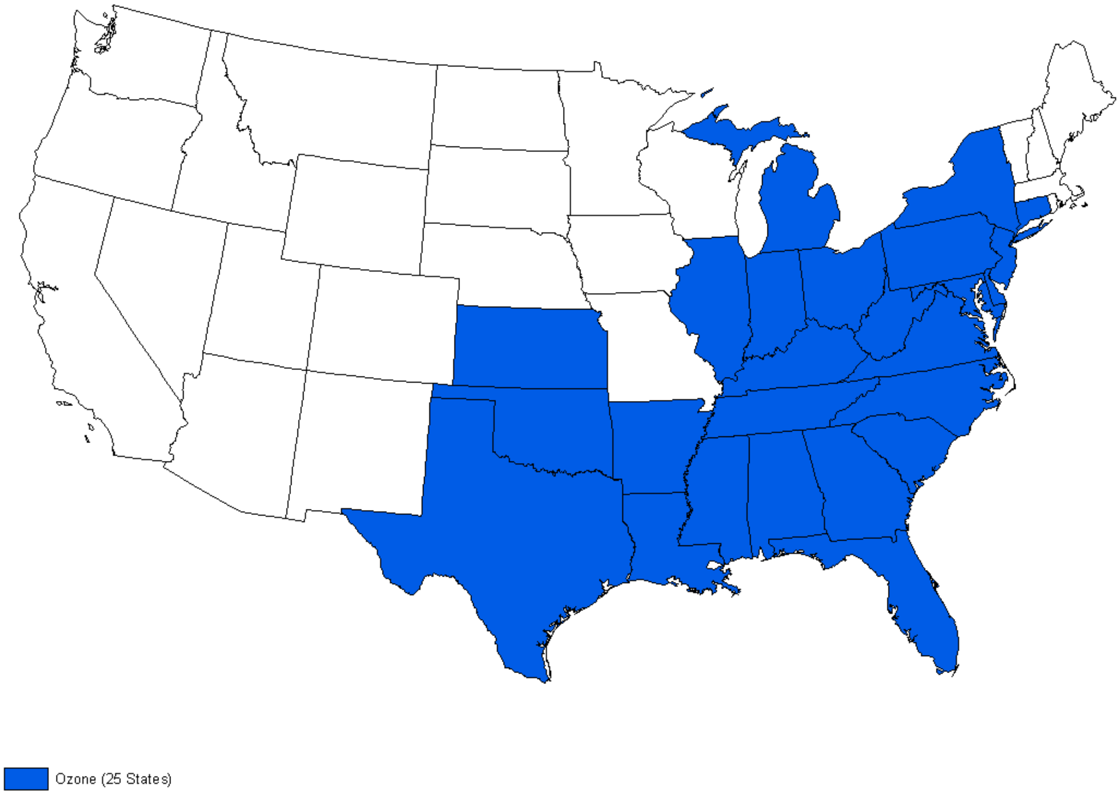

Table 7-2. Transport Rule Annual NO_x and SO_2 State Emission Budgets (tons)

State	SO_2, 2012 and 2013	SO_2, 2014 and Later	NO_x Annual, All Years
Alabama	161,871	161,871	69,169
Connecticut	3,059	3,059	2,775
Delaware	7,784	7,784	6,206
District of Columbia	337	337	170
Florida	161,739	161,739	120,001
Georgia	233,260	85,717	73,801
Illinois	208,957	151,530	56,040
Indiana	400,378	201,412	115,687
Iowa	94,052	86,088	46,068
Kansas	57,275	57,275	51,321
Kentucky	219,549	113,844	74,117
Louisiana	90,477	90,477	43,946
Maryland	39,665	39,665	17,044
Massachusetts	7,902	7,902	5,960
Michigan	251,337	155,675	64,932
Minnesota	47,101	47,101	41,322
Missouri	203,689	158,764	57,681
Nebraska	71,598	71,598	43,228
New Jersey	11,291	11,291	11,826
New York	66,542	42,041	23,341
North Carolina	111,485	81,859	51,800
Ohio	464,964	178,307	97,313
Pennsylvania	388,612	141,693	113,903
South Carolina	116,483	116,483	33,882
Tennessee	100,007	100,007	28,362
Virginia	72,595	40,785	29,581
West Virginia	205,422	119,016	51,990
Wisconsin	96,439	66,683	44,846
Group 1 SO_2 States	3,117,288	1,723,421	
Group 2 SO_2 States	776,582	776,582	
Total	3,893,870	2,500,003	1,376,312

Table 7-3. Ozone-season NO_x State Emission Budgets (tons)

State	NO_x Ozone Season, All Years
Alabama	29,738
Arkansas	16,660
Connecticut	1,315
Delaware	2,450
District of Columbia	105
Florida	56,939
Georgia	32,144
Illinois	23,570
Indiana	49,987
Kansas	21,433
Kentucky	30,908
Louisiana	21,220
Maryland	7,232
Michigan	28,253
Mississippi	16,530
New Jersey	5,269
New York	11,090
North Carolina	23,539
Ohio	40,661
Oklahoma	37,087
Pennsylvania	48,271
South Carolina	15,222
Tennessee	11,575
Texas	75,574
Virginia	12,608
West Virginia	22,234
Total	641,614

EPA modeling[54] shows that coal-fired and oil/gas-fired generation will continue to play an important part of the electricity generating portfolio in the United States. Electricity

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[54] EPA uses the IPM to make power-sector forecasts about emissions, costs, and other key factors of the power sector. Industry projections presented here are from EPA's base case scenario. For more information about IPM, see http://www.epa.gov/airmarkets/progsregs/epa-ipm/.

demand is anticipated to grow by roughly 1 percent a year, and total electricity demand is projected to be 4,333 billion kWh by 2014. Table 7-4 shows current electricity generation and projected levels in 2012, 2014, and 2020 using EPA IPM modeling. The increasing growth of coal-fired generation and decline of generation from units using natural gas and oil results primarily due to the relative prices of the fuels in EPA's IPM forecast and the energy efficiency of the generation technologies in producing electricity. IPM in essence is using EIA's *Annual Energy Outlook for 2008*'s electric demand forecast for the US and estimating the mix of fossil generation based on its on fuel price projections and assumptions on performance and costs of electric generation technologies. The base case assumption that CAIR is not in effect does have some modest influence on the fossil generation mix in this forecast, because CAIR had increased the costs of coal-fired generation relatively more than it had increased the costs to generation units that burned oil or natural gas.[55]

Table 7-4. 2008 Electricity Net Generation and EPA Base Case Projections for 2012, 2014 and 2020 for the Contiguous 48 States (Billion kWh)

	2008	2012	2014	2020
Coal	1,967	2,232	2,418	2,629
Oil	21	24	15	14
Natural Gas	798	743	632	570
Other	1,171	1,266	1,269	1,332
Total	**3,957**	**4,266**	**4,333**	**4,546**

Source: 2008 data from EIA Electric Power Annual 2008, Table 1.1 (adjusted to represent the Contiguous 48 States for consistency with projections, which are from the Integrated Planning Model run by EPA, 2010).

While EPA is proposing one particular remedy for the Transport Rule, State Budgets/Limited Trading, it is also requesting comment on the alternatives of State Budgets/Intrastate Trading and Direct Control. These two alternatives represent the range of myriad alternatives considered by EPA for this rule. While their features and impacts inevitably differ in some respects, the alternatives are distinct ways to try to achieve the same emissions reductions as the proposed remedy.

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[55] For the same time period, the Energy Information Administration's *Annual Energy Outlook for 2010* shows a small amount of growth in coal-fired electric generation, no growth in electric generation using oil, and a similar trend in generation from natural gas, which declines through 2014 and increases between 2014 and 2020.

The main difference between the three remedies lies in the kinds of flexibility they provide for compliance. State Budgets/Limited Trading allows sources to trade within state lines and, as long as state emissions remain within the budgets plus variability limits, across state lines as well. State Budgets/Intrastate Trading caps each state's emissions at its budget without variability and only allows trading within (and not between) states. Under Direct Control, each EGU must meet an emission rate limit, and a state's emissions must remain within its budget plus variability. Even though 2012 rate limits are designed to be met without installing unplanned controls, individual rates still offer less flexibility than trading. Details on the derivation of unit-specific rate limits for Direct Control can be found in the TSD, "State Budgets, Unit Allocations, and Unit Emissions Rates."

Even under Direct Control, the proposed rule provides some flexibility. Notably, in addition to allowing total emissions up to the budget plus variability limit, units in the same state within the same company are allowed to average their rates together. As described in section 7.11 below, intra-state, intra-company averaging is not modeled in IPM, while the variability provision is. Further details on Direct Control, State Budgets/Intrastate Trading, and the preferred State Budgets/Limited Trading can be found in Section V. of the Transport Rule preamble.

As noted above, IPM has been used for evaluating the economic and emission impacts of environmental policies for over a decade. The economic modeling presented in this chapter has been developed for specific analyses of the power sector. Thus, the model has been designed to reflect the industry as accurately as possible. As a result, EPA has used discount rates in IPM that are appropriate for the various types of investments and other costs that the power sector incurs. The primary real discount rate is 5.5 % for pollution control retrofits, fuel costs, allowance prices, and most generation technologies' capital and operating expenses.[56] The discount rates used in IPM differ from discount rates used in other RIA analyses done for the Transport Rule, particularly the discount rates used in the benefits and macroeconomic analyses that are assumed to be social discount rates. (See Chapters 5 and 8 where social discount rates of 3 % and 7 % are used.) EPA uses the best available information from utilities, financial institutions, debt rating agencies, and government

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[56] For renewable and most natural gas technologies a 6.1 % discount rate is used.

statistics as the basis for the discount rates used for power sector modeling in IPM. These discount rates have undergone review by the power sector and the Energy Information Administration.

More detail on IPM can be found in the model documentation, which provides additional information on the assumptions discussed here as well as all other assumptions and inputs to the model (http://www.epa.gov/airmarkets/progsregs/epa-ipm). Updates specific to Transport Rule modeling are also in the TSD “Updates to EPA Base Case v3.02 EISA Using the Integrated Planning Model.”

7.2 Projected SO_2 and NO_x Emissions and Reductions

Both the proposed Transport Rule remedy and the alternative remedies achieve substantial emissions reductions. Under each of these remedies, EPA projects annual SO_2 emission reductions of greater than 60 percent and annual NO_x emissions reductions of greater than 33 percent in the respective remedy regions by 2014 relative to the base case. Additionally, EPA projects ozone-season NO_x reductions of greater than 15 percent in the Transport Rule region (see Table 7-5). On the other hand, differences among the remedies the incentive for emissions banking can lead to slight differences in the timing of SO_2 reductions. The following section describes the emission results for each remedy.

In Figure 7-3 below, the results of EPA modeling of State Budgets/Limited Trading, the Transport Rule proposed remedy, show that substantial SO_2 emissions reductions occur in the Midwest and Mid-Atlantic regions of the country. Because banking of allowances is allowed to encourage early reductions, 2012 SO_2 reductions are greater overall than state budgets alone would require in that year. For many coal-fired electric generation units throughout the region it is economically advantageous to make extra emissions reductions in 2012 through fuel switching to have allowances to later use or sell in 2014 and beyond. This is when the Transport Rule becomes more stringent and when electric generators will also need to meet higher electric demand. Because of the banking provisions, the relative economics of making pollution reductions below the emissions cap levels in 2012 versus making emissions reductions later favor doing more in 2012. Annual NO_x emissions reductions occur across the Transport Rule region (see Figure 7-4), and with the Transport Rule, ozone-season NO_x emissions reductions are lower than they would have been with the NO_x SIP Call (base case) (see Figure 7-5).

Table 7-5. Projected Emissions of SO_2 and NO_x with the Base Case[a] (No Further Controls) and with Transport Rule Options (Million Tons)

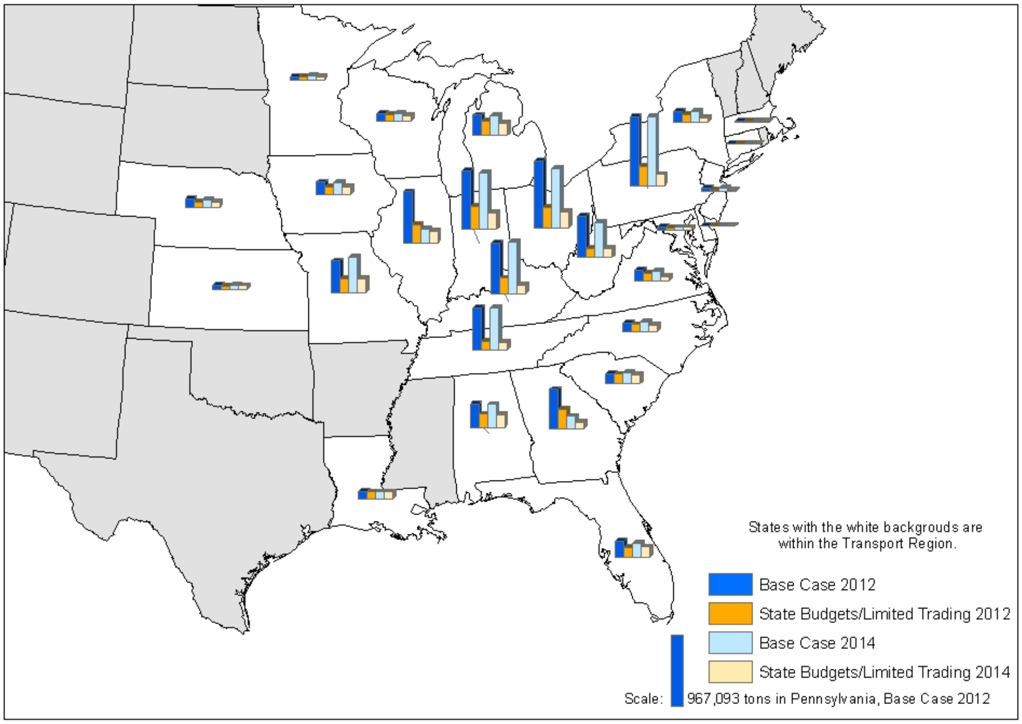
		Base Case		**State Budgets/Limited Trading**				**State Budgets/Intrastate Trading**				**Direct Control**			
		2012	**2014**	**2012**		**2014**		**2012**		**2014**		**2012**		**2014**	
	Coverage	2012 Base Case	2014 Base Case	Transport Rule	Emission Reductions	Transport Rule	Emission Reductions	Transport Rule	Emission Reductions	Transport Rule	Emission Reductions	Transport Rule	Emission Reductions	Transport Rule	Emission Reductions
SO_2 (annual)	Contiguous 48 States	9.5	8.5	4.8	4.7	4.1	4.4	4.6	4.9	4.2	4.3	5.1	4.4	4.1	4.4
	Transport Rule States	8.5	7.4	3.6	4.9	2.8	4.6	3.4	5.1	2.9	4.5	3.9	4.6	2.8	4.6
NO_x (annual)	Contiguous 48 States	3.0	3.0	2.2	0.8	2.2	0.8	2.2	0.8	2.2	0.8	2.2	0.8	2.2	0.9
	Transport Rule States	2.2	2.1	1.4	0.7	1.4	0.7	1.4	0.7	1.4	0.8	1.4	0.7	1.4	0.8
NO_x (summer)	Contiguous 48 States	1.1	1.1	1.0	0.1	1.0	0.1	1.0	0.1	1.0	0.2	1.0	0.1	1.0	0.2
	Transport Rule States	0.7	0.7	0.6	0.1	0.6	0.1	0.6	0.1	0.6	0.1	0.6	0.1	0.6	0.1

Note: Numbers may not add due to rounding. The emissions data presented here are EPA modeling results and the Transport Rule region includes States modeled for the annual SO_2 and NO_x requirements. “Summer” is from May 1–September 30, which is the ozone season.

[a] Base case includes Title IV Acid Rain Program, NO_x SIP Call, and State rules through February 3, 2009.

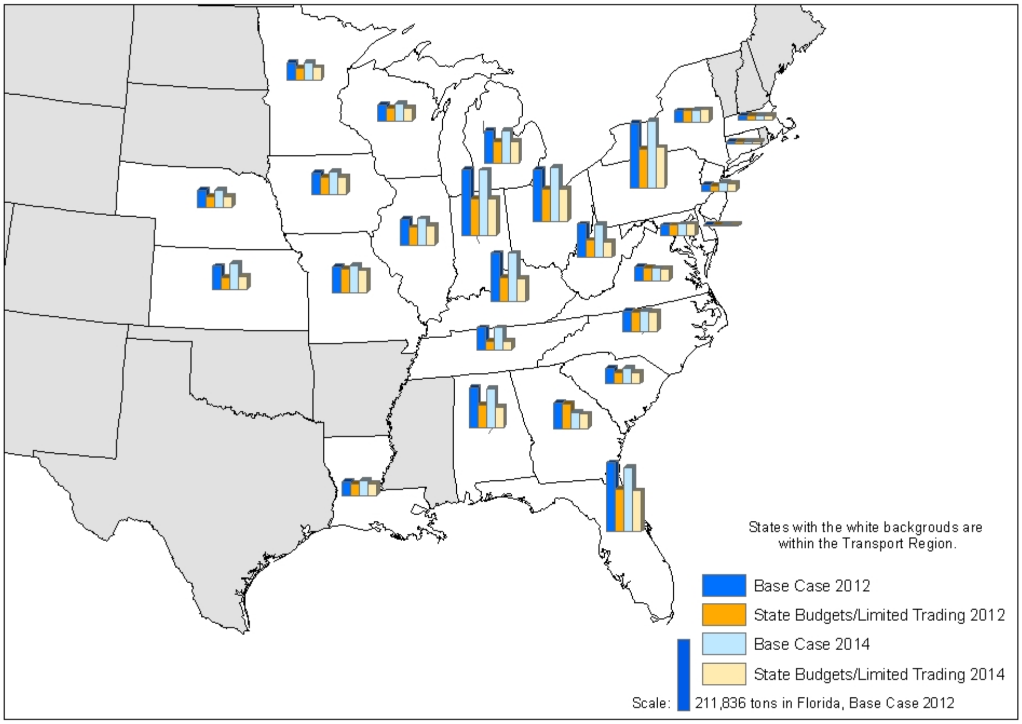
Source: Integrated Planning Model run by EPA, 2010.

Figure 7-3. SO_2 Emissions from the Power Sector in 2012 and 2014 with and without the Transport Rule (State Budgets/Limited Trading)



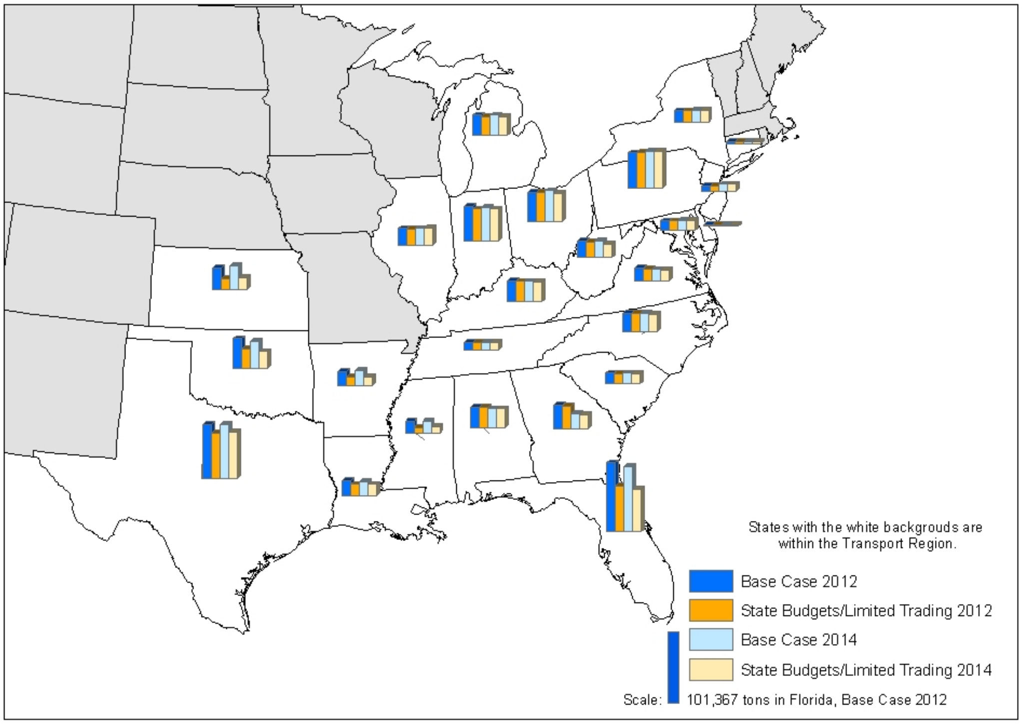
Source: EPA, IPM, 2010.

Figure 7-4. Annual NO_x Emissions from the Power Sector in 2012 and 2014 with and without the Transport Rule (State Budgets/Limited Trading)



Source: EPA, IPM, 2010.

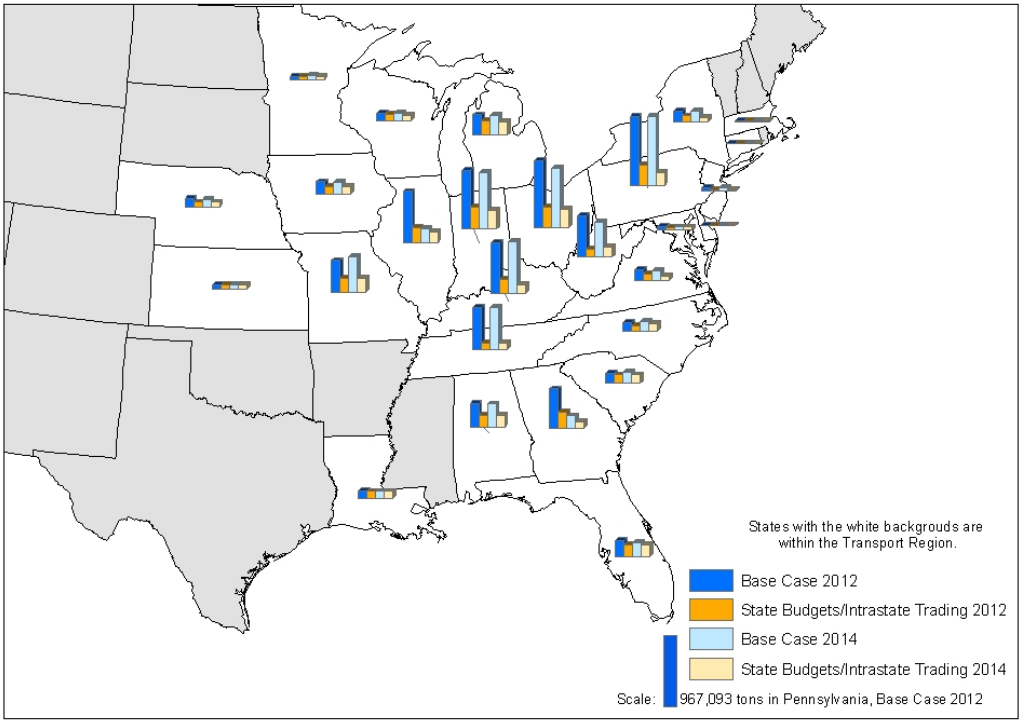
Figure 7-5. Ozone-season NO_x Emissions from the Power Sector in 2012 and 2014 with and without State Budgets/Limited Trading



Source: EPA, IPM, 2010.

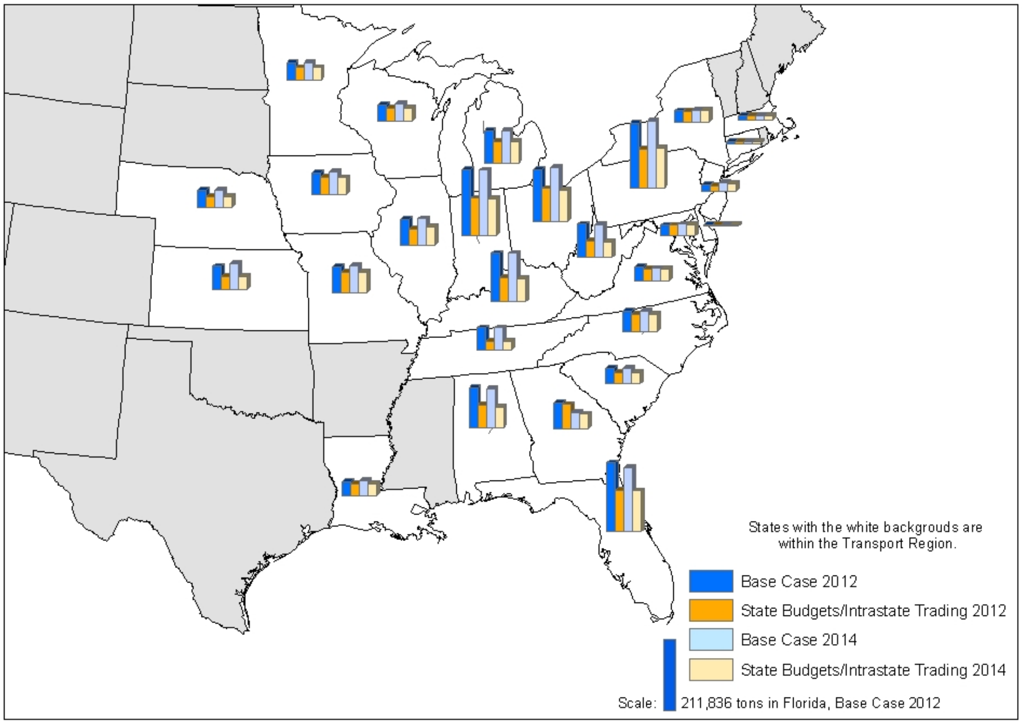
Emissions for State Budgets/Intrastate Trading are shown in Table 7-5 above and in Figures 7-6, 7-7, and 7-8 below. Compared to the proposed remedy, State Budgets/Intrastate Trading achieves slightly more SO_2 reduction in 2012 (and slightly less in 2014), as Table 7-5 shows above. For this remedy, each state's emissions were restricted to the state budget without variability. Without the opportunity for even limited trading of allowances across state borders, more banking was projected in some states than in the proposed remedy. In other states, more immediate emissions reductions (relative to the base case) are projected; because sources cannot purchase allowances from outside their own state, state budgets must be met exactly. Both of these factors lead to slightly greater SO_2 reductions in 2012 than in the State Budgets/Limited Trading.

Figure 7-6. SO_2 Emissions from the Power Sector in 2012 and 2014 with and without State Budgets/Intrastate Trading



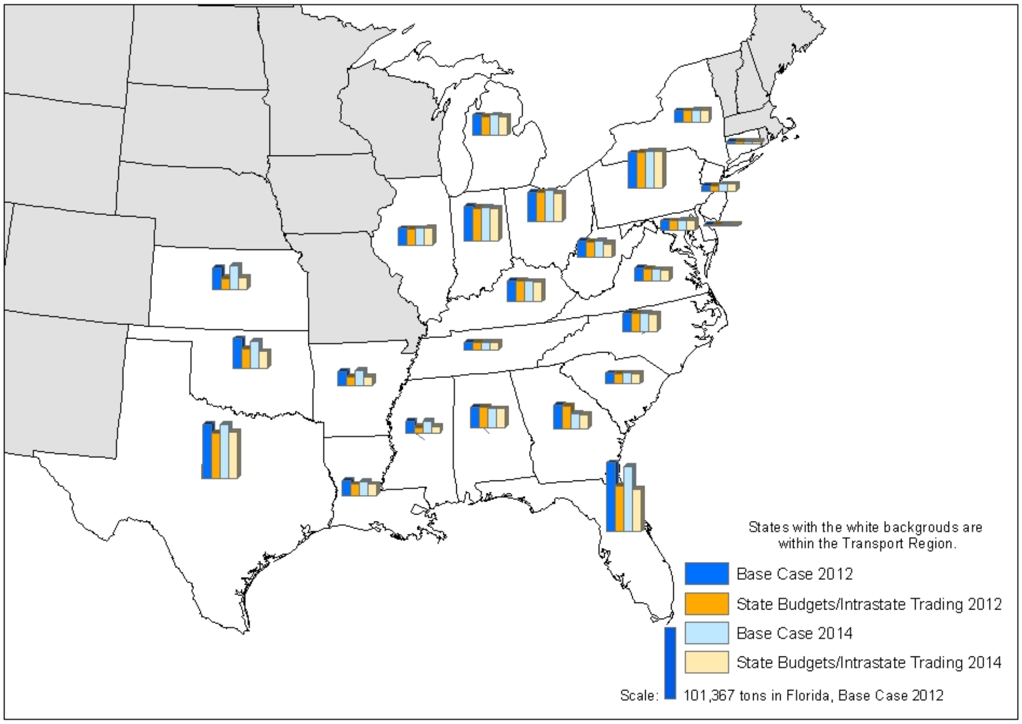
Source: EPA, IPM, 2010.

Figure 7-7. Annual NO_x Emissions from the Power Sector in 2012 and 2014 with and without State Budgets/Intrastate Trading



Source: EPA, IPM, 2010.

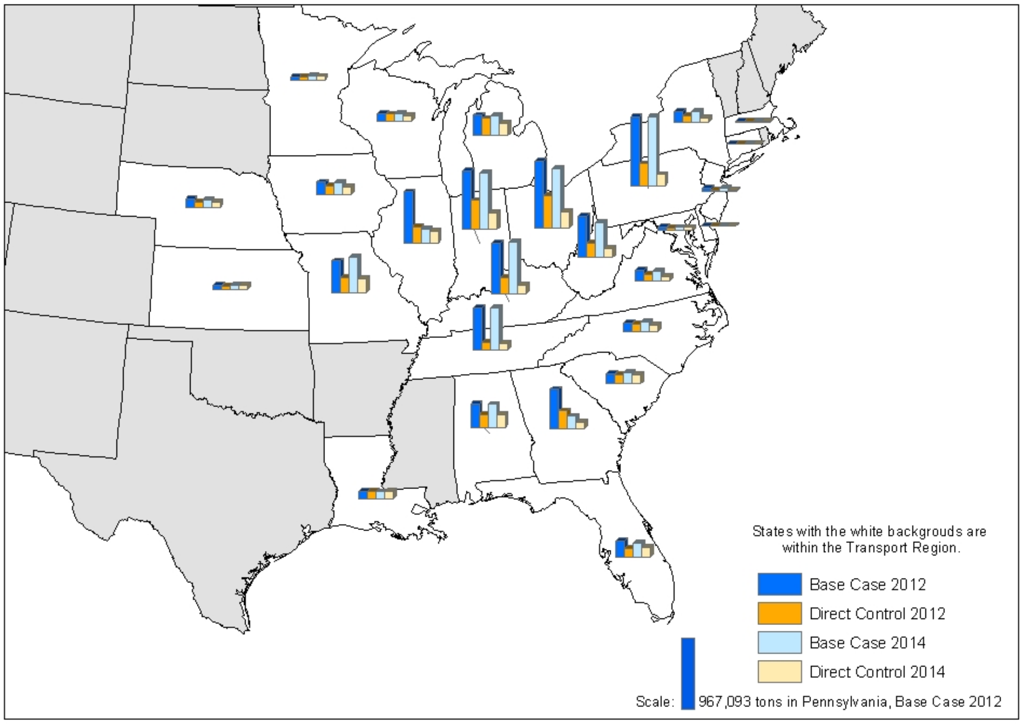
Figure 7-8. Ozone-season NO$_x$ Emissions from the Power Sector in 2012 and 2014 with and without State Budgets/Intrastate Trading



Source: EPA, IPM, 2010.

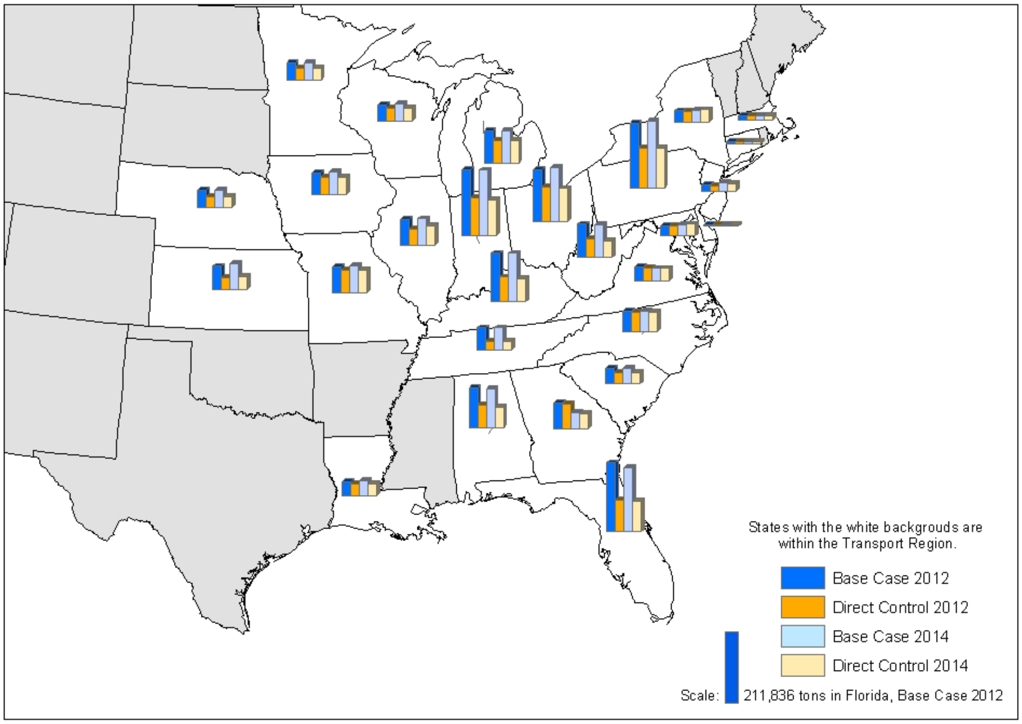
Emissions for Direct Control are shown in Table 7-5 above and in Figures 7-9, 7-10, and 7-11 below. Compared to State Budgets/Limited Trading, Direct Control results in less SO_2 reduction in 2012 (see Table 7-5). Because it does not allow banking for early reductions, the Direct Control alternative does not result in reductions below state budgets in 2012. The absence of banking does not lead to lower emissions relative to the other proposed remedies in 2014 because total emissions in each state can still be as high as the state's budget plus variability limit with the Direct Control option.

Figure 7-9. SO_2 Emissions from the Power Sector in 2012 and 2014 with and without Direct Control



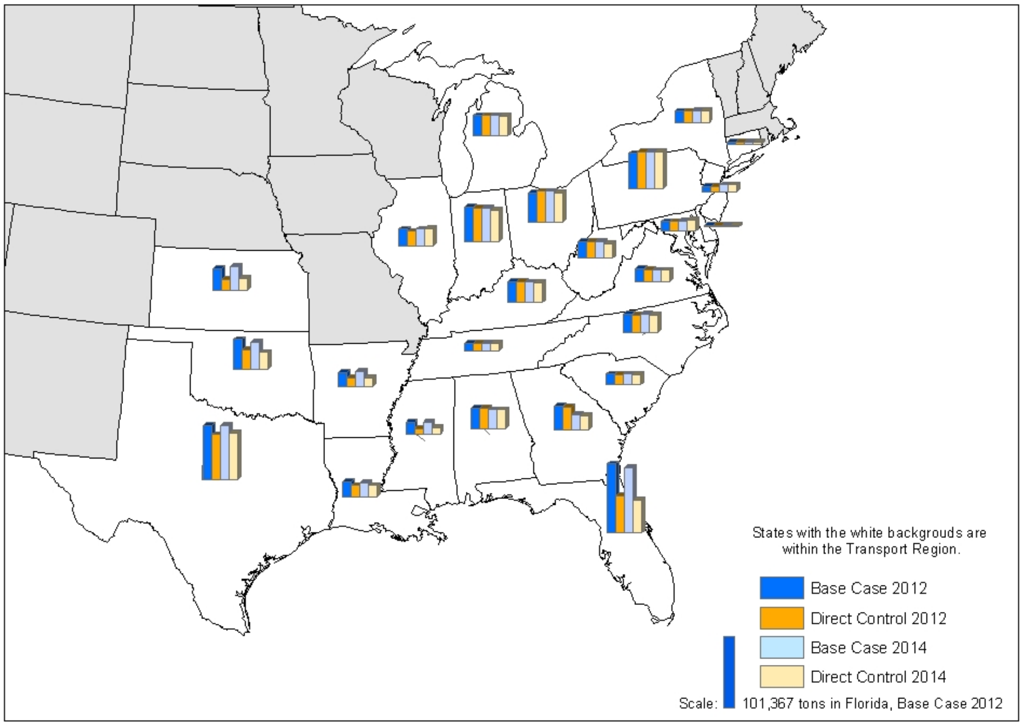
Source: EPA, IPM, 2010.

Figure 7-10. Annual NO_x Emissions from the Power Sector in 2012 and 2014 with and without Direct Control



Source: EPA, IPM, 2010.

Figure 7-11. Ozone-season NO_x Emissions from the Power Sector in 2012 and 2014 with and without Direct Control



Source: EPA, IPM, 2010.

7.3 Overview of Costs and Other Impacts

As shown above in Figures 7-1 and 7-2, the Transport Rule directly affects 27 states and the District of Columbia in controlling pollution related to fine particles. For ozone, it also affects a distinct but overlapping group of 25 states and the District of Columbia. The states in one or both of these regions constitute most of the fossil-fuel-fired generation and capacity in the contiguous US, especially coal-fired (see Tables 7-6 and 7-7 below).

Table 7-6. Fossil-fuel Generation Nationwide and in the Transport Region

	2008 Generation (Thousand GWh)		
	Contiguous 48 States	Transport Rule Fine Particle Area	Transport Rule Ozone Area
Coal-fired	1,967	1,477	1,487
Gas-fired	798	331	518
Oil-fired	21	20	18
Sum	2,786	1,828	2,022

Source: EIA Electric Power Monthly with data for December 2009, Tables 1.7.B, 1.8.B, 1.10.B.

Table 7-7. Fossil-fuel Capacity Nationwide and in the Transport Region

	2008 Capacity (GW)		
	Contiguous 48 States	Transport Rule Fine Particle Area	Transport Rule Ozone Area
Pulverized Coal	309	240	237
Combined Cycle	190	95	131
Other Oil/Gas	249	160	191
Sum	748	496	559

Source: EPA's NEEDS v3.02ARRA.

While most impacts of the Transport Rule affect the covered states themselves, national impacts are important. Because the electric grid is connected irrespective of state boundaries, effects on electrical generation in one state have spillover effects in other states. Likewise, because the Transport Rule states have the vast majority of coal-fired generation, changes in their coal consumption and demand affect coal prices nationwide. In some cases, such as retail electricity prices and the operation of pollution controls, nationwide information would not be as relevant as regional totals. But for most of the following sections, nationwide projections provide a more complete picture of the Transport Rule's impacts.

7.4 Projected Compliance Costs

The power industry's "compliance costs" are the changes in electric power generation costs in the base case and alternative pollution control approaches that are examined in this chapter. In simple terms, these costs are the resource costs of what the power industry will directly expend to comply with EPA's requirements. This is not the "social cost" of the control approaches, which is separately explained and estimated in EMPAX modeling in Chapter 8 that follows.

EPA projects that the annual incremental compliance costs of the Transport Rule proposed remedy (State Budgets/Limited Trading) are $3.7 billion in 2012 and $2.8 billion in 2014 (see Table 7-8 below). Another measure of this impact is the change in electricity prices (discussed in section 7.9). Costs generally are higher in 2012 than in 2014 because of reduced compliance flexibility in 2012, which is too soon for sources to retrofit new FGD and SCR that were not already planned.

Table 7-8. Annualized Compliance Cost of the Transport Rule

	State Budgets/Limited Trading		State Budgets/Intrastate Trading		Direct Control	
Annualized Cost (billions of 2006$)	**2012**	**2014**	**2012**	**2014**	**2012**	**2014**
	$3.7	$2.8	$4.2	$2.7	$4.3	$3.4

Note: Numbers rounded to the nearest hundred million for annualized cost.
Source: Integrated Planning Model run by EPA.

Though based on the same state budgets as State Budgets/Limited Trading, State Budgets/Intrastate Trading costs approximately $0.5 billion dollars more in 2012, as Table 7-8 shows above. As mentioned above in the context of emissions reductions, more banking is projected in some states and more immediate emissions reductions (relative to the base case) in others, both because the flexibility of trading between states is not available. These factors drive 2012 costs higher than those of State Budgets/Limited Trading.

The allowance prices of the two alternatives that allow emissions trading are provided in Appendix E.

The Direct Control alternative remedy costs $0.6 billion more than the proposed remedy in 2012 and 2014. Based on source-specific emissions rates, it provides less flexibility for compliance than either of the trading remedies. For example, demand for the lowest-sulfur grades of coal is higher in the Direct Control case even relative to the next-lowest grades. In this case, many unscrubbed units have rates that require either using the lowest-sulfur grade of coal or installing new controls, even though slightly higher-sulfur coal would likely be economical for some if trading were an option.

7.5 Projected Approaches to Emissions Reductions

Emission reductions of NO_x and SO_2 for the alternative pollution control remedies that EPA examined are achieved through a combination of compliance options by electric generation units using fossil fuels in the Transport Region. These actions include full operation of existing controls that are in place that were noneconomic to operate under the base case, additional pollution control installations, coal switching (including blending of coals), and generation shifts towards more efficient electricity producing units and lower-emitting generation technologies (e.g., some reduction of coal-fired generation with an increase of generation from natural gas). Notably, only coal-fired generation units actually install and operate added pollution controls in response to control alternatives. In 2012 and 2014, there are similar, but somewhat different sets of actions that EPA's modeling predicts will occur.

In 2012 in the State Budgets/Limited Trading option (preferred approach), a small shift from coal-fired and oil generation to greater use of natural gas lowers emissions a small amount (see Table 7-12). NO_x emissions reductions largely occur when on coal-fired units, selective catalytic reduction (SCR) controls that were only required to operate in the ozone season in the base case now operate throughout the year (see Table 7-11). Additionally, in states that were not in the original CAIR program but are covered in summer ozone season program of this rule there is a modest amount of NO_x reduction stemming from low- NO_x burners that are installed on coal-fired units. SO_2 emissions reductions result from the relatively high allowances values for the SO_2 programs, which makes it economic for all FGDs (scrubbers) that EPA estimates are in place by 2012 to operate under the preferred control option. High allowance values also lead to considerable coal switching to lower sulfur coals as the least cost compliance approach for many coal-fired units without FGDs.

Examining the changes in use of coals with differing sulfur contents as laid out in Table 7-9 below is helpful in considering what EPA estimates will occur from the base case to the preferred approach in the rule. For coal-fired generation units that do not operate FGDs in the base case, about 30 % of power generated by these units is from high sulfur coals; 10 % of the power generated is from high-medium sulfur coals; and 15 % of the power generated is from low-medium sulfur coals. The percentages for power generation from cleaner coals are close to 1 %, 15 %, and 30 % for low sulfur bituminous, low sulfur subbituminous, and very low sulfur subbituminous respectively. Under the preferred approach, for the units that do not have FGDs, power generation from the units using high, high-medium, and low-medium sulfur coals changes to about 2 %, 5 %, and 30 %, respectively. For the cleaner coals, the percentages of power generation from units without FGDs under the preferred approach were about 3 %, less than 1 %, and 60 % for low-sulfur bituminous, low sulfur subbituminous, and very low sulfur subbituminous respectively.

Table 7-9. Coal Sulfur Categories (lbs/mmBTU)

Subbituminous	High sulfur	> 1.2
	Low sulfur	0.7 to 1.2
	Very low sulfur	< 0.7
Bituminous	High sulfur	> 2
	High-medium sulfur	1.4 to 2
	Low-medium sulfur	0.9 to 1.4
	Low sulfur	< 0.9
Lignite	All grades	0.6 to 4

Table 7-10 shows total coal use among both scrubbed and unscrubbed EGUs in the states subject to the proposed SO_2 programs. The preferred remedy approach (State Budgets/Limited Trading) is associated with only a slight reduction (1% in 2014) in total coal use among these units compared to the base case. More importantly, the table reinforces that the preferred approach drives increased overall use of cleaner bituminous and subbituminous coals, especially very low sulfur subbituminous. This trend appears even when, as in this table, coal use of scrubbed units and very small units are included.

Table 7-10. Coal Use by Sulfur Category in the PM$_{2.5}$ Transport Region for the Base Case and Preferred Approach* (thousand short tons)

		Lignite	Subbituminous			Bituminous				Total
			High sulfur	Low sulfur	Very low sulfur	High sulfur	High-medium sulfur	Low-medium sulfur	Low sulfur	
2012	Base case	3,911	3,405	125,460	176,479	285,511	106,575	72,952	4,507	778,800
	Preferred approach	3,911	2,143	63,858	246,828	258,847	101,383	89,826	10,518	777,315
2014	Base case	3,883	6,664	110,357	193,885	331,913	69,060	85,248	5,143	806,153
	Preferred approach	3,883	4,823	64,434	242,821	294,305	84,519	91,672	11,558	798,014

*These coal usage results are for the 28 states covered by the rule in the trading program to reduce SO_2 emissions.

In 2014, the preferred approach to the Transport Rule is projected to result in the operation of an additional 40 GW of flue gas desulfurization (scrubbers) for SO_2 control on existing coal-fired generation capacity and the year-round operation of an additional 51 GW of selective catalytic reduction technology (SCR) for NO_x control on existing coal-fired generation capacity by 2014 (see Table 7-11). This accounts for the vast majority of the NO_x reduction. A small number of coal-fired units also install selective non-catalytic reduction technology (SNCR) for NO_x control under the preferred approach to the Transport Rule.

For SO_2, the reductions from the added FGDs are also supplemented by the continued and expanded use of relatively cleaner coals at uncontrolled units. Notably, many of the FGDs that are coming into operation are built due to control requirements other than the Transport Rule and only 14 GWs of capacity with newly constructed FGDs are resulting from the preferred approach.

Table 7-11. Advanced Pollution Controls on Coal-fired Generation by Technology with the Base Case (No Further Controls) and with Transport Rule Options (GW)

Control Technology	Base Case						State Budgets/Limited Trading				State Budgets/Intrastate Trading				Direct Control			
	2012				2014		2012		2014		2012		2014		2012		2014	
	Total Capacity (GW)		Capacity Controlled Year-round (GW)															
	Contiguous 48 States	Transport Region	Contiguous 48 States	Transport Region	Contiguous 48 States	Transport Region	Contiguous 48 States	Transport Region	Contiguous 48 States	Transport Region	Contiguous 48 States	Transport Region	Contiguous 48 States	Transport Region	Contiguous 48 States	Transport Region	Contiguous 48 States	Transport Region
FGD	194	146	162	117	202	139	193	147	241	179	191	145	238	176	191	145	247	185
SCR	140	136	88	83	117	106	133	128	167	157	134	129	168	157	137	132	171	160

Note: For FGD, the "Transport Region" comprises 28 states and District of Columbia as shown in Figure 7-1. For SCR, the "Transport Region" includes both these states and those under the ozone-season NO_x program as shown in Figure 7-2 above. All totals refer to coal-fired generating capacity.
Source: Parsed files from the Integrated Planning Model run by EPA, 2010.

7.6 Projected Generation Mix

Table 7-12 and Figure 7-12 show the generation mix with the Transport Rule. Coal-fired generation and natural-gas-fired generation are projected to remain relatively unchanged because of the phased-in nature of the Transport Rule, which allows industry the appropriate amount of time to install the necessary pollution controls. Both the base case and all three remedies show shifts away from oil and natural gas generation and toward increased coal generation between 2012 and 2014.

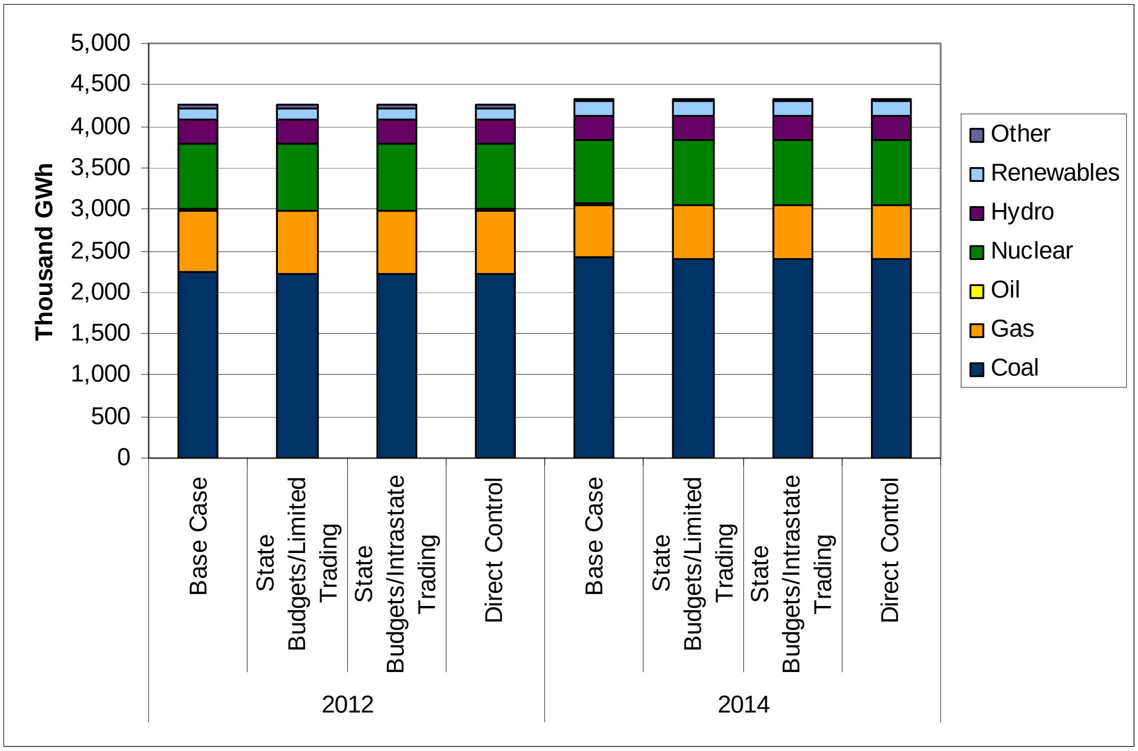
Table 7-12. Generation Mix with the Base Case (No Further Controls) and with Transport Rule Options (Thousand GWh)

	2008	2012							2014						
	Historical	Base Case	State Budgets/Limited Trading	Change from Base	State Budgets/Intrastate Trading	Change from Base	Direct Control	Change from Base	Base Case	State Budgets/Limited Trading	Change from Base	State Budgets/Intrastate Trading	Change from Base	Direct Control	Change from Base
Coal	1,967	2,232	2,223	-0.4%	2,211	-1%	2,226	-0.3%	2,418	2,400	-1%	2,396	-1%	2,407	-0.5%
Oil	21	24	19	-20%	16	-32%	15	-36%	15	11	-30%	10	-37%	11	-30%
Natural Gas	798	743	751	1%	761	2%	756	2%	632	641	1%	643	2%	639	1%
Other	1,171	1,266	1,273	1%	1,277	1%	1,269	0.2%	1,269	1,282	1%	1,284	1%	1,276	1%
Total	**3,957**	**4,266**	**4,266**	**0.0%**	**4,265**	**0.0%**	**4,266**	**0.0%**	**4,334**	**4,333**	**0.0%**	**4,333**	**0.0%**	**4,333**	**0.0%**

Note: Numbers may not add due to rounding.

Source: 2008 data from EIA Electric Power Monthly with data for December 2009, Tables 1.6.B 1.7.B, 1.8.B, 1.10.B; 2012 and 2014 projections are from the Integrated Planning Model run by EPA, 2010.

Figure 7-12. Generation Mix with the Base Case (No Further Controls) and with Transport Rule Options



Source: 2008 data derived from EIA U.S. Coal Supply and Demand: 2008 Review, Table 1; 2012 and 2014 projections from the Integrated Planning Model run by EPA, 2010.

Relative to the base case, about 1.2 GW of coal-fired capacity is projected to be uneconomic to maintain (less than 1 percent of all coal-fired capacity in the Transport Rule states) by 2014. Uneconomic units, for the most part, are small and infrequently used generating units that are dispersed throughout the states covered in the Transport Rule. In practice, units projected to be uneconomic to maintain may be "mothballed," retired, or kept in service to ensure transmission reliability in certain parts of the grid. EPA modeling is unable to distinguish between these potential outcomes. IPM can only predict that specific generating units are uneconomic to maintain, based on their fuel, operating and fixed costs, and whether they are needed to meet both demand and reliability reserve requirements.

Though similar to the proposed remedy, the alternative remedies result in slightly different projections of uneconomic units. State Budgets/Intrastate Trading results in 1.6

GW of such capacity by 2014, 0.4 GW more than in State Budgets/Limited Trading. In contrast, Direct Control yields only 0.5 GW in 2014 (0.7 GW less than the proposed remedy).

7.7 Projected Capacity Additions

In addition, EPA projects that most future growth in electric demand will be met with a combination of new natural gas- and coal-fired capacity (see Table 7-13). This occurs in the base case, under the proposed Transport Rule, and under both alternative remedies.

Table 7-13. Total Coal-fired, Natural Gas-fired, Oil-fired and Renewable Generation Capacity by 2025 (GW)

	2008	Base Case	State Budgets/Limited Trading	State Budgets/Intrastate Trading	Direct Control
Pulverized Coal	309	387	386	386	387
Combined Cycle Turbines	190	229	228	228	228
Other Oil/Gas	249	240	240	240	241
Renewables	30	49	49	49	49

Source: 2008 data from EPA's NEEDS v3.02ARRA. Projections from Integrated Planning Model run by EPA.
Note: “Renewables” include biomass, geothermal, solar, and wind electric generation capacity.

7.8 Projected Coal Production for the Electric Power Sector

Coal production for electricity generation is expected to increase relative to current levels, with or without the Transport Rule (see Table 7-14). The reductions in emissions from the power sector will be met through the installation and operation of pollution controls for SO_2 and NO_x removal. Many of these pollution controls can achieve SO_2 removal rates of 95 percent or greater, which allows industry to rely more heavily on local bituminous coal in the eastern and central parts of the country that has a higher sulfur content and is less expensive to transport than western subbituminous coal.

Table 7-14. Coal Production for the Electric Power Sector with the Base Case (No Further Controls) and with Transport Rule Options (Million Tons)

	Historical	Base Case		State Budgets/Limited Trading		State Budgets/Intrastate Trading		Direct Control	
Supply Area	2008	2012	2014	2012	2014	2012	2014	2012	2014
Appalachia	362	304	317	307	329	309	328	305	323
Interior	136	163	185	144	158	128	161	138	160
West	588	601	654	613	661	621	659	622	666
Total	1086	1068	1156	1064	1148	1058	1148	1065	1149

Source: 2008 data derived from EIA data. All projections from Integrated Planning Model run by EPA. http://www.eia.doe.gov/cneaf/coal/page/special/tbl1.html

7.9 Projected Retail Electricity Prices

Retail electricity prices for the Transport Rule region are projected to increase a small amount with the Transport Rule (see Table 7-15).

Table 7-15. Projected Regional Retail Electricity Prices with the Base Case (No Further Controls) and with Transport Rule Options (2006 Mills/kWh)

Year	Base Case	State Budgets/Limited Trading	Change from Base	State Budgets/Intrastate Trading	Percent Change	Direct Control	Change from Base
2012	86.2	88.3	2.5%	88.6	2.9%	88.0	2.1%
2014	85.6	86.9	1.5%	87.1	1.8%	86.7	1.4%

Source: EPA's Retail Electricity Price Model.

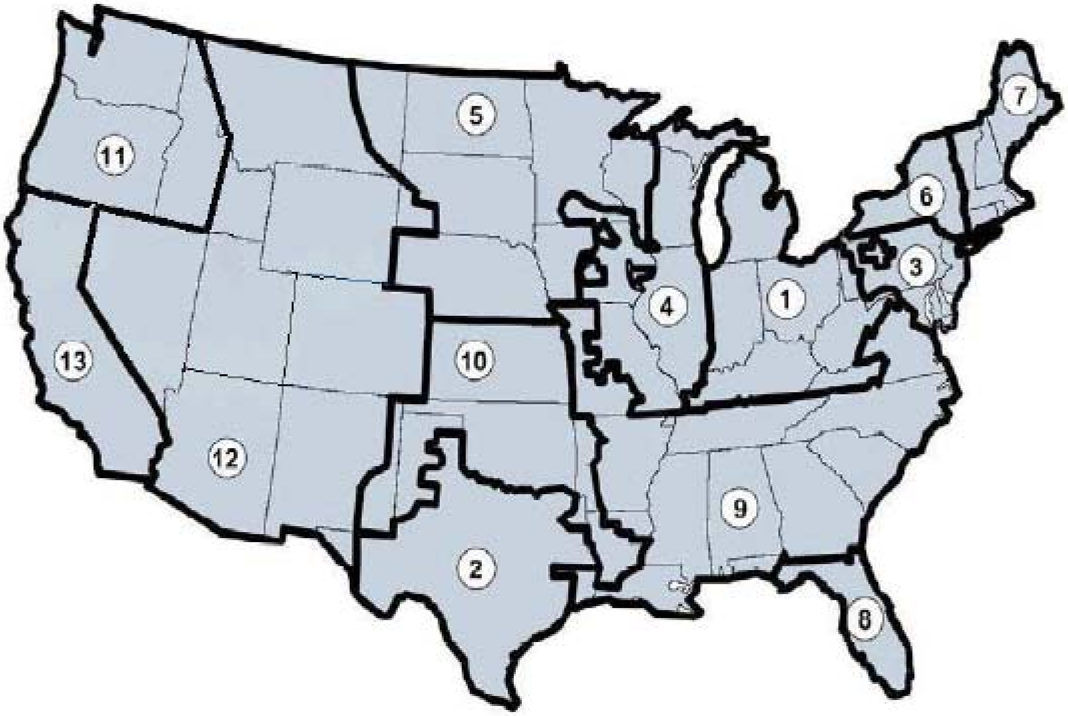
Regional retail electricity prices are projected to be 1 to 3 percent higher with the Transport Rule. Retail electricity prices by NERC region are provided in Table 7-16 (see Figure 7-13). These results show increases in retail prices for the NERC regions in the eastern part of the country. By 2014, retail electricity prices in the regions directly affected by the Transport Rule are projected to be roughly 1.5 percent higher with the Transport Rule (Table 7-15).

Table 7-16. Retail Electricity Prices by NERC Region with the Base Case (No Further Controls) and with Transport Rule Options (2006 Mills/kWh)

		Historical	Base Case		State Budgets/Limited Trading		State Budgets/Limited Trading Change from Base		State Budgets/ Intrastate Trading		State Budgets/ Intrastate Trading from Base		Direct Control		Direct Control Change from Base	
Power Region	**Primary States included**	**2000**	**2012**	**2014**	**2012**	**2014**	**2012**	**2014**	**2012**	**2014**	**2012**	**2014**	**2012**	**2014**	**2012**	**2014**
ECAR (1)	OH, MI, IN, KY, WV, PA	68	64	66	67	69	4%	5%	69	69	7%	5%	66	68	3%	3%
ERCOT (2)	TX	77	109	102	111	103	1%	1%	111	103	1%	1%	111	103	1%	1%
MAAC (3)	PA, NJ, MD, DC, DE	96	105	103	111	103	6%	0%	108	105	3%	2%	109	104	4%	1%
MAIN (4)	IL, MO, WI	73	66	70	69	73	4%	4%	70	73	6%	4%	68	72	3%	2%
MAPP (5)	MN, IA, SD, ND, NE	68	72	74	74	75	2%	1%	74	75	2%	1%	74	75	2%	2%
NY (6)	NY	124	143	143	144	144	1%	1%	145	144	1%	1%	144	144	1%	1%
NE (7)	VT, NH, ME, MA, CT, RI	107	126	123	127	124	1%	0%	127	124	1%	1%	127	124	1%	0%
FRCC (8)	FL	81	105	104	105	104	1%	0%	106	104	1%	0%	106	104	1%	0%
STV (9)	VA, NC, SC, GA, AL, MS, TN, AR, LA	71	72	73	74	74	2%	1%	74	74	2%	2%	74	74	2%	1%
SPP (10)	KS, OK, MO	71	90	84	93	85	4%	1%	93	85	3%	1%	93	85	3%	1%
Regionwide		79	86	86	88	87	3%	2%	89	87	3%	2%	88	87	2%	1%

Source: EPA's Retail Electricity Price Model based on the Integrated Planning Model run by EPA, 2010. 2000 prices are from EIA Annual Energy Outlook 2003 (adjusted to 2006 dollars).

Figure 7-13. NERC Power Regions



7.10 Projected Fuel Price Impacts

The impacts of the Transport Rule on coal prices and natural gas prices before shipment are shown below in Tables 7-17 and 7-18. The proposed remedy and two alternative remedies have the same effect on natural gas prices, but somewhat different effects on coal prices, reflecting differing effects of the remedies on mix of coal types used based on their sulfur content. Overall, average coal price changes are related to increased demand for a wide variety of coals, with the dominant factor being increased use of lower-sulfur coals. For example, under Direct Control, complying with unit-specific emission rates drives many uncontrolled units to demand only the lowest-sulfur grade of coal available. Conversely, in the proposed remedy, the incentive of the SO_2 allowance markets not only influences the relative demand for every coal grade (allowing cost-effective coal blending) but also affects other decisions such as dispatch, leading to an outcome with a less pronounced effect on any single coal grade. Commensurate with its flexibility relative to the other remedies, State Budgets/Intrastate Trading, in which units in different states have no opportunity to trade, leads to a greater average price increase than in the proposed remedy, but significantly less than under Direct Control.

Table 7-17. Henry Hub Natural Gas Prices and Minemouth Coal Prices with the Base Case (No Further Controls) and with Transport Rule Options (2006 $/MMBtu)

		Base Case		State Budgets/ Limited Trading		Percentage Change from Base		State Budgets/ Intrastate Trading		Percentage Change from Base		Direct Control		Percentage Change from Base	
Fuel	**2008**	**2012**	**2014**	**2012**	**2014**	**2012**	**2014**	**2012**	**2014**	**2012**	**2014**	**2012**	**2014**	**2012**	**2014**
Natural Gas	8.42	6.50	6.07	6.60	6.10	1.5%	0.5%	6.60	6.10	1.5%	0.5%	6.60	6.10	1.5%	0.5%
Coal	1.50	0.94	0.93	1.03	0.98	9.9%	4.7%	1.03	0.98	10.1%	4.8%	1.03	0.98	9.6%	5.3%

Source: Historical data from: Platts Gas Daily; EIA Electric Power Annual 2008, Table 3.5; EIA Annual Coal Report 2008 Table 28; 2012 and 2014 projections from the Integrated Planning Model run by EPA, 2010.

Table 7-18. Average Delivered Natural Gas and Coal Prices with the Base Case (No Further Controls) and with Transport Rule Options (2006 $/MMBtu)

		Base Case		State Budgets/ Limited Trading		Percentage Change from Base		State Budgets/ Intrastate Trading		Percentage Change from Base		Direct Control		Percentage Change from Base	
Fuel	**2008**	**2012**	**2014**	**2012**	**2014**	**2012**	**2014**	**2012**	**2014**	**2012**	**2014**	**2012**	**2014**	**2012**	**2014**
Natural Gas	9.06	6.57	6.09	6.68	6.12	1.7%	0.5%	6.68	6.12	1.7%	0.5%	6.69	6.12	1.8%	0.5%
Coal	1.97	1.59	1.56	1.70	1.62	6.9%	3.8%	1.71	1.62	7.5%	3.8%	1.73	1.63	8.8%	4.5%

Source: EIA Electric Power Annual 2008, Table 3.5; 2012 and 2014 projections from the Integrated Planning Model run by EPA, 2010.

7.11 Key Differences in EPA Model Runs for Transport Rule Modeling

As previously stated, the emissions, cost, air quality, and benefits analyses done for the Transport Rule are from a modeling scenario that requires annual SO_2 and NO_x reductions in 27 states and ozone season NO_x requirements in 25 states (See Figures 7-1 and 7-2). This modeling differs from the proposed Transport Rule because the District of Columbia is included neither in the annual SO_2 and NO_x requirements nor in the ozone season NO_x requirement. Modeled units in the District of Columbia include two small facilities, one of which has only units below 25 MW capacity. EPA believes the addition of emissions limits in the District of Columbia would have little to no effect on the modeling results.

Also, the modeling of the Direct Control alternative remedy did not include an emissions averaging provision, due to its dependence on EGU ownership information. Because this provision provides additional compliance flexibility, EPA believes it would not increase modeled compliance costs for this remedy and would have little to no effect on total emissions. 2012 budgets largely reflect installed controls, planned controls, and fuel switching that would lead to actual emission rates similar to allowable emission rates, so the effect of not modeling firm averaging on estimated cost impacts from IPM in the first phase are likely to not be substantial. In the second phase modeling, additional scrubbers are required based on which EGUs IPM projects have the lowest-cost opportunities for installing incremental scrubbers (which are also likely to be similar to utility projections of lowest-cost controls). Therefore, EPA believes that even if the flexibility provisions were modeled, they would not significantly impact modeled cost in either the first or second phase. Notably, in reality this flexibility could provide cost savings for dealing with unplanned outages, sudden price changes, and other dynamic costs of power-sector operation not modeled in IPM.

7.12 Projected Primary PM and Carbon Dioxide Emissions from Power Plants

IPM does not project primary PM emissions from power plants. These emissions are calculated using IPM outputs and emission factors. Fuel use (heat input) as projected by IPM is multiplied by PM emission factors to determine PM emissions. Primary PM emissions are calculated by adding the filterable PM and condensable PM emissions.

Filterable PM emissions for each unit are based on historical information regarding installed emissions controls and types of fuel burned and ash content. Condensable PM emission factors are based on existing SO_2 and PM controls, plant and fuel type.

This methodology tends to underpredict reductions in filterable PM emissions between the base case and the control case (especially when a unit does not have a high removal efficiency ESP or baghouse) because no changes are assumed in the emission factors even if a unit is projected to install a control such as an FGD, which could lead to a decrease in filterable PM emissions.

For condensable PM emissions, emission factors were changed between the base case and the control case to reflect SO_2 controls projected to be installed in the control case. Although EPA used the best emission factors available for its analysis, these emission factors did not account for the potential changes in condensable PM emissions due to the installation and operation of SCRs. The formation of additional condensable PM (in the form of SO_3 and H_2SO_4) in units with SCRs depends on a number of factors, including coal sulfur content, combustion conditions and characteristics of the catalyst used in the SCR, and is likely to vary widely from unit to unit. SCRs are generally designed and operated so that they minimize increases in condensable PM. This limitation leads to an overprediction of reductions in condensable PM emissions for units with SCRs. For a more complete description of the methodologies used to post-process PM emissions from IPM, see "IPM ORL File Generation Methodology," October 2007.

IPM provides EPA estimates of carbon dioxide (CO_2) emissions for fossil fuel electric generation as a standard output from the model, enabling consideration of the changes in CO_2 emissions that result from pollution control alternatives.[57] EPA found that the State Budget/Limited Trading option (preferred approach) lowered CO_2 emissions from the Base Case in 2014 by 15.3 million metric tons. This occurs due to reductions in coal and oil use and greater use of natural gas and non-fossil sources of electric generation (e.g., biomass cogeneration and nuclear generation, with one fewer unit retiring.)

EPA is not using IPM to project the impacts of this proposed rule on mercury. EPA recently commissioned an information collection request that will soon provide greatly improved power industry mercury emissions estimates that will enable the Agency to better estimate mercury emissions changes from its air emissions control actions. For this reason, the Agency did not estimate mercury changes in this rule and will instead wait for these new data, which will be available in the near future.

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[57] The CO_2 emissions factors for fossil fuels used in IPM are: oil = 173.9 lbs/mmbtu, natural gas = 117 lbs/mmbtu, bituminous coal = 202–205 lbs/mmbtu, subbituminous coal = 208–211 lbs/mmbtu, and lignite = 211–217 lbs/mmbtu.

7.13 Limitations of Analysis

EPA's modeling is based on its best judgment for various input assumptions that are uncertain. Assumptions for future fuel prices and electricity demand growth deserve particular attention because of the importance of these two key model inputs to the power sector. As a general matter, the Agency selects the best available information from available engineering studies of air pollution controls and has set up what it believes is the most reasonable modeling framework for analyzing the cost, emission changes, and other impacts of regulatory controls.

The annualized cost estimates of the private compliance costs that are provided in this analysis are meant to show the increase in production (engineering) costs to the power sector of the Transport Rule proposed remedy and major alternatives. In simple terms, the private compliance costs that are presented are the annual increase in revenues required for the industry to be as well off after the Transport Rule is implemented as before. To estimate these annualized costs, EPA uses a conventional and widely-accepted approach that is commonplace in economic analysis of power sector costs for estimating engineering costs in annual terms. For estimating annualized costs, EPA has applied a capital recovery factor (CRF) multiplier to capital investments and added that to the annual incremental operating expenses. The CRF is derived from estimates of the cost of capital (private discount rate), the amount of insurance coverage required, local property taxes, and the life of capital. The private compliance costs presented earlier are EPA's best estimate of the direct private compliance costs of the Transport Rule.

The annualized cost of the Transport Rule, as quantified here, is EPA's best assessment of the cost of implementing the Transport Rule. These costs are generated from rigorous economic modeling of changes in the power sector due to the Transport Rule. This type of analysis using IPM has undergone peer review and federal courts have upheld regulations covering the power sector that have relied on IPM's cost analysis.

The direct private compliance cost includes, but is not limited to, capital investments in pollution controls, operating expenses of the pollution controls, investments in new generating sources, and additional fuel expenditures. EPA believes that the cost assumptions used for the Transport Rule reflect, as closely as possible, the best information available to the Agency today. The relatively small cost associated with monitoring emissions, reporting, and record keeping for affected sources is not included in these annualized cost estimates, but EPA has done a separate analysis and estimated the cost to be approximately $28 million (see Section 9.3., Paperwork Reduction Act).

Cost estimates for the Transport Rule are based on results from ICF's Integrated Planning Model. The model minimizes the costs of producing electricity (including abatement costs) while meeting load demand and other constraints (full documentation for IPM can be found at http://www.epa.gov/airmarkets/progsregs/epa-ipm and in the TSD "Updates to EPA Base Case v3.02 EISA Using the Integrated Planning Model"). The structure of the model assumes that the electric utility industry will be able to meet the environmental emission caps at least cost. Montgomery (1972) has shown that this least cost solution corresponds to the equilibrium of an emission permit system. See also Atkinson and Tietenburg (1982), Krupnick et al. (1980), and McGartland and Oates (1985). However, to the extent that transaction and/or search costs, combined with institutional barriers, restrict the ability of utilities to exhaust all the gains from emissions trading, costs are underestimated by the model. Utilities in the IPM model also have "perfect foresight." To the extent that utilities misjudge future conditions affecting the economics of pollution control, costs may be understated as well.

The "perfect foresight" of the model is also relevant in the context of the assurance provisions required in the proposed remedy (State Budgets/Limited Trading) and Direct Control. Because of the sizeable penalties associated with violating assurance provisions, EPA believes it will be economical for units to comply with the provisions. EPA modeled these provisions, which restrict emissions from a state to the budget plus variability limits on a 1-year and 3-year rolling average basis, as state-specific emissions caps set at the budget plus 3-year average variability. The Power Sector Variability Technical Support Document contains further details on these assurance provisions.

Modeling the assurance provisions as caps means that the model must meet the same limit each year, but it also allows the model to optimize with perfect foresight of present and future limits. While the model minimizes production costs while meeting required generation and reserve margin, sources in reality may choose to make greater emissions reductions than required in exchange for more certainty about emissions variability. IPM captures the cost associated with making required reductions in each state, but because of its "perfect foresight," the model likely cannot capture the true benefit to sources of having a range of allowed variability.

From another vantage point, this modeling analysis does not take into account the potential for advancements in the capabilities of pollution control technologies for SO_2 and NO_x removal as well as reductions in their costs over time. Market-based cap and trade regulation serves to promote innovation and the development of new and cheaper technologies. As an example, cost estimates of the Acid Rain SO_2 trading program by Resources for the Future (RFF) and MIT's Center for Energy and Environmental Policy Research (CEEPR) have been as much as 83 percent lower than originally projected by the EPA (see Carlson et al., 2000; Ellerman, 2003). It is important to note that the original analysis for the Acid Rain Program done by EPA also relied on an optimization model like IPM. Ex ante, EPA cost estimates of roughly \$2.7 to \$6.2 billion[58] in 1989 were an overestimate of the costs of the program in part because of the limitation of economic modeling to predict technological improvement of pollution controls and other compliance options such as fuel switching. Ex post estimates of the annual cost of the Acid Rain SO_2 trading program range from \$1.0 to \$1.4 billion. Harrington et al. have examined cost analyses of EPA programs and found a tendency for predicted costs to overstate actual implementation costs in market-based programs (Harrington, Morgenstern, and Nelson, 2000). In recognition of this, EPA's mobile source program uses adjusted engineering cost estimates of pollution control equipment and installation costs to account for this fact, which EPA has not done in this case.[59] The Agency is considering approaches to make this adjustment in the future, or at least to be able to provide a sense of the rough amount by which costs could be overstated in the analysis that has occurred.

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[58] 2010 Phase II cost estimate in \$1995.

[59] See regulatory impact analysis for the Tier 2 Regulations for passenger vehicles (1999) and Heavy-Duty Diesel Vehicle Rules (2000).

EPA's latest update of IPM incorporates state rules or regulations and various NSR settlements adopted through February 3, 2009. Documentation for IPM can be found at http://www.epa.gov/airmarkets/progsregs/epa-ipm and in the TSD "Updates to EPA Base Case v3.02 EISA Using the Integrated Planning Model.". Any state or settlement action since that time has not been accounted for in our analysis in this chapter.

As configured in this application, IPM does not take into account demand response (i.e., consumer reaction to electricity prices). The increased retail electricity prices shown in Tables 7-15 and 7-16 would prompt end users to curtail (to some extent) their use of electricity and encourage them to use substitutes.[60] The response would lessen the demand for electricity, resulting in electricity price increases slightly lower than IPM predicts, which would also reduce generation and emissions. Because of demand response, certain unquantified negative costs (i.e., savings) result from the reduced resource costs of producing less electricity because of the lower quantity demanded. To some degree, these saved resource costs will offset the additional costs of pollution controls and fuel switching that we would anticipate with the Transport Rule. Although the reduction in electricity use is likely to be small, the cost savings from such a large industry[61] is not insignificant. EIA analysis examining multi-pollutant legislation under consideration in 2003 indicates that the annualized costs of the Transport Rule may be overstated substantially by not considering demand response, depending on the magnitude and coverage of the price increases.[62]

On balance, after consideration of various unquantified costs (and savings that are possible), EPA believes that the annual private compliance costs that we have estimated are more likely to overstate the future annual compliance costs that industry will incur, rather than understate those costs.

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[60] The degree of substitution/curtailment depends on the costs and performance of the goods that substitute for more energy consuming goods, which is reflected in the demand elasticity.

[61] Investor-owned utilities alone accounted for nearly $300 billion in revenue in 2008 (EIA).

[62] See "Analysis of S. 485, the Clear Skies Act of 2003, and S. 843, the Clean Air Planning Act of 2003." Energy Information Administration. September, 2003. EIA modeling indicated that the Clear Skies Act of 2003 (a nationwide cap and trade program for SO_2, NO_X, and mercury), demand response could lower present value costs by as much as 47% below what it would have been without an emission constraint similar to the Transport Rule.

7.14 Significant Energy Impact

The Transport Rule as proposed has significant impact according to *E.O. 13211: Actions that Significantly Affect Energy Supply, Distribution, or Use*. Under the provisions of this proposed rule, EPA projects that approximately 1.2 GW of coal-fired generation may be removed from operation by 2014 under the proposed remedy. In practice, however, the units projected to be uneconomic to maintain may be "mothballed," retired, or kept in service to ensure transmission reliability in certain parts of the grid. These units are predominantly small and infrequently-used generating units dispersed throughout the area affected by the rule. Assumptions of higher natural gas prices or electricity demand would create a greater incentive to keep these units operational.

The EPA estimates that there are several fuel price increases resulting from the proposed remedy in the Transport Rule. The EPA projects that the average retail electricity price could increase nationally by about 2.5 percent in 2012 and 1.5 percent in 2014. This is generally less of an increase than often occurs with fluctuating fuel prices and other market factors. Related to this, delivered coal prices increase by about 7 percent in 2012 and 4 percent in 2014 as a result of higher demand for lower-sulfur coals. The EPA also projects that delivered natural gas prices will increase by less than 1.7 percent in 2012 and 0.5 percent in 2014 and that natural gas use for electricity generation will increase by less than 73 million million cubic feet (mcf) by 2014. The price increase is also within the range we regularly see in delivered natural gas prices. Finally, the EPA projects coal production for use by the power sector, a large component of total coal production, will decrease by 3 million tons in 2012 and 9 million tons in 2014 from the base case levels, which is a relatively small amount compared to the more than one billion tons of coal produced for utility use each year. The EPA does not believe that this rule will have any other impacts that exceed the significance criteria.

The EPA believes that a number of features of the rulemaking serve to reduce its impact on energy supply. First, the trading programs in State Budgets/Limited Trading provide considerable flexibility to the power sector and enable industry to comply with the emission reduction requirements in the most cost-effective manner, thus minimizing overall costs and the ultimate impact on energy supply. Second, the more stringent budgets for SO_2 are set in two phases, providing adequate time for EGUs to install pollution controls. In addition, both the operational flexibility of trading and the ability to bank allowances for future years helps industry plan for and ensure reliability in the electrical system.

7.15 References

Atkinson, S., and T. Tietenberg. 1982. "The Empirical Properties of Two Classes of Design for Transferable Discharge Permit Markets." *Journal of Environmental Economics and Management* 9:101-121

Carlson, Curtis, Dallas R. Burtraw, Maureen, Cropper, and Karen L. Palmer. 2000. "Sulfur Dioxide Control by Electric Utilities: What Are the Gains from Trade?" *Journal of Political Economy* 108(#6):1292-1326.

Ellerman, Denny. January 2003. Ex Post Evaluation of Tradable Permits: The U.S. SO_2 Cap-and-Trade Program. Massachusetts Institute of Technology Center for Energy and Environmental Policy Research.

EIA Annual Coal Report 2008. DOE/EIA-0584 (2008). Available at: http://www.eia.doe.gov/cneaf/coal/page/acr/acr_sum.html

EIA Annual Energy Outlook 2003. DOE/EIA-0383 (2003). Available at: http://www.eia.doe.gov/oiaf/archive/aeo03/index.html

EIA Electric Power Annual 2008. DOE/EIA-0348 (2008). Available at: http://www.eia.doe.gov/cneaf/electricity/epa/epa_sum.htm

EIA Electric Power Monthly March 2010 with Data for December 2009. DOE/EIA-0226 (2010/03). Available at: http://www.eia.doe.gov/cneaf/electricity/epm/epm_sum.html

Freme, Fred. 2009. U.S. Coal Supply and Demand: 2008 Review. EIA. Available at: http://www.eia.doe.gov/cneaf/coal/page/special/tbl1.html

Harrington, W., R.D. Morgenstern, and P. Nelson. 2000. "On the Accuracy of Regulatory Cost Estimates." *Journal of Policy Analysis and Management* 19(2):297-322.

Keohane, Nathaniel O. 2009. "The Technocratic and Democratic Functions of the CAIR Regulatory Analysis." In *Reforming Regulatory Impact Analysis*, eds. Winston Harrington, Lisa Heinzerling, and Richard D. Morgenstern, 33–55. Washington, DC: Resources for the Future.

Krupnick, A., W. Oates, and E. Van De Verg. 1980. "On Marketable Air Pollution Permits: The Case for a System of Pollution Offsets." *Journal of Environmental Economics and Management* 10:233-47.

Manson, Nelson, and Neumann. 2002. "Assessing the Impact of Progress and Learning

Curves on Clean Air Act Compliance Costs." Industrial Economics Incorporated.

McGartland, A., and W. Oates. 1985. "Marketable Permits for the Prevention of Environmental Deterioration." *Journal of Environmental Economics and Management* 12:207-228.

Montgomery, W. David. 1972. "Markets in Licenses and Efficient Pollution Control Programs." *Journal of Economic Theory* 5(3):395-418.

Wagner, Wendy E. 2009. "The CAIR RIA: Advocacy Dressed Up as Policy Analysis." In *Reforming Regulatory Impact Analysis*, eds. Winston Harrington, Lisa Heinzerling, and Richard D. Morgenstern, 56–81. Washington, DC: Resources for the Future.

CHAPTER 8

MACROECONOMIC IMPACTS AND SOCIAL COSTS

EPA prepares an economic impact analysis (EIA) to provide decision makers with a measure of the social costs of using resources to comply with a program (EPA, 2000). The social costs can then be compared with estimated social benefits. As noted in EPA's (2000) *Guidelines for Preparing Economic Analyses*, several tools are available to estimate social costs and range from simple direct compliance cost methods to the development of a more complex market analysis. The Office of Air Quality Planning and Standards (OAQPS) adopted an economy-wide market analysis as described in the Office's Economic Resource Manual (EPA, 1999)[63] and uses the latest EMPAX computable general equilibrium modeling system.

The Economic Model for Policy Analysis (EMPAX) was first developed in 2000 to support economic analysis of EPA's maximum achievable control technology (MACT) rules for combustion sources (reciprocating internal combustion engines, industrial boilers, and turbines). The initial framework consisted of a national multi-market partial-equilibrium model with linkages only between manufacturing industries and the energy sector. Modified versions of EMPAX were subsequently used to analyze economic impacts of strategies for improving air quality in the Southern Appalachian mountain region as part of efforts in 2002 associated with the Southern Appalachian Mountain Initiative (SAMI). Later work extended its scope to cover all aspects of the U.S. economy with regional detail.

Since large-scale environmental policies also indirectly influence current and future input uses, income, and household consumption patterns, EPA subsequently updated the model system to include a complete set of economic linkages among all industrial and energy sectors as well as households that supply factors of production such as labor and purchase goods (i.e., a computable general equilibrium [CGE] framework). As a result, EMPAX is now a dynamic general equilibrium model that traces economic impacts as they are transmitted across time and throughout the economy. EMPAX-CGE underwent peer review

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[63] This document is available on the Internet at http://www.epa.gov/ttn/ecas/analguid.html.

in 2006; detailed model documentation and results of the peer review can be accessed at the following Web site: http://www.epa.gov/ttn/ecas/EMPAXCGE.htm.

8.1 EMPAX Computable General Equilibrium (CGE) Model: Overview

EMPAX-CGE is a dynamic, intertemporally optimizing model that solves in 5-year intervals from 2010 to 2050. It uses the classical Arrow-Debreu general equilibrium framework wherein households maximize utility subject to budget constraints, and firms maximize profits subject to technology constraints. The model structure, in which agents are assumed to have perfect foresight and to maximize utility across all time periods, allows agents to modify behavior in anticipation of future policy changes, unlike dynamic recursive models that assume agents do not react until a policy has been implemented.

Nested constant elasticity of substitution (CES) functions are used to portray substitution possibilities available to producers and consumers. Figure 8-1 illustrates this general framework and gives a broad characterization of the model.[64] Along with the underlying data, these nesting structures and associated substitution elasticities determine the effects that will be estimated for policies. These nesting structures and elasticities used in EMPAX-CGE are generally based on the Emissions Prediction and Policy Analysis (EPPA) Model developed at the Massachusetts Institute of Technology (Paltsev et al., 2005). This updated version of the EPPA model incorporates some extensions over the EPPA version documented in Babiker et al. (2001), such as specification of transportation purchases by households. These updates to transportation choices have been incorporated in this version of EMPAX-CGE as shown on the left-hand side of Figure 8-1. Although the two models continue to have different focuses (EPPA is a recursive dynamic, international model focused on national-level climate change policies), both are intended to simulate how agents will respond to environmental policies; thus, EPPA provides a strong basis to develop the theoretical structure of EMPAX-CGE.

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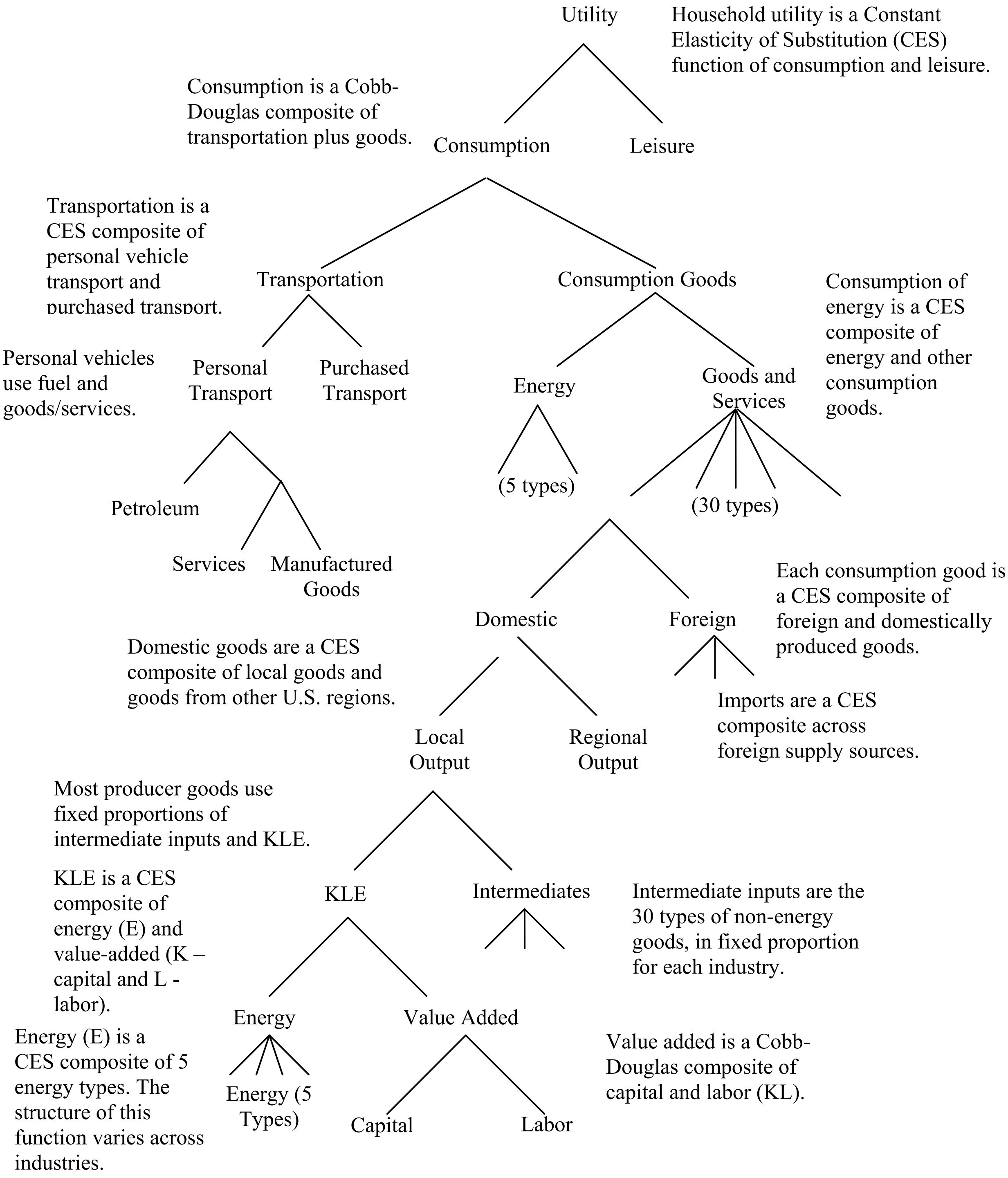
[64] Although it is not illustrated in Figure 8-1, some differences across industries exist in their handling of energy inputs. In addition, the agriculture and fossil fuel sectors in EMPAX-CGE contain equations that account for the presence of fixed inputs to production (land and fossil-fuel resources, respectively).

Figure 8-1. General Production and Consumption Nesting Structure in EMPAX-CGE



Given this basic similarity, EMPAX-CGE has adopted a comparable structure. EMPAX-CGE is programmed in the GAMS[65] language (Generalized Algebraic Modeling System) and solved as a mixed complementarity problem (MCP)[66] using MPSGE software (Mathematical Programming Subsystem for General Equilibrium).[67] The PATH solver from GAMS is used to solve the MCP equations generated by MPSGE.

8.1.1 Data Sources

The economic data come from state-level information provided by the Minnesota IMPLAN Group (2006),[68] and energy data come from the Energy Information Administration (EIA).[69] Forecasts for economic growth are taken from EIA's *Annual Energy Outlook 2009* Updated Reference Case (AEO) and Global Insight (2007).[70] Although IMPLAN data contain information on the value of energy production and consumption in dollars, these data are replaced with EIA data since the policies being investigated by EMPAX-CGE typically focus on energy markets, making it essential to include the best possible characterization of these markets in the model. Although the IMPLAN data are developed from a variety of government data sources at the U.S. Bureau of Economic Analysis and U.S. Bureau of Labor Statistics, these data do not always agree with energy information collected by EIA directly from manufacturers and electric utilities.

EMPAX-CGE combines these economic and energy data to create a balanced social accounting matrix (SAM) that provides a baseline characterization of the economy. The SAM contains data on the value of output in each sector, payments for factors of production

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[65] See Brooke, Kendrick, and Meeraus (1996) for a description of GAMS (http://www.gams.com/).

[66] Solving EMPAX-CGE as an MCP problem implies that complementary slackness is a feature of the equilibrium solution. In other words, any firm in operation will earn zero economic profits, and any unprofitable firms will cease operations. Similarly, for any commodity with a positive price, supply will equal demand, or conversely any good in excess supply will have a zero price.

[67] See Rutherford (1999) for MPSGE documentation (http://www.mpsge.org/).

[68] See http://www.implan.com/index.html for a description of the Minnesota IMPLAN Group and its data.

[69] These EIA sources include *AEO 2007*, the *Manufacturing Energy Consumption Survey*, *State Energy Data Report*, *State Energy Price and Expenditure Report*, and various annual industry profiles.

[70] See http://www.globalinsight.com/ProductsServices/ProductDetail1100.htm for a description of the Global Insight U.S. State Forecasting Service.

and intermediate inputs by each sector, household income and consumption, government purchases, investment, and trade flows. A balanced SAM for the baseline year consistent with the desired sectoral and regional aggregation is produced using procedures developed by Babiker and Rutherford (1997) and described in Rutherford and Paltsev (2000). This methodology relies on optimization techniques to maintain the calculated energy statistics (in both quantity and value terms) while minimizing any changes needed in the other economic data to create a new balanced SAM based on EIA/IMPLAN data for the baseline model year (in essence, industry production functions are adjusted, if necessary, to account for discrepancies between EIA energy data and IMPLAN economic data by matching the energy data and adjusting the use of nonenergy inputs so that the industry is in balance, that is, the value of inputs to production equals the value of output).

These data are used to define economic conditions in 50 states within the United States (plus the District of Columbia), each of which contains 80 industries. Prior to solving EMPAX-CGE, the states and industries are aggregated up to the categories to be included in the analysis. Aggregated regions have been selected to capture important differences across the country in electricity generation technologies, while industry aggregations are controlled by available energy consumption data.

Table 8-1 presents the 35 industry categories included in EMPAX-CGE for policy analysis. Their focus is on maintaining as much detail in the energy intensive and manufacturing sectors[71] as is allowed by available energy consumption data and computational limits of dynamic CGE models. In addition, the electricity industry is separated into fossil fuel generation and nonfossil generation, which is necessary because many electricity policies affect only fossil-fired electricity.

Figure 8-2 shows the five regions run in EMPAX-CGE in this analysis, which have been defined based on the expected regional distribution of policy impacts, availability of economic and energy data, and computational limits on model size. These regions have been constructed from the underlying state-level database designed to follow, as closely as

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[71] Energy-intensive industry categories are based on EIA definitions of energy-intensive manufacturers in the *Assumptions for the Annual Energy Outlook 2007*.

possible, the electricity market regions defined by the North American Electric Reliability Council (NERC).[72]

8.1.2 Production Functions

All productive markets are assumed to be perfectly competitive and have production technologies that exhibit constant returns to scale, except for the agriculture and natural resource extracting sectors, which have decreasing returns to scale because they use factors in fixed supply (land and fossil fuels, respectively). The electricity industry is separated into two distinct sectors: fossil fuel generation and nonfossil generation. This allows tracking of variables such as heat rates for fossil-fired utilities (in BTUs of energy input per kilowatt hour of electricity output).

All markets must clear (i.e., supply must equal demand in every sector) in every period, and the income of each agent in the model must equal their factor endowments plus any net transfers. Along with the underlying data, the nesting structures shown in Figure 8-1 and associated substitution elasticities define current production technologies and possible alternatives.

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[72] Economic data and information on nonelectricity energy markets are generally available only at the state level, which necessitates an approximation of the NERC regions that follows state boundaries.

Table 8-1. Industries in Dynamic EMPAX-CGE

EMPAX Industry	NAICS Classifications
Energy	
Coal	2121
Crude oil[a]	211111, 4861
Electricity (*fossil*)	2211
Electricity (*nonfossil*)	2211
Natural gas	211112, 2212, 4862
Petroleum refining[b]	324, 48691
General	
Agriculture	11
Mining (w/o coal, crude, gas)	21
Construction	23
Manufacturing	
Food products	311
Textiles and apparel	313, 314, 315, 316
Lumber	321
Paper and allied	322
Printing	323
Chemicals	325
Plastic & rubber	326
Glass	3272
Cement	3273
Other minerals	3271, 3274, 3279
Iron and steel	3311, 3312
Aluminum	3313
Other primary metals	3314, 3316
Fabricated metal products	332
Manufacturing equipment	333
Computers & communication equipment	334
Electronic equipment	335
Transportation equipment	336
Miscellaneous remaining	312, 337, 339
Services	
Wholesale & retail trade	42, 44, 45
Transportation[c]	481–488
Information	51
Finance & real estate	52, 54
Business/professional	53, 55, 56
Education (w/public)	61
Health care (w/public)	62
Other services	71, 72, 81, 92

[a] Although NAICS 211111 covers crude oil and gas extraction, the gas component of this sector is moved to the natural gas industry.

[b] Transportation does not include NAICS 4862 (natural gas distribution), which is part of the natural gas industry.

[c] The petroleum refining industry provided oil in delivered terms, which includes pipeline transport.

Figure 8-2. Regions Defined in Dynamic EMPAX-CGE

*"West" also includes Alaska and Hawaii.

8.1.3 Utility Functions

Each region in the dynamic version of EMPAX-CGE contains four representative households, classified by income, that maximize intertemporal utility over all time periods in the model subject to budget constraints, where the income groups are

- $0 to $14,999,
- $15,000 to $29,999,
- $30,000 to $49,999, and

- $50,000 and above.

These representative households are endowed with factors of production, including labor, capital, natural resources, and land inputs to agricultural production. Factor prices are equal to the marginal revenue received by firms from employing an additional unit of labor or capital. The value of factors owned by each representative household depends on factor use implied by production within each region. Income from sales of these productive factors is allocated to purchases of consumption goods to maximize welfare.

Within each time period, intratemporal utility received by a household is formed from consumption of goods and leisure. All consumption goods are combined using a Cobb-Douglas structure to form an aggregate consumption good. This composite good is then combined with leisure time to produce household utility. The elasticity of substitution between consumption goods and leisure depends on empirical estimates of labor supply elasticities and indicates how willing households are to trade off leisure time for consumption. Over time, households consider the discounted present value of utility received from all periods' consumption of goods and leisure.

Following standard conventions of CGE models, factors of production are assumed to be mobile among sectors within regions, but migration of productive factors is not allowed across regions. This assumption is necessary to calculate welfare changes for the representative household located in each region in EMPAX-CGE. EMPAX-CGE also assumes that ownership of natural resources and capital embodied in nonfossil electricity generation are spread across the United States through capital markets.

8.1.3.1 Welfare Measures

To analyze the social benefits and costs of policy alternatives, EMPAX uses a willingness-to-pay measure known as a Hicksian *equivalent variation* (EV). EV reflects the additional money that a household would need (at original prices p^0 and income m^0) to make it *as well off* with the new policy; the amount is "equivalent" to the changes in the utility households receive from consumption and leisure time.

$$EV = u(p^0; p',m') - u(p^0; p^0, m^0) = u(p^0; p',m') - m^0$$

where

p^0 = the baseline prices

m^0 = baseline income

p' = with policy prices

m' = with policy income

For example, under a policy that makes households worse off, EV represents the maximum amount of money the household would be willing to pay to avoid the policy. Through this analysis, we use this metric to measure the policy's social costs. It is important to emphasize the measure does not incorporate any environmental benefits associated with air quality improvements.

8.1.4 Treatment of Trade

In EMPAX-CGE, all goods and services are assumed to be composite, differentiated "Armington" goods made up of locally manufactured commodities and imported goods. Output of local industries is initially separated into output destined for local consumption by producers or households and output destined for export. This local output is then combined with goods from other regions in the United States using Armington trade elasticities that indicate agents make relatively little distinction between output from firms located within their region and output from firms in other regions within the United States. Finally, the domestic composite goods are aggregated with imports from foreign sources using lower trade elasticities to capture the fact that foreign imports are more differentiated from domestic output than are imports from other regional suppliers in the United States.

8.1.5 Tax Rates and Distortions

Taxes and associated distortions in economic behavior have been included in EMPAX-CGE because theoretical and empirical literature found that taxes can substantially alter estimated policy costs (e.g., Bovenberg and Goulder [1996]; Goulder and Williams

[2003]). For example, existing labor taxes distort economic choices because they encourage people to work below the levels they would choose in an economy without labor taxes and reduce economic efficiency.[73] When environmental policies raise production costs for firms and the price of goods and services, people may choose to work even less; the additional economic costs from this decision have been described as the "tax interaction" effect.

EMPAX-CGE considers these interaction effects by using tax data from several sources and by explicitly modeling household labor supply decisions. The IMPLAN economic database provides information on taxes such as indirect business taxes (all sales and excise taxes) and social security taxes. However, since IMPLAN reports factor payments for labor and capital at their gross of tax values, we use additional data sources to determine personal income and capital tax rates. Information from the TAXSIM model at the National Bureau of Economic Research (Feenberg and Coutts, 1993), along with user cost-of-capital calculations from Fullerton and Rogers (1993), are used to establish tax rates. Elasticity parameters describing labor supply choice ultimately determine how distortionary existing taxes are in the CGE model. EMPAX-CGE currently uses elasticities based on the relevant literature (i.e., 0.4 for the compensated labor supply elasticity and 0.15 for the uncompensated labor supply elasticity). These elasticity values give an overall marginal excess burden associated with the existing tax structure of approximately 0.3.

8.1.6 Intertemporal Dynamics and Economic Growth

EMPAX-CGE includes four sources of economic growth: technological change from improvements in energy efficiency, growth in the available labor supply (from both population growth and changes in labor productivity), increases in stocks of natural resources, and capital accumulation. Energy consumption per unit of output tends to decline over time because of improvements in production technologies and energy conservation. These changes in energy use per unit of output are modeled as autonomous energy efficiency improvements (AEEIs), which are used to replicate energy consumption forecasts by

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[73] These efficiency losses are often expressed in terms of overall marginal excess burden—the cost associated with raising an additional dollar of tax revenue. Estimates range from $0.10 to $0.35 per dollar (Ballard, Shoven, and Whalley, 1985).

industry and fuel from EIA.[74] The AEEI values provide the means for matching expected trends in energy consumption that have been taken from the AEO forecasts. They alter the amount of energy needed to produce a given quantity of output by incorporating improvements in energy efficiency and conservation. Labor force and regional economic growth, electricity generation, changes in available natural resources, and resource prices are also based on the AEO forecasts.

Savings provide the basis for capital formation and are motivated through people's expectations about future needs for capital. Savings and investment decisions made by households determine aggregate capital stocks in EMPAX-CGE. The IMPLAN data set provides details on the types of goods and services used to produce the investment goods underlying each region's capital stocks. Adjustment dynamics associated with formation of capital are controlled by using quadratic adjustment costs experienced when installing new capital, which imply that real costs are experienced to build and install new capital equipment.

Prior to investigating policy scenarios, it is necessary to establish a baseline path for the economy that incorporates economic growth and technology changes that are expected to occur in the absence of the policy actions. Beginning from the initial balanced SAM data set, the model is calibrated to replicate forecasts from the AEO 2009 (Updated Reference Case Version, March 2009). Upon incorporating these forecasts, EMPAX-CGE is solved to generate a baseline based on them through 2030. Once this baseline is established, it is possible to run the "counterfactual" policy experiments discussed below.

8.1.7 Linkage with the Integrated Planning Model

Although CGE models have been used extensively to analyze climate policies that limit carbon emissions from electricity production, some other types of utility-emissions policies are more difficult to consider. Unlike carbon dioxide, emissions of pollutants such as SO_2, NO_x, and mercury are not necessarily proportional to fuel use. These types of emissions

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[74] See Babiker et al. (2001) for a discussion of how this methodology was used in the EPPA model (EPPA assumes that AEEI parameters are the same across all industries in a country, while AEEI values in EMPAX-CGE are industry specific).

can be lowered by a variety of methods, but a CGE model cannot adequately capture the boiler-specific nature of these decisions and their costs and effects. Combining the strengths of the Integrated Planning Model (IPM) (disaggregated unit-level analyses of electricity policies) with the strengths of CGE models (macroeconomic effects of environmental policies) allows investigation of economy-wide implications of policies that would normally be hard to estimate consistently and effectively. IPM provides EMPAX with several electricity market outcomes needed to evaluate macroeconomic implications of policies.

IPM also provides information on generation costs in terms of capital costs, fixed operating costs, and variable operating costs. For EMPAX to effectively incorporate these IPM data on changes in costs, they have to be expressed in terms of the productive inputs used in CGE models (i.e., capital, labor, and material inputs produced by other industries). Rather than assume these costs represent a proportional scaling up of all inputs to the electricity industry in EMPAX, we use Nestor and Pasurka (1995) data on purchases made by industries for environmental protection reasons to allocate these additional expenditures across inputs within EMPAX (discussed in the EMPAX model documentation). Once these expenditures are specified, the incremental costs from IPM can be used to adjust the production technologies and input purchases by electricity generation in the CGE model.

Among the many results provided by IPM, several can potentially have significant implications for the rest of the economy including changes in electricity prices, fuel consumption by utilities, fuel prices, and changes in electricity production expenditures. EMPAX is capable of simultaneously incorporating some of all of these IPM findings, depending on the desired type and degree of linkage between the two models. At the regional level, EMPAX can match changes estimated by IPM for the following variables:

- electricity prices (percentage change in retail prices)
- coal and gas consumption for electricity (percentage changes in BTUs)
- coal and gas prices (percentage changes in prices)
- coal and gas expenditures ($ changes—BTUs of energy input times $/MMBTU)

- capital costs ($ changes)
- fixed operating costs ($ changes)
- variable operating costs ($ changes)

In addition, EMPAX-CGE can control electricity output to simulate the fixed demand used by IPM, or it can determine how changes in electricity prices will affect demand for electricity and hence electricity generation levels.

The IPM model calculates these variables for 26 NERC subregions. EMPAX uses information on generation levels for these subregions to aggregate the IPM results into the five regions used within EMPAX. Wholesale electricity prices are then matched to the changes shown by IPM. Fuel consumption by utilities in physical units (BTUs) is adjusted by the percentage changes in the IPM results. Fuel prices paid by both industries and households are also changed by the amounts estimated by IPM (the coal and gas market modules of IPM cover all fuel consumers, not merely utilities, so prices paid by all agents in EMPAX are adjusted).

8.1.8 Qualifications

Caveats that can typically be applied to CGE analyses, including this one, cover issues such as transitional dynamics in the economy. CGE models such as EMPAX, which assume foresight on the part of businesses and households, will allow agents to adapt to anticipated policy impacts coming in the future. These adaptations may occur more quickly than if agents adopted a wait-and-see approach to new regulations. The alternative, recursive-dynamic structure used in CGE models such as MIT EPPA imply that no anticipation or adjustments will occur until the policy is in place, which tends to overstate the costs of policies.

In addition to transition dynamics, while CGE models are ideally suited for analyzing broad, economy-wide impacts of policies, they are not able to examine firm-specific impacts on profits/losses or estimate how policies might affect particular types of disadvantaged households. Similarly, environmental justice and other distributional concerns cannot be adequately addressed using these types of models alone.

As noted above, the labor supply elasticities in the model have been chosen from the CGE literature on labor markets and tax distortions as discussed above. Other important assumptions about the production technologies and input substitution possibilities have been chosen from the MIT EPPA model. To ensure transparence of the assumptions, EMPAX-CGE underwent peer review in 2006, and detailed model documentation and results of the peer review can be accessed at the following Web site: http://www.epa.gov/ttnecas1/EMPAXCGE.htm.

8.2 EMPAX-CGE Model Results

8.2.1 Macroeconomic Variables and Social Costs

The transport rule will bring about changes in business and household behavior and will influence macroeconomic variables (gross domestic product [GDP] and consumption) and household economic welfare as estimated by the Hicksian EV method previously mentioned. Gross domestic product is the dollar value of all goods and services produced by the U.S. economy in a particular year. Consumption is defined in this analysis as the dollar value of goods and services consumed in the U.S. in a particular year. In 2015, EMPAX estimates that GDP and consumption levels are approximately 0.01% lower ($1.6 billion) (Figure 8-3).[75] Since the pollution controls vary by region, economic effects also vary by region; for example, Northeast GDP falls by 0.04% (Figure 8-4). There are small declines in GDP by region except for the Plains and West, where regional GDP increases as productive activities shift to these less regulated regions.

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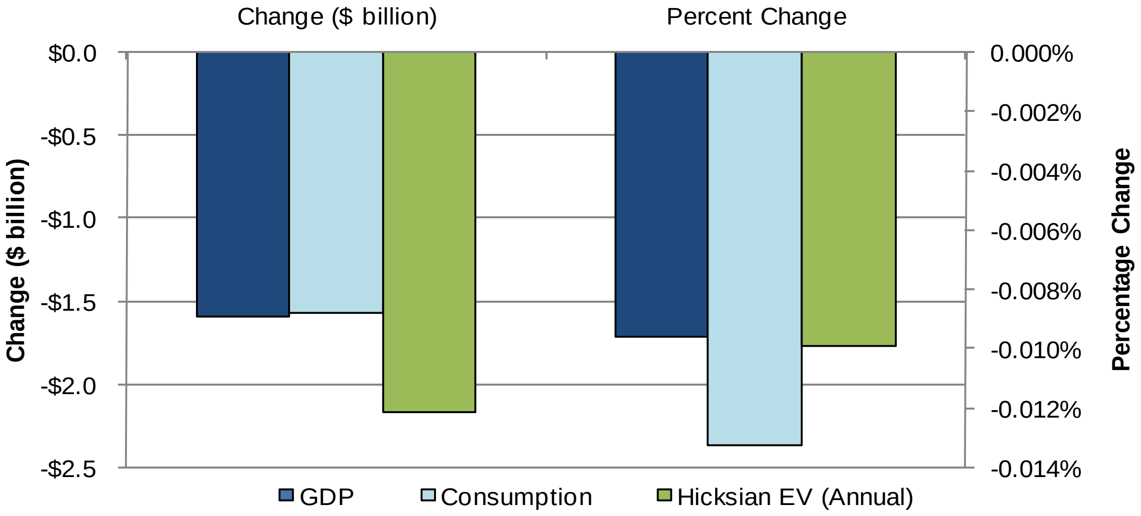
[75] We use 2015 estimates as a proxy for the impacts of compliance with the proposed rule in 2014.

Figure 8-3. Change in Macroeconomic Variables and Household Welfare (Percent change and Change in billion $2006) in 2015



Note: GDP represents the dollar value of all goods and services produced in the US in 2015. Consumption is the dollar value of all goods and services consumed within the US in 2015. Hicksian EV is the change in household economic welfare (defined in Section 8.1.3.1).

Average-annual social costs (as measured by Hicksian equivalent variation) are approximately 0.01% lower with the transport rule. Over the model's time horizon, the total present value of the losses is approximately $21.3 billion.[76] As noted in section 8.1.3.1 of this chapter, EMPAX-CGE does not incorporate any environmental benefits associated with air quality improvements. As a result, EMPAX welfare measures only approximate the rule's social cost. Using this interpretation, the annual social cost for 2015 is estimated to be $2.2

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[76] Values are discounted back to 2010 at the 5% interest rate used in the model. EPA uses a 5% interest rate based on the MIT Emissions Prediction and Policy Analysis (EPPA) model and SAB guidance from 2003 as discussed in U.S. EPA, Office of Policy Analysis and Review. 2003. "Benefits and Costs of the Clean Air Act 1990 - 2020: Revised Analytical Plan For EPA's Second Prospective Analysis." We recognize that this interest rate is not one of the interest rates (3 and 7%) that OMB's Circular A-4 guidance calls for in regulatory analyses. Detailed results for this EMPAX run for the proposed remedy can be found in the file "EMPAXresults_proposed transport remedy," that is available in the docket for this rule.

billion. With a 3 percent interest rate, the annual social cost for 2015 is estimated to be \$2.0 billion.

8.2.2 Industry Effects

The proposed rule directly influences the electricity sector's fuel use and private cost expenditures. As the electricity sector responds to these changes, other economy-wide changes occur. For example, higher electricity prices may encourage electricity-dependent sectors to reduce production levels, switch to other energy sources (e.g., oil) and/or seek energy efficiency improvements in their production process. Electricity sectors also make additional private cost expenditures in order to comply with the transport rule; these expenditures lead to other economy-wide changes. For example each dollar spent to comply with the program is used to buy environmental protection goods and services.[77] As a result, the demand for environmental protection goods and services will be higher with the transport rule. For sectors supplying environmental protection goods or services, the secondary effect may offset higher electricity costs. The following sections report and discuss output changes (i.e., changes in physical quantities of the goods/services each industry sector in each region produces) associated with the impacts of compliance in the year 2015, which serves as a proxy for compliance in 2014.

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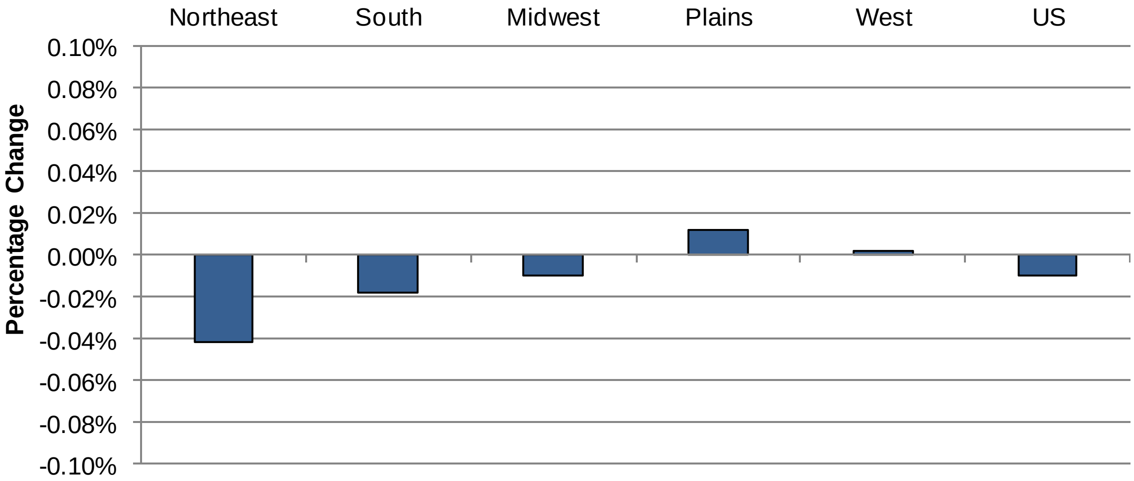
[77] Additional details are described in EMPAX-CGE model documentation (5-2 to 5-5).

Figure 8-4. Change in Regional Gross Domestic Product (GDP) (Percent) in 2015



Note: GDP in each region is the dollar value of goods and services produced in the region in 2015. See Figure 8-2 for a presentation of the states in each region

8.2.2.1 Energy Sectors

The EMPAX modeling system shows that the electricity sector experiences the most significant changes under the transport rule. Electricity output and fuel mix changes used to meet the transport rule also influence other energy sectors. For example, U.S. electricity and coal output both decline by approximately 0.3%. U.S. natural gas output changes for two reasons: 1) natural gas is used in electricity generation and electricity generation declines and 2) natural gas is a substitute for electricity, so gas use increases when electricity becomes more expensive. Overall natural gas output declines because the first effect (reduced electricity generation) is greater than the second effect (substitution from electricity use to natural gas use)). Crude oil and petroleum output decline, but the changes are small; these inputs are less critical to the electricity sector making them less sensitive to changes in electricity production (Figure 8-5).

Given the regional distribution of controls, there are differences in regional output quantity changes. For example, electricity production in the Northeast experiences the largest decline while the Plains and West electricity sectors see small output increases. Coal output changes to meet coal demand predictions from the IPM electricity model, and the IPM modeling system suggests that the Northeast's electricity sector uses additional coal inputs to meet the rule's requirements.

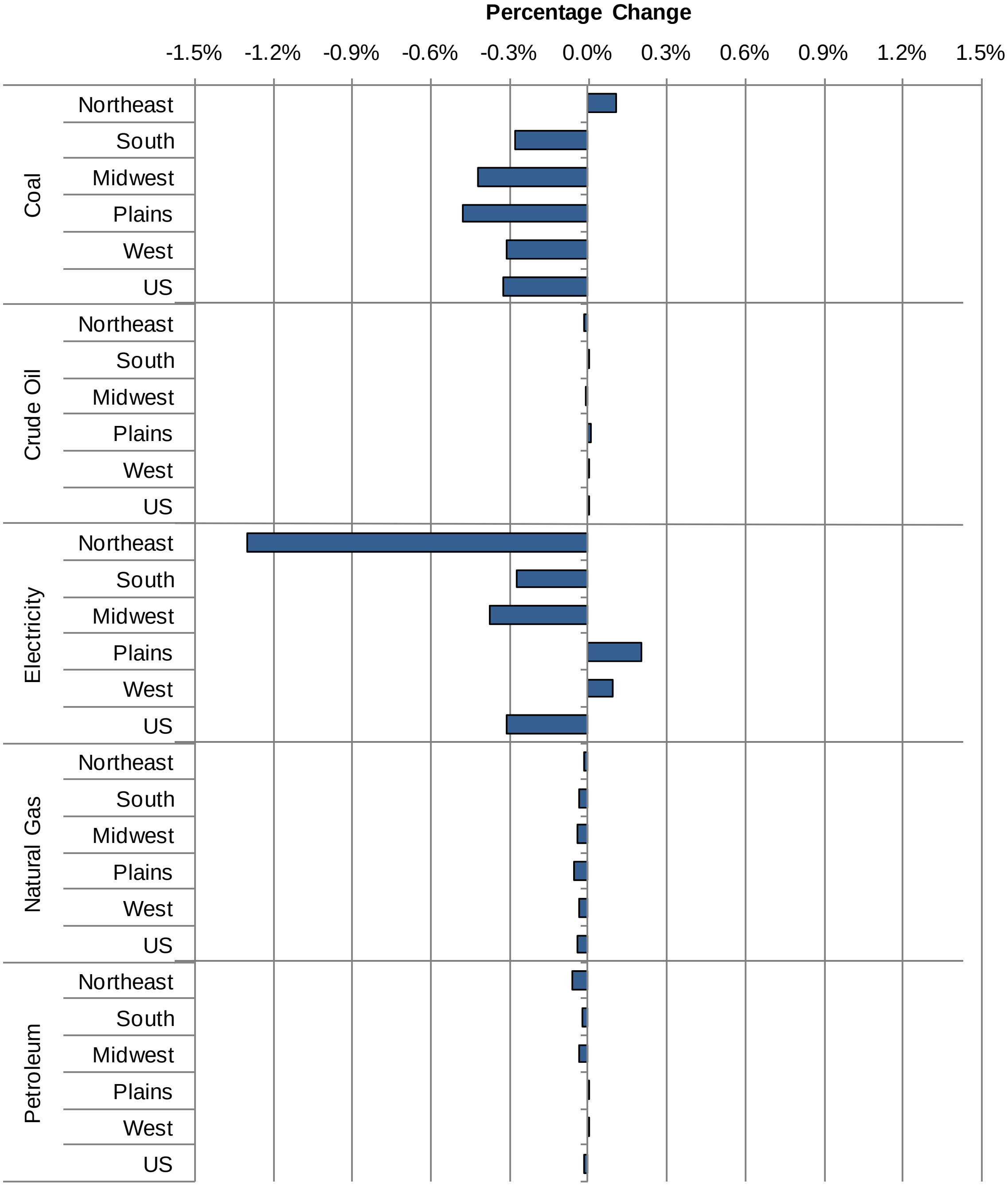
8.2.2.2 Energy-Intensive Sectors

Energy-intensive manufacturing industries are more sensitive to electricity and other energy price changes. Although the net U.S. output change for each energy-intensive industry is less than 0.1%, these sectors do show some (but economically small) regional variation. The most significant regional differences are seen in the aluminum sector, where production shifts from the Northeast, South, and Midwest regions to the Plains and West regions. Similar geographic shifts are observed in other energy-intensive industries (Figure 8-6).

8.2.2.3 Nonenergy Sectors

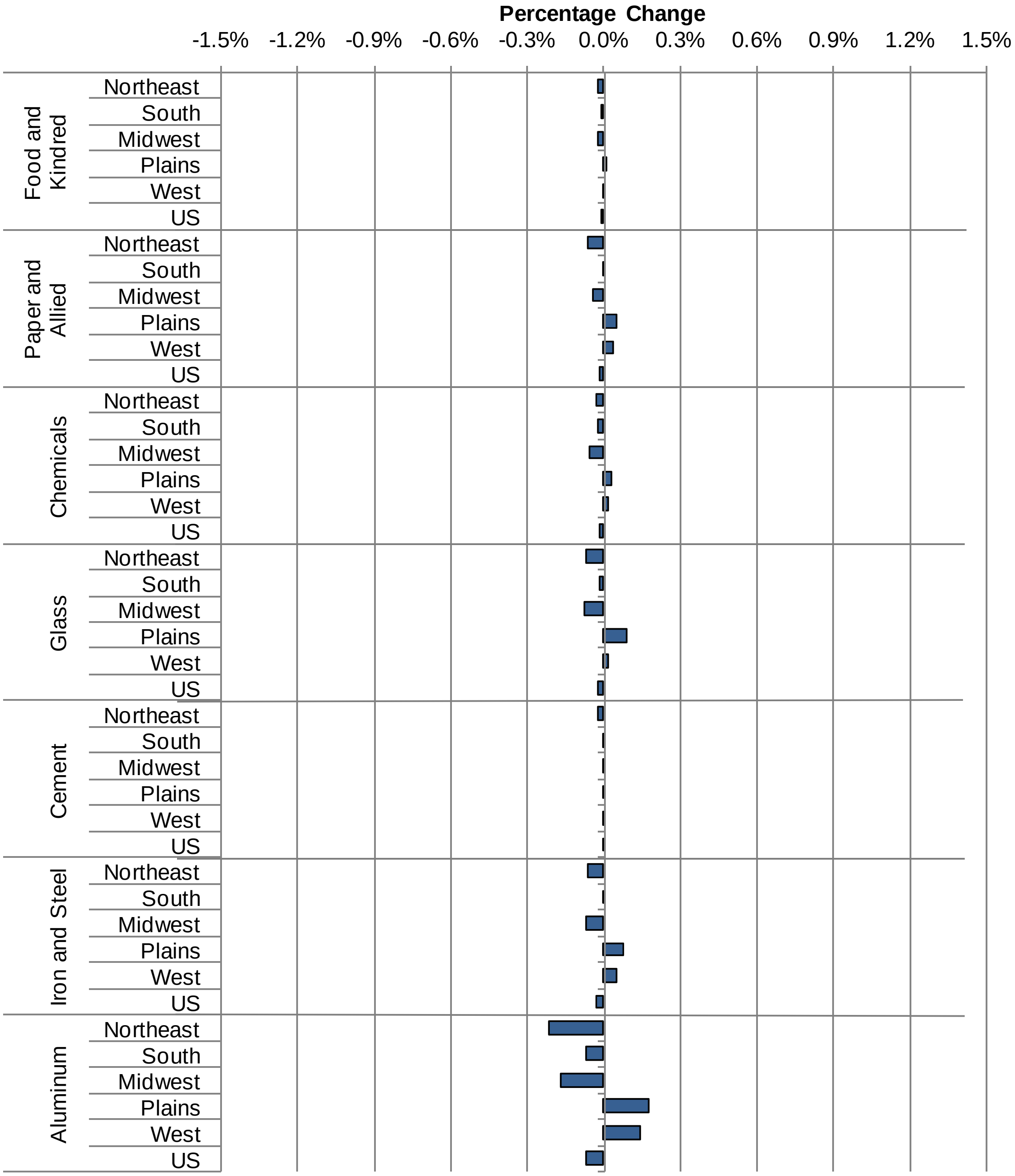
Although electricity expenditures represent a small fraction of nonenergy-sector production costs, higher electricity prices still influence nonenergy-sector production levels. However, nonenergy sector output effects are very small. National output levels for four broad nonenergy sectors: agriculture, other manufacturing, services, and transportation fall by less than one one-hundredth of a percent (0.01%). There is some regional variation as production shifts to areas with lower electricity costs (e.g., West, Plains), but the differences are not significant (Figure 8-7).

Figure 8-5. Output Changes in 2015: Energy Sectors (Percent)



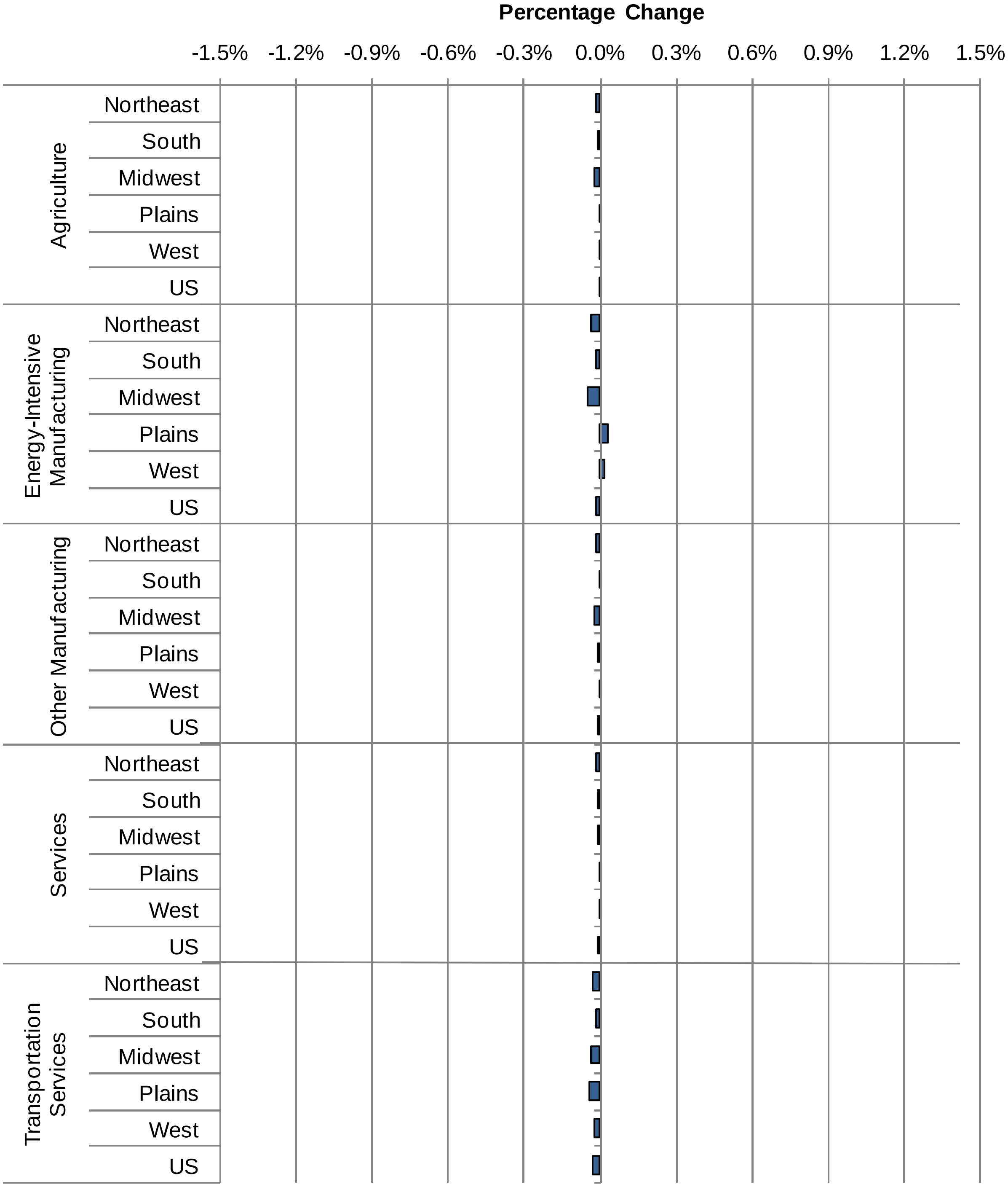
Note: Outcomes reflect changes in the physical quantities of goods/services each regional sector produces.

Figure 8-6. Output Changes in 2015: Energy-Intensive Sectors (Percent)



Note: Outcomes reflect changes in the physical quantities of goods/services each regional sector produces.

Figure 8-7. Output Changes in 2015: Nonenergy Sectors (Percent)



Note: Outcomes reflect changes in the physical quantities of goods/services each regional sector produces.

8.3 References

Babiker, M.H., and T.F. Rutherford. 1997. "Input Output and General Equilibrium Estimates of Embodied CO_2: A Data Set and Static Framework for Assessment." University of Colorado at Boulder, Working Paper 97-2. Available at http://www.mpsge.org/mainpage/mpsge.htm.

Babiker, M.H., J.M. Reilly, M. Mayer, R.S. Eckaus, I.S. Wing, and R.C. Hyman. 2001. "The MIT Emissions Prediction and CO_2 Policy Analysis (EPPA) Model: Revisions, Sensitivities, and Comparisons of Results." MIT Joint Program on the Science and Policy of Global Change, Report No. 71. Available at http://web.mit.edu/globalchange/www/eppa.html.

Ballard, C. J. Shoven, and J. Whalley. 1985. "General Equilibrium Computations of the Marginal Welfare Costs of Taxation in the United States." *American Economic Review* 75(1):128-138.

Bovenberg, L.A., and L.H. Goulder. 1996. "Optimal Environmental Taxation in the Presence of Other Taxes: General Equilibrium Analysis." *American Economic Review* 86(4):985–1000. Available at http://www.aeaweb.org/aer/.

Brooke, A., D. Kendrick, and A. Meeraus. 1996. GAMS Homepage. Available at http://www.gams.com.

Feenberg, D., and E. Coutts. 1993. "An Introduction to the TAXSIM Model." *Journal of Policy Analysis and Management* 12(1):189-194. Available at http://www.nber.org/~taxsim/.

Fullerton, D., and D. Rogers. 1993. "Who Bears the Lifetime Tax Burden?" Washington, DC: The Brookings Institute. Available at http://bookstore.brookings.edu/book_details.asp?product%5Fid=10403.

Global Insight. 2007. "U.S. State, Metropolitan Area, and County-level Forecasting Services." http://www.globalinsight.com/ProductsServices/ProductDetail1100.htm.

Goulder, L.H., and R.C. Williams. 2003. "The Substantial Bias from Ignoring General Equilibrium Effects in Estimating Excess Burden, and a Practical Solution." *Journal of Political Economy* 111:898-927. Available at http://www.journals.uchicago.edu/JPE/home.html.

Minnesota IMPLAN Group. 2006. State Level Data for 2004. Available from http://www.implan.com/index.html.

Nestor, D.V., and C.A. Pasurka. 1995. *The U.S. Environmental Protection Industry: A Proposed Framework for Assessment*. U.S. Environmental Protection Agency, Office of Policy, Planning, and Evaluation. EPA 230-R-95-001. Available at http://yosemite.epa.gov/ee/epa/eermfile.nsf/11f680ff78df42f585256b45007e6235/41b8b642ab9371df852564500004b543/$FILE/EE 0217A 1.pdf.

Paltsev, S., J.M. Reilly, H.D. Jacoby, R.S. Eckaus, J. McFarland, M. Sarofim, M. Asadoorian, and M. Babiker. 2005. "The MIT Emissions Prediction and Policy Analysis (EPPA) Model: Version 4." MIT Joint Program on the Science and Policy of Global Change, Report No. 125. Cambridge, MA. Available at http://web.mit.edu/globalchange/www/eppa.html.

Rutherford, T.F. 1999. "Applied General Equilibrium Modeling with MPSGE as a GAMS Subsystem: An Overview of the Modeling Framework and Syntax." *Computational Economics* 14(1):1-46. Available at http://www.gams.com/solvers/mpsge/syntax.htm.

Rutherford, T.F., and S.V. Paltsev. 2000. "GTAP Energy in GAMS: The Dataset and Static Model." University of Colorado at Boulder, Working Paper 00 2. Available at http://www.mpsge.org/mainpage/mpsge.htm.

U.S. Department of Energy, Energy Information Administration. Undated (a). *State Energy Data Report*. Washington, DC. Available at http://www.eia.doe.gov/emeu/states/_use_multistate.html.

U.S. Department of Energy, Energy Information Administration. Undated (b). *State Energy Price and Expenditure Report*. Washington, DC. Available at http://www.eia.doe.gov/emeu/states/price_multistate.html.

U.S. Department of Energy, Energy Information Administration. 2003. *Manufacturing Energy Consumption Survey 2002*. Washington, DC. Available at http://www.eia.doe.gov/emeu/mecs/.

U.S. Department of Energy, Energy Information Administration. January 2007. *Annual Energy Outlook 2007*. DOE/EIA-0383(2007). Washington, DC. Available at http://www.eia.doe.gov/oiaf/aeo/index.html.

U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards. 1999. *OAQPS Economic Analysis Resource Document*. http://www.epa.gov/ttn/ecas/analguid.html.

U.S. Environmental Protection Agency. September 2000. *Guidelines for Preparing Economic Analyses*. EPA 240-R-00-003. Washington, DC: EPA. http://yosemite.epa.gov/ee/epa/eed.nsf/webpages/Guidelines.html. Accessed August 2009.

CHAPTER 9

STATUTORY AND EXECUTIVE ORDER IMPACT ANALYSES

This chapter presents discussion and analyses relating to relevant Executive Orders and statutory requirements relevant for the Transport Rule. We discuss potential impacts to affected small entities as required by the Regulatory Flexibility Act (RFA), as amended by the Small Business Regulatory Enforcement Fairness Act (SBREFA). We also describe the analysis conducted to meet the requirements of the Unfunded Mandates Reform Act of 1995 (UMRA) that assess the impact of the Transport Rule for state, local and Tribal governments and the private sector. Analyses conducted to comply with the Paperwork Reduction Act (PRA) are also discussed. In addition, we address the requirements of Executive Order (EO) 13045: Protection of Children from Environmental Health and Safety Risks; EO 13175: Consultation and Coordination with Indian Tribal Governments; and EO 12898: Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations. The Discussion of Executive Order 13211: Actions that Significantly Affect Energy Supply, Distribution or Use is provided in Chapter 7 of this RIA.

9.1 Small Entity Impacts

The Regulatory Flexibility Act (5 U.S.C. § 601 et seq.), as amended by the Small Business Regulatory Enforcement Fairness Act (Public Law No. 104-121), provides that whenever an agency is required to publish a general notice of proposed rulemaking, it must prepare and make available an initial regulatory flexibility analysis, unless it certifies that the proposed rule, if promulgated, will not have “a significant economic impact on a substantial number of small entities” (5 U.S.C. § 605[b]). Small entities include small businesses, small organizations, and small governmental jurisdictions.

For the purposes of assessing the impacts of the Transport Rule on small entities, a small entity is defined as:

(1) A small business according to the Small Business Administration size standards by the North American Industry Classification System (NAICS) category of the owning entity. The range of small business size standards for

electric utilities is 4 billion kilowatt-hours of production or less;

(2) a small government jurisdiction that is a government of a city, county, town, district, or special district with a population of less than 50,000; and

(3) a small organization that is any not-for-profit enterprise that is independently owned and operated and is not dominant in its field.

Table 9-1 lists entities potentially affected by this proposed rule with the applicable NAICS code.

Table 9-1. Potentially Regulated Categories and Entities[a]

Category	NAICS Code[b]	Examples of Potentially Regulated Entities
Industry	221112	Fossil fuel-fired electric utility steam generating units.
Federal Government	221112[c]	Fossil fuel-fired electric utility steam generating units owned by the federal government.
State/Local/ Tribal Government	221112[c]	Fossil fuel-fired electric utility steam generating units owned by municipalities.
	921150	Fossil fuel-fired electric utility steam generating units in Indian Country.

[a] Include NAICS categories for source categories that own and operate electric generating units only.
[b] North American Industry Classification System.
[c] Federal, state, or local government-owned and operated establishments are classified according to the activity in which they are engaged.

EPA examined the potential economic impacts to small entities associated with this rulemaking based on assumptions of how the affected entities will implement control measures to meet their NO_x and SO_2 budgets. This analysis does not examine potential indirect economic impacts associated with the Transport Rule, such as employment effects in industries providing fuel and pollution control equipment, or the potential effects of electricity price increases on industries and households.

9.1.1 Identification of Small Entities

EPA used Velocity Suite's Ventyx data as a basis for identifying plant ownership and compiling the list of potentially affected small entities.[78] The data set contains detailed ownership and corporate affiliation information. For plants burning fossil fuel as the primary fuel, plant-level boiler and generator capacity, heat input, generation, and emissions data were aggregated by owner and then parent company. Entities with more than 4 billion kWh of annual electricity generation were removed from the list, as were municipal-owned entities serving a population greater than 50,000. Finally, for cooperatives, investor-owned utilities, and subdivisions that generate less than 4 billion kWh of electricity annually but may be part of a large entity, additional research on power sales, operating revenues, and other business activities was performed to make a final determination regarding size. Because the rule does not affect units with a generating capacity of less than 25 MW, small entities that do not own at least one generating unit with a capacity greater than or equal to 25 MW were dropped from the data set. According to EPA's analysis, nearly 600 small entities were exempted by this provision. Finally, small entities for which IPM does not project generation in 2014 in the base case were omitted from the analysis because they are not projected to be operating and thus will not face the costs of compliance with the Transport Rule. After omitting entities for the reasons above, EPA identified a total of 81 potentially affected small entities, out of a possible 760.[79] The number of potentially affected small entities by ownership type is listed in Table 9-2.

9.1.2 Overview of Analysis and Results

This section presents the methodology and results for estimating the impact on the Transport Rule to small entities in 2014 based on the following endpoints:

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[78] For details, see http://www.ventyx.com/

[79] There are 82 entities that are not technically electricity generating utilities so we applied other criteria that would apply to financial or industrial companies and found that they did not meet the definitions of small entities in this context.

- annual economic impacts of the Transport Rule on small entities and
- ratio of small entity impacts to revenues from electricity generation.

9.1.2.1 Methodology for Estimating Impacts of the Transport Rule on Small Entities

An entity can comply with the Transport Rule through some combination of the following: installing retrofit technologies, purchasing allowances, switching to a cleaner fuel, or reducing emissions through a reduction in generation or improved efficiency. Additionally, units with more allowances than needed can sell these allowances in the market. The chosen compliance strategy will be primarily a function of the unit's marginal control costs and its position relative to the marginal control costs of other units.

To attempt to account for each potential control strategy, EPA estimates compliance costs as follows:

$$C_{Compliance} = \Delta\, C_{Operating+Retrofit} + \Delta\, C_{Fuel} + \Delta\, C_{Allowances} + \Delta\, C_{Transaction} + \Delta\, R$$

where C represents a component of cost as labeled, and Δ R represents the value of foregone electricity generation, calculated as the difference in revenues between the base case and the Transport Rule.

In reality, compliance choices and market conditions can combine such that an entity may actually experience a savings in any of the individual components of cost. Under the Transport Rule, some units will forgo some level of electricity generation (and thus revenues) to comply and this impact will be lessened on these entities by the projected increase in electricity prices under the Transport Rule. On the other hand, those increasing generation levels will see an increase in electricity revenues and as a result, lower net compliance costs. If entities are able to increase revenue more than an increase in fuel cost and other operating costs, ultimately they will have negative net compliance costs (or savings). Elsewhere, units burning high or medium sulfur coal might decide to pay relatively more for low-sulfur coal under the Transport Rule and sell allowances on the market, in the hopes of negating some or all of their compliance cost. Overall, small entities are not projected to install relatively costly emissions control retrofits, but may choose to do so in some instances. Because this analysis evaluates the total costs along each of the compliance strategies laid out above for each entity, it inevitably captures savings or gains such as those described. As a result, what we describe as cost is really more of a measure of the net economic impact of the rule on small entities.

For this analysis, EPA used IPM-parsed output to estimate costs based on the parameters above, at the unit level. These impacts were then summed for each small entity, adjusting for ownership share. Net impact estimates were based on the following: operating and retrofit costs, sale or purchase of allowances, and the change in fuel costs or electricity generation revenues under the Transport Rule relative to the base case. These individual components of compliance cost were estimated as follows:

(1) **Operating and retrofit costs:** Using the IPM-parsed output for the base case and the Transport Rule, EPA identified units that install control technology under the Transport Rule and the technology installed. The equations for calculating retrofit costs were adopted from EPA's version of IPM. The model calculates the capital cost (in \$/MW); the fixed operation and maintenance (O&M) cost (in \$/MW-year); the variable O&M cost (in \$/MWh); and the total annualized retrofit cost for units projected to install FGD, SCR, or SNCR.

(2) **Sale or purchase of allowances:** EPA estimated the value of initial SO_2 and NO_x annual and NO_x ozone season allowance holdings. For both SO_2 and NO_x, the state emission budgets were assumed to be apportioned to units based on their share of the state's total emissions. EPA calculated each unit's SO_2 and NO_x annual and NO_x ozone season allowance allocations as the ratio of that unit's adjusted emissions (as determined in the state budget calculation methodology) to the sum of all units' emissions in the applicable state, times the final state budget. Thus each unit's allocation is the unit's proportional share of the state budget, based on emissions as determined in the state budget calculation. See State Budgets, Unit Allocations, and Unit Emissions Rates TSD.

To estimate the value of allowances holdings, allocated allowances were subtracted from projected emissions, and the difference was then multiplied by the allowance prices projected by IPM for 2014. Units were assumed to purchase or sell allowances to exactly cover their projected emissions under the Transport Rule.

(3) **Fuel costs:** Fuel costs were estimated by multiplying fuel input (MMBtu) by region and fuel-type-adjusted fuel prices (\$/MMBtu) from IPM. The change in fuel expenditures under the Transport Rule was then estimated by taking the difference in fuel costs between the Transport Rule and the base case.

(4) **Value of electricity generated:** EPA estimated electricity generation by first estimating unit capacity factor and maximum fuel capacity. Unit capacity factor is estimated by dividing fuel input (MMBtu) by maximum fuel capacity (MMBtu). The maximum fuel capacity was estimated by multiplying capacity (MW) * 8,760 operating hours * heat rate (MMBtu/MWh). The value of electricity generated is then estimated by multiplying capacity (MW) * capacity factor * 8,760 * regional-adjusted retail electricity price ($/MWh), for all entities except those categorized as "Private" in Ventyx. For private entities, EPA used wholesale electricity price instead retail electricity price because most of the private entities are independent power producers (IPP). IPPs sell their electricity to wholesale purchasers and do not own transmission facilities and thus their revenue was estimated with wholesale electricity prices.

As discussed later in this analysis, 75 percent of small entities projected to be affected by the Transport Rule do not have to operate in a competitive market environment and thus should be able to pass compliance costs on to consumers. We defined cost of service regions as regions with a deregulation percentage of less than 20 percent. The deregulation percentage is defined for this analysis as a percentage estimating the degree of competition in electricity market, as provided by EIA. The lower this percentage means that there are more areas with cost of service market characteristics. We have used the estimates published in AEO 2009.

To somewhat account for this cost pass-through, we incorporated the projected regional-adjusted retail electricity price calculated under the Transport Rule in our estimation of generation revenue under the Transport Rule.

(5) **Administrative costs:** Because most affected units are already monitored as a result of other regulatory requirements, EPA considered the primary administrative cost to be transaction costs related to purchasing or selling allowances. EPA assumed that transaction costs were equal to 1.5 percent of the total absolute value of a unit's allowances. This assumption is based on market research by ICF International.

9.1.2.2 Results

The potential impacts of the Transport Rule on small entities are summarized in Table 9-2. All costs are presented in $2006. EPA estimated the annualized net compliance cost to small entities to be approximately - $35.9 million in 2014 or savings of $35.9 million.[80] The fact that the net compliance costs for all entities are actually net savings does not mean that each small entity would benefit from the Transport Rule. The net savings are driven by a few entities that are able to increase their revenues by increasing generation and taking advantage of higher electricity prices.

Table 9-2. Projected Impact of the Transport Rule on Small Entities in 2014

EGU Ownership Type	Number of Potentially Affected Entities	Total Net Compliance Cost ($2006 millions)	Number of Small Entities with Compliance Costs >1% of Generation Revenues	Number of Small Entities with Compliance Costs >3% of Generation Revenues
Cooperative	16	-$27.4	5	3
Investor-Owned Utility	3	-$4.8	0	0
Municipal	46	-$8.2	19	7
Subdivision	5	-$3.3	1	0
Private	11	$7.7	5	4
Total	81	-$35.9	30	14

Note: The total number of potentially affected entities in this table excludes around 600 entities that have been dropped because they will not be affected by the Transport Rule. Also, the total number of entities with costs greater than 1 percent or 3 percent of revenues includes only entities experiencing positive costs. A negative cost value implies that the group of entities experiences a net savings under the Transport Rule.

Source: IPM analysis

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[80] Neither the costs nor the revenues of units that retire under the Transport Rule are included in the impact estimates. Because these units are better off retiring under the Transport Rule than continuing operation, the true cost of the rule on these units is not represented by our modeling. The true cost of the Transport Rule for these units is the differential between their costs in the base case and the costs of meeting their customers' demand under the rule.

EPA assessed the economic and financial impacts of the rule using the ratio of compliance costs to the value of revenues from electricity generation, focusing in particular on entities for which this measure is greater than 1 percent. Although this metric is commonly used in EPA impact analyses, it makes the most sense when as a general matter an analysis is looking at small businesses that operate in competitive environments. However, small businesses in the electric power industry often operate in a price-regulated environment where they are able to recover expenses through rate increases. Given this, EPA considers the 1 percent measure in this case a crude measure of the price increases these small entities will be asking of rate commissions or making at publicly owned companies.

Of the 81 small entities considered in this analysis, 30 entities may experience compliance costs greater than 1 percent of generation revenues in 2014. Entities that experience negative net costs under the Transport Rule are excluded from these totals. These results do not fully account for the reality that about three-quarters of these entities operate in cost of service markets and thus should be able to recover all of their costs of complying with the Transport Rule. Furthermore, of the approximately 550 units identified by EPA as being potentially owned by small entities, approximately two-thirds of the units that have higher costs are not expected to make operational changes as a result of this rule (e.g. install control equipment or switch fuels). Their increased costs are largely due to increased cost of the fuel they would be expected to use whether or not they had to comply with the proposed rule. Increased fuel costs are often passed through to rate-payers as common practice in many areas of the U.S. due to fuel adder arrangements instituted by state public utility commissions. Finally, EPA's decision to exclude units smaller than 25 MW has already significantly reduced the burden on small entities by nearly 600. Hence, EPA has concluded that there is no significant economic impact on a substantial number of small entities (No SISNOSE) for this rule. The number of entities with compliance costs exceeding 3 percent of generation revenues is also included in Table 9-2.

The distribution across entities of economic impacts as a share of base case revenue is summarized in Table 9-3. Although the distributions of economic impacts on each ownership type are in general fairly tight, there are a few outliers for which the percentage of economic impacts as a share of revenue is either very low or very high relative to the capacity-weighted average. In the cases where entities are projected to experience negative net impacts that are a high percentage of revenues, these entities have units that are able to increase generation with the Transport Rule, thus increasing revenues. In the cases where entities are projected to experience positive net impacts that are a high percentage of revenues, these entities do not find it economic to retrofit and are unable to switch to a lower sulfur coal. Thus, another reason for entities incurring impacts is that they are expected to reduce their generation under the Transport Rule which reduces revenues collected from electricity sales and inflates net costs.

Table 9-3. Summary of Distribution of Economic Impacts of the Transport Rule on Small Entities in 2014

EGU Ownership Type	Capacity-Weighted Average Economic Impacts as a % of Generation Revenues	Min	Max
Cooperative	-3.1%	-29.5%	14.4%
Investor-owned utility	-3.5%	-7.5%	0.3%
Municipal	2.7%	-19.7%	50.5%
Subdivision	0.0%	-2.4%	2.5%
Private	14.3%	-2.1%	86.1%
All	**3.5%**	**-29.5%**	**86.1%**

Source: IPM analysis

The separate components of annualized costs to small entities under the Transport Rule are summarized in Table 9-4. The most significant components of incremental cost to these entities under the Transport Rule are due to lower electricity revenues and increased fuel costs. Fuel costs increase over all ownership groups except the ones under the ownership type "Private" because an entity with the second largest generation under "Private" is projected to cut its generation by 25 percent under the Transport Rule, which translates to lower fuel costs for the whole group. Additionally, increases in electricity generation revenue, shown as cost savings or negative costs are experienced by cooperative, investor-owned utility, municipal and subdivision entities. This is due largely to the projected increase in electricity prices under the Transport Rule. Among the private category, however, reduced generation by the one entity with a large share of generation leads to higher net costs for the entire category. Our data suggests this entity owns a group of combined cycle units and which are presumably marginal units in their respective load segments under the base case.

Table 9-4. Incremental Annualized Costs under the Transport Rule Summarized by Ownership Group and Cost Category in 2014 ($2006 millions)

EGU Ownership Type	Retrofit + Operating Cost	Net Purchase of Allowances	Fuel Cost	Lost Electricity Revenue	Administrative Cost
Cooperative	$6.1	$1.8	$11.2	-$46.5	$0.1
Investor-Owned Utility	$0.7	$0.0	$4.7	-$10.2	$0.0
Municipal	$5.8	-$4.0	$13.0	-$23.0	$0.0
Subdivision	$1.6	$0.6	$4.0	-$9.6	$0.0
Private	$4.5	$0.1	-$23.6	$26.9	$0.0

Source: IPM analysis.

Furthermore, 26 MW of total small entity capacity, or 0.04 percent of total small entity capacity in the Transport Rule region, is projected to be uneconomic to maintain under the Transport Rule relative to the base case. To put these numbers in context, of all affected capacity under the Transport Rule, about 1.9 GW (0.5 percent) of coal-fired capacity is projected to be uneconomic to maintain relative to the base case. This comparison suggests that small entities should not be disproportionately affected by the Transport Rule. In practice, units projected to be uneconomic to maintain may be mothballed, retired, or kept in service to ensure transmission reliability in certain parts of the grid. Our IPM modeling is unable to distinguish between these potential outcomes.

9.1.3 Summary of Small Entity Impacts

EPA examined the potential economic impacts to small entities associated with this rulemaking based on assumptions of how the affected states will implement control measures to meet their emissions. To summarize, of the 81 small entities potentially affected, and the 760 small entities in the Transport Rule region that are included in EPA's modeling, 30 may experience compliance costs in excess of 1 percent of revenues in 2014, based on assumptions of how the affected states implement control measures to meet their emissions budgets as set forth in this rulemaking. Potentially affected small entities experiencing compliance costs in excess of 1 percent of revenues have some potential for significant impact resulting from implementation of the Transport Rule. However, as noted above, it is EPA's position that because very few of the affected entities currently operate in a competitive market environment, they should generally be able to pass the costs of complying with the Transport Rule on to rate-payers. Furthermore, the decision to include only units greater than 25 MW in size exempts around 600 small entities that would otherwise be potentially affected by the Transport Rule.

9.2 Unfunded Mandates Reform Act (UMRA) Analysis

Title II of the UMRA of 1995 (Public Law 104-4)(UMRA) establishes requirements for federal agencies to assess the effects of their regulatory actions on state, local, and Tribal governments and the private sector. Under Section 202 of the UMRA, 2 U.S.C. 1532, EPA generally must prepare a written statement, including a cost-benefit analysis, for any proposed or final rule that includes any Federal mandate that may result in the expenditure by State, local, and Tribal governments, in the aggregate, or by the private sector, of $100,000,000 or more ... in any one year. A Federal mandate is defined under Section 421(6), 2 U.S.C. 658(6), to include a Federal intergovernmental mandate and a Federal private sector mandate. A Federal intergovernmental mandate, in turn, is defined to include a regulation that would impose an enforceable duty upon State, Local, or Tribal governments, Section 421(5)(A)(i), 2 U.S.C. 658(5)(A)(i), except for, among other things, a duty that is a condition of Federal assistance, Section 421(5)(A)(i)(I). A Federal private sector mandate includes a regulation that would impose an enforceable duty upon the private sector, with certain exceptions, Section 421(7)(A), 2 U.S.C. 658(7)(A).

Before promulgating an EPA rule for which a written statement is needed under Section 202 of the UMRA, Section 205, 2 U.S.C. 1535, of the UMRA generally requires EPA to identify and consider a reasonable number of regulatory alternatives and adopt the least costly, most cost-effective, or least burdensome alternative that achieves the objectives of the rule. EPA included descriptions of three remedy options that it considered when developing its proposed rule: (1) the proposed remedy of State Budgets/Limited Trading, (2) State Budgets/Intrastate Trading, and (3) Direct Controls. Moreover, section 205 allows EPA to adopt an alternative other than the least costly, most cost-effective or least burdensome alternative if the Administrator publishes with the final rule an explanation why that alternative was not adopted.

Furthermore, as EPA stated in the proposal, EPA is not directly establishing any regulatory requirements that may significantly or uniquely affect small governments, including Tribal governments. Thus, under the proposed Transport Rule, EPA is not obligated to develop under Section 203 of the UMRA a small government agency plan.

EPA analyzed the economic impacts of the Transport Rule on government entities. This analysis does not examine potential indirect economic impacts associated with the Transport Rule, such as employment effects in industries providing fuel and pollution control equipment, or the potential effects of electricity price increases on industries and households.

9.2.1 Identification of Government-Owned Entities

Using Ventyx data, EPA identified state- and municipality-owned utilities and subdivisions in the Transport Rule region. EPA then used IPM-parsed output to associate these plants with individual generating units. Entities that did not own at least one unit with a generating capacity of greater than 25 MW were omitted from the analysis because of their exemption from the rule. This exempts 380 entities owned by state or local governments. Additionally, government-owned entities for which IPM does not project generation in 2014 under the base case or the Transport Rule were exempted from this analysis, because they are not projected to be operating and thus will not face the costs of compliance with the Transport Rule. Twenty-five entities were dropped from the analysis for this reason. Out of the 380 and 25 dropped entities, 7 of them are both less than 25 MW and not projected to operate in 2014. Thus, EPA identified 84 state and municipality-owned utilities that are potentially affected by the Transport Rule, out of a possible 482, which are summarized in Table 9-5.

9.2.2 Overview of Analysis and Results

After identifying potentially affected government entities, EPA estimated the impact of the Transport Rule in 2014 based on the following:

- total impacts of compliance on government entities and
- ratio of government entity impacts to revenues from electricity generation.

The financial burden to owners of EGUs under the Transport Rule is composed of compliance and administrative costs. This section outlines the compliance and administrative costs for the 84 potentially affected government-owned units in the Transport Rule region.

9.2.2.1 Methodology for Estimating Impacts of the Transport Rule on Government Entities

The primary burden on state and municipal governments that operate utilities under the Transport Rule is the cost of installing control technology on units to meet SO_2 and NO_x emission limits or the cost of purchasing allowances. However, an entity can comply with the Transport Rule through any combination of the following: installing retrofit technologies, purchasing allowances, switching to a cleaner fuel, or reducing emissions through a reduction in generation. Additionally, units with more allowances than needed can sell these allowances on the market. The chosen compliance strategy will be primarily a function of the unit's marginal control costs and its position relative to the marginal control costs of other units.

To attempt to account for each potential control strategy, EPA estimates compliance costs as follows:

$$C_{Compliance} = \Delta\ C_{Operating+Retrofit} + \Delta\ C_{Fuel} + \Delta\ C_{Allowances} + \Delta\ C_{Transaction} + \Delta\ R$$

where C represents a component of cost as labeled, and $\Delta\ R$ represents the retail value of foregone electricity generation.

In reality, compliance choices and market conditions can combine such that an entity may actually experience a savings in any of the individual components of cost. Under the Transport Rule, for example, some units will forgo some level of electricity generation (and thus revenues) to comply, this impact will be lessened on these entities by the projected increase in electricity prices under the Transport Rule, while those not reducing generation levels will see an increase in electricity revenues. Because this analysis evaluates the total costs along each of the four compliance strategies laid out above for each entity, it inevitably captures savings or gains such as those described. As a result, what we describe as cost is really more of a measure of the net economic impact of the rule on small entities.

In this analysis, EPA used IPM-parsed output for the base case and the Transport Rule to estimate compliance cost at the unit level. These costs were then summed for each small entity, adjusting for ownership share. Compliance cost estimates were based on the following: operating and retrofit costs, sale or purchase of allowances, and the change in fuel costs or electricity generation revenues under the Transport Rule relative to the base case. These components of compliance cost were estimated as follows:

(1) **Retrofit and operating costs:** Using the IPM-parsed output for the base case and the Transport Rule, EPA identified units that install control technology under the Transport Rule and the technology installed. The equations for calculating retrofit costs for SCR, SNCR, and FGD were adopted from EPA's TRUM. The model calculates the capital cost (in \$/MW), the fixed O&M cost (in \$/MW-year), the variable O&M cost (in \$/MWh), and the total annualized retrofit and operating cost by unit.

(2) **Sale or purchase of allowances:** EPA estimated the value of initial SO_2 and NO_x annual and NO_x ozone season allowance holdings. For both SO_2 and NO_x, the state emission budgets were assumed to be apportioned to units based on their share of the state's total emissions. EPA calculated each unit's

SO_2 and NO_x annual and NO_x ozone season allowance allocations as the ratio of that unit's adjusted emissions (as determined in the state budget calculation methodology) to the sum of all units' emissions in the applicable state, times the final state budget. Thus each unit's allocation is the unit's proportional share of the state budget, based on emissions as determined in the state budget calculation. See State Budgets, Unit Allocations, and Unit Emissions Rates TSD.

To estimate the value of allowances holdings, allocated allowances were subtracted from projected emissions, and the difference was then multiplied by the allowance price projected by IPM. Units were assumed to purchase or sell allowances to exactly cover their projected emissions under the Transport Rule.

(3) **Fuel costs:** Fuel costs were estimated by multiplying fuel input (MMBtu) by region and fuel type-adjusted fuel prices ($/MMBtu) from TRUM. The change in fuel expenditures under the Transport Rule was then estimated by taking the difference in fuel costs between the Transport Rule and the base case.

(4) **Value of electricity generated:** EPA estimated electricity generation by first estimating the unit capacity factor and maximum fuel capacity. The unit capacity factor is estimated by dividing fuel input (MMBtu) by maximum fuel capacity (MMBtu). The maximum fuel capacity was estimated by multiplying capacity (MW) * 8,760 operating hours * heat rate (MMBtu/MWh). The value of electricity generated was then estimated by multiplying capacity (MW) * capacity factor * 8,760 * regional-adjusted retail electricity price ($/MWh).

(5) **Administrative costs:** Because most affected units are already monitored as a result of other regulatory requirements, EPA considered the primary administrative cost to be transaction costs related to purchasing or selling allowances. EPA assumed that transaction costs were equal to 1.5 percent of the total absolute value of a unit’s allowances. This assumption is based on market research by ICF International.

9.2.2.2 Results

A summary of economic impacts on government-owned entities is presented in Table 9-5. According to EPA’s analysis, the total net economic impact on each category of government-owned entity (state- and municipality-owned utilities and subdivisions) is expected to be negative in 2014.[81]

Table 9-5. Summary of Potential Impacts on Government Entities under the Transport Rule in 2014

EGU Ownership Type	Potentially Affected Entities	Projected Annualized Costs ($2006 millions)	Number of Government Entities with Compliance Costs >1% of Generation Revenues	Number of Government Entities with Compliance Costs >3% of Generation Revenues
Subdivision	7	$8.1	2	0
State	3	-$12.3	1	1
Municipal	74	-$11.4	24	7
Total	**84**	**-$15.7**	**27**	**8**

Note: The total number of potentially affected entities in this table excludes the 482 entities that have been dropped because they will not be affected by the Transport Rule. Also, the total number of entities with costs greater than 1 percent or 3 percent of revenues includes only entities experiencing positive costs. A negative cost value implies that the group of entities experiences a net savings under the Transport Rule.

Source: IPM analysis

As was done for the small entities analysis, EPA further assessed the economic and financial impacts of the rule using the ratio of compliance costs to the value of revenues from electricity generation in the base case, also focusing specifically on entities for which this measure is greater than 1 percent.[82] EPA projects that 27 government entities will have

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[81] All costs are reported in 2006 dollars.

[82] Neither the costs nor the revenues of units that retire under the Transport Rule are included in this portion of the analysis. Because these units are better off retiring under the Transport Rule than continuing operation, the true cost of the rule on these units is not represented by our modeling. The true cost of the Transport Rule for these units is the differential between their costs in the base case and the costs of meeting their customers' demand under the rule.

compliance costs greater than 1 percent of revenues from electricity generation in 2014. Also similar to the small entity analysis, the majority of the units that have higher costs are not expected to make operational changes as a result of this rule (e.g., install control equipment or switch fuels). Their increased costs are largely due to increased cost of the fuel they would be expected to use whether or not they had to comply with the proposed rule. Further, increased fuel costs are often passed through to rate-payers as common practice in many areas of the U.S. due to fuel adder arrangements instituted by state public utility commissions. Entities that are projected to experience negative compliance costs under the Transport Rule are not included in those totals. This approach is more indicative of a significant impact when an analysis is looking at entities operating in a competitive market environment. Government-owned entities do not operate in a competitive market environment and therefore will be able to recover expenses under the Transport Rule through rate increases. Given this, EPA considers the 1 percent measure in this case a crude measure of the extent to which rate increases will be made at publicly owned companies.

The distribution across entities of economic impacts as a share of base case revenue is summarized in Table 9-6. For municipality-owned entities and subdivisions, the maximum economic impact as a share of base case revenues is approximately 50.5 and 2.5 percent, respectively. A few municipality-owned entities experience economic impacts that are significantly higher than the capacity-weighted average for this group. In the cases where entities are projected to experience positive net costs that are a high percentage of revenues, these entities do not find it economic to retrofit and are unable to switch to a lower-sulfur coal. Thus, another reason for entities incurring impacts is that they are expected to reduce their generation under the Transport Rule which reduces revenues collected from electricity sales and inflates net costs.

Table 9-6. Distribution of Economic Impacts on Government Entities under the Transport Rule in 2014

EGU Ownership Type	Capacity-Weighted Average Economic Impacts as a % of Generation Revenues	Min	Max
Sub-division	0.7%	-2.4%	2.5%
State	-2.7%	-4.0%	4.3%
Municipal	-0.2%	-54.5%	50.5%
All	**-0.0%**	**-54.5%**	**50.5%**

Source: IPM analysis

Additionally, a few entities are projected to experience negative net costs that are a high percentage of base case revenues. These entities have units that are able to increase generation levels, thus increasing revenues. Additionally, entities in regions for which we project large electricity price increases relative to other regions tend to be among those at the lower end of the distribution.

The various components of annualized incremental cost under the Transport Rule to each group of government entities are summarized in Table 9-7. In 2014, subdivisions are a net purchaser of allowances, while states and municipalities sell allowances. Additionally, each group experiences both an increase in fuel expenditures and an increase in electricity revenue under the Transport Rule. Incremental fuel costs are positive because these entities are projected to increase generation and face higher fuel prices. Overall, increases in total electricity revenue by government entities under the Transport Rule exceed the increases in fuel and operating costs.

Table 9-7. Incremental Annualized Costs under the Transport Rule Summarized by Ownership Group and Cost Category ($2006 millions) in 2014

EGU Ownership Type	Retrofit + Operating Cost	Net Purchase of Allowances	Fuel Cost	Lost Electricity Revenue	Administrative Cost
Subdivision	$2.9	$3.0	$20.3	-$18.2	$0.1
State	$9.0	$0.2	$12.6	-$34.0	$0.0
Municipal	$30.2	$6.2	$49.6	-$97.8	$0.4

Source: ICF International analysis based on IPM analysis

IPM modeling of the Transport Rule projects that approximately 60 MW (2 units of 347 in this analysis) of municipality-owned capacity would be uneconomic to maintain under the Transport Rule, beyond what is projected in the base case. This represents about 0.1 percent of all subdivision, state, and municipality capacity in the Transport Rule region. For comparison, overall affected capacity under the Transport Rule, about 1.9 GW, or 0.5 percent of all coal-fired capacity is projected to be uneconomic to maintain relative to the base case. This comparison suggests that government entities should not face a disproportionate burden under the Transport Rule. In practice, units projected to be uneconomic to maintain may be "mothballed," retired, or kept in service to ensure transmission reliability in certain parts of the grid. Our IPM modeling is unable to distinguish between these potential outcomes.

9.2.3 Summary of Government Entity Impacts

EPA examined the potential economic impacts on state and municipality-owned entities associated with this rulemaking based on assumptions of how the affected states will implement control measures to meet their emissions. According to EPA's analysis, the total net economic impact on government-owned entities is expected to be over -$15 million in 2014 or a net savings of more than $15 million. This does not mean that each government entity will experience net savings as the overall net savings is driven by several entities garnering large savings. Of the 84 government entities considered in this analysis and the 482 government entities in the Transport Rule region that are included in EPA's modeling, 27 may experience compliance costs in excess of 1 percent of revenues in 2014, based on our assumptions of how the affected states implement control measures to meet their emissions budgets as set forth in this rulemaking.

Government entities projected to experience compliance costs in excess of 1 percent of revenues have some potential for significant impact resulting from implementation of the Transport Rule. However, as noted above, it is EPA's position that because these government entities can pass on their costs of compliance to rate-payers, they will not be significantly affected. Furthermore, the decision to include only units greater than 25 MW in size exempts 380 government entities that would otherwise be potentially affected by the Transport Rule.

9.3 Paperwork Reduction Act

In compliance with the Paperwork Reduction Act (44 U.S.C. 3501 et seq.), EPA submitted a proposed Information Collection Request (ICR) (EPA ICR number 2512.01) to the Office of Management and Budget (OMB) for review and approval on July 19, 2004 (FR 42720-42722). The ICR describes the nature of the information collection and its estimated burden and cost associated with the final rule. In cases where information is already collected by a related program, the ICR takes into account only the additional burden. This situation arises in states that are also subject to requirements of the Consolidated Emissions Reporting Rule (EPA ICR number 0916.10; OMB control number 2060-0088) or for sources that are subject to the Acid Rain Program (EPA ICR number 1633.13; OMB control number 2060-0258) or NO_x SIP Call (EPA ICR number 1857.03; OMB number 2060-0445) requirements.

EPA solicited comments on specific aspects of the information collection. The purpose of the ICR is to estimate the anticipated monitoring, reporting, and record-keeping burden estimates and associated costs for states, local governments, and sources that are expected to result from the Transport Rule.

The record-keeping and reporting burden to sources resulting from states choosing to participate in a regional cap-and-trade program is approximately $28 million annually. This estimate includes the annualized cost of installing and operating appropriate SO_2 and NO_x

emissions monitoring equipment to measure and report the total emissions of these pollutants from affected EGUs (serving generators greater than 25 megawatts capacity). The burden to state and local air agencies includes any necessary SIP revisions, performance of monitoring certification, and fulfilling of audit responsibilities. More information on the ICR analysis is included in the official the Transport Rule docket.

In accordance with the Paperwork Reduction Act on July 19, 2004, an ICR was made available to the public for comment. The 60-day comment period expired September 19, 2004, with no public comments received specific to the ICR.

9.4 Environmental Justice

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

9.4.1. Consideration of Environmental Justice Issues in the Rule Development Process

In the rulemaking process EPA considers whether there are positive or negative impacts of the action that appear to affect low-income, minority, or Tribal communities disproportionately, and, regardless of whether a disproportionate effect exists, whether there is a chance for these communities to meaningfully participate in the rulemaking process. EPA expects that this rule "Federal Implementation Plans to Reduce Interstate Transport of Fine Particulate Matter and Ozone" will provide significant health and environmental benefits to, among others, people with asthma, people with heart disease, and people living in ozone or fine particle ($PM_{2.5}$) nonattainment areas. This rule also has the potential to affect the cost structure of the utility industry and could lead to regional shifts in electricity generation and/or emissions of various pollutants. Therefore we expect this rule to be of interest to many environmental justice communities. EPA's analysis of the effects of this proposed rule, including information on air quality changes and the resulting health benefits, is presented both in section IX of the preamble and in more detail in the air quality modeling Technical Support Document and chapters 3 and 4 of this Regulatory Impact Analysis (RIA). These documents can be accessed through the rule docket No. EPA-HQ-OAR-2009-0491 and from the main EPA webpage for the rule www.epa.gov/airtransport. This section summarizes the legal basis for this rule, and provides background information on how this rule fits into the larger regulatory strategy for controlling pollution from the power sector. A summary of the emissions, air quality, and health benefit estimates for this rule then follows.

This rule is replacing an earlier rule (the 2005 Clean Air Interstate Rule (CAIR)) that was first vacated and then remanded to EPA by the U.S. Court of Appeals for the District of Columbia Circuit. CAIR was vacated by the U.S. Court of Appeals for the District of Columbia Circuit in July 2008 in a case known as North Carolina v EPA. In December 2008, the vacatur was altered to a remand based on the likely environmental harms of vacating the rule and EPA's stated intent to replace the rule promptly. At the time of the 2008 court ruling, many sources had already begun to install and run emissions control devices or otherwise alter their operations and had successfully begun reducing their emissions. The court decision has led to significant uncertainty among affected sources as to what emission reductions will be required and among states and communities as to what air quality benefits will be achieved. By proposing this aggressive replacement rule that meets the legal requirements of the Clean Air Act (CAA) as interpreted by the Court in the North Carolina decision promptly, EPA is both maximizing the likelihood that the goals of the CAA will be met, and helping communities receive the air quality benefits they need as quickly as possible by minimizing the chance that any emission reductions achieved under CAIR would be lost.

It is important to note that CAA § 110(a)(2)(d), which addresses transport of criteria pollutants between states and is the authority for this rule, is only one of many provisions of the CAA that provide EPA, states, and local governments with authorities to reduce exposure to ozone and $PM_{2.5}$ in communities. These legal authorities work together to reduce exposure to these pollutants in communities, including environmental justice communities, and provide substantial health benefits to both the general public and sensitive sub-populations.

9.4.2. Potential Environmental and Public Health Impacts to Vulnerable Populations

There are several considerations to take into account when assessing the effects of this proposed rule on minority, low-income, and Tribal populations. These include: amount of emissions reductions and where they take place (including any potential for areas of increased emissions); the changes in ambient concentrations across the affected area; and the health benefits expected from the rules.

Emission reductions. This proposed rule will reduce exposure to $PM_{2.5}$ and ozone pollution in most eastern states by reducing interstate transport of these pollutants and their chemical precursors (sulfur dioxide (SO_2) and nitrogen oxides (NO_x)). This rule has the effect of reducing emissions of these pollutants that affect the most-contaminated areas (i.e. areas that are not meeting the 1997 and 2006 ozone and $PM_{2.5}$ National Ambient Air Quality Standards (NAAQS)). This rule separately identifies both nonattainment areas and maintenance areas (maintenance areas are those that currently meet the NAAQS but that, based on past data, are in danger of exceeding the standards in the future). This approach of requiring emission reductions to protect maintenance areas as well as nonattainment areas reduces the likelihood that any areas close to the level of the standard will exceed the *current* health-based standards in the future.

Ozone and $PM_{2.5}$ concentrations in both nonattainment and maintenance areas identified in this rule are the result of both local emissions and long-range transport of pollution. This rule requires upwind states to reduce or eliminate their significant contribution to nonattainment or maintenance problems in downwind states. Even when the significant contributions of upwind states are fully eliminated, additional emissions reductions within the nonattainment area and/or the downwind state will be needed for some areas to attain and maintain the NAAQS.

The proposed remedy option for this rule would use a limited emissions trading mechanism among power plants to achieve significant emission reductions in states covered by the rule. EPA recognizes that many environmental justice communities have voiced concerns about emissions trading and any resulting potential for any emissions increases in any location.

The proposed rule uses EPA's authority in CAA §110(a)(2)(d) to require states to eliminate emissions from power plants in their state that contribute significantly to downwind $PM_{2.5}$ or ozone nonattainment or maintenance areas. EPA's proposed mechanism for achieving these emission reductions is to use a tightly constrained trading program that requires a strict emission ceiling in each state while allowing a limited ability to shift emissions between facilities or states. This approach ensures that emissions in each state that significantly contribute to downwind nonattainment or maintenance areas are controlled, while allowing power companies to adjust generation based on fluctuations in electricity demand, weather, availability of low-emitting power sources (e.g. temporary shut-down of a nuclear power plant for maintenance or repairs), or other unanticipated factors affecting the interconnected electricity grid.

Any emissions above the state's allocated level must be offset by emission reductions from another state in the region below that state's budget or by using extra "banked" allowances from earlier years. All sources must hold enough allowances to cover their emissions; therefore, if they emit more than their allocation they must buy allowances from another source that emitted less than its allocation. $PM_{2.5}$ and ozone pollution from power plants have both local and regional components: part of the pollution in a given location – even in locations near emissions sources – is due to emissions from nearby sources and part is due to emissions that travel hundreds of miles and mix with emissions from other sources. Therefore, in many instances the exact location of the upwind reductions does not affect the levels of air pollution downwind.

It is important to recognize that the section of the Clean Air Act providing authority for this rule, 110(a)(2)(D), unlike some other provisions, does not dictate levels of control for particular facilities. None of EPA's alternatives within this proposal can ensure there will be no emission increases at any facility. Under the direct control alternative, the emission rate for each facility is reduced but each facility could emit more by increasing their power output in order to meet electricity reliability or other goals. Under the intrastate trading option, state emissions must stay constant but individual facilities within each state could increase their emissions as long as another facility in the state had decreased theirs. By strictly setting state budgets to eliminate the portion of significant contributions to non-attainment and maintenance areas that EPA has identified in today's action, by limiting the amount of interstate trading possible and by requiring any emissions above the level of the allocations to be offset by emission decreases elsewhere in the region, the proposed remedy options reduce ambient concentrations where they are most needed.

EPA's emissions modeling data indicate that nationwide SO_2 emissions from electric generating units (EGUs) will be approximately 6.4 million tons (60%) lower in 2014 than they were in 2005 (which is the year that the Clean Air Interstate Rule was finalized). Emissions would also decrease when compared to the base case (the base case estimates of SO_2 emissions in 2014 in the absence of this proposed rule or the Clean Air Interstate Rule it is replacing). SO_2 emissions under this proposed rule are projected to be approximately 4.4 million tons (50%) lower than they would have been in 2014 in the base case (i.e. without this rule).

EPA's modeling does project that some states not covered by one or more aspects of the program may experience increases of SO_2 emissions (i.e., their emissions are greater in the control case modeling than in the base case modeling). These emission increases are the result of forecasted changes in operation of units outside of the controlled region (due to the interconnected nature of the utility grid or influence of the rule on the market for lower sulfur coal). As shown in Table IV.D.6 of the preamble, Arkansas, Mississippi, North Dakota, South Dakota, and Texas all exhibit 2012 SO_2 emissions increases over the base case of more than 5,000 tons. Texas is projected to have by far the largest increase (136,000 tons), while the other states' increases range from 6,000 to 32,000 tons. Further analysis with the simplified air quality assessment tool indicates that these projected increases in the Texas SO_2 emissions would increase Texas's contribution to an amount that would exceed the 0.15 $\mu g/m^3$ threshold for annual $PM_{2.5}$. For this reason, EPA requests comment on whether Texas should be included in the program as a group 2 state. For additional details, <u>see</u> section IV.D of the preamble for this rule.

EPA's emissions modeling data indicates that nationwide ozone season NO_x emissions from EGUs will be approximately 400,000 tons (30%) lower in 2014 than they were in 2005 (before implementation of the Clean Air Interstate Rule). Emissions would also decrease compared to the base case. Ozone season NO_x emissions from EGUs under this proposed rule are projected to be approximately 150,000 tons (15%) lower than they would have been in 2014 in the base case (i.e. without this rule). EPA anticipates that additional upcoming actions, and likely additional interstate transport reductions to help states attain the proposed 2010 ozone NAAQS, will result in significant additional NO_x reductions.

EPA anticipates that this proposed action will significantly reduce, but not eliminate, the number of nonattainment and maintenance areas for the 1997 ozone and $PM_{2.5}$ and 2006 $PM_{2.5}$ NAAQS. Table IX-1 of the preamble lists the changes in number of nonattainment sites. Most of these sites are located in urban areas. A single nonattainment area usually contains multiple monitoring sites; therefore there are more nonattainment sites than nonattainment counties or areas. As discussed in detail in section IV.D of the preamble, where this proposal does not fully quantify all of the significant contribution and interference with maintenance, EPA intends to address these additional requirements quickly. To the extent possible, EPA will supplement this proposed notice with additional information so that we can provide downwind states with all the certainty about upwind emission reductions they need to address their own local nonattainment concerns. In addition, as stated above, elimination of these nonattainment areas may require both local and regional emission reductions and this proposed action seeks only to address the regional transport component.

As a result of these SO_2 and NO_x reductions, EPA's air quality modeling indicates that concentrations of fine particles will decline throughout the eastern U.S. and in all the states affected by this rule. These reductions are largest in the area of the Ohio River valley and neighboring states and extend east through New England, west to Texas, south to Florida, and north through the Great Lakes states. "Border" states immediately outside the transport region are also predicted to see reductions in air concentrations, even though emissions increase in some of these states. This is because concentrations of fine particles in most locations are composed of both local emissions and those transported over hundreds of miles and emission reductions far away can cause significant improvements in local air quality.

The modeling suggests also that there may be some small increases in $PM_{2.5}$ near locations in the western U.S. where SO_2 emissions are forecast to increase. These increases are small compared to the reductions predicted to take place in the eastern U.S. The increases are due to the regional nature of this rule (i.e. these states are not covered because sources in these states have not been found to contribute significantly to downwind nonattainment or maintenance areas) and the national nature of both coal markets and the Acid Rain Program allowance market. They are not the result of any particular type of remedy option (e.g. trading). EPA anticipates that future rulemakings, such as CAA section 112(d) standards and anticipated revisions to the 2006 fine particulate standards, are likely to reduce emissions in the areas not covered by this rule.

EPA's air quality modeling also indicates that concentrations of ozone will decline in much of the eastern U.S. These reductions are largest along much of the Gulf Coast and in Florida and in a region encompassing western Wisconsin, Iowa, Kansas, Missouri, Arkansas, and northeastern Oklahoma. These areas with the largest reductions are roughly the area immediately outside the boundaries of the NO_x SIP Call region. States in the SIP Call region were required to make significant reductions in NO_x beginning in 2003 and these emission reductions are included in the baseline modeling for this proposed Transport Rule and therefore not captured as additional benefits of this rulemaking.

As is common when modeling many NO_x control strategies, the air quality modeling for this proposed rule also suggests there may be a few small, localized areas in the Eastern U.S. where there are small increases in ozone concentrations. These generally small increases are a result of *reductions* in NO_x emissions in these local areas; they do not appear to represent a lack of NO_x emissions reductions or be the result of any specific emission control strategy (e.g. any type of trading). Rather, this phenomenon can result from complex atmospheric chemistry reactions taking place among chemical constituents of air pollution in these areas. Due to the complex photochemistry of ozone production, NO_x emissions lead to both the formation and destruction of ozone, depending on the relative quantities of NO_x, volatile organic compounds, and ozone formation catalysts. In the 2014 base case, NO_x emissions from sources in a few locations act to "quench" (i.e., lower) ozone compared to ozone concentrations in surrounding areas. The application of NO_x controls in these areas reduces this quenching effect, thereby increasing ozone to levels generally on par with those of the surrounding area. In this case it is uncertain whether the structure of the model itself is potentially exacerbating the spatial extent or magnitude of any ozone increases which might actually occur as a result of this rule. It should be noted that these same NO_x emission reductions that might be causing extremely localized ozone increases are certainly causing larger, more widespread improvements in ozone concentrations in downwind areas. Finally, as stated in the preamble, it is important to note that EPA intends to promulgate additional rules over the next few years that will further reduce concentrations of ozone and $PM_{2.5}$ and that the federal government and the states can and do use many different legal authorities to limit exposure to ozone.

Health benefits. This rule reduces concentrations of $PM_{2.5}$ and ozone pollution, exposure to which can cause, or contribute to, adverse health effects including premature mortality and many types of heart and lung diseases that affect many minority and low-income individuals, and Tribal communities. $PM_{2.5}$ and ozone are particularly (but not exclusively) harmful to children, the elderly, and people with existing heart and lung diseases, including asthma. Exposure to these pollutants can cause premature death and trigger heart attacks, asthma attacks in those with asthma, chronic and acute bronchitis, emergency room visits and hospitalizations, as well as milder illnesses that keep children home from school and adults home from work. High rates of both heart disease and asthma are a cause for concern in many environmental justice communities, making these populations more susceptible to air pollution health impacts. In addition, many individuals in these communities also lack access to high quality health care to treat these illnesses.

We estimate that in 2014 the PM-related annual benefits of the proposed remedy option include approximately 14,000 to 36,000 fewer premature mortalities, 9,200 fewer cases of chronic bronchitis, 22,000 fewer non-fatal heart attacks, 11,000 fewer hospitalizations (for respiratory and cardiovascular disease combined), 10 million fewer days of restricted activity due to respiratory illness and approximately 1.8 million fewer lost work days. We also estimate substantial health improvements for children in the form of fewer cases of upper and lower respiratory illness, acute bronchitis, and asthma attacks.

Ozone health-related benefits are expected to occur during the summer ozone season (usually ranging from May to September in the Eastern U.S.). Based upon modeling for 2014, annual ozone related health benefits are expected to include between 50 and 230 fewer premature mortalities, 690 fewer hospital admissions for respiratory illnesses, 230 fewer emergency room admissions for asthma, 300,000 fewer days with restricted activity levels, and 110,000 fewer days where children are absent from school due to illnesses. When adding the PM and ozone-related mortalities together, we find that the proposed remedy option for this rule will yield between 14,000 and 36,000 fewer premature mortalities. EPA has also estimated the benefits of the alternate remedies in this proposal using a benefit-per-ton estimation approach and found they would provide similar benefits.

It should be noted that, as discussed in the RIA for this action, there are other benefits to the emission reductions discussed here, such as improved visibility and, indirectly, reduced mercury deposition. Additional benefits of reducing emissions of SO_2 include reduced acidification of lakes and streams, and reduced mercury methylation; additional benefits of NO_x reductions include reduced acidification of lakes and streams and reduced coastal eutrophication. Conversely, it is possible that the modest increases in emissions modeled for this rule in some western areas could result in limited increases of one or more of these effects in these locations.

9.4.3. Meaningful Public Participation

During the comment period for this proposed rule, EPA intends to reach out specifically to environmental justice communities and organizations to notify them of the opportunity to provide comments on this rule and to solicit their comments on both this rule and the upcoming actions described above and in section III.E of the preamble. EPA will hold public hearings on this rule; see the information at the very beginning of the preamble for locations, times and dates. Comments can also be submitted in writing or electronically by following the instructions at the beginning of the preamble.

9.4.4 Summary

EPA believes that the vast majority of communities and individuals in areas covered by this rule, including numerous low-income, minority, and Tribal communities in both rural areas and inner cities in the East, will see significant improvements in air quality and resulting improvements in health. EPA also recognizes that there is the potential for a number of communities or individuals outside the region covered by this rule to experience slightly worse air quality as an indirect result of emission reductions required under this proposal. EPA requests comment on the impacts of this proposed action on low income, minority, and Tribal communities. EPA will further analyze environmental justice issues related to the impacts of the rule on those communities based both on additional data that may be developed and on comments on those issues prior to final action on this rule.

CHAPTER 10

COMPARISON OF BENEFITS AND COSTS

10.1 Comparison of Benefits and Costs

The estimated social costs to implement the proposed Transport Rule, as described in this document, are approximately \$2.03 or \$2.23 billion annually for 2014 (2006 dollars, 3 percent and 7 percent discount rate, respectively). Thus, the net benefits (social benefits minus social costs) of the program in 2014 are approximately \$120 to 292 + B billion or \$109 to 264 + B billion annually (2006 dollars, based on a discount rate of 3 percent and 7 percent, respectively and rounded to three significant figures). (B represents the sum of all unquantified benefits and disbenefits of the regulation.) Therefore, implementation of this rule is expected, based purely on economic efficiency criteria, to provide society with a significant net gain in social welfare, even given the limited set of health and environmental effects we were able to quantify. Addition of directly emitted $PM_{2.5}$-, mercury-, acidification-, and eutrophication-related impacts would likely increase the net benefits of the rule. Table 10-1 presents a summary of the benefits, costs, and net benefits of the final rule.

The capital costs spent for pollution controls installed for CAIR were not included in the annual social costs reported here since the Transport Rule did not lead to their installation. Those CAIR-related capital investments are roughly estimated to have an annual social cost less than 1.15 -1.29 billion dollars (under the two discount rates). EPA developed this estimate using the annual capital costs for compliance for the original CAIR rule in 2005 from its RIA modeling with IPM. That modeling estimated capital costs for 2010 and 2015 in 1999\$ that we first converted to 2006\$ and then interpolated to a 2012 value (\$1.6 billion). This value represents a rough estimate of the cost of the CAIR pollution controls that EPA recognizes were in place in this Transport Rule RIA. We then converted this estimate of compliance costs to social costs appropriate for comparison to the above costs used in the benefit cost analysis. Notably, several states and EPA settlement agreements in recent years have actually required the pollution controls that we projected in CAIR by 2014, so the range offered above should be viewed as a likely upper bound of the capital costs solely attributable to the original CAIR rules.

Air quality modeling was not conducted for the two alternative remedy options (state budgets/intrastate trading and direct control). We estimate the benefits of these alternatives by applying a benefit per-ton approach described in Chapter 5 of this RIA. While these benefit per ton estimates quantify the health impacts and monetized benefits of reductions in $PM_{2.5}$, they omit important welfare benefits including changes in recreational visibility. Table 10-2 below presents the social costs and health benefits, including net social benefit, of these two alternative remedies alongside that of the proposed Transport Rule remedy.

EPA also analyzed the costs and benefits of two scenarios that differ from the proposed State Budgets/Limited Trading in the stringency of SO_2 budgets beginning in 2014. Unlike the proposed Transport Rule, these scenarios are not the result of extensive analysis of each state's significant contribution. Nor do they necessarily represent EPA's view of what emission reductions are required by section 110(a)(2)(D)(i)(I) of the Clean Air Act. Rather, they are designed to show the effects of more or less stringent reduction requirements in a structure otherwise the same as the proposed State Budgets/Limited Trading remedy.

Both scenarios represent the same type of remedy as in the proposed FIPs and have the same 2012 requirements. In addition to requirements beginning in 2012 for annual NO_x, ozone-season NO_x, and SO_2, the proposed Transport Rule requires greater SO_2 reductions in 15 states (Group 1) beginning in 2014. The less stringent of the two scenarios considered here removes these 2014 requirements; instead, the 15 states maintain their 2012 requirements in all subsequent years. This results in budgets that, relative to the proposed Transport Rule, allow about 1.4 million tons more SO_2 to be emitted annually.

The more stringent of the two scenarios changes the requirement for Group 1 SO_2 emissions reductions beginning in 2014 and moves 8 additional states to Group 1 from Group 2. Connecticut, Florida, Kansas, Maryland, Massachusetts, Minnesota, Nebraska, and New Jersey join the 15 Group 1 states of the proposed rule, making 23 states in all and leaving 4 states and the District of Columbia in Group 2. Also, an additional 200,000 tons of SO_2 reduction beyond that required in the proposed rule is required in the 23 Group 1 states.

Air quality modeling was not conducted for these alternatives; thus we estimated the benefits of these alternatives by applying the same benefit per-ton approach as done for the alternative remedy options. The compliance costs of these alternatives are estimated using IPM. The social costs of these alternatives are estimated using EMPAX.[83] Table 10-3 presents the social costs and health benefits, including net social benefit, of the two scenarios alongside that of the proposed Transport Rule remedy.

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[83] Detailed results for this EMPAX run for these two alternatives can be found in the files "EMPAXresults_more stringent SO2 option" and "EMPAXresults_less stringent SO2 option," respectively, and these files are available in the docket for this rule.

Table 10-1. Summary of Annual Benefits, Costs, and Net Benefits of the Proposed Transport Rule in 2014[a] (billions of 2006 dollars)

Description	Proposed Remedy
Social costs[b]	
3 percent discount rate	\$2.03
7 percent discount rate	\$2.23
Social benefits[c,d,e]	
3 percent discount rate	\$122 to \$294 + B
7 percent discount rate	\$111 to \$266 + B
Health-related benefits:	
3 percent discount rate	\$119 to \$290 + B
7 percent discount rate	\$108 to \$262 + B
Visibility benefits	\$3.56
Net benefits (benefits-costs)[e,f]	
3 percent discount rate	\$120 to \$292
7 percent discount rate	\$109 to \$264

[a] When presenting benefits and net benefits, EPA traditionally rounds all estimates to two significant figures. In this case we have rounded to three significant digits to facilitate comparison of the benefits and costs among the preferred remedy, less and more stringent scenarios.

[b] Note that costs are the annualized total social costs of reducing pollutants including NO_x and SO_2 for the EGU source category in the proposed Transport Rule region in 2014. The social costs are the loss of household utility as measured in Hicksian equivalent variation. More information on the social costs can be found in Chapter 8 of this RIA.

[c] Total benefits are comprised primarily of monetized PM-related health benefits. The reduction in premature fatalities each year accounts for over 90 percent of total monetized benefits. Benefits in this table are nationwide (with the exception of ozone and visibility) and are associated with NO_x and SO_2 reductions. Ozone benefits represent benefits nationwide. Visibility benefits represent benefits in Class I areas in the southeast, southwest and California. The estimate of social benefits also includes CO_2-related benefits calculated using the social cost of carbon, discussed further in chapter 5.

[d] Not all possible benefits or disbenefits are quantified and monetized in this analysis. B is the sum of all unquantified benefits and disbenefits. Potential benefit categories that have not been quantified and monetized are listed in Table 1-4.

[e] Valuation assumes discounting over the SAB-recommended 20-year segmented lag structure. Results reflect the use of 3 percent and 7 percent discount rates consistent with EPA and OMB guidelines for preparing economic analyses (EPA, 2000; OMB, 2003).

[f] Net benefits are rounded to three significant figures. Columnar totals may not sum due to rounding.

Table 10-2. Summary of Annual Benefits, Costs, and Net Benefits of Versions of the Proposed Remedy Option in 2014[a] (billions of 2006 dollars)

Description	*Proposed Remedy*	*Less Stringent Scenario*	*More Stringent Scenario*
Social costs[b]			
3 percent discount rate	\$2.03	\$1.12*	\$2.21*
7 percent discount rate	\$2.23	\$1.23*	\$2.43*
Health-related benefits[c,d]			
3 percent discount rate	\$118 to \$288 + B	\$82 to 200 + B	\$120 to 292 + B
7 percent discount rate	\$107 to \$260 + B	\$76 to 184 + B	\$110 to 267 + B
Net benefits (benefits-costs)[e,f]			
3 percent discount rate	\$116 to \$286	\$81 to 200	\$118 to 290
7 percent discount rate	\$105 to \$258	\$74 to 182	\$107 to 265

[a] When presenting benefits and net benefits, EPA traditionally rounds all estimates to two significant figures. In this case we have rounded to three significant digits to facilitate comparison of the benefits and costs among the proposed remedy and the less and more stringent scenarios.

[b] Note that costs are the annualized total social costs of reducing pollutants including NO_x and SO_2 for the EGU source category in the proposed Transport Rule region in 2014. The social costs are the loss of household utility as measured in Hicksian equivalent variation. More information on the social costs can be found in Chapter 8 of this RIA.

[c] Due to methodological limitations, the health benefits of the direct control and intrastate trading remedies include $PM_{2.5}$ –related benefits but omit visibility, ozone, and CO_2-related benefits. We present the $PM_{2.5}$ –related benefits of the proposed remedy, omitting these other important benefits, so that readers may compare directly the benefits of the proposed and alternate remedies. Total benefits are comprised primarily of the value of PM-related avoided premature mortalities. The reduction in these premature mortalities in each year account for over 90 percent of total $PM_{2.5}$-related monetized benefits. Benefits in this table are nationwide and are associated with NO_x and SO_2 reductions. To ensure that the benefits of the proposed remedy and the more and less stringent scenarios are directly comparable, we exclude the visibility-related benefits of the proposed remedy from this table; these visibility-related benefits are approximately \$3.6 billion (2006\$).

[d] Not all possible benefits or disbenefits are quantified and monetized in this analysis. B is the sum of all unquantified benefits and disbenefits. Potential benefit categories that have not been quantified and monetized are listed in Table 1-6.

[e] Valuation assumes discounting over the SAB-recommended 20-year segmented lag structure. Results reflect the use of 3 percent and 7 percent discount rates consistent with EPA and OMB guidelines for preparing economic analyses (EPA, 2000; OMB, 2003).

[f] Net benefits are rounded to three significant figures. Columnar totals may not sum due to rounding.

* The 2014 compliance costs (incremental to the base case) for the proposed remedy, less stringent scenario, and more stringent scenario are approximately \$2.76, \$1.19, and \$2.81 billion 2006 dollars.

Table 10-3. Summary of Annual Benefits, Costs, and Net Benefits of Versions of the Proposed Remedy Option in 2014[a] (billions of 2006 dollars)

Description	*Proposed Remedy*	*Direct Control*	*Intrastate Trading*
Social costs[b]			
3 percent discount rate	$2.03	$2.68	$2.49
7 percent discount rate	$2.23	$2.91	$2.70
Health-related benefits[c,d]			
3 percent discount rate	$118 to $288 + B	$117 to $286 + B	$113 to $276 + B
7 percent discount rate	$107 to $260 + B	$108 to $262 + B	$104 to $252 + B
Net benefits (benefits-costs)[e,f]			
3 percent discount rate	$116 to $286	$115 to $283	$110 to $273
7 percent discount rate	$105 to $258	$105 to $259	$101 to $249

[a] When presenting benefits and net benefits, EPA traditionally rounds all estimates to two significant figures. In this case we have rounded to three significant digits to facilitate comparison of the benefits and costs among the proposed remedy and the less and more stringent scenarios.

[b] Note that costs are the annualized total social costs of reducing pollutants including NO_x and SO_2 for the EGU source category in the proposed Transport Rule region in 2014. The social costs are the loss of household utility as measured in Hicksian equivalent variation. More information on the social costs can be found in Chapter 8 of this RIA.

[c] Due to methodological limitations, the health benefits of the direct control and intrastate trading remedies include $PM_{2.5}$ –related benefits but omit visibility, ozone, and CO_2-related benefits. We present the $PM_{2.5}$ –related benefits of the proposed remedy, omitting these other important benefits, so that readers may compare directly the benefits of the proposed and alternate remedies. Total benefits are comprised primarily of the value of PM-related avoided premature mortalities. The reduction in these premature mortalities in each year account for over 90 percent of total $PM_{2.5}$-related monetized benefits. Benefits in this table are nationwide and are associated with NO_x and SO_2 reductions. To ensure that the benefits of the proposed remedy and the more and less stringent scenarios are directly comparable, we exclude the visibility-related benefits of the proposed remedy from this table; these visibility-related benefits are approximately $3.6 billion (2006$).

[d] Not all possible benefits or disbenefits are quantified and monetized in this analysis. B is the sum of all unquantified benefits and disbenefits. Potential benefit categories that have not been quantified and monetized are listed in Table 1-6.

[e] Valuation assumes discounting over the SAB-recommended 20-year segmented lag structure. Results reflect the use of 3 percent and 7 percent discount rates consistent with EPA and OMB guidelines for preparing economic analyses (EPA, 2000; OMB, 2003).

[f] Net benefits are rounded to three significant figures. Columnar totals may not sum due to rounding.

* The 2014 compliance costs (incremental to the base case) for the proposed remedy, less stringent scenario, and more stringent scenario are approximately $2.76, $1.19, and $2.81 billion 2006 dollars.

As with any complex analysis of this scope, there are several uncertainties inherent in the final estimate of benefits and costs that are described fully in Chapters 5 and 7. In addition to the uncertainty characterization provided in these chapters, we also present two types of probabilistic approaches to characterize uncertainty in the benefit estimate of the proposed Transport Rule. The first approach generates a distribution of benefits based on the classical statistical error expressed in the underlying health and economic valuation studies used in the benefits modeling framework. The second approach uses the results from a pilot expert elicitation project designed to characterize key aspects of uncertainty in the ambient $PM_{2.5}$/mortality relationship, and augments the uncertainties in the mortality estimate with the statistical error reported for other endpoints in the benefit analysis.

10.2 References

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.

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.

U.S. Environmental Protection Agency (EPA). September 2000. *Guidelines for Preparing Economic Analyses.* EPA 240-R-00-003.

U.S. Office of Management and Budget (OMB). 2003. Circular A-4 Guidance to Federal Agencies on Preparation of Regulatory Analysis.

Woodruff, T.J., J. Grillo, and K.C. Schoendorf. 1997. "The Relationship Between Selected Causes of Postneonatal Infant Mortality and Particulate Infant Mortality and Particulate Air Pollution in the United States." *Environmental Health Perspectives* 105(6):608-612.

APPENDIX A

HUMAN HEALTH BENEFITS OF DIRECT CONTROL AND INTRASTATE TRADING REMEDIES AND PRESENTATION OF STATE-LEVEL BENEFITS OF PROPOSED REMEDY

Synopsis

This appendix summarizes the results of the human health benefits assessment of the direct control and intrastate trading remedies using $PM_{2.5}$ benefit per ton estimates. The $PM_{2.5}$-related benefits of the direct control remedy are between \$120 and \$290 billion (2006\$) discounted at 3% and between \$110 and \$260 billion (2006\$) discounted at 7%. The benefits of the intrastate trading remedy are between \$110 and \$280 billion (2006\$) discounted at 3% and between \$100 and \$250 billion (2006\$) discounted at 7%. In addition, this appendix includes state-level estimates of benefits of the proposed remedy. Due to methodological limitations, these estimates omit important benefits categories, including benefits from reduced ozone exposure, visibility improvement and ecological improvements.

A.1 Methods

In section 5.2.3 of the Benefits chapter we describe our approach to estimating a $PM_{2.5}$ benefit per ton metric. In the interest of completeness, we repeat this discussion here. Benefit per-ton (BPT) estimates quantify the health impacts and monetized human health benefits of an incremental change in air pollution precursor emissions. In situations when we are unable to perform air quality modeling because of resource or time constraints, this approach can provide a reliable estimate of the benefits of emission reduction scenarios. EPA has used the benefit per-ton technique in previous RIAs, including the recent Ozone NAAQS RIA (U.S. EPA, 2008) and NO_2 NAAQS RIA (U.S. EPA, 2010). Time constraints prevented the Agency from modeling the air quality changes resulting from either the intrastate and direct control remedies or the more and less stringent SO_2 caps and so we estimate a subset of these health benefits using $PM_{2.5}$ benefit per-ton estimates. The assessment of the alternate scenarios omits ozone-related benefits for two reasons. First, the overall level of ozone-related benefits in the modeled case is relatively small compared to those associated with $PM_{2.5}$ reductions (see table 5-17 in the Benefits chapter), due in part to the fairly modest summer time NO_x emission reductions under this scenario. The level of summertime NO_x emission reductions of the alternate scenarios are very similar to the modeled scenario, suggesting that the omission of ozone-related impacts would not greatly influence the overall level of benefits. Second, the complex non-linear chemistry of ozone formation introduces uncertainty to the development and application of a benefit per ton estimate. Taken together, these factors argued against developing an ozone benefit per ton estimate for this RIA.

For this analysis, EPA applies $PM_{2.5}$ BPT estimates that are methodologically consistent with those reported in Fann et al. (2009), but have been adjusted for this analysis to better match the spatial distribution of air quality changes projected for the Transport Rule. To derive the BPT estimates for this analysis, we:

1. *Quantified the $PM_{2.5}$-related human and monetized health benefits of the SO_2 emission reductions of the proposed remedy*. We first quantified the health impacts and monetized benefits of total $PM_{2.5}$ mass formed from the SO_2 reductions of the proposed remedy, allowing us to isolate the PM air quality impacts from SO_2 reductions alone.[84] This procedure allowed us to develop $PM_{2.5}$ BPT estimates that quantified the $PM_{2.5}$-related benefits of incremental changes in SO_2 emissions. Because reductions in NO_x emissions are relatively small in each scenario, and previous EPA modeling indicates that $PM_{2.5}$ formation is less sensitive to NO_x emission reductions on a per-$\mu g/m^3$ basis (U.S. EPA, 2006d), we did not quantify the NO_x -related $PM_{2.5}$ changes.

2. *Divided the health impacts and monetized benefits by the emission reduction.* This calculation yields BPT estimates for PM-related SO_2.

The resulting BPT estimates were then multiplied by the projected SO_2 emission reductions for the Direct Control and Intrastate Trading remedy options to produce an estimate of the PM- and ozone-related health impacts and monetized benefits. There is no analogous approach for estimating a BPT for visibility, and so the benefits of the alternative

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[84] The Transport Rule includes both SO_2 and NOx emissions reductions. In general SO_2 is a precursor to particulate sulfate and NOx is a precursor to particulate nitrate. However, there are also several interactions between the $PM_{2.5}$ precursors which cannot be easily quantified. For example, under conditions in which SO_2 levels are reduced by a substantial margin, "nitrate replacement" may occur. This occurs when particulate ammonium sulfate concentrations are reduced, thereby freeing up excess gaseous ammonia. The excess ammonia is then available to react with gaseous nitric acid to form particulate nitrate. The impact of nitrate replacement is also affected by concurrent NOx reductions. NOx reductions can lead to decreases in nitrate, which competes with the process of nitrate replacement. NOx reductions can also lead to reductions in photochemical by-products which can reduce both particulate sulfate and secondary organic carbon PM concentrations. Due to the complex nature of these interactions, EPA performed a sensitivity modeling analysis in which only SO_2 emissions were reduced. We calculated benefits from this air quality modeling run to generate an SO_2-only benefit per ton estimate. The results of the SO_2-only sensitivity run may be found in the EPA Benefits TSD [Docket No. EPA-HQ-OAR-2009-0491]

remedies omit this important monetized benefit. The PM-related benefits of the two alternative remedies are roughly commensurate with the total benefits of the proposed remedy; this is due in large part to the roughly similar emission reductions achieved under each of the preferred and alternate remedies and the fact that EPA used air quality modeling for the proposed remedy to generate as the basis for the benefit per ton estimates subsequently used to quantify the benefits of the two alternate remedies not modeled.

A.2 Results

Following the procedure described above, we calculated $PM_{2.5}$ benefit per-ton estimates summarized in Table A-1. We then calculated the reduction in incidence of $PM_{2.5}$ adverse health effects for each remedy option and present the results in Table A-2. Finally, we present the benefits associated with the reduction in incidence of these adverse health effects for each remedy option and present the results in Table A-3. For comparison purposes these tables also includes the PM-related benefits of the proposed remedy.

Table A-1: The Benefit per-ton of Reducing a Ton of SO_2 from EGU's in the Transport Rule Trading Region in 2014

	Discount rate	
$PM_{2.5}$ mortality estimate[A]	3%	7%
Pope et al. (2002)	$26,400	$65,000
Laden et al. (2006)	$24,000	$59,000

[A] Values represent the sum of PM-related mortality using each mortality estimate and the value of all avoided morbidities. Estimates rounded to two significant figures.

Table A-2: Estimated Reduction in Incidence of $PM_{2.5}$-related Adverse Health Effects of Direct Control and Intrastate Trading Remedy Options

Health Effect	*Direct control*[A]	*Intrastate trading*[A]	*Proposed Remedy*[B]
PM-Related endpoints			
Premature Mortality			
Pope et al. (2002) (age >30)	14,000	14,000	14,000 (4,000—25,000)
Laden et al. (2006) (age >25)	36,000	35,000	36,000 (17,000—56,000)
Infant (< 1 year)	58	56	59 (-66—180)[c]
Chronic Bronchitis	9,100	8,800	9,200 (320—18,000)
Non-fatal heart attacks (age > 18)	22,000	22,000	22,000 (5,800—39,000)
Hospital admissions—respiratory (all ages)	3,500	3,300	3,500 (1,400—5,500)
Hospital admissions—cardiovascular (age > 18)	7,400	7,200	7,500 (5,200—8,900)
Emergency room visits for asthma (age < 18)	14,000	13,000	14,000 (7,200—21,000)
Acute bronchitis (age 8-12)	21,000	20,000	21,000 (-4,800—46,000)[c]
Lower respiratory symptoms (age 7-14)	250,000	240,000	250,000 (98,000—400,000)
Upper respiratory symptoms (asthmatics age 9-18)	190,000	180,000	190,000 (36,000—350,000)
Asthma exacerbation (asthmatics 6-18)	230,000	230,000	240,000 (8,300—800,000)
Lost work days (ages 18-65)	1,800,000	1,700,000	1,800,000 (1,500,000—2,000,000)
Minor restricted-activity days (ages 18-65)	11,000,000	10,000,000	10,000,000 (8,600,000—12,000,000)

[A] Estimates rounded to two significant figures; column values will not sum to total value. Confidence intervals unavailable for estimate calculated using benefit per ton estimates.

[B] $PM_{2.5}$-related health impacts only. Excludes ozone impacts.

[C] The negative estimates for certain endpoints are the result of the weak statistical power of the study used to calculate these health impacts and do not suggest that increases in air pollution exposure result in decreased health impacts.

Table A-3: Estimated PM$_{2.5}$-related Monetized Benefits of Direct Control and Intrastate Trading Options

Health Effect	*Direct control*[A]	*Intrastate trading*[A]	*Proposed Remedy*[B]
Premature mortality (Pope et al. 2002 mortality estimate)			
3% discount rate	$110	$110	$110 ($8.8—$340)
7% discount rate	$99	$96	$99 ($7.9—$300)
Premature mortality (Laden et al. 2006 mortality estimate)			
3% discount rate	$280	$270	$280 ($25—$820)
7% discount rate	$250	$240	$250 ($22—$310)
Chronic Bronchitis	$4.2	$4.1	$4.3 ($0.2—$20)
Non-fatal heart attacks			
3% discount rate	$2.4	$2.4	$2.5 ($0.4—$6)
7% discount rate	$2.4	$2.3	$2.4 ($0.4—$5.9)
Hospital admissions—respiratory	$0.05	$0.05	$0.06 ($0.03—$0.1)
Hospital admissions—cardiovascular	$0.2	$0.2	$0.2 ($0.1—$0.3)
Emergency room visits for asthma	$0.005	$0.005	$0.005 ($0.002—$0.008)
Acute bronchitis	$0.009	$0.009	$0.009 (-$0.0004—$0.03)[C]
Lower respiratory symptoms	$0.005	$0.004	$0.005 ($0.002—$0.009)
Upper respiratory symptoms	$0.005	$0.005	$0.006 ($0.001—$0.014)
Asthma exacerbation	$0.01	$0.01	$0.012 ($0.001--$0.046)
Lost work days	$0.2	$0.2	$0.2 ($0.19—$0.24)
Minor restricted-activity days	$0.6	$0.6	$0.64 ($0.34—$0.97)

Monetized total PM$_{2.5}$-related benefits (Pope et al. 2002 mortality estimate)			
3% discount rate	**$120**	**$110**	**$120** **($10—$360)**
7% discount rate	**$110**	**$100**	**$110** **($9.2—$330)**
Monetized total PM$_{2.5}$-related benefits (Laden et al. 2006 mortality benefits)			
3% discount rate	**$290**	**$280**	**$290** **($26—$840)**
7% discount rate	**$260**	**$250**	**$260** **($23—$760)**

[A] Estimates rounded to two significant figures. Confidence intervals unavailable for estimate calculated using benefit per ton estimates.

[B] PM$_{2.5}$-related health impacts only. Excludes ozone and visibility-related benefits.

[C] The negative 5th percentile estimates for certain endpoints are the result of the weak statistical power of the study used to calculate these health impacts and do not suggest that increases in air pollution exposure result in decreased health impacts.

Table A-4: Avoided $PM_{2.5}$-Related Premature Mortalities for the Proposed Remedy in 2014 Summarized at the State Level (90 percent confidence intervals)

State	Pope et al. (2002) mortality estimate	Laden et al. (2006) mortality estimate
Alabama	360 (140--580)	930 (510--1300)
Arizona	1.5 (0.57--2.3)	3.8 (2--5.5)
Arkansas	220 (87--360)	570 (310--830)
California	-0.81 (-0.32---1.3)	-2.1 (-1.1---3)
Colorado	-7.6 (-3---12)	-20 (-11---29)
Connecticut	170 (67--280)	440 (240--640)
Delaware	64 (25--100)	160 (90--240)
District of Columbia	38 (15--62)	98 (54--140)
Florida	600 (230--960)	1,500 (840--2200)
Georgia	560 (220--900)	1,400 (780--2100)
Idaho	-0.61 (-0.24---0.98)	-1.6 (-0.85---2.3)
Illinois	720 (280--1100)	1,800 (1000--2700)
Indiana	640 (250--1000)	1600 (890--2400)
Iowa	100 (40--160)	260 (140--380)
Kansas	74 (29--120)	190 (100--280)
Kentucky	610 (240--970)	1,500 (850--2200)
Louisiana	210 (81--330)	530 (290--770)
Maine	26 (10--42)	67 (36--97)
Maryland	490 (190--790)	1,300 (690--1800)
Massachusetts	190 (74--300)	480 (260--700)

Michigan	600 (240--960)	1,500 (840--2200)
Minnesota	80 (31--130)	200 (110--300)
Mississippi	230 (90--370)	590 (320--850)
Missouri	370 (150--600)	950 (520--1400)
Montana	-0.33 (-0.13---0.53)	-0.85 (-0.46---1.2)
Nebraska	33 (13--53)	84 (46--120)
Nevada	-0.0003 (-0.0001---0.0005)	-0.0003 (-0.0002---0.0004)
New Hampshire	37 (15--60)	95 (52--140)
New Jersey	550 (220--880)	1,400 (770--2000)
New Mexico	5.9 (2.3--9.5)	15 (8.3--22)
New York	950 (370--1500)	2,400 (1300--3500)
North Carolina	720 (280--1200)	1,800 (1000--2700)
North Dakota	2.7 (1.1--4.4)	7 (3.8--10)
Ohio	1,300 (520--2100)	3,300 (1800--4800)
Oklahoma	120 (48--200)	320 (170--460)
Oregon	-1 (-0.4---1.7)	-2.6 (-1.4---3.8)
Pennsylvania	1,400 (560--2300)	3,600 (2000--5200)
Rhode Island	38 (15--61)	97 (53--140)
South Carolina	330 (130--530)	850 (460--1200)
South Dakota	8.4 (3.3--13)	21 (12--31)
Tennessee	710 (280--1100)	1,800 (990--2600)
Texas	520 (200--840)	1,300 (730--1900)

Utah	-1.4 (-0.56---2.3)	-3.7 (-2---5.4)
Vermont	22 (8.5--35)	56 (30--81)
Virginia	700 (270--1100)	1,800 (970--2600)
Washington	-1.7 (-0.68---2.8)	-4.5 (-2.4---6.5)
West Virginia	280 (110--450)	720 (390--1000)
Wisconsin	190 (74--300)	480 (260--700)
Wyoming	-0.12 (-0.048---0.2)	-0.31 (-0.17---0.46)

Table A-5: Avoided Ozone-Related Premature Mortalities for the Proposed Remedy in 2014 Summarized at the State Level (90 percent confidence intervals)

State	Bell et al. (2004) mortality estimate	Levy et al. (2005) mortality estimate
Alabama	5.8 (3.3--8.4)	8.2 (6--10)
Arizona	0.31 (0.17--0.44)	0.43 (0.32--0.55)
Arkansas	5.7 (3.2--8.2)	8 (5.9--10)
California	0.12 (0.056--0.18)	0.17 (0.11--0.22)
Colorado	0.03 (-0.042--0.1)	0.042 (-0.019--0.1)
Connecticut	0.44 (0.25--0.63)	0.62 (0.46--0.78)
Delaware	0.19 (0.1--0.27)	0.26 (0.19--0.33)
District of Columbia	0.11 (0.062--0.16)	0.15 (0.11--0.2)
Florida	22 (12--32)	31 (22--39)
Georgia	6.7 (3.7--9.6)	9.4 (6.9--12)
Idaho	0.007 (0.0039--0.01)	0.0099 (0.0073--0.013)
Illinois	7.7 (4.3--11)	11 (8--14)
Indiana	4.3 (2.4--6.2)	6.1 (4.5--7.7)
Iowa	4.5 (2.5--6.5)	6.3 (4.7--8)
Kansas	6 (3.4--8.6)	8.4 (6.2--11)
Kentucky	3 (1.7--4.3)	4.2 (3.1--5.3)
Louisiana	5.4 (3--7.8)	7.6 (5.6--9.7)
Maine	0.067 (0.038--0.097)	0.095 (0.07--0.12)
Maryland	1.6 (0.87--2.2)	2.2 (1.6--2.8)
Massachusetts	0.37 (0.21--0.54)	0.53 (0.39--0.67)

Michigan	5.1 (2.8--7.3)	7.1 (5.3--9)
Minnesota	4.2 (2.4--6.1)	6 (4.4--7.5)
Mississippi	4.2 (2.4--6.1)	5.9 (4.4--7.5)
Missouri	12 (6.7--17)	17 (12--21)
Montana	0.012 (0.007--0.018)	0.018 (0.013--0.022)
Nebraska	2 (1.1--2.8)	2.8 (2--3.5)
Nevada	0.038 (0.021--0.054)	0.053 (0.039--0.067)
New Hampshire	0.11 (0.063--0.17)	0.16 (0.12--0.21)
New Jersey	1.6 (0.91--2.3)	2.3 (1.7--2.9)
New Mexico	0.37 (0.21--0.54)	0.53 (0.39--0.67)
New York	4.2 (2.4--6.1)	6 (4.4--7.6)
North Carolina	3.6 (2--5.2)	5.1 (3.7--6.4)
North Dakota	0.18 (0.1--0.26)	0.26 (0.19--0.33)
Ohio	7.9 (4.4--11)	11 (8.2--14)
Oklahoma	7.2 (4--10)	10 (7.5--13)
Oregon	0.0032 (0.0009--0.0054)	0.0045 (0.0025--0.0064)
Pennsylvania	8 (4.5--11)	11 (8.3--14)
Rhode Island	0.087 (0.049--0.13)	0.12 (0.09--0.15)
South Carolina	2.9 (1.6--4.2)	4.1 (3--5.2)
South Dakota	0.48 (0.27--0.69)	0.68 (0.5--0.85)
Tennessee	4.2 (2.3--6)	5.9 (4.3--7.4)
Texas	11 (6.3--16)	16 (12--20)

Utah	0.02 (0.011--0.029)	0.028 (0.021--0.036)
Vermont	0.092 (0.052--0.13)	0.13 (0.095--0.16)
Virginia	2.3 (1.3--3.3)	3.3 (2.4--4.1)
Washington	0.0018 (0.0003--0.0033)	0.0025 (0.0013--0.0038)
West Virginia	2.1 (1.2--3)	3 (2.2--3.7)
Wisconsin	4.8 (2.6--6.9)	6.7 (4.9--8.5)
Wyoming	0.0088 (0.0042--0.014)	0.013 (0.0085--0.016)

References

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 2:169–176.

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). 2010. Final Regulatory Impact Analysis (RIA) for the NO_2 National Ambient Air Quality Standards (NAAQS). Office of Air Quality Planning and Standards, Research Triangle Park, NC. January. Available on the Internet at <http://www.epa.gov/ttn/ecas/regdata/RIAs/FinalNO2RIAfulldocument.pdf>.

APPENDIX B

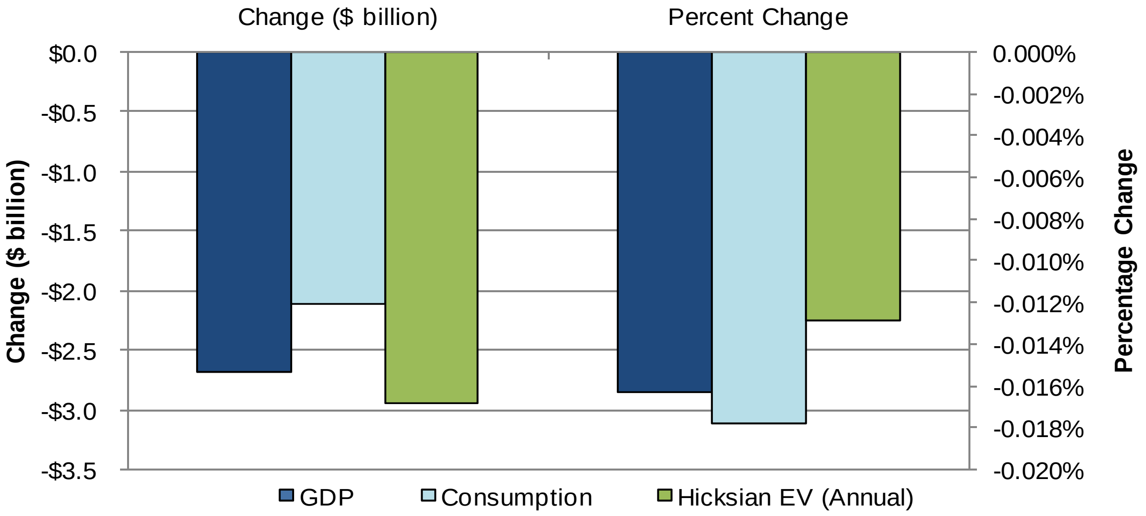
ECONOMIC IMPACT ANALYSES OUTSIDE OF ELECTRIC POWER SECTOR AND SOCIAL COSTS -– ALTERNATE REMEDIES

B.1 Direct Control Remedy Option

B.1.1 Macroeconomic Variables and Social Costs

The transport rule will bring about changes in business and household behavior and will influence macroeconomic variables (gross domestic product [GDP] and consumption) and household economic welfare. In 2015, EMPAX estimates that GDP and consumption levels are approximately 0.02% lower ($2.7 billion) (Figure B-1) [85] Since the pollution controls vary by region, economic effects also vary by region; for example, Northeast GDP falls by 0.04% (Figure B-2).

Figure B-1. Change in Macroeconomic Variables and Household Welfare (Percent change and Change in billion $2006) in 2015



Note: GDP represents the dollar value of all goods and services produced in the US in 2015. Consumption is the dollar value of all goods and services consumed within the US in 2015. Hicksian EV is the change in household economic welfare (defined in Section 8.1.3.1).

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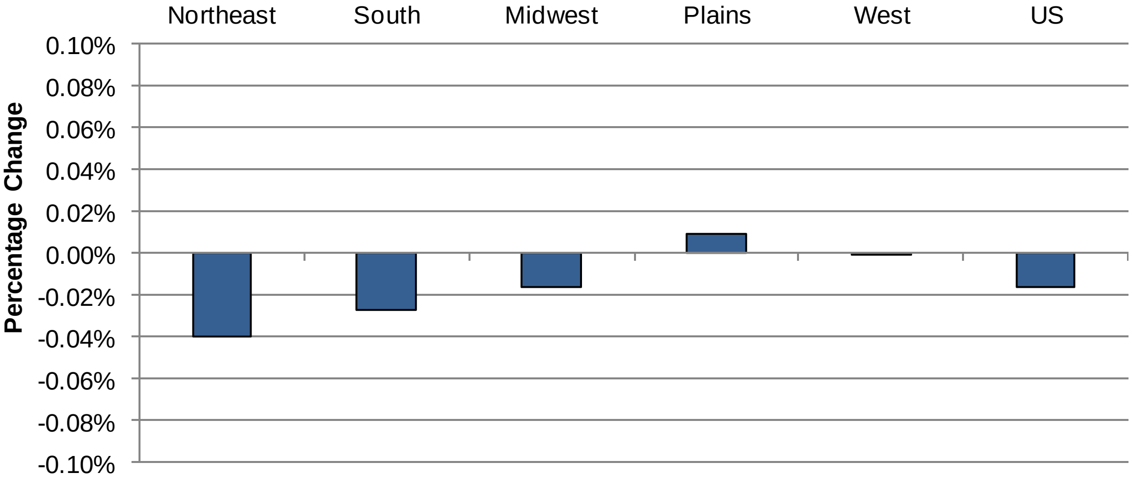
[85] We use 2015 estimates as a proxy for the impacts of compliance with the proposed rule in 2014.

Figure B-2. Change in Regional Gross Domestic Product (GDP) (Percent) in 2015



Note: GDP in each region is the dollar value of goods and services produced in the region in 2015. See Figure 8-2 for a presentation of the states in each region.

Average-annual social costs (as measured by Hicksian equivalent variation) are approximately 0.01% lower with the transport rule.[86] As noted in Chapter 8, EMPAX-CGE does not incorporate any environmental benefits associated with air quality improvements. As a result, EMPAX welfare measures only approximate the rule's social cost. Using this interpretation, the annual social cost for 2015 is estimated to be $2.9 billion.

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[86] Values are discounted back to 2010 at the 5% interest rate used in the model. EPA uses a 5% interest rate based on the MIT Emissions Prediction and Policy Analysis (EPPA) model and SAB guidance from 2003 as discussed in U.S. EPA, Office of Policy Analysis and Review. 2003. "Benefits and Costs of the Clean Air Act 1990 - 2020: Revised Analytical Plan for EPA's Second Prospective Analysis." We recognize that this interest rate is not one of the interest rates (3 and 7%) that OMB's Circular A-4 guidance calls for in regulatory analyses. . Detailed results for this EMPAX run for the direct control remedy can be found in the file "EMPAXresults_direct control remedy," that is available in the docket for this rule.

B.1.2 Industry Effects

The proposed rule directly influences the electricity sector's fuel use and private cost expenditures. As the electricity sector responds to these changes, other economy-wide changes occur. For example, higher electricity prices may encourage electricity-dependent sectors to reduce production levels, switch to other energy sources (e.g., oil) and/or seek energy efficiency improvements in their production process. Electricity sectors also make additional private cost expenditures in order to comply with the transport rule; these expenditures lead to other economy-wide changes. For example each dollar spent to comply with the program is used to buy environmental protection goods and services.[87] As a result, the demand for environmental protection goods and services will be higher with the transport rule. For sectors supplying environmental protection goods or services, the secondary effect may offset higher electricity costs. The following sections report and discuss output changes associated with the impacts of compliance in the year 2015, which serves as a proxy for compliance in 2014.

B.1.2.1 Energy Sectors

The EMPAX modeling system shows that the electricity sector experiences the most significant changes under the transport rule. Electricity output and fuel mix changes used to meet the transport rule also influence other energy sectors. For example, U.S. electricity output declines by approximately 0.5%, while coal output declines by 0.2%. Similarly, U.S. natural gas output declines. Crude oil and petroleum output decline but the changes are small; these inputs are less critical to the electricity sector, making them less sensitive to changes in electricity production (Figure B-3).

Given the regional distribution of controls, there are differences in regional output changes. For example, electricity production in the Northeast experiences the largest decline while the Plains and West electricity sectors see small output increases. Very few States in the Plains and no Western States are included in the Transport Rule region, and lack of emission controls applied in these regions may mean lower electricity generation and dispatch costs relative to such costs to States within the Transport Rule region. This is an

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[87] Additional details are described in EMPAX-CGE model documentation (5-2 to 5-5).

explanation for the small output increases in the Plains and West. Coal output changes to meet coal demand predictions from the IPM electricity model and the IPM modeling system suggest that the Northeast's electricity sector uses additional coal inputs to meet the rule's requirements.

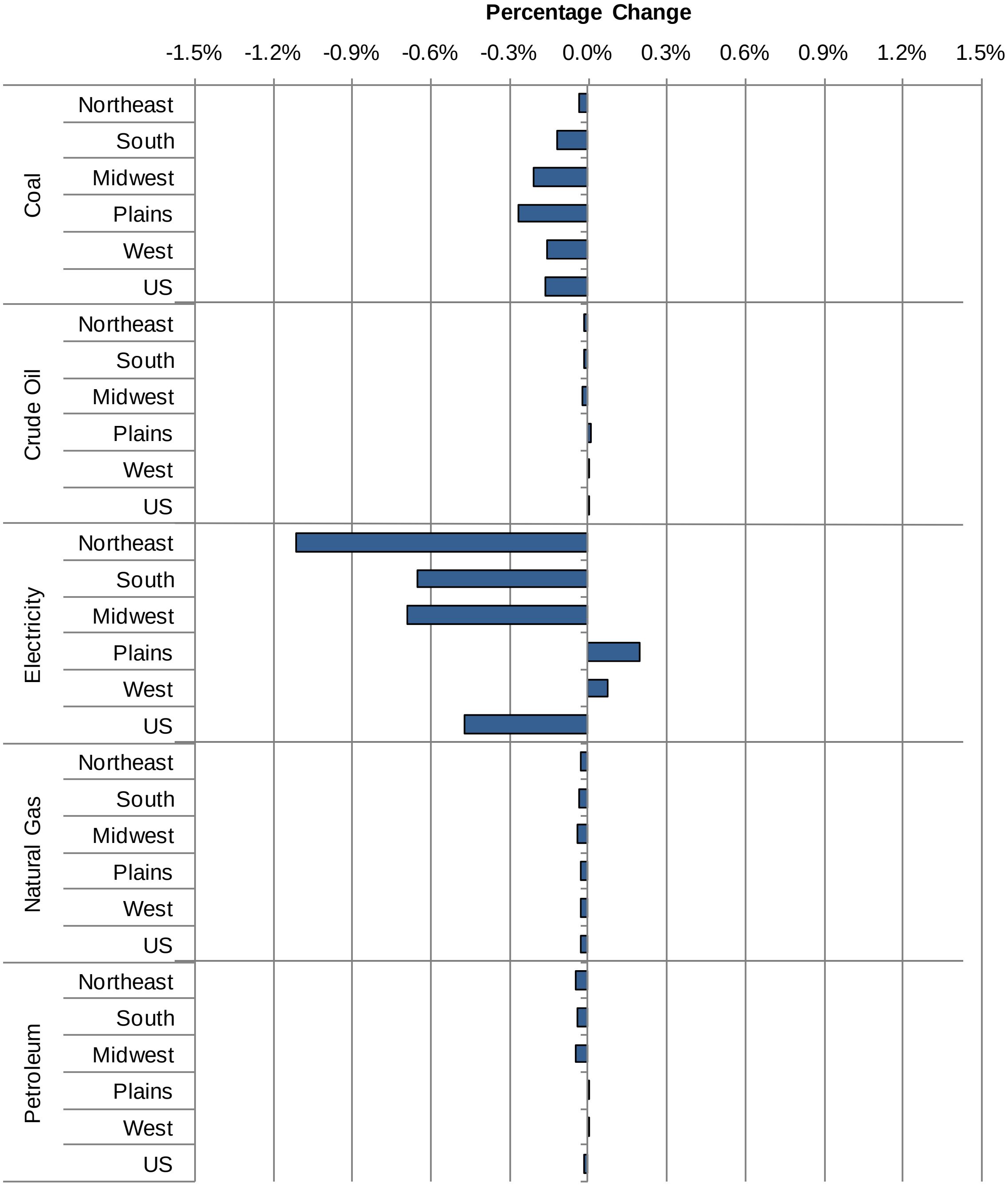
B.1.2.2 Energy-Intensive Sectors

Energy-intensive manufacturing industries are more sensitive to electricity and other energy price changes. Although the net U.S. output change for each energy-intensive industry is less than 0.2%, these sectors do show some (economically small) regional variation. The most significant regional differences are seen in the aluminum sector, where production shifts from the Northeast, South, and Midwest regions to the Plains and West regions. Similar geographic shifts are observed in other energy-intensive industries (Figure B-4).

B.1.2.3 Nonenergy Sectors

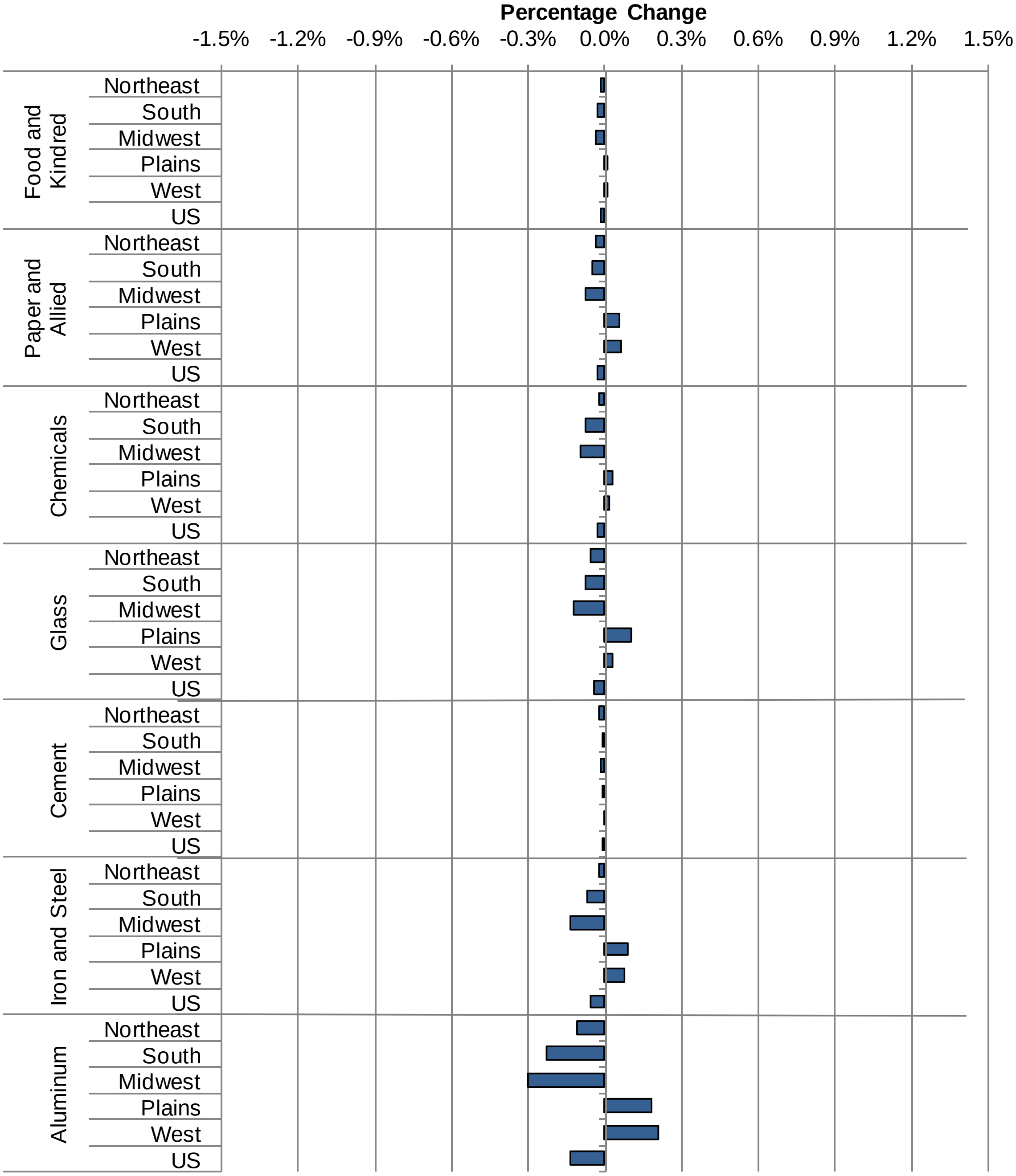
Although electricity expenditures represent a small fraction of non-energy sector production costs, higher electricity prices still influence non-energy sector production levels. However, non-energy sector output effects are very small. National output levels for four broad non-energy sectors: agriculture, other manufacturing, services, and transportation fall by less than one one-hundredth of a percent (0.01%). There is some regional variation as production shifts to areas with lower electricity costs (e.g., West, Plains), but the differences are not significant (Figure B-5).

Figure B-3. Output Changes in 2015: Energy Sectors (Percent)



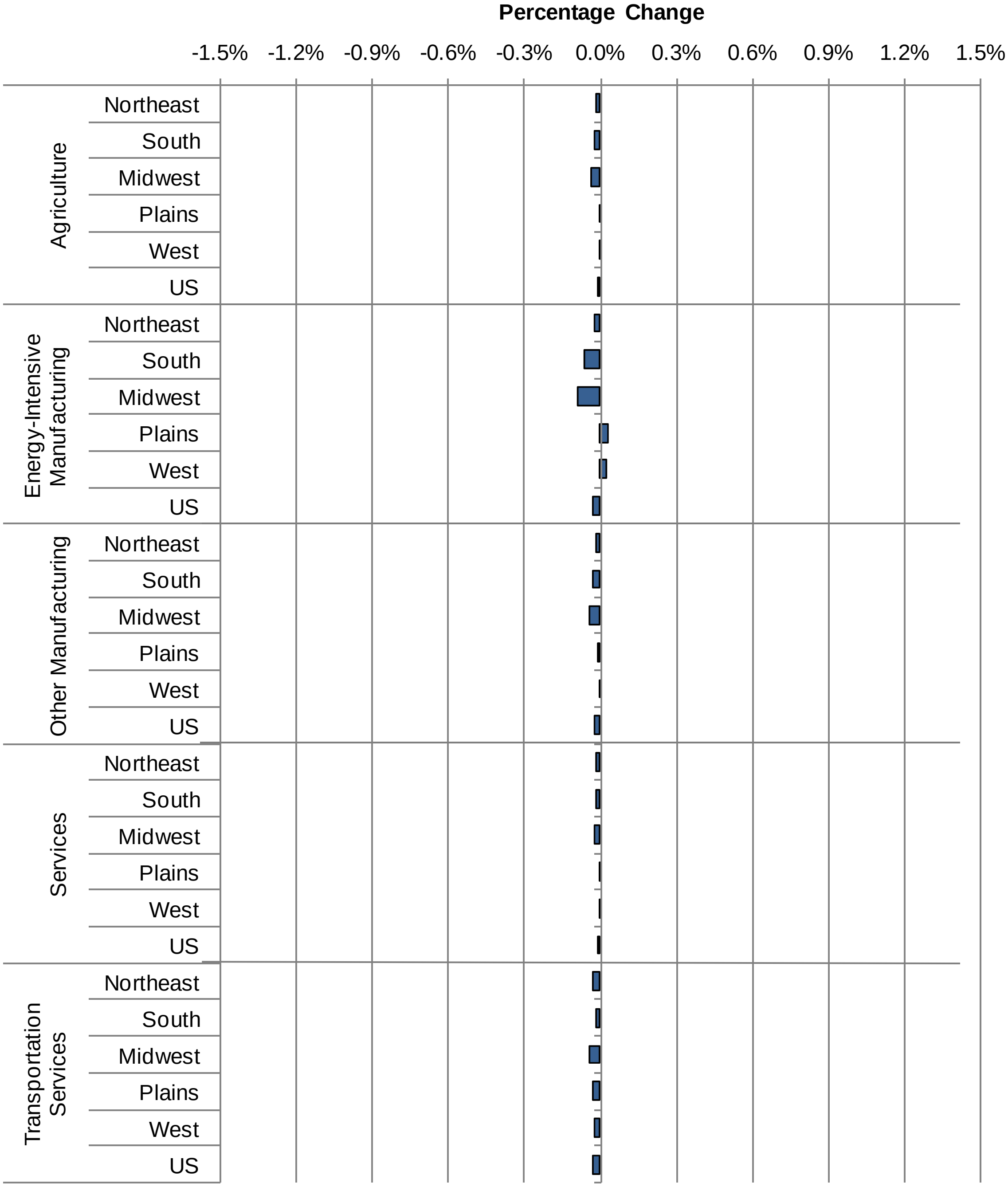
Note: Outcomes reflect percent changes in the physical quantities of goods/services that each regional sector produces.

Figure B-4. Output Changes in 2015: Energy-Intensive Sectors (Percent)



Note: Outcomes reflect percent changes in the physical quantities of goods/services that each regional sector produces.

Figure B-5. Output Changes in 2015: Nonenergy Sectors (Percent)



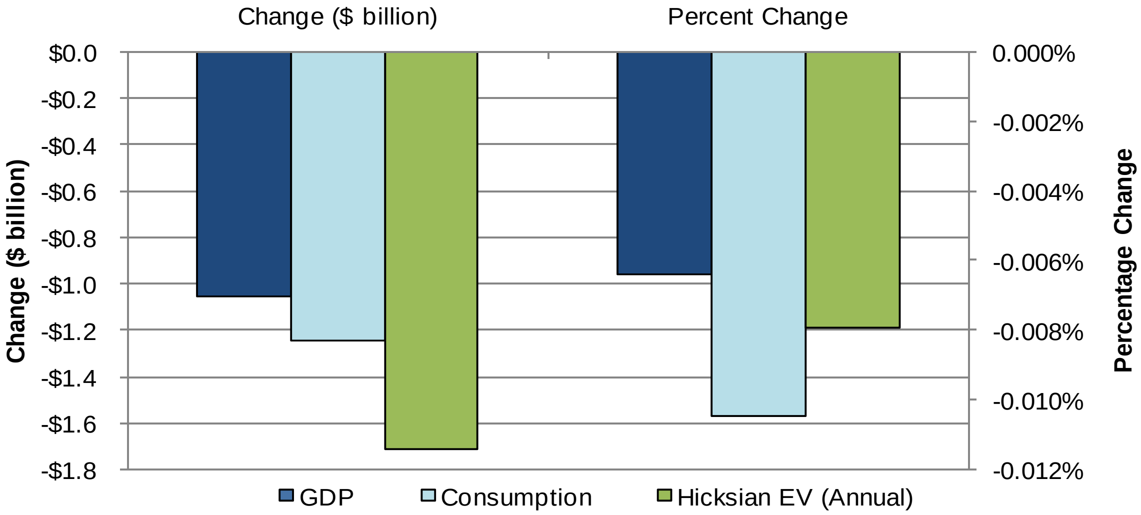
Note: Outcomes reflect percent changes in the physical quantities of goods/services that each regional sector produces.

B.2 Intrastate Trading Remedy Option

B.2.1 Macroeconomic Variables and Social Costs

The transport rule will bring about changes in business and household behavior and will influence macroeconomic variables (gross domestic product [GDP] and consumption) and household economic welfare. In 2015, EMPAX estimates that GDP and consumption levels are approximately 0.01% lower ($1.1 billion change in GDP and $1.2 billion change in consumption) (Figure B-6).[88] Since the pollution controls vary by region, economic effects also vary by region; for example, Northeast GDP falls by 0.04% (Figure B-7).

Figure B-6. Change in Macroeconomic Variables and Household Welfare (Percent change and Change in billion $2006) in 2015



Note: GDP represents the dollar value of all goods and services produced in the US in 2015. Consumption is the dollar value of all goods and services consumed within the US in 2015. Hicksian EV is the change in household economic welfare (defined in Section 8.1.3.1 of this RIA).

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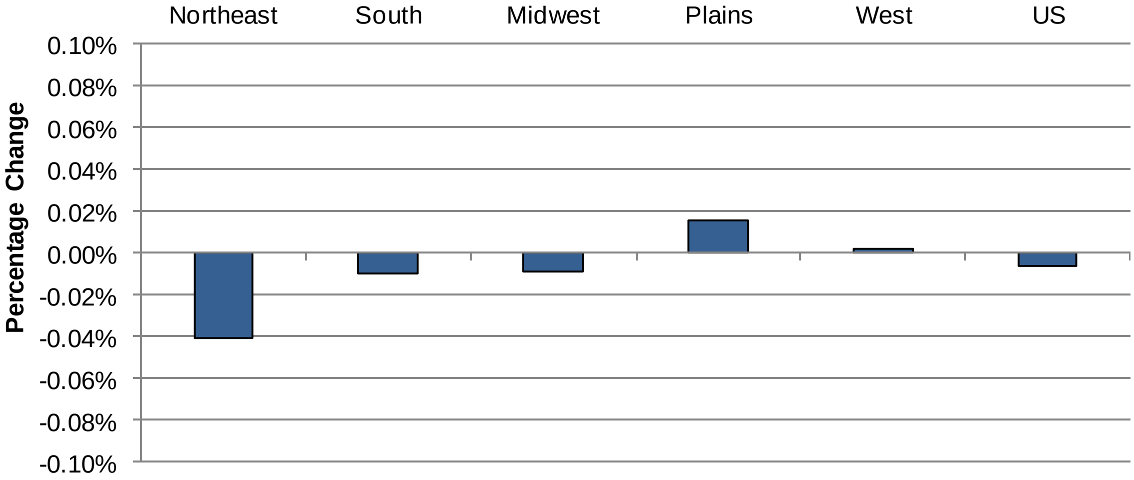
[88] We use 2015 estimates as a proxy for the impacts of compliance with the proposed rule in 2014.

Figure B-7. Change in Regional Gross Domestic Product (GDP) (Percent) in 2015



Note: GDP in each region is the dollar value of goods and services produced in the region in 2015. See Figure 8-2 for a presentation of the states in each region.

Average-annual social costs (as measured by Hicksian equivalent variation) are approximately 0.01% lower with the transport rule.[89] As noted in Chapter 8, EMPAX-CGE does not incorporate any environmental benefits associated with air quality improvements. As a result, EMPAX welfare measures only approximate the rule's social cost. Using this interpretation, the annual social cost for 2015 is estimated to be $2.7 billion.

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[89] Values are discounted back to 2010 at the 5% interest rate used in the model. EPA uses a 5% interest rate based on the MIT Emissions Prediction and Policy Analysis (EPPA) model and SAB guidance from 2003 as discussed in U.S. EPA, Office of Policy Analysis and Review. 2003. "Benefits and Costs of the Clean Air Act 1990 - 2020: Revised Analytical Plan for EPA's Second Prospective Analysis." We recognize that this interest rate is not one of the interest rates (3 and 7%) that OMB's Circular A-4 guidance calls for in regulatory analyses. Detailed results for this EMPAX run for the intrastate trading remedy can be found in the file "EMPAXresults_intrastate trading remedy," that is available in the docket for this rule.

B.2.2 Industry Effects

The proposed rule directly influences the electricity sector's fuel use and private cost expenditures. As the electricity sector responds to these changes, other economy-wide changes occur. For example, higher electricity prices may encourage electricity-dependent sectors to reduce production levels, switch to other energy sources (e.g., oil) and/or seek energy efficiency improvements in their production process. Electricity sectors also make additional private cost expenditures in order to comply with the transport rule; these expenditures lead to other economy-wide changes. For example each dollar spent to comply with the program is used to buy environmental protection goods and services.[90] As a result, the demand for environmental protection goods and services will be higher with the transport rule. For sectors supplying environmental protection goods or services, the secondary effect may offset higher electricity costs. The following sections report and discuss output changes associated with the impacts of compliance in the year 2015, which serves as a proxy for compliance in 2014.

B.2.2.1 Energy Sectors

The EMPAX modeling system shows that the electricity sector experiences the most significant changes under the transport rule. Electricity output and fuel mix changes used to meet the transport rule also influence other energy sectors. For example, U.S. electricity output declines by approximately 0.3%, while coal output declines by 0.4%. Similarly, U.S. natural gas output declines. Crude oil and petroleum output decline but the changes are small; these inputs are less critical to the electricity sector, making them less sensitive to changes in electricity production (Figure B-8).

Given the regional distribution of controls, there are differences in regional output changes. For example, electricity production in the Northeast experiences the largest decline while the Plains and West electricity sectors see small output increases. Very few States in the Plains and no Western States are included in the Transport Rule region, and lack of emission controls applied in these regions may mean lower electricity generation and

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[90] Additional details are described in EMPAX-CGE model documentation (5-2 to 5-5).

dispatch costs relative to such costs to States within the Transport Rule region. This is an explanation for the small output increases in the Plains and West. Coal output changes to meet coal demand predictions from the IPM electricity model and the IPM modeling system suggest that the Northeast's electricity sector uses additional coal inputs to meet the rule's requirements.

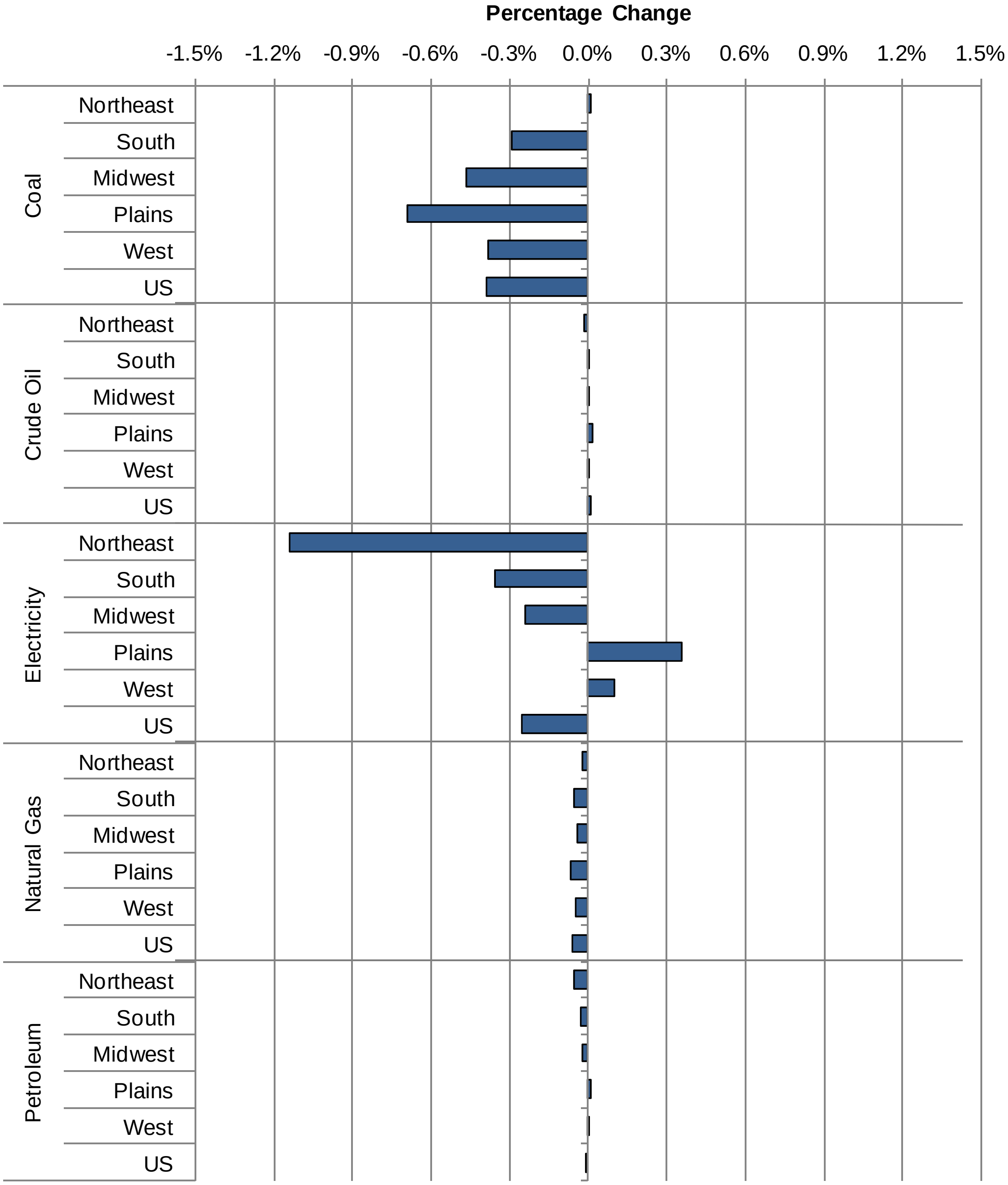
B.2.2.2 Energy-Intensive Sectors

Energy-intensive manufacturing industries are more sensitive to electricity and other energy price changes. Although the net U.S. output change for each energy-intensive industry is less than 0.2%, these sectors do show some (economically small) regional variation. The most significant regional differences are seen in the aluminum sector, where production shifts from the Northeast, South, and Midwest regions to the Plains and West regions. Similar geographic shifts are observed in other energy-intensive industries (Figure B-9).

B.2.2.3 Nonenergy Sectors

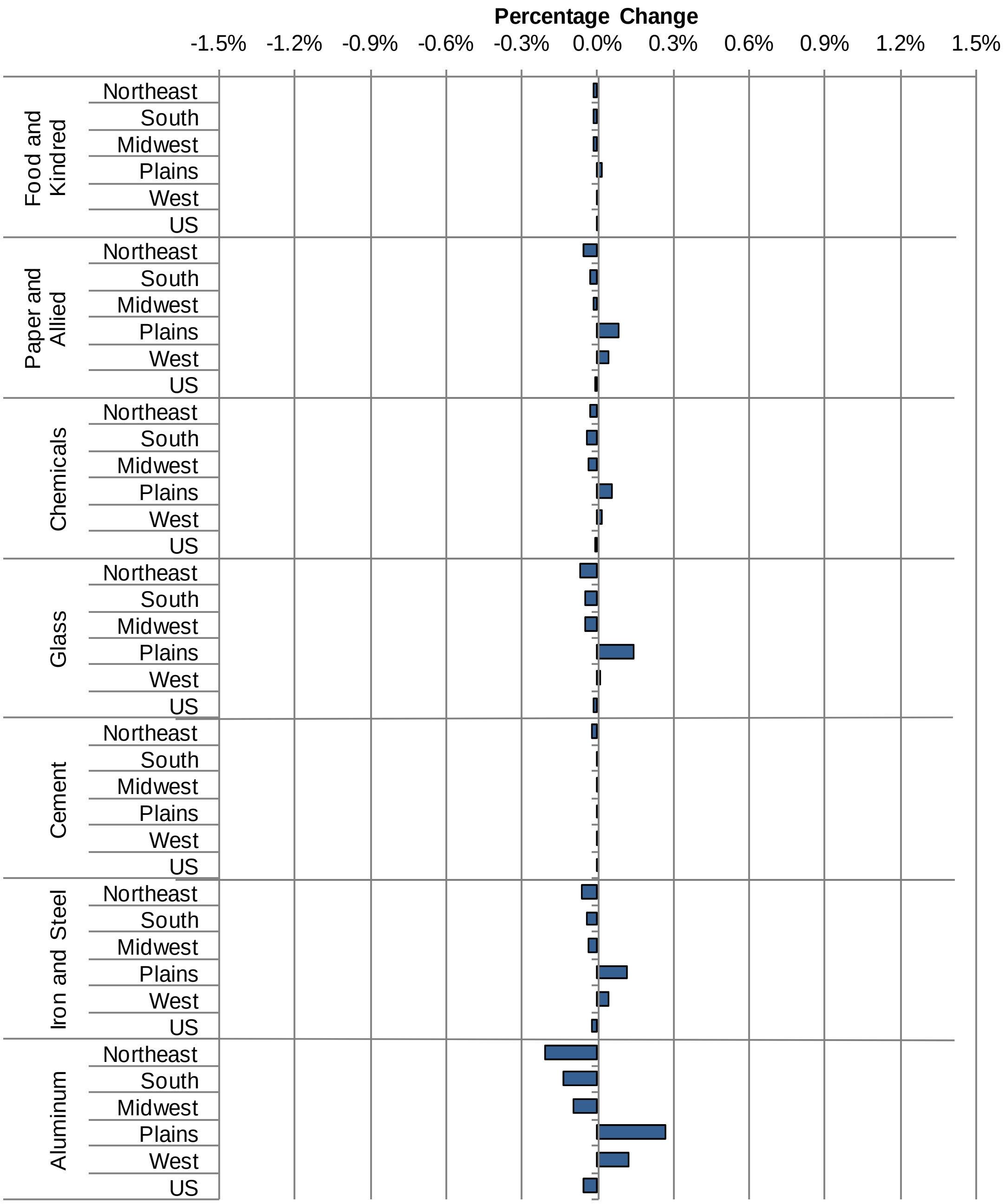
Although electricity expenditures represent a small fraction of non-energy sector production costs, higher electricity prices still influence non-energy sector production levels. However, non-energy sector output effects are very small. National output levels for four broad non-energy sectors: agriculture, other manufacturing, services, and transportation fall by less than one one-hundredth of a percent (0.01%). There is some regional variation as production shifts to areas with lower electricity costs (e.g., West, Plains), but the differences are not significant (Figure B-10).

Figure B-8. Output Changes in 2015: Energy Sectors (Percent)



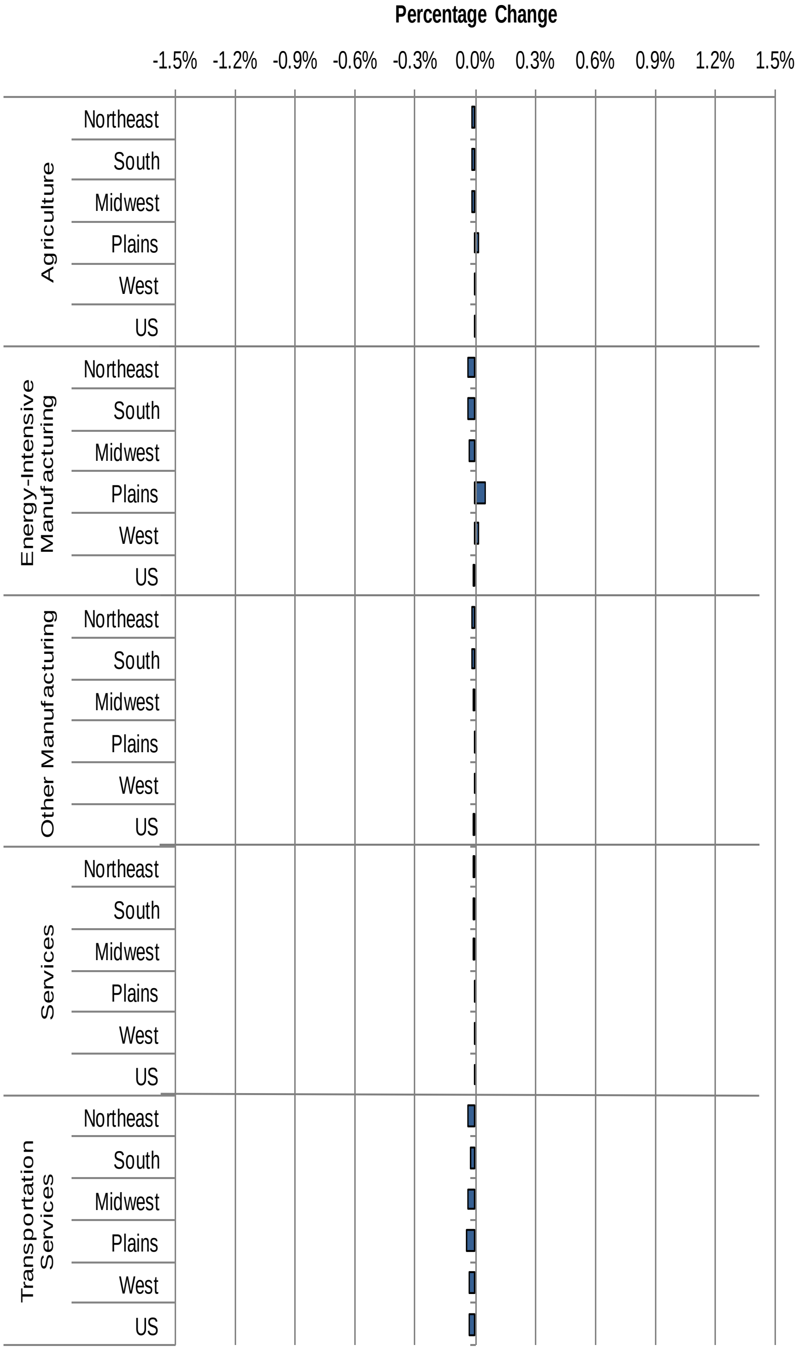
Note: Outcomes reflect percent changes in the physical quantities of goods/services that each regional sector produces.

Figure B-9. Output Changes in 2015: Energy-Intensive Sectors (Percent)



Note: Outcomes reflect percent changes in the physical quantities of goods/services that each regional sector produces.

Figure B-10. Output Changes in 2015: Nonenergy Sectors (Percent)



Note: Outcomes reflect percent changes in the physical quantities of goods/services that each regional sector produces.

APPENDIX C

COMPARISON OF STATE LEVEL ELECTRICAL GENERATING UNIT EMISSIONS UNDER VARIOUS REGULATORY ALTERNATIVES TO REDUCE SO_2 AND NO_x EMISSIONS UNDER THE TRANSPORT RULE

ELECTRICAL GENERATING UNIT SO$_2$ Emissions for Base Case and Regulatory Alternatives (tons)

State		Base Case		State Budgets/ Limited Trading		State Budgets/ Intrastate Trading		Direct Control	
		2012	*2014*	*2012*	*2014*	*2012*	*2014*	*2012*	*2014*
Alabama	AL	335,734	322,130	185,518	172,197	161,871	161,870	172,198	172,198
Connecticut	CT	5,493	5,512	2,560	2,586	2,713	2,560	2,713	2,560
Delaware	DE	6,918	6,883	7,979	7,996	7,784	7,784	7,991	7,920
District of Columbia	DC	0	0	0	0	0	0	0	0
Florida	FL	228,360	192,903	136,373	137,985	161,739	157,300	120,257	133,546
Georgia	GA	551,326	172,529	267,644	91,648	226,736	88,979	247,709	91,648
Illinois	IL	721,155	197,265	251,084	160,197	208,957	151,530	222,003	159,939
Indiana	IN	806,825	786,593	296,182	214,022	281,693	240,855	393,403	214,022
Iowa	IA	158,443	149,271	91,536	86,892	92,045	87,092	100,464	92,040
Kansas	KS	59,492	65,035	44,527	51,150	57,275	52,403	49,089	53,185
Kentucky	KY	716,807	737,962	222,246	121,399	195,480	125,879	232,552	121,398
Louisiana	LA	98,110	92,695	93,169	92,763	90,477	90,477	93,579	94,220
Maryland	MD	49,078	42,636	39,566	42,756	34,451	42,604	38,138	43,051
Massachusetts	MA	16,300	16,300	8,987	9,340	7,902	7,630	8,064	7,943
Michigan	MI	283,616	268,916	200,547	165,644	200,848	180,919	239,676	165,645
Minnesota	MN	47,090	53,910	40,142	41,103	41,471	41,525	46,979	50,802
Missouri	MO	428,394	481,531	171,496	168,911	168,282	167,587	185,948	168,911
Nebraska	NE	119,258	114,163	73,937	73,473	71,598	71,598	75,466	73,529
New Jersey	NJ	36,116	36,038	13,031	12,925	11,291	11,291	12,925	12,925
New York	NY	131,360	129,757	82,450	45,450	65,968	40,368	71,366	45,450
North Carolina	NC	117,264	131,291	96,166	87,567	75,825	99,689	108,931	87,567
Ohio	OH	936,919	802,942	270,343	189,582	275,549	226,395	440,986	189,583
Pennsylvania	PA	963,947	970,705	270,570	150,855	281,189	179,277	302,447	150,855
South Carolina	SC	145,171	149,157	130,457	124,190	116,483	116,483	124,189	124,189
Tennessee	TN	596,987	600,066	127,175	106,762	100,007	100,007	106,762	94,073
Virginia	VA	132,093	122,393	93,143	44,136	72,595	37,439	77,768	44,137
West Virginia	WV	587,667	495,573	125,333	126,869	119,546	146,804	209,488	126,869
Wisconsin	WI	99,464	105,230	72,392	71,514	75,933	73,438	88,716	71,514
Arizona	AZ	22,773	20,944	24,927	23,477	26,072	23,477	25,457	23,477
Arkansas	AR	85,068	88,187	117,046	119,945	123,920	120,427	116,494	115,389
California	CA	3,307	3,307	3,307	3,307	3,307	3,307	3,307	3,307
Colorado	CO	69,273	69,184	82,964	84,835	82,950	85,042	82,964	88,779
Idaho	ID	0	1	0	0	0	0	0	0
Maine	ME	15,375	11,650	15,921	11,669	16,658	11,669	16,244	11,675
Mississippi	MS	41,304	43,020	59,568	57,228	59,550	57,228	57,147	54,307
Montana	MT	15,892	16,863	18,128	19,093	18,128	19,093	18,097	18,274
Nevada	NV	13,323	20,155	13,288	20,531	13,288	20,531	13,288	21,416
New Hampshire	NH	7,290	6,608	7,290	7,290	7,290	7,290	7,290	7,290
New Mexico	NM	12,684	13,210	12,391	12,529	12,684	12,754	12,617	11,845
North Dakota	ND	77,383	80,320	88,321	88,321	88,321	85,649	84,835	85,649
Oklahoma	OK	156,032	165,773	159,773	165,994	159,773	165,994	159,773	165,905

Oregon	OR	14,381	13,366	20,306	20,187	20,028	20,187	14,381	20,187
Rhode Island	RI	0	0	0	0	0	0	0	0
South Dakota	SD	12,121	12,127	18,377	27,565	18,377	27,565	12,121	27,565
Texas	TX	327,726	373,803	463,908	467,617	463,538	514,641	442,319	481,621
Utah	UT	24,972	25,414	26,124	29,117	26,476	29,266	26,476	27,807
Vermont	VT	0	0	0	0	0	0	0	0
Washington	WA	19,663	19,155	19,663	18,863	19,663	18,863	19,663	18,793
Wyoming	WY	52,774	51,254	57,169	56,276	57,229	57,711	57,223	56,276
Nationwide total		**9,350,726**	**8,283,726**	**4,623,022**	**3,833,752**	**4,422,958**	**3,990,475**	**4,949,497**	**3,839,283**

This table shows the SO_2 emissions for each state in the contiguous US that result from the base case and the main control options. Notably, states adjacent to the Transport region states can have emissions increase to some degree due to change in relative dispatch economics from where the border exists for the Transport region. Note that in the West the increased emissions result from the Court’s decision to not allow the use of Title IV allowances in this program and the resulting collapse of the ARP trading market. This occurs in both direct control and trading cases.

Emissions are for fossil EGUs with capacity greater than 25 MW.

Electrical Generating Unit Annual NO_x Emissions for Base Case and Regulatory Alternatives (tons)[A]

State		Base Case		State Budgets/ Limited Trading		State Budgets/ Intrastate Trading		Direct Control	
		2012	*2014*	*2012*	*2014*	*2012*	*2014*	*2012*	*2014*
Alabama	AL	121,772	118,376	68,589	61,214	68,785	61,039	68,814	61,286
Connecticut	CT	2,753	2,793	2,722	2,805	2,750	2,787	2,725	2,805
Delaware	DE	4,277	4,151	4,464	4,572	4,557	4,517	4,470	4,464
District of Columbia	DC	2	1	2	1	2	1	2	1
Florida	FL	194,872	179,796	112,954	109,578	110,610	109,489	81,614	78,069
Georgia	GA	77,815	47,897	74,486	44,092	73,586	44,093	75,646	44,242
Illinois	IL	77,545	79,795	53,669	56,905	48,404	54,090	49,128	57,294
Indiana	IN	201,042	198,802	109,788	110,015	112,544	110,234	113,942	106,203
Iowa	IA	62,241	63,426	48,330	48,522	46,068	45,567	47,395	47,612
Kansas	KS	70,779	78,672	36,291	39,660	38,907	39,715	36,521	39,212
Kentucky	KY	149,032	148,360	73,614	71,148	74,046	71,469	75,800	69,673
Louisiana	LA	43,892	44,607	35,068	36,310	34,989	35,649	35,078	36,219
Maryland	MD	17,063	19,772	17,065	19,849	16,967	17,268	17,064	20,022
Massachusetts	MA	6,201	6,431	6,616	6,797	5,960	5,960	6,624	6,811
Michigan	MI	95,021	95,073	62,226	60,614	62,257	61,163	64,728	63,565
Minnesota	MN	48,892	49,444	32,589	32,970	33,546	33,115	31,868	32,214
Missouri	MO	74,492	79,515	69,168	63,475	57,681	57,682	63,898	63,898
Nebraska	NE	51,597	51,711	32,612	33,718	31,842	32,722	33,591	34,140
New Jersey	NJ	15,285	15,548	11,816	12,002	11,525	11,956	11,780	11,914
New York	NY	22,456	24,850	22,659	24,961	22,031	22,917	22,171	24,963
North Carolina	NC	59,714	59,781	59,688	57,678	51,800	51,800	57,678	57,678
Ohio	OH	156,728	161,040	95,594	94,882	96,662	91,566	102,774	96,927
Pennsylvania	PA	191,749	194,916	112,909	113,620	113,455	113,016	115,684	113,381
South Carolina	SC	46,560	46,310	34,121	33,302	32,703	32,349	33,202	33,184
Tennessee	TN	68,543	68,890	28,460	28,188	27,655	26,850	28,482	28,849
Virginia	VA	32,571	28,705	31,415	26,999	27,423	26,932	29,664	28,191
West Virginia	WV	102,251	99,623	52,587	47,683	51,990	48,489	57,878	50,057
Wisconsin	WI	45,904	49,489	35,559	35,946	36,384	36,576	37,095	36,940
Arizona	AZ	80,943	73,100	80,943	73,053	80,943	73,058	80,943	73,065
Arkansas	AR	43,134	44,703	25,255	26,173	26,536	26,308	25,162	25,812
California	CA	17,539	15,872	17,535	15,905	17,535	15,903	17,535	15,902
Colorado	CO	59,357	59,454	59,754	59,755	59,789	59,779	59,754	59,686
Idaho	ID	397	398	397	397	397	397	397	397
Maine	ME	3,036	2,534	3,104	2,530	3,246	2,594	3,081	2,552
Mississippi	MS	37,016	36,634	22,510	22,598	22,540	22,599	22,510	22,596
Montana	MT	36,761	36,800	36,764	36,789	36,764	36,789	36,762	36,790
Nevada	NV	19,893	29,115	19,891	29,117	19,891	29,117	19,891	29,117
New Hampshire	NH	2,515	2,515	2,444	2,456	2,265	2,456	2,144	2,378
New Mexico	NM	51,134	51,160	51,134	51,178	51,134	51,178	51,134	51,171
North Dakota	ND	59,551	59,559	59,551	59,551	59,551	59,551	59,551	59,551

Oklahoma	OK	86,661	80,886	54,584	50,151	54,494	50,170	54,584	50,105
Oregon	OR	13,780	13,889	13,782	13,889	13,782	13,889	13,782	13,889
Rhode Island	RI	220	280	208	276	254	305	222	262
South Dakota	SD	15,116	15,137	15,116	15,132	15,116	15,132	15,116	15,132
Texas	TX	159,170	165,765	140,046	147,556	140,225	147,654	139,281	145,248
Utah	UT	64,074	64,088	64,074	64,070	64,074	64,070	64,074	64,070
Vermont	VT	0	0	0	0	0	0	0	0
Washington	WA	18,214	18,374	18,213	18,359	18,213	18,362	18,213	18,350
Wyoming	WY	72,956	72,963	72,956	72,953	72,965	72,953	72,955	72,953
Nationwide total		**2,882,511**	**2,860,999**	**2,083,320**	**2,039,392**	**2,054,845**	**2,011,274**	**2,062,406**	**2,008,837**

[A] Emissions are for fossil EGUs with capacity greater than 25 MW.

Electrical Generating Unit Ozone Season NO_x Emissions for Base Case and Regulatory Alternatives (tons)[A]

State		Base Case		State Budgets/ Limited Trading		State Budgets/ Intrastate Trading		Direct Control	
		2012	***2014***	***2012***	***2014***	***2012***	***2014***	***2012***	***2014***
Alabama	AL	29,676	26,730	29,428	26,461	29,655	26,287	29,552	26,557
Arkansas	AR	20,420	21,529	11,715	11,943	11,772	11,957	11,626	11,939
Connecticut	CT	1,198	1,203	1,169	1,210	1,177	1,203	1,172	1,203
Delaware	DE	1,767	1,675	1,876	1,991	1,973	1,991	1,881	1,932
District of Columbia	DC	1	1	1	1	1	1	1	1
Florida	FL	94,007	87,324	59,509	54,860	56,939	54,675	46,142	40,808
Georgia	GA	35,036	21,789	32,615	19,529	32,144	19,530	33,131	19,591
Illinois	IL	24,085	23,881	22,393	24,644	19,796	22,547	20,754	24,415
Indiana	IN	49,967	48,053	46,204	46,482	47,827	46,817	48,035	44,594
Kansas	KS	30,535	34,284	15,477	17,218	16,672	17,337	15,485	17,029
Kentucky	KY	30,907	29,843	30,161	29,286	30,697	29,581	31,189	29,023
Louisiana	LA	21,188	20,980	16,617	16,924	16,693	16,643	16,622	16,844
Maryland	MD	7,219	8,310	7,186	8,391	7,089	7,198	7,186	8,435
Michigan	MI	28,038	28,119	25,917	25,498	25,819	25,773	28,013	27,600
Mississippi	MS	16,482	16,547	8,080	8,116	8,110	8,118	8,080	8,114
New Jersey	NJ	5,254	5,501	5,209	5,441	5,007	5,400	5,257	5,363
New York	NY	10,622	11,859	10,686	12,012	10,399	11,049	10,457	12,010
North Carolina	NC	25,831	25,765	25,800	24,852	22,498	22,194	24,799	24,739
Ohio	OH	40,641	43,099	40,631	39,337	40,490	38,263	43,592	41,113
Oklahoma	OK	42,871	38,249	27,503	24,175	27,503	24,174	27,503	24,129
Pennsylvania	PA	47,841	48,900	48,531	48,745	48,271	48,271	49,934	48,219
South Carolina	SC	15,223	15,111	14,584	14,199	14,251	14,047	14,196	14,134
Tennessee	TN	11,623	12,010	11,612	11,858	11,272	10,945	11,634	11,997
Texas	TX	78,315	79,118	66,442	68,314	66,437	68,420	65,676	67,377
Virginia	VA	13,861	12,494	13,648	11,601	12,607	11,491	13,417	12,294
West Virginia	WV	23,803	24,149	22,948	20,034	22,234	21,251	24,112	21,285
Arizona	AZ	35,296	32,672	35,296	32,625	35,296	32,630	35,296	32,637
California	CA	8,679	7,087	8,676	7,105	8,676	7,103	8,676	7,102
Colorado	CO	25,852	25,861	25,949	26,087	25,949	26,111	25,949	26,019
Idaho	ID	135	135	135	135	135	135	135	135
Iowa	IA	26,663	27,523	20,712	20,829	19,649	19,648	20,417	20,452
Maine	ME	964	963	1,022	958	1,152	994	1,043	964
Massachusetts	MA	2,489	2,646	2,836	2,825	2,472	2,547	2,844	2,829
Minnesota	MN	21,153	21,544	13,993	14,082	14,662	14,224	13,865	14,054
Missouri	MO	32,584	34,641	30,760	27,881	27,691	25,439	30,189	28,516
Nebraska	NE	22,551	22,715	14,071	14,537	13,709	14,216	14,381	14,960
Montana	MT	16,077	16,109	16,078	16,104	16,078	16,104	16,078	16,105
Nevada	NV	9,216	13,226	9,215	13,223	9,215	13,223	9,215	13,223
New Hampshire	NH	1,134	1,134	1,063	1,068	893	1,068	772	1,008
New Mexico	NM	22,561	22,438	22,561	22,446	22,561	22,446	22,561	22,443

North Dakota	ND	26,029	26,033	26,029	26,029	26,029	26,029	26,029	26,029
Oregon	OR	5,398	5,537	5,401	5,537	5,401	5,537	5,401	5,537
Rhode Island	RI	93	103	89	103	111	105	89	103
South Dakota	SD	6,626	6,644	6,626	6,642	6,626	6,642	6,626	6,642
Utah	UT	28,076	28,084	28,076	28,076	28,076	28,076	28,076	28,076
Vermont	VT	0	0	0	0	0	0	0	0
Washington	WA	7,152	7,437	7,152	7,424	7,152	7,428	7,152	7,417
Wisconsin	WI	19,422	21,329	15,465	15,484	15,445	15,736	15,754	16,074
Wyoming	WY	31,848	31,851	31,848	31,847	31,854	31,847	31,848	31,847
Nationwide total		**1,056,410**	**1,042,235**	**918,994**	**894,168**	**906,160**	**882,453**	**911,841**	**882,917**

[A] Emissions are for fossil EGUs with capacity greater than 25 MW.

APPENDIX D

INTEGRATED PLANNING MODEL RUNS

Table A-1 lists the Integrated Planning Model (IPM) runs used in analyses presented in Chapters 7 and 10. Table A-2 lists the IPM parsed files used in air quality and health benefits analyses.[91] Chapters 7 and 10 describe the IPM runs in greater detail. The IPM runs and parsed files can be found in the docket for this rulemaking (Docket ID No. EPA-HQ-OAR-2009-0491).

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[91] Whereas IPM output files report aggregated results for "model" plants (i.e., aggregates of generating units with similar operating characteristics), parsed files show IPM results at the generating unit level.

Table A-1. IPM Runs Used in Transport Rule Analyses

Run Name	**Run Description**
Transport Rule Base Case	
TR_Base_Case	Base Case model run, which includes the national Title IV SO_2 cap-and-trade program; NO_x SIP Call regional ozone season cap-and-trade program; and settlements and state rules through February 3, 2009. This run represents conditions without the proposed Transport Rule and without the rule it would replace (CAIR).
Transport Rule remedy options†*	
TR_SB_Limited_Trading	This run models the State Budgets/Limited Trading proposed remedy described in the Transport Rule preamble.
TR_SB_Intrastate_Trading	This run models the State Budgets/Intrastate Trading alternative remedy described in the Transport Rule preamble.
TR_Direct_Control	This run models the Direct Control alternative remedy described in the Transport Rule preamble.
Additional runs used for analysis of scenarios in Regulatory Impact Analysis†*	
TR_A-4_less_stringent	This run models the less stringent scenario described in Chapter 10 of the proposed Transport Rule RIA.
TR_A-4_more_stringent	This run models the more stringent scenario described in Chapter 10 of the proposed Transport Rule RIA.

[*] In addition to base case assumptions, these runs include additional control options for units between 25 and 100 MW capacity. See IPM documentation for more details.

[†] In addition to base case assumptions, these runs include unit-specific adjustments based on recent emissions data for 33 units. See IPM documentation for more details.

Table A-2. IPM Parsed Files Used in Transport Rule Analyses

Run with Parsed Results	Years Parsed
TR_Base_Case	2012, 2014
TR_SB_Limited_Trading	2012, 2014
TR_SB_Intrastate_Trading	2012, 2014
TR_Direct_Control	2012, 2014

APPENDIX E
ALLOWANCE VALUES FOR EMISSIONS TRADING PROGRAMS

Allowance Values for Emissions Trading Programs

As discussed in Chapter 7 above, the proposed State Budgets/Limited Trading remedy and alternative State Budgets/Intrastate Trading remedy both include emissions trading programs. State Budgets/Limited Trading features programs for annual NO_x,[92] ozone-season NO_x,[93] annual SO_2 for the 15 Group 1 states,[94] and annual SO_2 for the 13 Group 2 states.[95] In contrast, State Budgets/Intrastate Trading includes separate emissions trading programs for each pollutant in each affected state, a total of 82 programs.

Tables E-1 through E-3 below show the projected allowance values resulting from modeling of these remedy options using the Integrated Planning Model (IPM). Values for SO_2 reflect the marginal cost of reducing SO_2, including the operation of dispatchable flue gas desulfurization controls (FGD). Section 6 of the TSD "Updates to EPA Base Case v3.02 EISA Using the Integrated Planning Model" contains details on the definition and determination of dispatchable controls.

Similarly, NO_x allowance values reflect the marginal cost of reducing NO_x, including the variable operation and maintenance costs of existing controls.[96] For example, an allowance price of $500 per ton (reflecting variable cost of operating SCRs) for annual NO_x

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[92] The 28 states in this program are Alabama, Connecticut, Delaware, District of Columbia, Florida, Georgia, Illinois, Indiana, Iowa, Kansas, Kentucky, Louisiana, Maryland, Massachusetts, Michigan, Minnesota, Missouri, Nebraska, New Jersey, New York, North Carolina, Ohio, Pennsylvania, South Carolina, Tennessee, Virginia, West Virginia, and Wisconsin.

[93] The 26 states in this program are Alabama, Arkansas, Connecticut, Delaware, District of Columbia, Florida, Georgia, Illinois, Indiana, Kansas, Kentucky, Louisiana, Maryland, Michigan, Mississippi, New Jersey, New York, North Carolina, Ohio, Oklahoma, Pennsylvania, South Carolina, Tennessee, Texas, Virginia, and West Virginia.

[94] Group 1 states are Georgia, Illinois, Indiana, Iowa, Kentucky, Michigan, Missouri, New York, North Carolina, Ohio, Pennsylvania, Tennessee, Virginia, West Virginia, and Wisconsin.

[95] Group 2 states are Alabama, Connecticut, Delaware, District of Columbia, Florida, Kansas, Louisiana, Maryland, Massachusetts, Minnesota, Nebraska, New Jersey, and South Carolina.

[96] Because the variable operating cost of an SCR is very similar for most SCRs (about $500/ton), EPA's modeling of dispatchable SCRs assumes that those SCRs are operated under a cap, rather than modeling each economic decision independently. Because these costs are not then factored into IPM's marginal cost calculation, EPA exogenously assigns an allowance price of $500/ton if the projected NO_x allowance price in IPM is less than $500/ton.

indicates that compliance can be achieved through year-round operation of existing advanced NO_x controls without the use of more expensive compliance options, such as incrementally installing new post-combustion controls. Rules for determining whether NO_x controls in the model operate year-round can be found in the Chapter 3 Appendix of the Documentation for EPA Base Case 2004 Using IPM (http://www.epa.gov/airmarkets/progsregs/epa-ipm/docs/bc3appendix.pdf).

Table E-1. Projected Regional Allowance Prices for State Budgets/Limited Trading Preferred Approach
(2006$)

		2012	**2014**
Annual NO_x		$500	$500
Ozone-season NO_x		$500	$500
Annual SO_2	Group 1	$1,000	$1,100
	Group 2	$800	$300

Source: EPA 2010

For SO_2 allowances, in group 1 states, allowance prices are lower than $2,000/ton in 2014 for two reasons (note that all allowance prices are across all states in the applicable trading region). First, because of banking allowances in 2012 and 2013, less reduction is necessary in 2014. Second, because of the interstate trading companies have more flexibility to take advantage of the lowest cost reduction opportunities. Group 2 allowances are lower than $500/ton in 2014, because in some states, additional SO_2 controls are installed between 2012 and 2014 due to requirements outside of this rule. Allowance prices are higher than $500/ton in 2012 because EPA state budgets were based on the performance of units in 2009 (which reflects improved performance of units complying with Phase I of the CAIR NO_x program and banking substantially for the CAIR SO_2 program), while modeling in IPM did not reflect all of those recent improvements. Actual improved performance that was used to develop the budgets is therefore not always reflected in the model (e.g. better performance of low NO_x burners and scrubbers and access to lower sulfur coals). In its final modeling, EPA will update IPM to reflect the better unit performance seen in 2009. This will likely result in lower allowance prices.

Table E-2. Projected Annual NO$_x$ and SO$_2$ Allowance Prices for State Budgets/Intrastate Trading Alternative (2006$)

		Annual NO$_x$		Annual SO$_2$	
		2012	**2014**	**2012**	**2014**
Alabama	AL	$500	$500	$1,400	$1,100
Connecticut	CT	$2,900	$3,400	$0[97]	$0
Delaware	DE	$500	$500	$4,400	$3,100
Florida	FL	$500	$500	$700	$300
Georgia	GA	$500	$500	$1,700	$2,000
Illinois	IL	$1,500	$1,800	$2,500	$1,500
Indiana	IN	$500	$500	$1,300	$1,500
Iowa	IA	$500	$600	$900	$1,000
Kansas	KS	$500	$500	$100	$100
Kentucky	KY	$500	$500	$1,500	$1,700
Louisiana	LA	$500	$500	$1,900	$1,900
Maryland	MD	$6,800	$8,000	$1,400	$1,700
Massachusetts	MA	$17,700	$5,600	$3,200	$2,200
Michigan	MI	$500	$500	$1,500	$1,800
Minnesota	MN	$500	$500	$200	$200
Missouri	MO	$1,700	$1,200	$1,000	$1,200
Nebraska	NE	$500	$500	$1,200	$800
New Jersey	NJ	$500	$500	$6,000	$2,200
New York	NY	$500	$500	$2,200	$2,600
North Carolina	NC	$7,100	$2,700	$1,200	$1,400
Ohio	OH	$600	$700	$1,100	$1,300
Pennsylvania	PA	$500	$600	$1,100	$1,300
South Carolina	SC	$700	$800	$1,700	$1,300
Tennessee	TN	$1,600	$1,800	$1,800	$1,700

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[97] In Connecticut IPM's original modeling (which was used in the determination of significant contribution) did not account for the fact that Connecticut's largest coal unit uses a significantly lower sulfur coal than normally projected by IPM. In the final cost modeling, EPA adjusted the coals assigned to this unit to reflect the sulfur content of the coal actually used. This results in Connecticut meeting its budget with an allowance cost of $0/ton (e.g. without making any emission reductions). In modeling Connecticut in the final rule, EPA will account for the lower sulfur content coal throughout the entire analytic process. This could result in a lower SO$_2$ budget for Connecticut.

		Annual NO_x		Annual SO_2	
		2012	**2014**	**2012**	**2014**
Virginia	VA	$1,000	$1,200	$2,600	$1,800
West Virginia	WV	$1,300	$600	$1,100	$1,300
Wisconsin	WI	$500	$500	$1,300	$1,500

Source: EPA, 2010

While most of the allowance prices seen in the state-by-state modeling for the Intrastate Trading alternative are consistent with the allowance prices that would be expected based on EPA's significant contribution modeling, there are some exceptions. For example, while two-thirds of the states have NO_x allowance prices near $500/ton, a number of states do see higher allowance prices. This is because the budgets were based on the lower of IPM emission projections or projections of emissions using actual unit performance data (as explained in the Technical Support Document – "State Budgets, Unit Allocations and Unit Emission Rates"). Because IPM has not been updated to reflect actual NO_x emission rates seen in 2009, IPM projects that it will be harder to meet those budgets than the most recent real world data shows it will be (this is why EPA considered the most current data as well as the IPM projections in setting the budgets). Between proposal and final rulemaking, EPA will be updating IPM to reflect these lower emission rates. This also occurs in some of the SO_2 budgets. It is most prevalent in either smaller states or states with a larger percentage of well-controlled units, where the marginal cost curve for the remaining uncontrolled units is steeper and allowance prices are most sensitive to small changes in budgets. EPA will be revising IPM to reflect the most recent real world data between proposal and final.

Table E-3. Projected Ozone-season NO$_x$ Allowance Prices for State Budgets/Intrastate Trading Alternative (2006$)

		2012	2014
Alabama	AL	$500	$500
Arkansas	AR	$500	$500
Connecticut	CT	$500	$500
Delaware	DE	$500	$500
Florida	FL	$500	$500
Georgia	GA	$700	$500
Illinois	IL	$1,000	$1,200
Indiana	IN	$500	$500
Kansas	KS	$500	$500
Kentucky	KY	$500	$500
Louisiana	LA	$500	$500
Maryland	MD	$500	$500
Michigan	MI	$500	$500
Mississippi	MS	$500	$500
New Jersey	NJ	$500	$500
New York	NY	$1,100	$1,300
North Carolina	NC	$500	$500
Ohio	OH	$500	$500
Oklahoma	OK	$500	$500
Pennsylvania	PA	$1,300	$700
South Carolina	SC	$500	$500
Tennessee	TN	$1,100	$1,300
Texas	TX	$500	$500
Virginia	VA	$500	$500
West Virginia	WV	$500	$500

Source: EPA 2010