

Chapter 4 BENCHMARK AND BASELINE EMISSIONS, AND AIR QUALITY

4.1 Results in Brief

The foundation for this analysis is a 1990 emissions inventory which is projected to a future year. In actuality, that year is 2010. However, for purposes of this analysis and to comport with legislative changes regarding implementation of the Regional Haze and PM_{2.5} National Ambient Air Quality Standards (NAAQS) rules, the year is 2015. That year, 2015, is near the end of the first long-term strategy, 2018. For that projection, we find that emissions of VOC, NO_x, SO₂, and secondary organic aerosols (SOA) are estimated to decrease relative to 1990 levels. This is due in part to other Clean Air Act (CAA) programs such as the Maximum Achievable Control Technology (MACT), Federal Motor Vehicle Control Program (FMVCP), New Source Performance Standards (NSPS), and the final NO_x State Implementation Plan (SIP) which are not part of the implementation program for the PM_{2.5} and Ozone NAAQS that were promulgated in 1997. Other emissions are projected to increase over this period due to increases in economic activity despite the emission reduction requirements in the CAA.

The air quality modeling associated with this projected emission inventory leads to air quality projections that are used to determine the amount of air quality improvement to meet the illustrative progress goals explained in Chapter 3. The projected emissions and air quality levels serve as the benchmark for this analysis.

With implementation of the PM_{2.5} and Ozone NAAQS and the Tier II program, there will be emission reductions and improved air quality, including visibility. The particulate matter concentration reductions and visibility improvements associated with partial attainment of the NAAQS and implementation of a Tier II program are estimated using the Phase II Climatological Regional Dispersion Model (CRDM). The resulting air quality improvement will bring several counties with Class I areas into achieving with the illustrative visibility progress goals. In particular, between 27 and 55 counties with Class I areas will achieve the progress goals incidental to partial attainment of the new NAAQS and a Tier II program.

The air quality levels after partial attainment of the new PM_{2.5} and Ozone NAAQS and the Tier II program serve as the baseline for the incremental benefit, cost, and economic impact analyses contained in this regulatory impact analysis (RIA). In particular, any deciview goals not met in the baseline are addressed with additional or incremental control measures and strategies. The control measures applied in this RIA will result in an additional 26 to 60 counties having Class I areas meeting the illustrative progress goals under the emissions control case in which fugitive dust emissions controls are allowed (Case A), and 11 to 53 counties having Class I areas meeting the illustrative progress goals under the emissions control case in which fugitive dust emissions controls are not allowed (Case B). The range is due to differences in illustrative progress goals as well as differences in control strategies.

4.2 Introduction

This chapter describes the methods used to estimate baseline emissions and air quality in 2015 in order to assess the incremental costs, benefits and economic impacts of the illustrative RH progress goals.¹ The assessments are conducted from a consistent analytical baseline that is benchmarked to available 2010 CAA projections for emissions growth, levels of controls, and their contribution towards visibility improvement. A single emissions inventory employing consistent methods is used as the basis for the RH analyses. The year 2015 is selected as the year of analysis to provide an appropriate period in which 1) major programs of the CAA of 1990 should be reaching full implementation, and 2) the Ozone and PM standards promulgated in 1997 are to be achieved. Considerable progress is expected in attaining the new criteria air pollutant standards. The year 2015 is best understood for purposes of this report as a nominal “snapshot” year for presenting estimated visibility, costs, economic impacts and benefits; it should be noted that these impacts are based on 2010 emission projections, projections that serve as a proxy for projections for the year 2015.

The RH analyses have been constructed such that benefits, economic impacts, and costs are estimated incremental to those derived from the combined effects of implementing both the CAA of 1990 and the 8-hour Ozone and PM_{2.5} 15/65 standards as of the year 2015. The effects of implementing the CAA of 1990 are called the benchmark for determining the starting point for analyzing the potential visibility improvements associated with the RH rule. The effects of implementing the 8-hour Ozone and PM_{2.5} 15/65 standards promulgated in 1997 including a modest version of the Tier II program are called the baseline from which the incremental effects of meeting these illustrative RH progress goals under both emission control cases (Cases A and B) are measured. These analyses provide a “snapshot” of air quality impacts, costs, economic impacts, and benefits associated with implementation of these illustrative RH progress goals from a baseline of partial attainment of the Ozone and PM_{2.5} 15/65 standards that is benchmarked to future CAA implementation.

Some Class I area counties are not expected to reach the illustrative progress goals as a result of controls put in place to achieve the Ozone and PM standards. Once these Class I area counties have been identified within the set of PM monitored areas, the analysis assumes additional control strategies on a local, regional, and national basis for the purpose of allowing these Class I area counties to meet the goals. It should be noted that while these areas are identified from within monitored areas only, control requirements, costs, benefits, and other economic impacts are estimated for both PM monitored and unmonitored areas. This results

¹ 2018 is the end of the period for the first long-term strategy. The term “long-term strategy” refers to the set of emission reduction measures the State includes in its SIP in order to meet the reasonable progress goal it has set. 2015 is a nominal “snapshot” year that reflects the partial attainment control cases for the Ozone and PM_{2.5} NAAQS included in the baseline, and is near the end of the period for the first long-term strategy.

from the fact that controls are expected to be applied to emission sources outside of Class I area counties so that these counties can meet the illustrative progress goals.

The EPA believes that the monitored counties' analytic approach for identifying Class I area counties that cannot meet the illustrative progress goals is most appropriate because 1) the likelihood of significant inability to comply with visibility progress goals in unmonitored areas after modeled emission controls are assumed is small; 2) serious modeling difficulties exist that prevent reliable prediction of visibility progress in unmonitored areas; and 3) any such inability to meet RH progress goals in unmonitored areas may not be detected (U.S. EPA, 1997c). It is possible, however, that the placement of new PM monitors in the future may affect the estimates of counties' ability to meet these illustrative progress goals.

Figure 4-1 illustrates the analytical approach employed for this assessment. Base year emissions for 1990 are projected to 2010 by applying sector-specific growth factors. The CAA-mandated controls (i.e., control efficiencies or control-specific emission factors) then are applied to these future emissions to capture implementation of the 1990 CAA (our "benchmark"). The 2010 post-CAA control emissions are input to air quality models to predict baseline visibility levels from which Class I areas that cannot meet the progress goals subsequently are identified. Control measures to bring these areas to the point of meeting these progress goals are evaluated and applied in the cost analyses. Emission reductions achieved by these control measures determine the "post-control" visibility in these areas. The methodologies used to estimate visibility for assessing the RH progress goals are discussed in Chapter 6.

4.3 Estimation of 1990 Emissions and 2010 Emissions Projections

The initial step in the assessment of RH illustrative progress goals is the development of the 2010 CAA emission estimates. These emissions and associated air quality modeling serve as the benchmark for determining the starting point for analyzing creditable visibility improvements associated with the new NAAQS. The emissions estimation and projection methodologies build upon work conducted for the July 1997 Ozone, PM, and proposed RH rule Regulatory Impact Analysis (RIA) (U.S. EPA, 1997).

The major data sources and estimates for these RH analyses are as follows:

- ! Version 3 of the 1990 National Particulates Inventory (NPI v.3)(Pechan, 1996c)
- ! Bureau of Economic Analysis (BEA) projections of Gross State Product (GSP) (BEA, 1995) are used to estimate 2010 emissions.
- ! Utility sector CAA-control emission projections incorporate future utility deregulation and a 0.15 lb/MMBtu nitrogen oxides (NO_x) cap with trading and banking;

! The following CAA-mandated control assumptions are updated in the 2010 benchmark emissions estimates:

- OTAG Level 2 NO_x controls on industrial point sources in 37 OTAG States are applied (it should be noted that the methodology for this analysis was completed before the final NO_x SIP call of September 1998 was completed so the analysis in this RIA assumed the OTAG Level 2 NO_x controls across the OTAG States as a surrogate for the NO_x controls in the actual NO_x SIP call);
- Estimated emission reductions from 7/10 year Maximum Achievable Control Technology (MACT) standards are included;
- Proposed control requirements for Architectural and Industrial Maintenance (AIM) coatings and consumer and commercial products rules are incorporated.

Figure 4-1 Overview of Emissions and Air Quality Analytical Approach

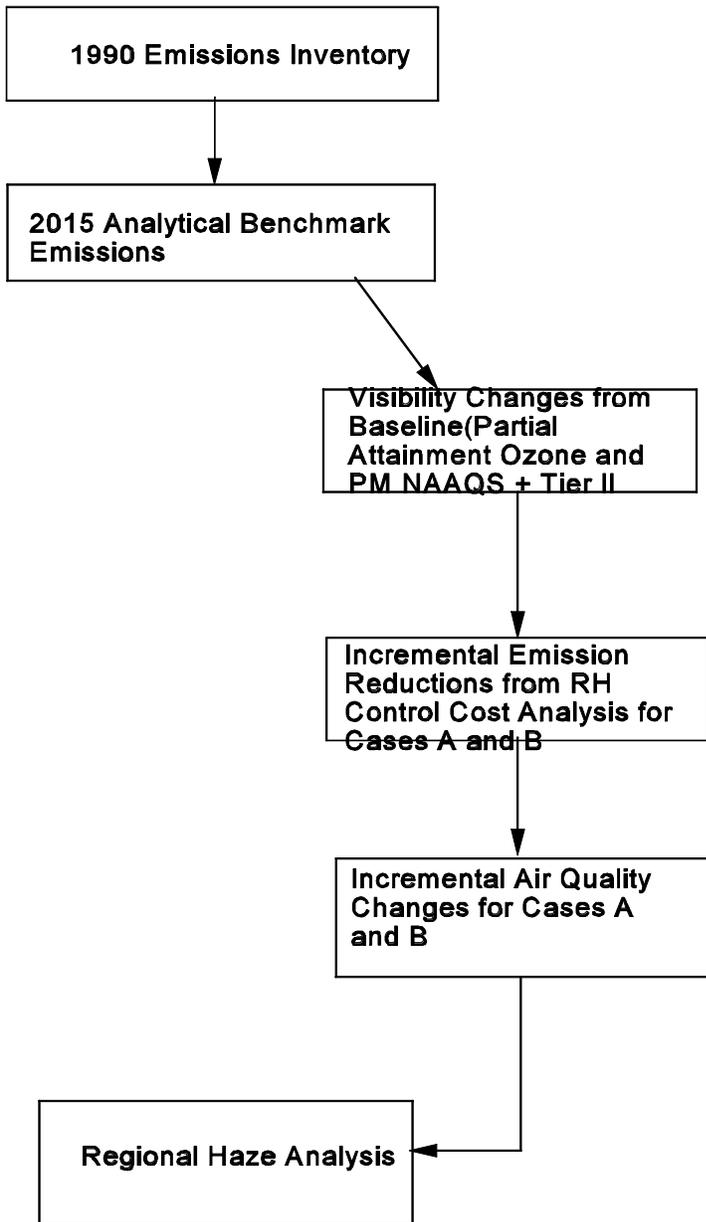
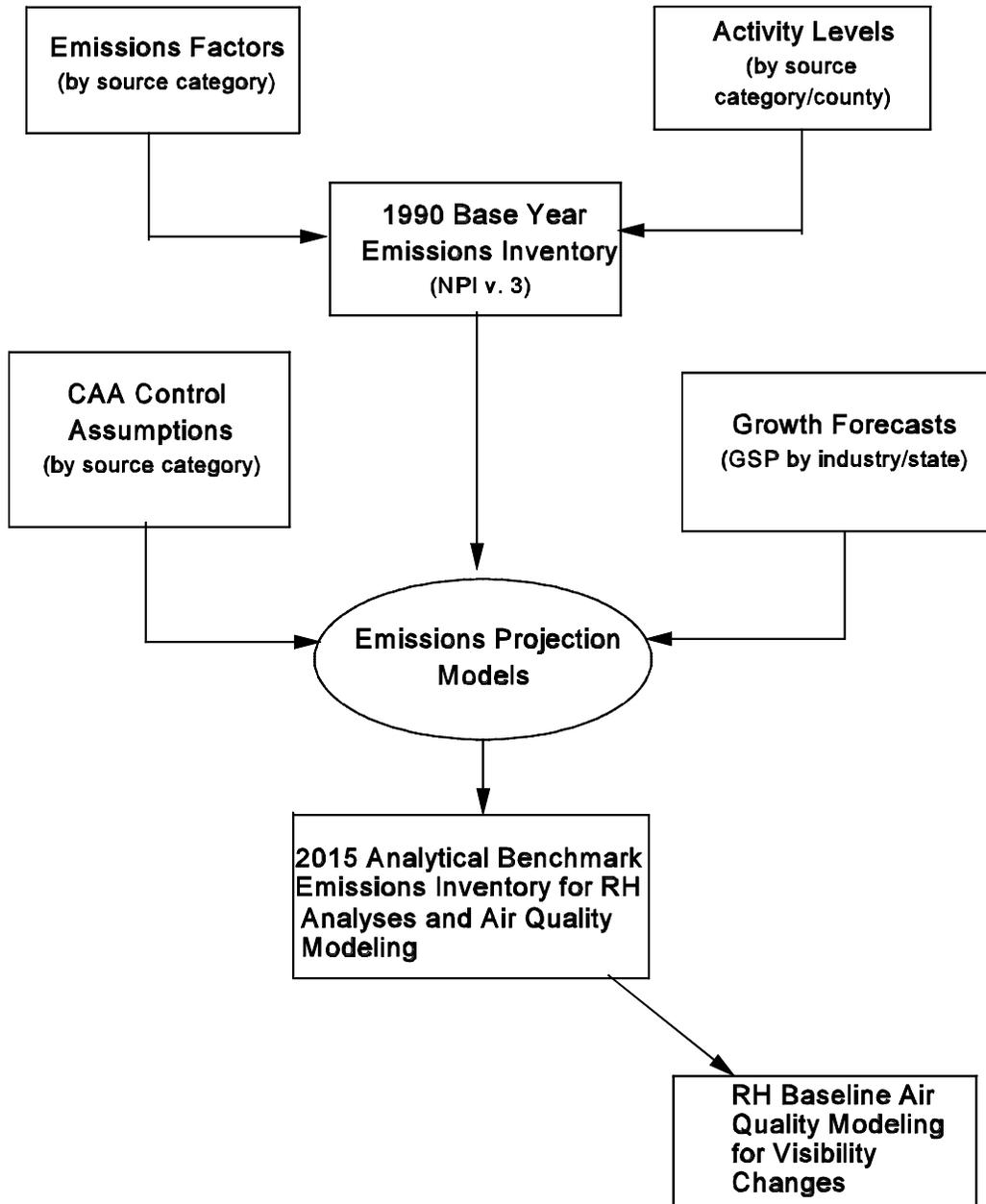


Figure 4-2 illustrates the steps followed in the development of 2010 benchmark emissions. First, source category-specific activity levels and emissions factors are used to estimate emissions for the base year 1990. Any pollution controls in place prior to 1990 are reflected in these base year values. Emissions are estimated for VOC, NO_x, sulfur dioxide (SO₂), primary PM₁₀ and PM_{2.5}, SOA, organic carbon (OC), elemental carbon (EC) and ammonia. As described in the introduction, certain VOC species, based on the reactivity of these organic compounds with atmospheric oxidants, form SOA (Grosjean and Seinfeld, 1989). To estimate SOA emissions, fractional aerosol coefficients (FACs) based on VOC species profiles for each Source Classification Code (SCC) are applied to 1990 VOC emissions (Pechan, 1997a). Biogenic VOC emissions are involved in ozone and SOA formation and are estimated for the base year inventory.

Additionally, ammonia plays a role in the formation of particulate ammonium sulfate and ammonium nitrate. However, anthropogenic emissions of ammonia are a small component of total ammonia emissions. The majority of the ammonia that enters the atmosphere is produced by the biological decomposition of organic material in soils, plant residues, and wastes from animals and humans (NAPAP, 1991). Given that ammonia is not a limiting factor in the formation of secondary particles, ammonia emissions are not considered in the RH control strategy analyses.

Because air quality modeling is conducted on the county level, emissions are estimated for all counties in the contiguous 48 States. The 1990 emissions are then input to an emissions projection model (e.g., Emission Reduction and Cost Analysis Model (ERCAM) for VOC and NO_x) that predicts emissions in 2010 based on State-level growth forecasts and control assumptions reflective of implementation of CAA-mandated programs. The resultant 2010 emissions, which serve as a proxy for emissions in 2015, then serve as inputs to the air quality modeling.

Figure 4-2 Development of 2010 Benchmark Emissions



4.3.1 Development of 1990 Benchmark Emissions Inventory

The 1990 emissions inventory that is the benchmark emissions inventory for the analyses of incremental effects contained in this RIA based on Version 3 of the National Particulate Inventory (NPI) (Pechan, 1996c; Pechan, 1997a).

The NPI is developed using a “top-down” approach to estimate national emissions at the county level. Top-down methods rely on existing data sources and use estimation techniques that are comprehensive but with less area-specific detail. In general, emissions factors for individual source types are applied to activity levels for source categories within the major emitting sectors (i.e., utility, industrial point, area, nonroad engines/vehicles, mobile sources and biogenics/natural sources). Emissions factors are expressed in terms of amount of a pollutant emitted for a given activity level (e.g., per ton of fuel consumed, per vehicle mile travelled). Emission factors developed by EPA are available for VOC, NO_x, SO₂, and PM₁₀. Because there are no emission factors for PM_{2.5}, a PM calculator program containing particle size distribution data for various source categories is used to develop these estimates (Pechan, 1994). The program estimates the fraction of PM emissions from both controlled and uncontrolled sources that are within the fine particle fraction (i.e., < 2.5 microns in diameter) and coarse particle fraction (i.e., between 2.5 and 10 microns in diameter). Finally, anthropogenic ammonia emission factors are a compilation of estimates based primarily on recent European studies (Asman, 1992; Battye et al., 1994).

For the States of California and Oregon and for prescribed burning and wildfire emissions in the 11 western States, emissions estimates based on a bottom-up assessment conducted by the Grand Canyon Visibility Transport Commission (GCVTC) are used (Radian, 1995). These emission estimates are derived from more recent and detailed surveys of emissions from various source categories.

Biogenic VOC emissions are developed based on EPA’s Biogenic Emissions Inventory System (BEIS) (Pierce et al., 1990). Biogenic SOA is estimated from application of VOC species-specific FACs to biogenic VOC emissions (Pechan, 1997a). Natural sources of PM emissions (i.e., wind erosion) are taken from the National Emission Trends Inventory (U.S. EPA, 1996h).

Table 4-1 summarizes the approaches used in development of the benchmark inventory.

4.3.2 1990 Benchmark Emissions Inventory Results and Discussion

Table 4-2 presents a summary of 1990 emissions by pollutant and major sector. Area sources are the largest contributor to anthropogenic VOC emissions in 1990 (45 percent of total national anthropogenic VOC emissions). Biogenic and natural sources of VOC emissions are estimated to be roughly equivalent in magnitude to the anthropogenic total. Motor vehicles account for 33 percent of total national NO_x emissions with 46 percent of the motor vehicle emissions contributed by cars (i.e., light-duty gasoline vehicles). With regard to national SO₂ emissions, the utility sector is the largest emitter (71 percent). Area sources account for the bulk of PM₁₀ and PM_{2.5} emissions. Anthropogenic fugitive dust sources contribute the majority of primary PM₁₀ and PM_{2.5} emissions. More recent emission inventory efforts indicate that these estimates are overstated. Refer to Section 4.3.3 for a discussion of the potential biases in these estimates.

It should be noted that the ambient air quality impacts of emissions on visibility levels from any individual sector may not be proportional to their contribution to national emissions. The reader is referred to the RH air quality modeling sections (Chapter 4) and the cost chapter (Chapter 6) to understand how emissions from various source categories impact visibility levels.

**Table 4-1
Benchmark Emission Inventory - Summary of Approach**

Major Source Type	Modeling Approach/Data Sources
Industrial Point Sources	1985 National Acid Precipitation Assessment Program (U.S. EPA, 1989) emissions inventory grown to 1990 based on historical BEA earnings data (BEA, 1990). PM ₁₀ and PM _{2.5} emissions based on total suspended particulate (TSP) emissions and particle-size multipliers (U.S. EPA, 1994b). California and Oregon State data substituted (Radian, 1995).
Electric Utilities	Based on EIA-767 fuel use for 1990 and unit-specific emission limits (DOE, 1991b) and AP-42 emission rates (U.S. EPA, 1995a)
Nonroad	Internal Combustion Engines/Vehicles (VOC, NO _x , PM _{2.5} , PM ₁₀): 1991 Office of Mobile Sources (OMS) Nonroad Inventory (U.S. EPA, 1991b) Internal Combustion Engines/Vehicles (SO ₂) and Aircraft, Commercial Marine Vessels, Railroads: 1985 NAPAP (U.S. EPA, 1989) grown to 1990 based on historical BEA earnings data (BEA, 1990).
Motor Vehicles	Federal Highway Administration travel data (FHWA, 1992), MOBILE5a/PART5 emission factors (U.S. EPA, 1993a).
Area Sources	1985 NAPAP inventory grown to 1990 based on historical BEA earnings data (BEA, 1990) and State Energy Data System (SEDS) fuel use data (DOE, 1991a); emission factor changes for selected categories (U.S. EPA, 1995a). California and Oregon State data substituted (Radian, 1995).
Solvents	National solvent usage estimates by end-use category from U.S. Paint Industry Data Base and industrial solvent marketing reports (Connolly et al., 1990). Allocated to county level based on industry employment and population (BOC, 1987, 1988a, 1988b).
Fugitive Dust (PM ₁₀ , PM _{2.5}) Agricultural Tilling Construction Unpaved and Paved Roads Livestock	U.S. Department of Agriculture data (USDA, 1991), U.S. EPA PM ₁₀ emission factors (U.S. EPA, 1995a). Census Bureau Construction Expenditures (BOC, 1992), EPA PM ₁₀ emission factors (U.S. EPA, 1995a). EPA PART5 emission factors (U.S. EPA, 1994c), FHWA travel data (FHWA, 1992). USDA farming activity levels (USDA, 1991), EPA PM ₁₀ emission factors (U.S. EPA, 1995a). Particle size multipliers are applied to PM ₁₀ emissions to estimate PM _{2.5} emissions (U.S. EPA, 1994b).
Biogenic VOC	Emissions for eight landcover types based on a forest canopy model which was used to account for the effects of solar radiation, temperature, humidity, and wind speed on predicted VOC emission rates (Lamb et al., 1993).
Wind Erosion	PM wind erosion emissions from agricultural lands based on acres of spring- or fall-planted crops in each State from the USDA and the expected dust flux (emission rate) based on a simplified version of the NAPAP method (Gillette, 1991). Emissions were distributed to the county level based on rural land area.
Agricultural Ammonia (NH ₃)	NH ₃ emissions for livestock feedlots and fertilizers based on Census of Agriculture data (BOC, 1992) and EPA-recommended emission factors (Battye et al., 1994).

4.3.3 Key Uncertainties Associated with 1990 Emissions

Given the on-going nature of emissions research, improvements to emissions estimation methodologies will continue to be made. However, there will be uncertainties associated with top-down approaches that rely on existing data sources and less source-specific data.

Because development of 1990 emissions employs emission factors as primary inputs, more uncertain emission estimates result than if source-specific stack tests, load-curve based factors or continuous emissions monitoring (CEM) data are used. The differences in utility SO₂ and NO_x emissions between alternative estimation methodologies, however, are not that large. Comparisons of SO₂ CEM data with estimates based on SO₂ emission factors and fuel consumption for a sample of plants showed that the two techniques produced emission estimates within an average of 8 percent at the State level (Schott, 1996). A comparison of NO_x emissions based on CEM data and NO_x emissions based on EPA emission factors for a sample of utilities in Louisiana resulted in a difference of 22 percent between the two methods (Schott, 1996). However, for area, non-road and motor vehicle sources where source-specific data are mostly unavailable, emission factors are applied to activity levels for each county. Thus, the potential uncertainties are greater for these sources than the better inventoried utility and industrial point sources (Pechan, 1996a). Finally, any possible biases in national emissions estimates from using emissions factors is unclear.

Use of particle size multipliers to estimate PM₁₀ and PM_{2.5} emissions from TSP data yields uncertain results relative to application of PM₁₀ or PM_{2.5} emission factors. The degree of uncertainty may vary by source category; however, there is no known bias in these factors.

**Table 4-2
Summary of 1990 National Emissions Estimates by Major Sector**

Major Sector	VOC (1000 tpy)	NOx (1000 tpy)	SO₂ (1000 tpy)	PM₁₀ (1000 tpy)	PM_{2.5} (1000 tpy)	SOA (1000 tpy)	OC (1000 tpy)	EC (1000 tpy)
Utility	37	7,426	15,865	283	109	1	10	25
Industrial Point Area	3,467	2,850	4,644	926	589	35	58	13
Nonroad	10,098	2,100	1042	35,290	7,639	92	1,066	139
Motor Vehicle	2,054	2,836	242	336	293	23	99	160
	6,811	7,446	568	355	291	48	57	59
Anthropogenic Subtotal	22,466	22,656	22,359	37,190	8,921	198	1,290	396
Biogenics	25,988					3,325		
Natural Sources	248	89	1	5,429	995			
TOTAL	48,702	22,745	22,360	42,619	9,916	3,523	1,290	396

Note: Emissions estimates may not sum due to rounding.
1990 fugitive dust emissions have not been adjusted here as described in Section 4.4.
Air quality impacts from major emitting sectors are not necessarily proportional to their contribution to national emissions estimates. See Sections 4.4 and 4.5 and Chapter 6 of this RIA.

The more recent biogenic emissions estimates from BEIS2 (Geron et al., 1994) are not incorporated in version 3 of the NPI. VOC emissions estimated using BEIS2 are 28 percent higher nationally than biogenics included in the base year emissions. These higher VOC estimates also lead to higher biogenic SOA nationally. However, given that BEIS2 emission estimates have better spatial resolution, higher or lower biogenic VOC emissions for specific counties may result relative to the NPI estimates. Thus at the national level, the estimates of biogenic VOC and SOA may be understated. However, due to the better spatial resolution in the BEIS2 compared to the resolution of the VOC and SOA emissions in the NPI version 3, the bias is less clear in any particular county (Pechan, 1997a).

The most recent fugitive dust emissions estimates developed for the National Emissions Trends Inventory (U.S. EPA, 1997h) indicate that NPI version 3 PM_{10} fugitive dust emissions may be overestimated by 40 percent and $PM_{2.5}$ fugitive dust emissions may be overestimated by 72 percent relative to the Trends estimates. The Trends fugitive dust information was available after PM air quality modeling had been completed and therefore could not be incorporated into this analysis. See Section 4.4 for a discussion of the implications of this overestimate of fugitive dust emissions on modeled visibility levels. Of particular interest is that the $PM_{2.5}$ emission estimate for agricultural operations (tilling and windblown dust) was decreased by about 50 percent, or 1 million tons per year. The emissions decrease from farming operations is clearly concentrated in the farm belt of the central US. Thus, the RH air quality analysis is likely biased toward overestimating fugitive dust impacts on visibility impairment in farming areas, relative to other areas. While some other categories of fugitive dust emissions were also decreased, the net effect of those changes on the RH air quality analysis is unclear.

Fractional aerosol coefficients are used to estimate the percentage of VOCs that may react in the atmosphere and form secondary organic aerosols. There is considerable uncertainty associated with this estimation approach. This assessment assumes that 100 percent of all photochemically-reactive VOC species released eventually react to form SOA. This assumption may lead to overstated modeled SOA concentrations in areas close to the emission sources of organic species having long reaction times (Pechan, 1997a).

For the nonroad emissions category, the extrapolation of the nonroad inventory for 27 $PM_{2.5}$ nonattainment areas to the rest of the country introduces uncertainty to the nonroad emissions estimates, however, with no known bias.

Because the 1985 National Acid Precipitation Activity Project (NAPAP) inventory serves as the basis for the 1990 base year inventory for some source categories, a number of factors are not accounted for. New plant construction, control equipment installation and retirement of emissions sources between 1985 and 1990 are not incorporated in the 1990 inventory. The magnitude of the uncertainty and direction of potential bias in national 1990 emission estimates as a result of these factors is unclear. Additionally, State-level industry earnings data are used to

grow emissions from 1985 to 1990 rather than applying the more recent Bureau of Economic Analysis (BEA) Gross State Product (GSP) estimates. This may result in a small underestimate of 1990 emissions (Pechan, 1997a).²

Considering relative uncertainty across emissions of individual pollutants, SO₂ emission estimates are the most certain. The SO₂ is generated during combustion of any sulfur-containing fuel and is emitted by industrial processes that consume sulfur-containing raw materials. Apart from control efforts, sulfur emissions are directly related to the fuel sulfur content. As long as fuel usage and fuel sulfur content are measured, SO₂ emissions can be estimated within a relatively narrow range. For example, as part of the Grand Canyon Visibility Transport Commission (GCVTC) emission inventory, uncertainty estimates were developed for various major SO₂ sources (Balentine and Dickson, 1995). The uncertainty estimate calculated for SO₂ emissions from copper smelting is ± 50 percent. However, associated uncertainty for emissions estimates from diesel and gasoline vehicles are assessed at ± 150 percent. Most of this uncertainty is due to the variability in the sulfur content of the fuels.

The NO_x estimates are the next most certain category of emissions. Like SO₂, NO_x is a product of fuel combustion. Since NO_x formation is somewhat more complicated than SO₂, emission estimates are more variable, and uncertain, as well.

The level of uncertainty in PM₁₀ emission estimates varies widely by source category. The largest component of the 1990 PM₁₀ emission estimates is fugitive dust including fugitive emissions from paved and unpaved roads, construction activities, agricultural tilling, and windblown dust. The GCVTC study estimated the uncertainty for unpaved road emissions to be ± 400 percent. The estimated uncertainty for PM_{2.5} emissions from paved road dust is ± 180 percent (Ballentine and Dickson, 1995). The PM₁₀ emission estimates for large point sources, such as utility boilers, are more certain than the fugitive dust source estimates, because these stacks are typically controlled using baghouses or electrostatic precipitators, the outlets of which are frequently tested to ensure compliance with regulations.

The VOC emissions are uncertain because organics are emitted both as a product of fuel combustion and through evaporation. Evaporative emissions are difficult to quantify due to measurement problems. The GCVTC study estimated VOC emissions uncertainty for motor vehicles to be ± 150 percent (Ballentine and Dickson, 1995).

Table 4-3 summarizes the key uncertainties associated with estimation of 1990 emissions (Pechan, 1997a). For each potential source of uncertainty in the base year emissions, the direction of bias is provided. “Positive bias” indicates that 1990 emissions may be overestimated; “negative bias” indicates that they may be underestimated; and “bias unclear” indicates that the direction of potential bias in the emission estimates is unknown.

² The State-level industry earnings data provided a slight underestimate of production activity for 1990 compared to the 1990 BEA GSP estimates due to a more precise methodology for estimating production activity.

**Table 4-3
Uncertainties and Possible Biases in Estimating 1990 Emissions**

Potential Source of Uncertainty	Positive Bias? (Overestimate)	Negative Bias? (Underestimate)	Bias Unclear
Use of emission factors rather than stack test, load-curve, or CEM data			✓
Use of particle-size multipliers to estimate PM ₁₀ and PM _{2.5} emissions from TSP emissions			✓
Extrapolation of nonroad inventory from 27 PM _{2.5} nonattainment areas to nation			✓
Use BEIS rather than more recent BEIS2 for biogenic VOC		✓ (total biogenic VOC and SOA)	✓ (county-level biogenic VOC and SOA)
Use NPI version 3 for fugitive dust emissions rather than more recent data from National Emissions Trends	✓		
Use FACs to estimate SOA from VOC emissions	✓		
Use of 1985 NAPAP inventory for some source categories: - lack data to incorporate for 1985-1990 new plant construction, control equipment installation, retirement of sources. - used state-level earnings data rather than recent BEA GSP to grow emissions from 1985 to 1990.		✓ (small)	✓

4.3.4 1990 Emissions Inventory Results and Discussion

Table 4-4 presents a summary of 1990 baseline emissions by pollutant and major sector. These emissions estimates reflect the partial attainment of the 8-hour Ozone and PM_{2.5} standards modeled in the 1997 RIA. Area sources are the largest contributor to anthropogenic VOC emissions in 1990 (45 percent of total national anthropogenic VOC emissions). Biogenic and natural sources of VOC emissions are estimated to be roughly equivalent in magnitude to the anthropogenic total. Motor vehicles account for 33 percent of total national NO_x emissions with

46 percent of the motor vehicle emissions contributed by cars (i.e., light-duty gasoline vehicles). With regard to national SO₂ emissions, the utility sector is the largest emitter (71 percent). Area sources account for the bulk of PM₁₀ and PM_{2.5} emissions. Anthropogenic fugitive dust sources contribute the majority of primary PM₁₀ and PM_{2.5} emissions. More recent emission inventory efforts indicate that these estimates are overestimated. Refer to Section 4.3.3 for a discussion of the potential biases in these estimates.

Although biogenic and anthropogenic VOC are approximately equivalent, biogenic SOA is almost 17 times greater than anthropogenic SOA. This difference is due to the FACs used to estimate SOA. The FAC for terpenes, which account for 15 - 60 percent of biogenic VOCs, is 30 percent, while the average FAC for anthropogenic VOC sources is less than 1 percent.

Anthropogenic ammonia emissions are estimated to be approximately 4 million tons per year in 1990, but are believed to be a small component relative to natural sources of ammonia. Given that ammonia is not a limiting factor in the formation of secondary particles, ammonia emissions are not considered for control in these RH analyses.

It should be noted, as is noted earlier in Section 4.3, that the ambient air quality impacts of emissions from any individual sector may not be proportional to their contribution to national emissions. The reader is referred to the air quality modeling sections later in this chapter and the cost chapter (Chapter 6) to understand how emissions from various source categories impact air quality modeling and estimated visibility levels.

**Table 4-4
Summary of National 1990 Baseline Emissions Estimates by Major Sector**

Major Sector	VOC (1000 tpy)	NO_x (1000 tpy)	SO₂ (1000 tpy)	PM₁₀ (1000 tpy)	PM_{2.5} (1000 tpy)	SOA (1000 tpy)	OC (1000 tpy)	EC (1000 tpy)
Utility	53	3,548	5,235	246	108	1	2	5
Industrial Point Area	2,158	1,735	4,668	1,004	651	26	18	4
Nonroad	7,046	2,872	1,518	42,601	9,061	57	374	58
Motor Vehicle	1,888	2,061	237	351	373	24	37	58
	3,688	5,331	408	204	142	26	26	27
Anthropogenic Subtotal	14,833	15,547	12,066	44,406	10,335	134	457	152
Biogenics	25,988					3,325		
Natural Sources	248	89	1	5,429	995			
TOTAL	41,609	15,636	12,061	49,835	11,330	3,459	457	152

Note: Emissions estimates may not sum due to rounding.
 Fugitive dust emissions have been adjusted here as described in Section 4.4.
 Air quality impacts from major emitting sectors are not necessarily proportional to their contribution to national emissions estimates. See Sections 4.4 and 4.5 and Chapter 6 of this RIA.

4.3.5 Development of 2015 Analytical Emission Projections

The 1990 emissions are projected to 2010 (as a proxy for emissions in 2015) to develop the emissions baseline from which to evaluate additional control measures needed to meet the illustrative RH progress goals. In general, emissions are projected by applying expected increases in 1990 emissions or activity levels and incorporating the effects of 2010 CAA-mandated controls through application of control efficiencies or emission factors, respectively.

4.3.6 Growth Assumptions by Major Sector

This section describes the sector-specific growth assumptions used to project emissions to 2010, which serves as a proxy for emissions in 2015. Table 4-5 summarizes the emissions projection modeling approach by major sector. Version 3 of the NPI employs 1995 BEA GSP 2010 projections by State/Industry for industrial point sources and, in combination with BEA population projections, for nonroad and area source categories. In the absence of product output projections, value added projections such as GSP are superior than earnings or employment projections for estimating future emissions (U.S. EPA, 1991a). Value added is the difference between the value of industry outputs and inputs. The BEA GSP projections are a fuller measure of growth given that future changes in production processes, efficiency, and technological changes are captured.

For the utility sector, outputs from the Integrated Planning Model (IPM) are used to predict how the electric power industry will operate in the future given deregulation (i.e., movement from cost-of-service pricing to competitive pricing) and consequent industry restructuring (U.S. EPA, 1996j). National Electric Reliability Council (NERC) forecasts of regional electricity demand are used to reflect the assumption that utility deregulation will likely lead to lower electricity prices for many users and therefore increased electricity demand. Additional major assumptions included in the utility modeling are the following: 1) technology will continue to improve for coal and natural gas production so that energy prices for these fuels will not substantially increase between 1990 and 2010; 2) the large steam electric generation stock fueled by coal, oil, and gas will be the source of a large amount of power in the future; 3) improvement of the performance and reduction of the costs of electric generation technologies will continue; and 4) movement of power will be primarily constrained at the 16 NERC regions modeled in the analysis (U.S. EPA, 1997a).

**Table 4-5
2015 Analytical Growth Assumptions by Major Sector^a**

Sector	Growth Forecast	Modeling Approach
Industrial Point	BEA Gross State Product (GSP) Projections by State/Industry (BEA, 1995)	VOC, NO _x - Emission Reduction and Cost Analysis Model (ERCAM): applies BEA growth projections to base year emissions and applies future year controls as selected by the user (Pechan, 1994, 1996b). PM ₁₀ , PM _{2.5} , SO ₂ , NH ₃ - While no formal model exists, the same basic approach applied in ERCAM was used for these pollutants (Pechan, 1997a).
Utility	Projections of heat input by unit based on National Electric Reliability Council (NERC) data, price and demand forecasts, and technology assumptions.	SO ₂ , NO _x - Integrated Planning Model (IPM) (U.S. EPA, 1996i). VOC, PM ₁₀ , PM _{2.5} - base year emission rates or AP-42 emission factors applied to IPM projected heat input by unit (Pechan, 1997a). NH ₃ - NH ₃ slippage for units controlling with selective catalytic reduction (SCR) (Pechan, 1997a).
Nonroad	BEA GSP and Population Projections by State/Industry (BEA, 1995)	VOC, NO _x - ERCAM (Pechan, 1994, 1996b). PM ₁₀ , PM _{2.5} , SO ₂ , NH ₃ - ERCAM approach (no formal model)(Pechan, 1997a).
Motor Vehicle	National Vehicle Miles Traveled (VMT) Projections from the EPA OMS MOBILE Fuel Consumption Model (FCM) Scaled to Metropolitan/Rest-of-State Areas by Population (U.S. EPA, 1993)	NO _x , VOC - ERCAM: applies MOBILE5a emission factors to projected VMT by month and county/vehicle type/roadway classification (U.S. EPA, 1991c, 1993a). PM ₁₀ , PM _{2.5} , SO ₂ - PART5 emission factors(U.S. EPA, 1994c) applied to projected VMT (U.S. EPA, 1991c). NH ₃ - special study emission factors applied to projected VMT (Pechan, 1997a).
Area	BEA GSP and Population Projections by State/Industry (BEA, 1995)	VOC, NO _x - ERCAM (Pechan, 1994, 1996b). PM ₁₀ , PM _{2.5} , SO ₂ , NH ₃ - ERCAM approach (no formal model)(Pechan, 1997a).
Biogenic VOC and PM Wind Erosion	Emissions held at 1990 levels	--

^a Actual growth in emissions is to 2010. 2015 is the nominal “snapshot” year that reflects the partial attainment costs for the Ozone and PM_{2.5} NAAQS included in the baseline, and is near the end of the period for the first long-term strategy.

Mobile source 1990 emissions are projected to 2010 based on growth in VMT. The EPA’s MOBILE4.1 Fuel Consumption Model (FCM) is used as the basis for the VMT projections (U.S. EPA, 1991c).

There is no growth assumed in nationwide biogenic emissions of VOC or SOA. Similarly, 2010 PM emissions from natural

sources are assumed equal to 1990 levels.

4.3.7 2010 CAA Control Emissions by Major Sector

In order to capture the effects in 2015 (using 2010 emissions projections) of implementation of the CAA as a benchmark for these analyses, future year control efficiencies or emission factors are applied to projected 2010 emissions or activity levels respectively. Table 4-6 summarizes the major CAA requirements that are modeled for the benchmark case. These control requirements are discussed in Appendix A for each major sector.

For the 2010 CAA-control emissions, refined control measure effectiveness (CME) estimates are employed in combination with control efficiencies. The CME reflects the degree to which individual control measures achieve their intended effect. For this assessment, CME is assumed to be 95 percent for this subset. The refined CME estimate is based upon a recent study of historical EPA monitoring and enforcement data that indicate that, on average, control measures achieve 95 - 100 percent of the intended impact (PQA, 1997). The new CME is applied to the appropriate CMEs in place prior to 1990 and those controls assumed in the 2010 CAA-control emissions projections.

Rate of Progress (ROP) and Reasonable Further Progress (RFP) requirements are not modeled for the emissions benchmark; instead, the emission reductions and costs are assessed for future attainment of the 8-hour ozone standard and are therefore in the baseline. Appendix A discusses the methodology and results of this analysis.

Additionally, updated information regarding proposed Title I Architectural Coatings and Consumer and Commercial Products rules and Title III 7 and 10-year Maximum Achievable Control Technology (MACT) rules are incorporated in the 2010 CAA-control emissions.

Ozone air quality modeling analyses show that NO_x emissions must be substantially reduced in broad areas of the country in order for areas that are not meeting the current ozone standard to meet that standard (U.S. EPA, 1996b). Efforts to address long-range ozone transport issues have been undertaken by the Northeast Ozone Transport Commission (OTC, 1994) and the Ozone Transport Assessment Group (OTAG). These efforts will likely result in implementation of regional NO_x control measures far in advance of the 2015 air quality assessment undertaken for this RIA. These control measures are included in the benchmark case for this RIA.

**Table 4-6
CAA 2015 Projection Scenario Summary by Major Sector**

Major Sector	Major CAA Scenario Requirements
Industrial Point	<p>VOC and NO_x RACT for all NAAs (except NO_x waivers). New control technique guidelines (CTGs). 0.15 pounds per million British thermal unit (lb/MMBtu) Ozone Transport Assessment Group (OTAG)-wide NO_x cap on fuel combustors ≥ 250 MW. OTAG Level 2 NO_x controls across OTAG States. MACT standards (primarily VOC).</p>
Utility	<p>Title IV Phase I and Phase II limits for all boiler types. 250 ton Prevention of Significant Deterioration (PSD) and New Source Performance Standards (NSPS). RACT and New Source Review (NSR) for all non-waived (NO_x waiver) NAAs. Phase II of the Ozone Transport Commission (OTC) NO_x memorandum of understanding (MOU). 0.15 lb/MMBtu OTAG-wide seasonal NO_x cap utility boilers with banking/trading.</p>
Nonroad	<p>Federal Phase I and II compression ignition (CI) engine standards. Federal Phase I and II spark ignition (SI) engine standards. Federal locomotive standards. Federal commercial marine vessel standards. Federal recreational marine vessel standards.</p>
Motor Vehicles	<p>Tier 1 tailpipe standards. 49-State LEV program. Phase 2 Reid vapor pressure (RVP) limits. I/M programs for O₃ and carbon monoxide (CO) NAAs. Federal reformulated gasoline for O₃ NAAs. California LEV (California only). California reformulated gasoline (California only). Diesel fuel sulfur content limits. Oxygenated fuel in CO NAAs.</p>
Area	<p>VOC and NO_x RACT requirements. New CTGs (VOC). MACT Standards (VOC). PM NAA controls. Onboard vapor recovery (vehicle refueling). Stage II vapor recovery systems. Federal rules (consumer/commercial product limits, architectural and industrial maintenance (AIM) coating limits).</p>

The 2010 benchmark reflects the application of regional NO_x reductions that are intended to approximate the reductions EPA would propose based upon OTAG recommendations. The regional NO_x controls applied for this analysis include: 1) OTAG-wide 0.15 lb/MMBtu NO_x emission limit on utilities and on non-utility boilers \geq 250 MW; 2) OTAG Level 2 NO_x controls on non-utility point sources across OTAG States; National Low Emission Vehicle (LEV) emissions standards on light duty vehicles in 49 States, beginning with the 1999 model year. The OTAG recommendation covers a broader universe of sources and provides for an emissions trading program. In addition, OTAG's recommendation does not include uniform control measures across the entire 37-State region. For purposes of comparison, the final NO_x SIP call rulemaking promulgated in September 1998 requires States to implement sufficient levels of control to achieve a 0.15 lb/MMBtu NO_x emission limit applied to utility boilers; 60 percent control applied to non-utility boilers and combustion turbines; and additional controls applied to cement kilns and stationary internal combustion engines.³

The LEV program is included in the baseline based on negotiations with the automobile industry that were initiated several years ago in order to help meet the current standard. Although no agreement has yet been reached, additional reductions from mobile sources likely will be required, either nationally or on a State-by-State basis, in order to meet the current standard. Therefore, inclusion of reductions from this program in the baseline is appropriate. This analysis, however, does not prejudge the outcome of negotiations with the automobile industry.

A version of the Tier II rule scheduled to be proposed this year is included in the baseline. This version reflects prior expectations as to type of standards that would be included in the proposal. The version in the baseline includes standards applicable to light-duty trucks and other light-duty vehicles, but does not include a sulfur standard applicable to refiners nor additional control of hydrocarbon exhaust and evaporative emissions. Since this rule is currently close to proposal, inclusion of reductions from a Tier II program in the baseline where analytically possible is appropriate. This analysis, however, does not prejudge the eventual form of the Tier II rule.

4.3.8 Benchmark Emissions Results and Discussion

Table 4-7 summarizes national 2010 CAA emissions by major sector. Total emissions of VOC, NO_x, SO₂, and SOA are estimated to decrease from 1990 levels; however, emissions of PM₁₀ and PM_{2.5} are estimated to increase between 1990 and 2010. The increases in PM emissions are due primarily to growth in anthropogenic sources of fugitive dust (i.e., paved roads and construction activity).

Emission reductions in 2010 attributable to individual CAA programs are also estimated

² The estimated level of emissions reductions from implementation of the final NO_x SIP call is 1.16 million tons in 2007. This is roughly two-thirds of the reductions estimated under the NO_x control programs presently in the benchmark case.

(U.S. EPA, 1997j). These emission reductions reflect the change in emissions between projected 2010 emissions (i.e., incorporating growth between 1990 and 2010) with and without the application of CAA-mandated controls. National VOC emission reductions estimated to be achieved in 2010 due to Titles I and III point source controls are 1.0 million tons of VOC per year. The 2010 Title I and III area source controls are projected to achieve 5.7 million tons of VOC emission reductions per year.

National NO_x emission reductions for Title I industrial point source controls are estimated to total 1.6 million tons per year: the CAA-mandated controls and the NO_x cap account for approximately 500,000 tons and 100,000 tons of NO_x reductions respectively and OTAG-wide Level 2 NO_x controls contribute an additional 1 million tons per year of NO_x reductions (U.S. EPA, 1997j). Title I area source NO_x controls account for reductions of 1.4 million tons of NO_x per year. Title I mandated controls, Title IV Acid Rain NO_x requirements, and the OTAG-wide NO_x cap result in an estimated 3 million tons of summertime NO_x reductions from the utility sector (U.S. EPA, 1997a).

Title II mobile source VOC and NO_x controls including a national LEV program are estimated to result in annual reductions of 2.8 million tons of VOC and 3.5 million tons of NO_x nationally in 2010 (U.S. EPA, 1997j).

The Title IV Acid Rain Program accounts for an 8 million ton reduction in utility SO₂ emissions from 2010 no-control levels (U.S. EPA, 1997a).

4.3.9 Key Uncertainties Associated with Benchmark Emissions

Table 4-9 summarizes the key uncertainties associated with the 2010 benchmark emissions. Because 1990 emissions and activity levels are the basis from which 2010 emissions are projected, the uncertainties associated with 1990 emissions estimates are carried through to the 2010 emission estimates. These uncertainties are discussed in Section 4.3.4.

There are uncertainties associated with the activity surrogates and projections data used to make 2010 growth forecasts for each source sector. However, there are no known biases in either of these data inputs.

Table 4-7
Summary of National 2010 CAA Emissions Estimates by Major Sector

Major Sector	VOC (1000 tpy)	NOx (1000 tpy)	SO₂ (1000 tpy)	PM₁₀ (1000 tpy)	(1000 tpy)
Utility	50	3,755	9,746	277	
Industrial Point	2,164	1,958	5,990	1,170	
Area	7,533	2,932	1,518	41,051	
Nonroad	1,888	2,063	236	336	
Motor Vehicle	3,946	5,574	409	204	
Anthropogenic Subtotal	15,581	16,282	17,899	43,038	
Biogenics	25,988				
Natural Sources	248	89	1	5,429	
TOTAL	41,817	16,371	17,900	48,467	

Note: Emissions estimates may not sum due to rounding.
1990 fugitive dust emissions have not been adjusted.
Air quality impacts from major emitting sectors are not necessarily proportional to their contribution to national emissions estimates. See sections 4.4 and 4.5 and Chapter 6 of this RIA.
Organic carbon and elemental carbon emissions were not estimated for 2010 for the CAA baseline scenario.

The 2010 control assumptions used to incorporate the effects of CAA-mandated controls also have related uncertainties. Potential revisions to existing rules or rules that are currently in draft form but would be implemented in 2010 are not incorporated in the 2010 emissions baseline. It is unclear the net effect of these omissions on baseline emissions. Because RFP and ROP are not incorporated in the baseline, 2010 emissions could be underestimated. There may be an overestimate in baseline emissions given that the co-control emission reductions (e.g., PM, NOx) from MACT standards and off-set requirements in the Ozone Transport Region (OTR) and ozone nonattainment areas have not been estimated. Finally, because the NPI is a top-down inventory, area-specific control measures as outlined in nonattainment State implementation plans (SIPs) have not been incorporated in the baseline emissions. The potential bias is unclear for this potential source of uncertainty.

Table 4-8
Uncertainties and Possible Biases in Estimating 2010 Emissions^a

Potential Source of Uncertainty	Positive Bias? (Overestimate)	Negative Bias? (Underestimate)	Bias Unclear
1990 Emissions	✓ (fugitive dust)	✓ (total biogenic VOC and SOA)	✓
Growth Forecasts: - activity surrogates - projections data			✓ ✓

Potential Source of Uncertainty	Positive Bias? (Overestimate)	Negative Bias? (Underestimate)	Bias Unclear
<p>2010 Control Assumptions:</p> <ul style="list-style-type: none"> - Potential revisions to existing rules or rules in draft form not incorporated; - RFP/ROP for individual ozone nonattainment areas not estimated; - Co-control from MACT standards not estimated; - Off-set requirements in OTR and ozone nonattainment areas not estimated; - Area-specific reductions as reflected in SIPs not incorporated. 	<p style="text-align: center;">✓</p> <p style="text-align: center;">✓</p>	<p style="text-align: center;">✓</p>	<p style="text-align: center;">✓</p> <p style="text-align: center;">✓</p>

^a The projection of emissions to 2010 is a proxy for emissions in 2015, the analysis year in this RIA.

4.4 Estimation of Benchmark Visibility Levels

The methodology for estimation of benchmark visibility levels for this assessment builds upon the previous method used in the 1997 RIA. The CRDM is used to estimate ambient PM concentrations in 2010. This model predicts quantitative relationships (i.e., source-receptor relationships) between county-level emissions of primary particles and secondary particle precursors and annual concentrations of PM_{10} and $PM_{2.5}$ at county-level receptors. The following data inputs are implemented for this assessment:

- ! Phase II CRDM air quality modeling results are employed;
- ! The source-receptor (S-R) matrix is calibrated using 1993 -1995 Aerometric Information Retrieval System (AIRS) monitoring data for all 711 counties monitored for PM_{10} in the 48 contiguous States during this 3-year period;

The following refinements are employed in this analysis that are not employed in the modeling for the proposed RH rule in 1997:

- ! Contribution to visibility improvements from reduction of OC and EC emissions is not accounted for;
- ! Estimates of visibility improvements in non-Class I area counties are now estimated and serve as an input to the benefits analyses in Chapter 9.

4.4.1 Overview of Phase II Air Quality Modeling

This section provides a general overview of the Phase II air quality modeling analysis. More detailed information follows in subsequent sections. For Phase II, the Lagrangian Regional Model is used to guide the refinement of the Climatological Regional Dispersion Model (CRDM) to correct for misestimation of fugitive dust emissions (Latimer, 1996). Using 1990 meteorology, the refined CRDM is applied to 1990 emissions to calculate a transfer matrix of S-R relationships for all relevant primary and precursor emissions to estimate cumulative regional ambient concentrations of $PM_{2.5}$ and PM_{10} , as well as the important chemical constituents of secondary particulates: sulfate, nitrate, secondary organics and ammonium. As described in section 4.4.2, the refined CRDM, when used with adjusted primary PM fugitive dust emissions, provides more representative estimates of the spatial distribution of annual PM concentrations in the United States (Pechan, 1997b).

The S-R matrix next is calibrated using 1993-1995 PM_{10} and $PM_{2.5}$ annual monitoring data to benchmark the modeling to ambient air quality values. Additionally, this calibration provides a way to capture the 3-year and spatial averaging aspects of the $PM_{2.5}$ annual standard alternatives.

In order to predict ambient PM concentrations in 2010, emissions projections as described in Section 4.3 are input to the calibrated S-R matrix to produce annual PM_{10} and $PM_{2.5}$ concentration values at county-level receptors. Finally, 1993-1995 peak-to-mean ratios (i.e., ratio of 24-hour value to annual average value) for each monitored county in the analysis are used to estimate the 24-hour PM concentration (i.e., 4th highest daily maximum for the current PM_{10} daily form and 98th percentile value for the $PM_{2.5}$ daily form alternatives) from the model-predicted annual PM concentration. Nonmonitored counties are calibrated using regional average normalization factors. Additionally, regional peak-to-mean ratios are used to derive the 24-hour PM concentration in the nonmonitored counties.

Once 2010 baseline air quality is developed, monitored counties in class I areas are evaluated to determine if they can meet the illustrative RH progress goals. Figure 4-3 illustrates the development of 2010 baseline visibility levels.

4.4.2 Elements of Visibility Modeling

4.4.2.1 Lagrangian Regional Model

The Lagrangian Regional Model (LRM) is used to guide the refinement of the CRDM through the estimation of the transport, diffusion, deposition, and chemical conversion of emissions using a spatially and temporally varying wind field. Because the computer memory and run times are excessive to run the LRM for the entire country with 6,000 sources and 3,000 receptors, the LRM was tested for a single point source for a few days of 1990 meteorological data from the Meteorological Model-4 (MM-4) mesoscale model. The LRM simulates the hourly release of puffs which are transported by the averaged winds appropriate for the time and location of the puff. In general, puff-type air quality models are better than Gaussian dispersion models at handling transport and diffusion of pollutants at low wind speeds and therefore show a greater air quality impact from emissions in the local area. A single uniform concentration of each particulate chemical constituent for each hourly puff is calculated based on standard vertical diffusion coefficients, limited by the mixed layer height, and mesoscale diffusion coefficients. Results from the LRM are subsequently used to refine CRDM assumptions to take into account long-range transport of secondary particles and impacts of a county's primary emissions on its air quality (Latimer, 1996).

4.4.2.2 Climatological Regional Dispersion Model

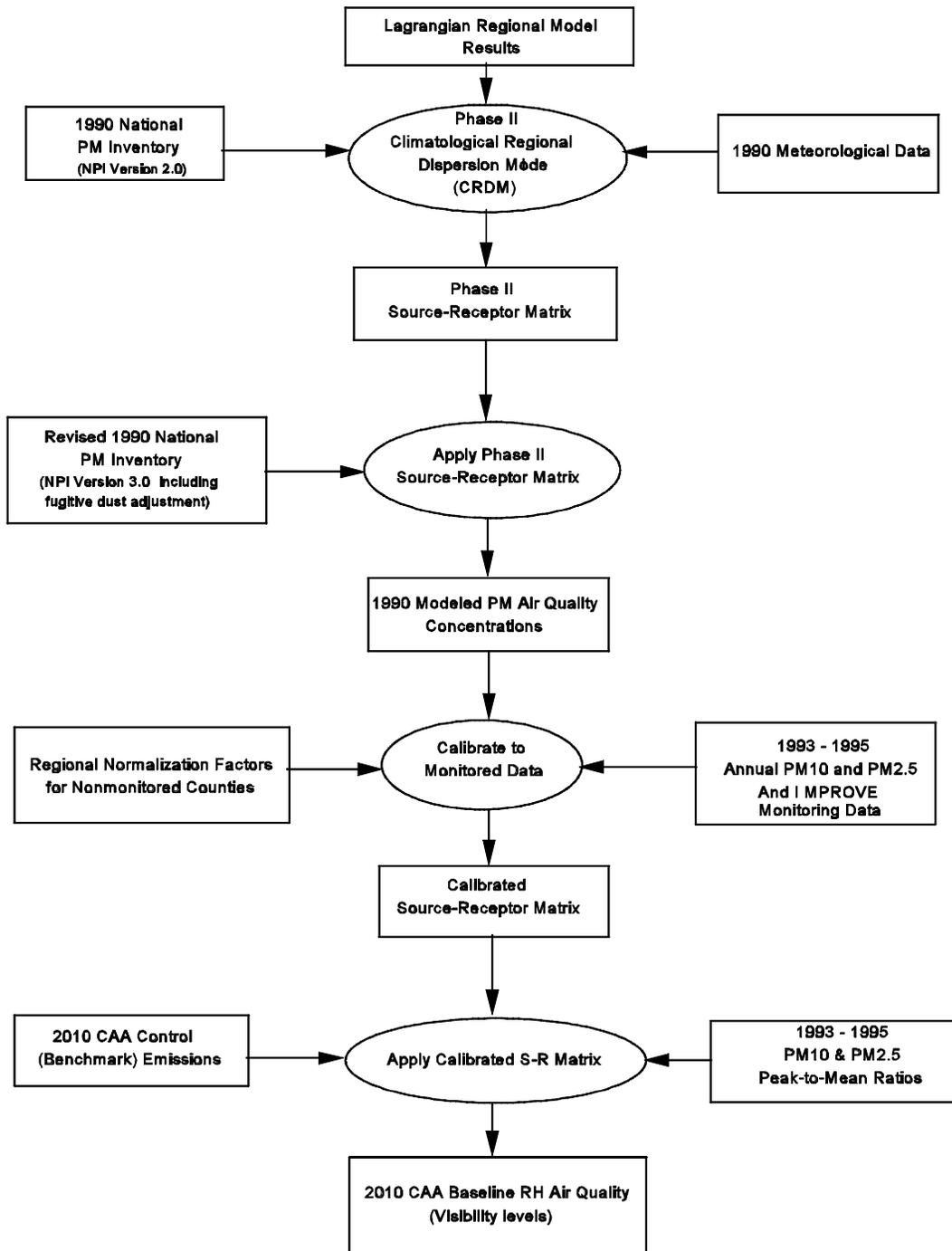
The CRDM is used to generate a matrix of S-R relationships that relate emissions of direct PM_{10} and $PM_{2.5}$ and particle precursors to annual average PM_{10} and $PM_{2.5}$ concentrations (Pechan, 1997b). The S-R matrix reflects the relationship between PM concentration values at a single receptor in each county (a hypothetical design value monitor sited at the county population centroid) and the contribution by PM species to this concentration from each emission source.

The CRDM uses assumptions similar to the Industrial Source Complex Short Term (ISCST3), an EPA-recommended short range Gaussian dispersion model (U.S. EPA, 1995b). The CRDM incorporates terms for wet and dry deposition and chemical conversion of SO₂ and NO_x, and uses climatological summaries (annual average mixing heights and joint frequency distributions of wind speed and direction) from 100 upper air meteorological sites throughout North America. For this analysis, meteorological data for 1990 are used.

The CRDM uses Turner's sector-average approach, a probabilistic method in which the frequencies of occurrence of various wind and stability conditions are used to calculate the frequencies of transport of pollutants in various sectors. This method is recommended for estimation of long-term average pollutant concentrations and is discussed more fully in a contractor report (Pechan, 1997b). The assumptions related to chemical conversion of secondary particle precursors, long-range transport of secondary particles and the impact of a county's primary emissions on itself are refined based upon the LRM results. For the Phase II modeling, chemical conversion, transport and deposition equations are updated. Additionally, it was assumed that all primary emissions from the county are evenly distributed over a square with the same area as the county. It is also assumed that primary emissions from the county are always impacting the county. A simple box model is used for each wind speed and stability category. The vertical diffusion coefficient is calculated at a downwind distance corresponding to the length of the side of the square. These assumptions are necessary since spatial variation of emissions within a county cannot be provided for a national scale model.

Emissions data from version 3.0 of the 1990 NPI are input to the CRDM. Stationary and mobile source emissions, as well as ground-level area source emissions, for 3,081 counties in the contiguous United States are contained in the 1990 NPI. The high number of point sources in the inventory (61,619 point sources) made it impractical to model each point source individually. As a result, elevated point source emissions are aggregated at the county level by plume height. The effective stack height of each of these sources was calculated for an average wind speed (5 meters/second) using the plume rise algorithm for ISCST3. Two aggregated elevated point source groupings are made: one for sources with effective stack heights less than 250 m, and one for sources with effective stack heights between 250 and 500 m. Sources with effective stack heights greater than 500 m are modeled as separate sources. In addition to point sources, the modeled emission sources also include total area/mobile sources for each county and emissions for 10 Canadian provinces and 29 Mexican cities/States. Receptors modeled include all county centroids plus receptors in Canada and Mexico.

**Figure 4-3
Development of 2015 Baseline RH Air Quality**



A total of 5,944 sources (i.e., industrial point, utility, area, nonroad, and motor vehicle) of primary and precursor emissions are modeled. In addition, secondary organic aerosols formed from anthropogenic and biogenic VOC emissions are modeled. Natural sources of PM₁₀ and PM_{2.5} (i.e., wind erosion and wild fires) are also included. Emissions of SO₂, NO_x, and ammonia are modeled in order to calculate ammonium sulfate and ammonium nitrate concentrations, the primary particulate forms of sulfate and nitrate. The CRDM produces an S-R matrix of transfer coefficients for each of these primary and particulate precursor pollutants. These coefficients can be applied to the emissions of any unit (area source or individual point source) to calculate a particular source's contribution to a county receptor's total annual PM₁₀ or PM_{2.5} concentration. Each individual unit in the inventory is associated with one of the source types (i.e., area, point sources with effective stack height of 0 to 250 m, 250 m to 500 m, and individual point sources with effective stack height above 500 m) for each county.

Once the S-R matrix is developed, the transfer coefficients must be adjusted to reflect concentrations of secondarily-formed particulates (Latimer, 1996). First, the transfer coefficients for SO₂, NO_x, and ammonia are multiplied by the ratios of the molecular weights of sulfate/SO₂, nitrate/nitrogen dioxide and ammonium/ammonia to obtain concentrations of sulfate, nitrate and ammonium.¹ The relative concentrations in the atmosphere of ammonium sulfate and ammonium nitrate depend on complex chemical reactions. In the presence of sulfate and nitric acid (the gas phase oxidation product of NO_x), ammonia reacts preferentially with sulfate to form particulate ammonium sulfate rather than react with nitric acid to form particulate ammonium nitrate. Under conditions of excess ammonium and low temperatures, ammonium nitrate forms. For each county receptor, the sulfate-nitrate-ammonium equilibrium is estimated based on the following simplifying assumptions:

- All sulfate is neutralized by ammonium;
- Ammonium nitrate forms only when there is excess ammonium;
- Because ammonium nitrate forms only under low temperatures, annual average particle nitrate concentrations are divided by four assuming that sufficiently low temperatures are present only one-quarter of the year.

Finally, the total particle mass of ammonium sulfate and ammonium nitrate is calculated.²

¹ Ratio of molecular weights: Sulfate/SO₂= 1.5; nitrate/nitrogen dioxide = 1.35; ammonium/ammonia = 1.06.

² To calculate total particle mass of ammonium sulfate and ammonium nitrate, the anion concentrations of sulfate and nitrate are multiplied by 1.375 and 1.29 respectively.

4.4.2.3 *Comparison of Modeled and Measured PM Concentrations*

In order to evaluate the performance of the Phase II CRDM, model-predicted PM concentrations and measured ambient PM concentrations are compared. Measured annual average PM concentrations by chemical species from the Interagency Monitoring for Protection of Visual Environments (IMPROVE) network are examined for the 3-year period March 1988 - February 1991. This period is chosen because it relates closely to 1990 emissions and meteorological data used in the CRDM. Given that IMPROVE network monitors visibility impairment in predominantly rural Class I areas, these comparisons are incomplete due to the lack of coverage in urban areas. With the exception of the fugitive dust component of PM_{2.5} and PM₁₀, modeled and measured concentrations of sulfate, nitrate and organics are comparable (Latimer, 1996).

This PM air quality modeling effort attempts to model the “background” contribution to ambient PM concentrations. Background PM is defined as the distribution of PM concentrations that would be observed in the U.S. in the absence of anthropogenic emissions of PM and precursor emissions of VOC, NO_x and SO_x in North America (U.S. EPA, 1996). Estimating background PM concentrations is important for the cost analysis as it represents that portion of PM mass that is uncontrollable. Background PM levels vary by geographic location and season. The natural component of background arises from physical processes of the atmosphere that entrain small particles of crustal material (i.e., soil from wind erosion), as well as emissions of organic particles and nitrate precursors resulting from natural combustion sources, such as wildfire. In addition, certain vegetation can emit SOA. Biogenic sources and volcanos also emit sulfate precursors. The exact magnitude of this natural portion of PM for a given geographic location cannot be precisely determined because it is difficult to distinguish from the long-range transport of anthropogenic particles and precursors. The PM Criteria Document (U.S. EPA, 1996a) reports that annual average PM_{2.5} concentrations range from 1 - 4 ug/m³ in the West and from 2 - 5 ug/m³ in the East.

Given the uncertainties in estimating biogenic VOC and SOA emissions and primary PM emissions from natural sources as well as the uncertainties in the PM air quality model, there is considerable uncertainty in the modeled predictions of the background contribution to PM mass. For some nonattainment counties, apparent overpredictions in the background contribution to PM mass reduces the relative contribution of anthropogenic sources to PM mass. This in turn can significantly diminish the modeled effectiveness of control measures on anthropogenic sources in reducing estimated PM concentration levels.

Although the bulk of primary PM emissions are from anthropogenic and natural fugitive dust sources¹, available speciated monitoring data indicate that fugitive dust contributes substantially less to total PM_{2.5} levels relative to other particle species such as sulfates and

¹ Natural and anthropogenic fugitive dust emissions account for 93 percent of PM₁₀ emissions and 76 percent of PM_{2.5} emissions in the 1990 base year inventory (NPI version 3).

nitrates. The CRDM-predicted average fugitive dust contribution to $PM_{2.5}$ mass is 31 percent in the East and 32 percent in the West (Pechan, 1997b). Speciated monitoring data show that minerals (i.e., crustal material) comprise approximately 5 percent of $PM_{2.5}$ mass in the East and approximately 15 percent of $PM_{2.5}$ mass in the West (U.S. EPA, 1996a). The 1990 model predictions therefore are not consistent with ambient data. These disparate results may suggest a systematic overbias in the fugitive dust emission estimates. Subsequent PM emission inventory efforts indicate that fugitive dust emissions are overestimated in the baseline emissions inventory. The NPI version 3 fugitive dust PM_{10} and $PM_{2.5}$ emissions used in this analysis are 40 percent and 73 percent greater, respectively, than the most recent NET Inventory estimates¹ (U.S. EPA, 1997h). Furthermore, this overestimate in the contribution of fugitive dust to modeled ambient fine particle concentrations relative to speciated monitoring data is likely to be compounded by uncertainties in the air quality modeling (U.S. EPA, 1996c).

To address this bias, a multiplicative factor is applied nationally to fugitive dust emissions as a reasonable first-order attempt to reconcile differences between modeled predictions of $PM_{2.5}$ and actual ambient data. The multiplicative adjustment of 0.25 is applied under Case A. The 0.25 multiplicative adjustment results in a fugitive dust contribution to modeled ambient $PM_{2.5}$ concentrations of 10 - 17 percent. Given the uncertainties noted in the fugitive dust emissions inventory, however (U.S. EPA, 1998), the multiplicative factor of 0.0 is applied to nationally to these emissions under Case B.

4.4.2.4 Application of Phase II S-R Matrix to Updated 1990 National Particulate Emissions Inventory

As described in section 4.3, version 3 of the NPI is used as the base year 1990 inventory. The Phase II S-R matrix next is applied to the revised PM emissions inventory to predict 1990 PM air quality concentrations.

4.4.2.5 Normalization of S-R Matrix for Annual Estimates of PM_{10} and $PM_{2.5}$

The resulting 1990 annual PM_{10} and $PM_{2.5}$ values are compared and calibrated to monitored annual PM_{10} and $PM_{2.5}$ concentrations. All predictions are normalized regardless of over prediction or under prediction relative to monitored values. This is done by application of a “normalization factor”, calculated as the monitored value divided by the modeled value. This factor was applied consistently across particle species contributing to the air quality value at a county-level receptor. Calibration is conducted for county-level modeled PM_{10} and $PM_{2.5}$ estimates falling into one of four air quality data tiers. The tiering scheme reflects increasing relaxation of data completeness criteria and therefore increasing uncertainty for the annual design

² Natural and anthropogenic fugitive dust emissions account for 86 percent of PM_{10} emissions and 59 percent of $PM_{2.5}$ emissions in the most recent 1990 National Emission Trends Inventory.

value (U.S. EPA, 1997k). Tier 1 monitored counties cover the 504 counties with at least 50 percent data completeness and therefore have the highest level of certainty associated with the annual design value. Tier 2 monitored counties cover 100 additional counties with at least one data point (i.e., one 24-hour value) for each of the 3 years during the period 1993 -1995. Tier 3 monitored counties cover 107 additional counties with missing monitoring data for one or two of the 3 years 1993 - 1995. In total, Tiers 1, 2 and 3 cover 711 counties currently monitored for PM_{10} in the 48 contiguous States.¹ Tier 4 covers the remaining 2369 nonmonitored counties. Normalization factors are calculated and applied to the respective counties for Tiers 1 through 3. Tier 4 nonmonitored counties are calibrated using the appropriate regional normalization factor calculated as the average of Tier 1 normalization factors across a given modeling region².

The calibration procedure is conducted employing 1993-1995 PM_{10} ambient monitoring data from the AIRS database following the air quality tier data completeness parameters discussed above. The PM_{10} data represent the annual average of design value monitors averaged over 3 years (U.S. EPA, 1996i). The standardization for temperature and pressure was eliminated from this concentration data based upon proposed revisions to the reference method for PM_{10} .³

Because there is little $PM_{2.5}$ monitoring data available, a general linear model is developed to predict $PM_{2.5}$ concentrations directly from the 1993-1995 PM_{10} values (U.S. EPA, 1996e). A SASTM general linear model (i.e., GLM) procedure is used to predict $PM_{2.5}$ values (dependent variable) as a function of independent variables for season, region, and measured PM_{10} value. These derived $PM_{2.5}$ data are used to calibrate model predictions of annual average $PM_{2.5}$. Given the $PM_{2.5}$ annual standard alternatives allow for spatial averaging, model-predicted annual average $PM_{2.5}$ air quality data are calibrated to the spatially-averaged annual $PM_{2.5}$ value⁴ from the derived $PM_{2.5}$ dataset. Additionally, the proposed form of the standard allows for averaging over 3 years of air quality data. These derived, annual $PM_{2.5}$ data represent the annual average value over a 3-year period. These $PM_{2.5}$ concentrations also reflect the elimination of the temperature and pressure standardization, given that they are developed from the previously discussed PM_{10} dataset.

¹ The current PM_{10} monitoring network consists of approximately 1600 individual monitors with a coverage of approximately 711 counties in the 48 contiguous States.

² As presented in Chapter 6, the contiguous 48 States are divided into six modeling regions for the control strategy-cost analysis. See p. 6-5.

³ See Proposed Revisions to Appendix J - Reference Method for PM_{10} , Proposed Rule for National Ambient Air Quality Standards for Particulate Matter (Federal Register, Vol. 61, No. 241, p. 65666, December 13, 1996).

⁴ County-level spatial averaging is used for this analysis.

4.4.2.6 Application of Calibrated Phase II S-R Matrix to 2015 CAA Control Emissions

The calibrated Phase II S-R matrix is next applied to the 2015 CAA control emissions to predict baseline annual air quality and visibility levels at the county level. This baseline air quality reflects the fugitive dust emissions adjustment of 0.25. This is the baseline air quality used in the calculations of results under emissions control case A. Adjusting the fugitive dust emissions by 0 (i.e., zeroing them out) leads to the baseline air quality used in the calculation of results under emissions control case B.

4.4.2.7 Peak-to-mean Ratios for Calculating 24-hour Average Concentration Value

Since the CRDM predicts only annual average PM_{10} and $PM_{2.5}$ concentrations, peak-to-mean ratios are employed to derive these values. For each annual PM concentration for the Tier 1 through 3 monitored counties, three sets of peak-to-mean ratios are used to predict 24-hour peak PM_{10} and $PM_{2.5}$ concentrations reflective of the forms of the alternatives being analyzed.¹ The first peak-to-mean ratio is the 3-year average fourth highest 24-hour maximum PM_{10} value to the annual arithmetic mean PM_{10} value. This ratio is applied to the modeled annual average PM_{10} value to predict the fourth highest daily maximum PM_{10} value, the form of the current PM_{10} daily standard. The ratio of annual mean PM_{10} to 99th percentile 24-hour PM_{10} is used to predict the 3-year average 99th percentile PM_{10} value (i.e., form of the selected PM_{10} standard) from the annual mean PM_{10} . The $PM_{2.5}$ peak-to-mean ratio is calculated as the 3-year average 98th percentile 24-hour peak $PM_{2.5}$ value to the spatially averaged annual arithmetic mean $PM_{2.5}$ value. This ratio is applied to the annual mean $PM_{2.5}$ value to predict the 3-year average 98th percentile 24-hour peak $PM_{2.5}$ value (U.S. EPA, 1996e).

4.4.3 Class I Area Counties Meeting each RH Progress Goal

The model-predicted visibility levels reflecting the 2010 CAA-control baseline are used to determine county air quality status. Predicted visibility levels are the most certain for the Tier 1 counties since the estimates are calibrated using 50 percent complete AIRS data as described in Section 4.4.2.5. This set represents approximately 70 percent of the counties within the 48 contiguous States monitored for PM_{10} during 1993-1995, covering approximately 150 million people.

² Used 1993-1995 AIRS monitoring data following air quality data tiering scheme discussed in section 4.3.2.4.

4.4.4 Uncertainties in Air Quality Modeling for Visibility Improvements

The methodology used to estimate visibility improvements in 2015 from 1990 emissions and ambient concentration data introduces several sources of uncertainty to the control strategy-cost and benefits analyses. Table 4-9 presents potential sources of uncertainty and associated biases in estimating the number of 2015 counties not initially meeting the illustrative progress goals. “Positive bias” indicates that estimated number of 2015 counties not meeting the illustrative progress goals may be overestimated; “negative bias” indicates that estimated number of 2015 counties not meeting the illustrative progress goals may be underestimated; “bias unclear” indicates that the direction of impact from a given potential source of uncertainty on 2015 counties not meeting the illustrative progress goals is unknown. The level of uncertainty associated with a particular input variable to the air quality projection procedure has been quantified to the extent possible based on information from published literature or internal EPA studies.

Because 1990 emissions are an input to the CRDM model, the uncertainties associated with the emissions inventory are carried through to the air quality modeling. As discussed in section 4.3.3, apart from the fugitive dust and biogenic VOC and SOA categories, emissions of primary PM and PM precursors are uncertain although with no known bias. Fugitive dust PM emissions appear to be overestimated by 40 percent for PM₁₀ and 73 percent for PM_{2.5} relative to the more recent NET Inventory. The biogenic VOC emissions are underestimated relative to the more recent BEIS2 estimates. Finally, the methodology used to estimate SOA formation from reactive VOCs may overestimate SOA emissions and therefore ambient concentrations of SOA.

There is uncertainty associated with the 1993 - 1995 monitored annual average and 24-hour PM₁₀ concentration values that are used to calibrate the ambient concentrations generated by the CRDM at the county-level receptors. These monitoring values are taken from the AIRS data base, which has a performance requirement of 5 $\mu\text{g}/\text{m}^3$ for concentrations less than 80 $\mu\text{g}/\text{m}^3$ and ± 7 percent for concentrations greater than 80 $\mu\text{g}/\text{m}^3$. However, a comparison of AIRS data obtained from side-by-side samplers of the same and different types indicated measurement differences ranging from 10 to 14 percent for like samplers to 16 to 26 percent for dissimilar samplers (U.S. EPA, 1996k). However, there is no known bias associated with these values.

Since the PM_{2.5} data are derived from monitored PM₁₀ concentrations, they too have associated uncertainty due to instrument measurement error, as described above. Additionally, and more importantly, the PM_{2.5} values are predicted from a regression model (U.S. EPA, 1996e), and therefore are subject to uncertainty associated with this model. Subsequent reanalysis of the model has shown that there is no systematic bias to the PM_{2.5} estimates (U.S. EPA, 1997i).

The CRDM used to generate a matrix of S-R transfer coefficients employs a large number

of input variables in its calculations, including meteorological data (i.e., wind speed, wind velocity, and stability conditions). While there have been no studies of uncertainty associated with CRDM output, Freeman *et al.* (1986) used error propagation and Monte Carlo simulation to study the uncertainty of short range concentration estimates calculated by a similar model, EPA's ISCST Gaussian dispersion model for a single point source. Freeman *et al.* found that for relatively low values of uncertainty assigned to input values (1 to 10 percent), the uncertainty of the concentration at distances from 3 to 15 kilometers downwind of a source averaged 16 percent. When input data uncertainties were increased by a factor of 4, however, the output uncertainty ranged from about 75 - 160 percent.

Despite application of the fugitive dust adjustment factor, comparisons of modeled PM predictions to ambient data indicate that the CRDM overpredicts the contribution of fugitive dust to total PM_{2.5} mass and therefore to visibility impairment. The CRDM may overestimate or underestimate other fine particle species when evaluating county-level model predictions relative to PM_{2.5} ambient data. For example, in some PM residual nonattainment counties, the predicted biogenic organic contribution to PM_{2.5} mass appears to be overestimated relative to speciated monitoring data. However, at the national level, there appears to be no systematic bias to the modeled air quality predictions for the non-fugitive dust particle species.

The uncertainties and biases in the 1990 modeled predictions combined with uncertainties in 2010 emission projections bring about similar uncertainties and biases in the 2015 visibility improvement predictions. Table 4-9 lists these uncertainties and biases.

Although the CRDM S-R matrix serves as a useful tool in the design of cost-effective PM control strategies, the modeling approach does not reflect application of state-of-the-art techniques. Many of the physical and chemical formulations in the CRDM are crude representations of actual mixing and reaction phenomena required to address aerosol formation, transport and removal phenomena. Where available, more scientifically credible RADM results are used to complement the CRDM results, particularly with regard to nitrogen deposition. However, even with the anticipated delivery of more comprehensive modeling techniques, the scarcity of speciated ambient data in both urban and rural environments to evaluate model behavior will continue to compromise the certainty of model-derived conclusions.

Table 4-9
Uncertainties and Possible Biases in Estimating the Number of 2015 Counties
that Cannot Meet Illustrative RH Progress Goals

Potential Source of Uncertainty	Positive Bias? (Overestimate)	Negative Bias? (Underestimate)	Bias Unclear
<u>Base Year 1990</u> - 1990 emissions - 1993 - 1995 PM10 ambient data - 1993 - 1995 PM2.5 derived data - CRDM 1990 adjusted S-R matrix	✓ (fugitive dust, SOA) ✓ (fugitive dust)	✓ (total biogenic VOC and SOA)	✓ (other emissions) ✓ ✓ ✓ (other emissions)
<u>Projection Year 2010</u> (proxy for 2015 analysis year) - Uncertainties from 1990 adjusted S-R matrix - 2010 emissions projections - 2010 air quality predictions	✓ (fugitive dust) ✓ (fugitive dust, SOA) ✓ (fugitive dust)	✓ (total biogenic VOC and SOA)	✓ ✓ (other emissions) ✓ (other particle species)
<u>2010 Nonattainment Counties</u> - Tier 1 geographic scope assumption		✓ (small)	

It should be noted that an air quality adjustment procedure is used to account for CAA-control emissions inventory changes between 2007 and 2010. This adjustment procedure is applied to ozone nonattainment areas that are affected under the cap-and-trade program within the baseline for the final RH rule. For the most part, emissions are projected to decrease between 2007 and 2010. It is therefore reasonable to assume that air quality would improve as a result of these reductions. Because it is not possible to account for the air quality impacts of these changes outside of the nonattainment area, there may be a small overestimate in baseline air quality. Similarly, the centroid model used to predict ozone concentrations in nonmonitored counties cannot fully account for ozone transport from nonattainment areas to downwind areas. The centroid model employs geographic interpolation between ozone concentration values in

monitored counties to derive ozone concentrations in nonmonitored counties. The centroid model is not an air quality model and therefore any transport impacts from emission changes between 2007 and 2010 cannot be assessed.

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