

US EPA ARCHIVE DOCUMENT

## 4.0 BASELINE EMISSIONS AND AIR QUALITY

### 4.1 RESULTS IN BRIEF

Baseline 2010 emissions are projected from 1990 by application of sector-specific growth factors and Clean Air Act (CAA)-mandated controls to 1990 base year emissions. Total 2010 emissions of VOC, NO<sub>x</sub>, SO<sub>2</sub> and secondary organic aerosols are estimated to decrease from 1990 levels; however, emissions of primary PM<sub>10</sub> and PM<sub>2.5</sub> are estimated to increase.

Baseline particulate matter (PM) air quality concentrations in 2010 are estimated using the Phase II Climatological Regional Dispersion Model (CRDM). Initial nonattainment counties (i.e., prior to application of controls) for each PM<sub>10</sub> and PM<sub>2.5</sub> standard alternative are estimated based on these modeled air quality predictions for counties with PM monitors during 1993 - 1995. At the national level, 45 counties are estimated to be in initial nonattainment of the current PM<sub>10</sub> standard (50/150- 1 expected exceedance). Before applying the National PM Strategy, 102 counties are estimated to initially violate the selected PM<sub>2.5</sub> standard (15/65- 98th percentile) incremental to the current PM<sub>10</sub> standard. These projections are for purposes of estimating costs and benefits; specific nonattainment designations will be based on monitoring data collected in the future for each area. As discussed in Chapter 6, the National PM Strategy brings 35 counties into attainment of the selected PM<sub>2.5</sub> standard, leaving 67 counties requiring additional control for attainment. At the national level, 11 counties are estimated to be in initial nonattainment of the selected PM<sub>10</sub> standard (50/150- 99th percentile).

Baseline ozone air quality concentrations in 2010 are estimated using a Regional Oxidant Model (ROM) extrapolation methodology. Initial nonattainment areas for alternative ozone standards are identified based on these modeled values for counties with ozone monitors in 1990. At the national level, nine areas are predicted to be in initial nonattainment of the current one-hour ozone standard; an additional 10 areas (19 total areas) are predicted to violate the 0.08 ppm/8-hr/4th max ozone alternative. These projections are for purposes of estimating costs and benefits; specific nonattainment designations will be based on monitoring data collected in the

future for each area.

## 4.2 INTRODUCTION

This chapter describes the methods used to estimate baseline emissions and air quality in 2010 in order to assess the costs, benefits and economic impacts of alternative ozone and PM standards and regional haze goals. The assessments are conducted from a consistent analytical baseline. A single emissions inventory employing consistent methods is used as the basis for the ozone, PM and RH analyses. The year 2010 is selected as the year of analysis to provide an appropriate period in which 1) major programs of the CAA of 1990 will be reaching full implementation; 2) current standards are to be achieved; and 3) new standards are being implemented.

The PM and ozone analyses have been constructed such that benefits and costs are estimated incremental to those derived from the combined effects of implementing both the CAA of 1990 and the current ozone and PM standards as of the year 2010. These analyses provide a “snapshot” of air quality impacts, costs, and benefits associated with implementation of the new ozone and PM standards from a baseline of future CAA implementation and attainment of current standards. RH visibility goals are evaluated incremental to implementation of the new ozone and PM standards.

For the purpose of identifying the nonattainment areas associated with alternative NAAQS, this RIA excludes areas that did not have monitors during 1990 - 1995. Once nonattainment areas have been identified within the set of monitored areas, the analysis assumes control strategies on a local, regional, and national basis for the purpose of bringing identified nonattainment areas into attainment. Therefore, while the nonattainment areas are identified from within monitored areas only, control requirements, costs, benefits, and other economic impacts are estimated for both monitored and unmonitored areas.

EPA believes that the monitored counties analytic approach for identifying nonattainment

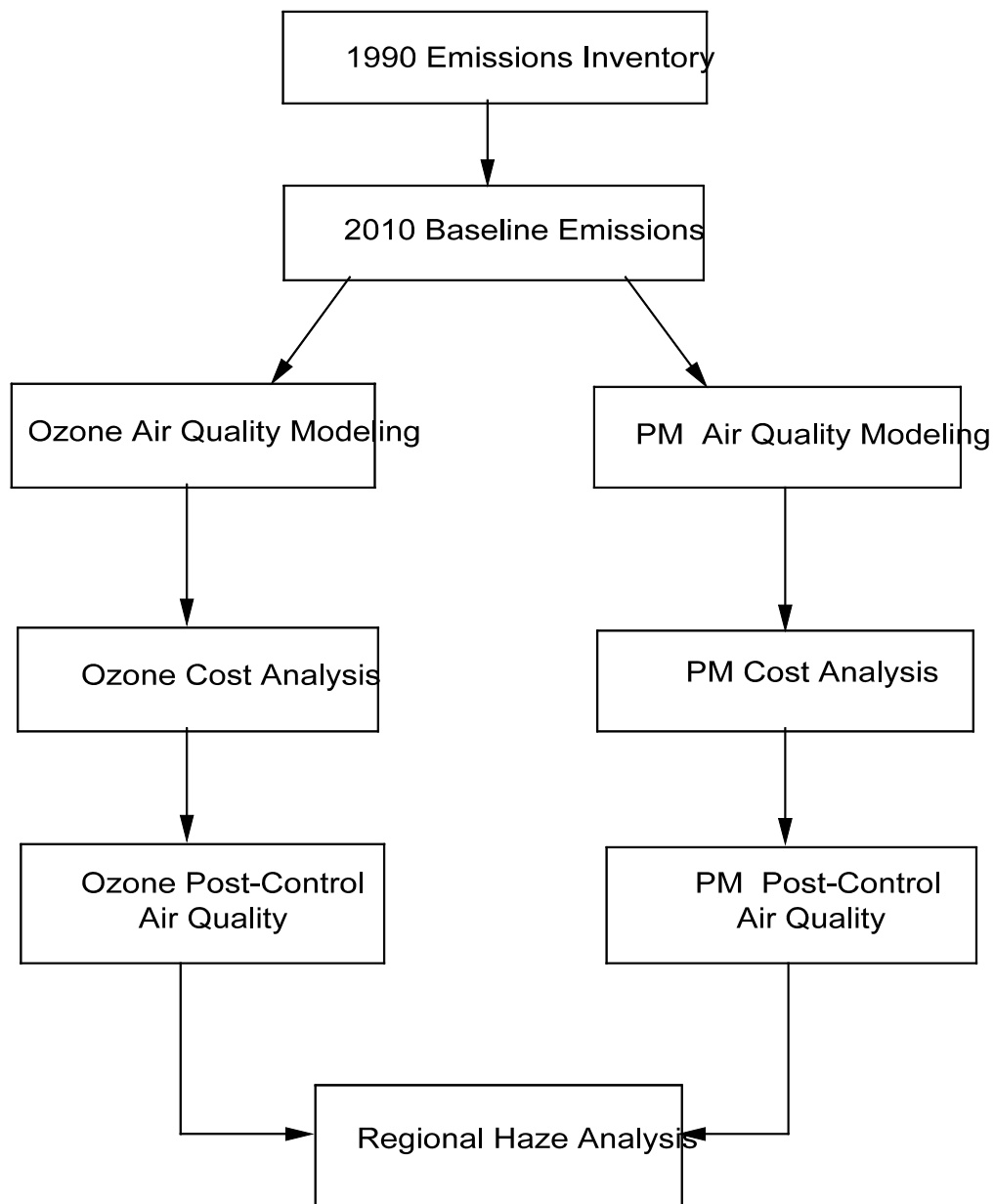
areas is most appropriate because 1) the likelihood of significant nonattainment in unmonitored areas after RIA controls are imposed is small; 2) serious modeling difficulties exist that prevent reliable prediction of nonattainment in unmonitored areas; and 3) any such nonattainment in unmonitored areas may not be detected (U.S. EPA, 1997c). It is possible, however, that even after all controls are imposed, nonattainment areas may exist that are not identified in the RIA, but may be identified in the future by the placement of new monitors.

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. CAA-mandated controls (i.e., control efficiencies or control-specific emission factors) then are applied to these future emissions to capture implementation of the CAA. The 2010 post-CAA control emissions are input to air quality models to predict baseline PM and ozone air quality from which PM and ozone nonattainment areas subsequently are identified. Control measures to bring these areas into attainment of alternative PM and ozone standards are evaluated and applied in the cost analyses. Emission reductions achieved by these control measures determine the “post-control” PM and ozone air quality in these areas. Given that regional haze goals are to be evaluated by areas after implementation of proposed PM and ozone standards, the post-control PM and ozone air quality serve as the baseline from which regional haze goals are analyzed. The methodologies used to estimate visibility for assessing the RH targets are discussed in Chapter 8.

#### **4.3 ESTIMATION OF 1990 EMISSIONS AND 2010 EMISSIONS PROJECTIONS**

The initial step in the assessment of alternative ozone and PM standards and RH goals is the development of the 2010 CAA emission estimates. These emissions serve as the baseline for

Figure 4.1 Overview of Emissions and Air Quality Analytical Approach

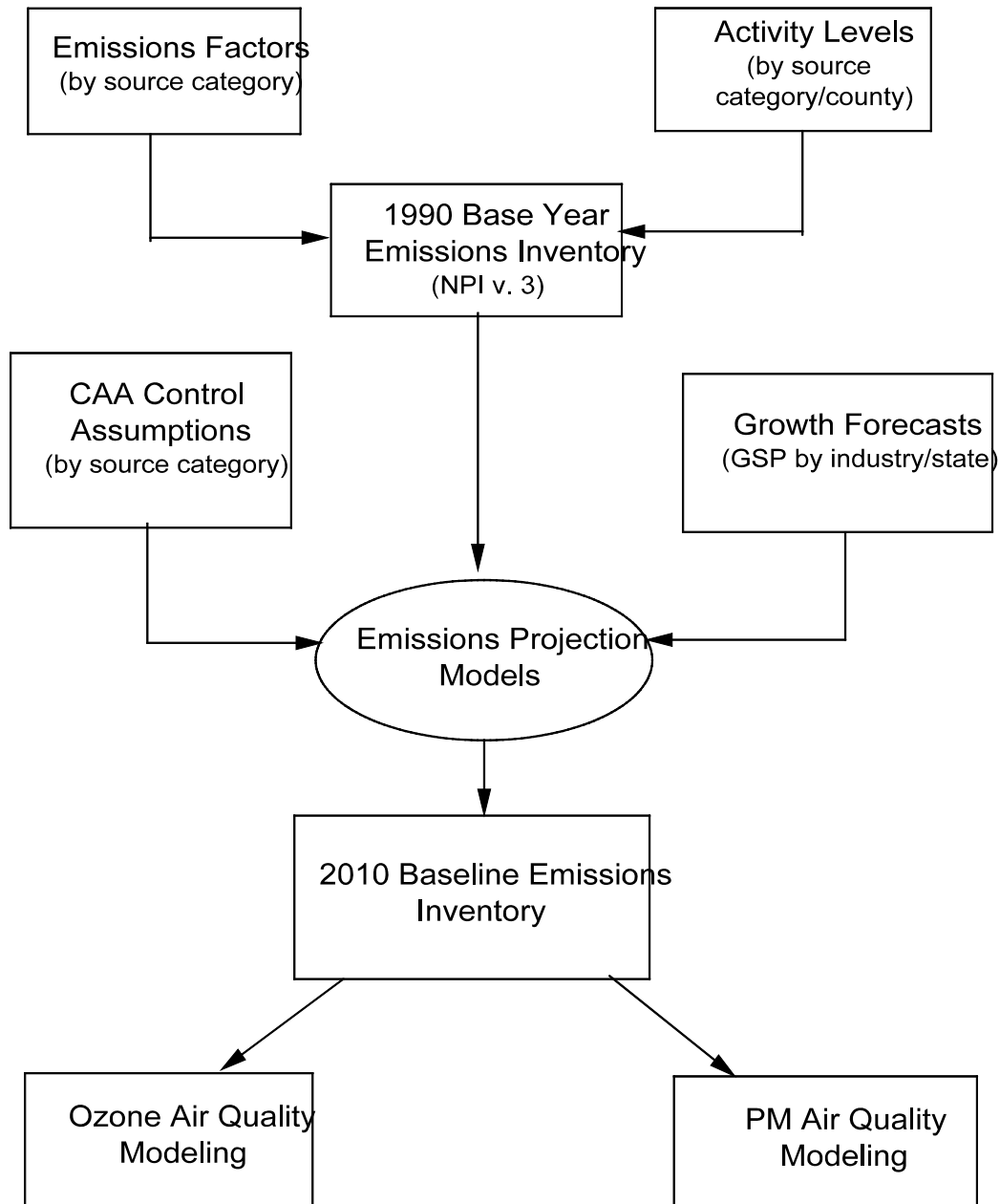


evaluation of alternative control measures for reducing ozone and PM precursor emissions for attainment of ozone and PM standards. The emissions estimation and projection methodologies build upon work conducted for the December 1996 ozone and PM Regulatory Impact Analyses (RIAs) (U.S. EPA, 1996f, 1996g). Major updates and refinements to the December 1996 emissions estimation methodologies are listed below.

- Version 3 of the 1990 National Particulates Inventory (NPI v.3)(Pechan, 1996c) is used as the base year emissions inventory (Version 2 was used in the December 1996 assessments (Pechan, 1995));
- Bureau of Economic Analysis (BEA) projections of Gross State Product (GSP) (BEA, 1995) are used to estimate 2010 emissions (BEA earnings data were employed in the December 1996 assessments (BEA, 1990));
- Utility sector CAA-control emission projections incorporate future utility deregulation and a 0.15 lb/MMBtu nitrogen oxides (NO<sub>x</sub>) cap with trading and banking;
- The following CAA-mandated control assumptions are updated in the 2010 baseline emissions:
  - Control measure effectiveness for volatile organic compounds (VOC) and NO<sub>x</sub> control measures is increased from 80% to 95%;
  - OTAG Level 2 NO<sub>x</sub> controls on industrial point sources in 37 OTAG states are applied;
  - 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 4.2 illustrates the steps followed in the development of 2010 baseline 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

Figure 4.2 Development of 2010 Baseline Emissions



year values. Emissions are estimated for VOC, NO<sub>x</sub>, sulfur dioxide (SO<sub>2</sub>), primary PM<sub>10</sub> and PM<sub>2.5</sub>, secondary organic aerosols (SOA) 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 PM NAAQS control strategy analysis.

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<sub>x</sub>) 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 then serve as inputs to the ozone and PM air quality modeling.

#### **4.3.1 Development of 1990 Base Year Emissions Inventory**

The 1990 base year emissions inventory is based on Version 3 of the NPI (Pechan, 1996c; Pechan, 1997a). This is a more recent version of the NPI than was used in the December 1996 RIAs (i.e., NPI version 2 (Pechan, 1995)). The major difference in the inventories is in the



fugitive dust PM emissions estimates: version 3 fugitive dust emissions estimates are lower than version 2.

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). EPA emission factors are available for VOC, NO<sub>x</sub>, SO<sub>2</sub>, and PM<sub>10</sub>. Because there are no emission factors for PM<sub>2.5</sub>, 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 base year inventory. Appendix A.1 describes in more detail the emissions estimation methodologies used for each major emitting sector.

#### 4.3.2 1990 Emissions Inventory Results and Discussion

Table 4.2 presents a summary of 1990 emissions by pollutant and major sector. Appendix A.2 presents 1990 emissions by source category and major sector. Area sources are the largest contributor to anthropogenic VOC emissions in 1990 (45% 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% of total national  $\text{NO}_x$  emissions with 46% of the motor vehicle emissions contributed by cars (i.e., light-duty gasoline vehicles). With regard to national  $\text{SO}_2$  emissions, the utility sector is the largest emitter (71%). Area sources account for the bulk of  $\text{PM}_{10}$  and  $\text{PM}_{2.5}$  emissions. Anthropogenic fugitive dust sources contribute the majority of primary  $\text{PM}_{10}$  and  $\text{PM}_{2.5}$  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% of biogenic VOCs, is 30%, while the average FAC for anthropogenic VOC sources is less than 1%.

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

**Table 4.1 Base Year 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 <sub>10</sub> and PM <sub>2.5</sub> 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 <sub>x</sub> , PM <sub>2.5</sub> , PM <sub>10</sub> ): 1991 Office of Mobile Sources (OMS) Nonroad Inventory (U.S. EPA, 1991b) Internal Combustion Engines/Vehicles (SO <sub>2</sub> ) 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 <sub>10</sub> , PM <sub>2.5</sub> ) Agricultural Tilling Construction Unpaved and Paved Roads Livestock	U.S. Department of Agriculture data (USDA, 1991), U.S. EPA PM <sub>10</sub> emission factors (U.S. EPA, 1995a). Census Bureau Construction Expenditures (BOC, 1992), EPA PM <sub>10</sub> 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 <sub>10</sub> emission factors (U.S. EPA, 1995a). Particle size multipliers are applied to PM <sub>10</sub> emissions to estimate PM <sub>2.5</sub> 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 <sub>3</sub> )	NH <sub>3</sub> emissions for livestock feedlots and fertilizers based on Census of Agriculture data (BOC, 1992) and EPA-recommended emission factors (Battye et al., 1994).

emissions are not considered for control in the PM control strategy-cost analysis.

It should be noted 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 PM and ozone air quality modeling sections (Chapter 4) and the PM and ozone cost chapters (Chapters 5 and 6) to understand how emissions from various source categories impact PM and ozone air quality.

### **4.3.3 Key Uncertainties Associated with 1990 Base Year 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 emissions 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 SO<sub>2</sub> and NO<sub>x</sub> emissions between alternative estimation methodologies, however, are not that large. Recent comparisons of SO<sub>2</sub> CEM data with estimates based on SO<sub>2</sub> 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<sub>x</sub> emissions based on CEM data and NO<sub>x</sub> 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 is 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.

**Table 4.2 Summary of National 1990 Base Year Emissions Estimates by Major Sector**

<b>Major Sector</b>	<b>VOC</b> (1000 tpy)	<b>NOx</b> (1000 tpy)	<b>SO<sub>2</sub></b> (1000 tpy)	<b>PM<sub>10</sub></b> (1000 tpy)	<b>PM<sub>2.5</sub></b> (1000 tpy)	<b>SOA</b> (1000 tpy)
Utility	37	7,426	15,865	283	109	1
Industrial Point	3,467	2,850	4,644	926	589	35
Area	10,098	2,100	1042	35,290	7,639	92
Nonroad	2,054	2,836	242	336	293	23
Motor Vehicle	6,811	7,446	568	355	291	48
<b>Anthropogenic</b>	<b>22,466</b>	<b>22,656</b>	<b>22,359</b>	<b>37,190</b>	<b>8,921</b>	<b>198</b>
<b>Subtotal</b>						
Biogenics	25,988					3,325
Natural Sources	248	89	1	5,429	995	
<b>TOTAL</b>	<b>48,702</b>	<b>22,745</b>	<b>22,360</b>	<b>42,619</b>	<b>9,916</b>	<b>3,525</b>

Note: Emissions estimates may not sum due to rounding.

1990 fugitive dust emissions have not been adjusted here as described in Section 4.4.2.3.

Air quality impacts from major emitting sectors are not necessarily proportional to their contribution to national emissions estimates. See PM and Ozone Air Quality Modeling Sections 4.4 and 4.5 and Chapters 6 and 7.

Use of particle size multipliers to estimate  $PM_{10}$  and  $PM_{2.5}$  emissions from TSP data yields uncertain results relative to application of  $PM_{10}$  or  $PM_{2.5}$  emission factors. The degree of uncertainty may vary by source category; however, there is no known bias in these factors.

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 than biogenics included in the base year emissions. These higher VOC estimates also lead to higher biogenic SOA. 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, biogenic VOC and SOA may be underestimated, but in any individual county the bias is unclear (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% and  $PM_{2.5}$  fugitive dust emissions may be overestimated by 72% 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 4.4.2.3 for a discussion of the implications of this overestimate of fugitive dust emissions on modeled PM air quality. Of particular interest is that the  $PM_{2.5}$  emission estimate for agricultural operations (tilling and windblown dust) was decreased by about 50%, 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 PM air quality analysis is likely biased toward overestimating fugitive dust impacts 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 PM 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% 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 nonattainment areas to the rest of the country introduces uncertainty to the nonroad emissions estimates, however, with no known bias.

Because the 1985 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 is used to grow emissions from 1985 to 1990 rather than applying the more recent BEA GSP estimates. This may result in a small underestimate of 1990 emissions (Pechan, 1997a).

Considering relative uncertainty across emissions of individual pollutants, SO<sub>2</sub> emission estimates are the most certain. SO<sub>2</sub> 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, SO<sub>2</sub> emissions can be estimated within a relatively narrow range. For example, as part of the GCVTC emission inventory, uncertainty estimates were developed for various major SO<sub>2</sub> sources (Balentine and Dickson, 1995). The uncertainty estimate calculated for SO<sub>2</sub> emissions from copper smelting is  $\pm 50$  percent. However, associated uncertainty for emissions estimates from diesel and gasoline vehicles are assessed at  $\pm 150$  percent. Most of this uncertainty is due to the variability in the sulfur content of the fuels.

The NO<sub>x</sub> estimates are the next most certain category of emissions. Like SO<sub>2</sub>, NO<sub>x</sub> is a product of fuel combustion. Since NO<sub>x</sub> formation is somewhat more complicated than SO<sub>2</sub>, emission estimates are more variable, and uncertain, as well.

The level of uncertainty in  $PM_{10}$  emission estimates varies widely by source category. The largest component of the 1990  $PM_{10}$  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  $\pm 400$  percent. The estimated uncertainty for  $PM_{2.5}$  emissions from paved road dust is  $\pm 180$  percent (Ballentine and Dickson, 1995).  $PM_{10}$  emission estimates for large point sources such as utility boilers are likely 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.

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  $\pm 150$  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.



**Table 4.3 Uncertainties and Possible Biases in Estimating 1990 Emissions**

<b>Potential Source of Uncertainty</b>	<b>Positive Bias? (Overestimate)</b>	<b>Negative Bias? (Underestimate)</b>	<b>Bias Unclear</b>
Use of emission factors rather than stack test, load-curve, or CEM data			✓
Use of particle-size multipliers to estimate PM <sub>10</sub> and PM <sub>2.5</sub> emissions from TSP emissions			✓
Extrapolation of nonroad inventory from 27 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 Development of 2010 Emission Projections**

The base year emissions are projected to 2010 to develop the emissions baseline from which to evaluate additional control measures needed to meet alternative ozone and PM standards and RH 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.

#### **4.3.5 Growth Assumptions by Major Sector**

This section describes the sector-specific growth assumptions used to project emissions to 2010. Table 4.4 summarizes the emissions projection modeling approach by major sector. Version 3 of the NPI employs 1995 BEA Gross State Product (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. 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, the Integrated Planning Model (IPM) is 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). Utility deregulation was not accounted for in the December 1996 RIAs. 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

**Table 4.4 2010 Growth Assumptions by Major Sector**

Sector	Growth Forecast	Modeling Approach
Industrial Point	BEA Gross State Product (GSP) Projections by State/Industry (BEA, 1995)	<p>VOC, NO<sub>x</sub> - 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).</p> <p>PM<sub>10</sub>, PM<sub>2.5</sub>, SO<sub>2</sub>, NH<sub>3</sub> - While no formal model exists, the same basic approach applied in ERCAM was used for these pollutants (Pechan, 1997a).</p>
Utility	Projections of heat input by unit based on National Electric Reliability Council (NERC) data, price and demand forecasts, and technology assumptions.	<p>SO<sub>2</sub>, NO<sub>x</sub> - Integrated Planning Model (IPM) (U.S. EPA, 1996i).</p> <p>VOC, PM<sub>10</sub>, PM<sub>2.5</sub> - base year emission rates or AP-42 emission factors applied to IPM projected heat input by unit (Pechan, 1997a).</p> <p>NH<sub>3</sub> - NH<sub>3</sub> slippage for units controlling with selective catalytic reduction (SCR) (Pechan, 1997a).</p>
Nonroad	BEA GSP and Population Projections by State/Industry (BEA, 1995)	<p>VOC, NO<sub>x</sub> - ERCAM (Pechan, 1994, 1996b).</p> <p>PM<sub>10</sub>, PM<sub>2.5</sub>, SO<sub>2</sub>, NH<sub>3</sub> - ERCAM approach (no formal model)(Pechan, 1997a).</p>
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)	<p>NO<sub>x</sub>, VOC - ERCAM: applies MOBILE5a emission factors to projected VMT by month and county/vehicle type/roadway classification (U.S. EPA, 1991c, 1993a).</p> <p>PM<sub>10</sub>, PM<sub>2.5</sub>, SO<sub>2</sub> - PART5 emission factors(U.S. EPA, 1994c) applied to projected VMT (U.S. EPA, 1991c).</p> <p>NH<sub>3</sub> - special study emission factors applied to projected VMT (Pechan, 1997a).</p>
Area	BEA GSP and Population Projections by State/Industry (BEA, 1995)	<p>VOC, NO<sub>x</sub> - ERCAM (Pechan, 1994, 1996b).</p> <p>PM<sub>10</sub>, PM<sub>2.5</sub>, SO<sub>2</sub>, NH<sub>3</sub> - ERCAM approach (no formal model)(Pechan, 1997a).</p>
Biogenic VOC and PM Wind Erosion	Emissions held at 1990 levels	--

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).

Mobile source 1990 emissions are projected to 2010 based on growth in VMT. 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 biogenic emissions of VOC or SOA. Similarly, 2010 PM emissions from natural sources are assumed equal to 1990 levels.

#### **4.3.6 2010 CAA Control Emissions by Major Sector**

In order to capture the effects in 2010 of implementation of the CAA, future year control efficiencies or emission factors are applied to projected 2010 emissions or activity levels respectively. Table 4.5 summarizes the major CAA requirements that are modeled for the 2010 baseline. These control requirements are discussed in Appendix A.3 for each major sector.

For the 2010 CAA-control emissions, refined control measure effectiveness (CME) estimates are employed in combination with control efficiencies. CME reflects the degree to which individual control measures achieve their intended effect. An 80% CME was applied in the December 1996 RIAs for a subset of primarily VOC source category-control measure combinations. For this assessment, CME is assumed to be 95% 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 control measure efficiencies in place prior to

**Table 4.5 CAA 2010 Projection Scenario Summary by Major Sector**

Major Sector	Major CAA Scenario Requirements
Industrial Point	<p>VOC and NO<sub>x</sub> RACT for all NAAs (except NO<sub>x</sub> waivers).                      New control technique guidelines (CTGs).                      0.15 pounds per million British thermal unit (lb/MMBtu) Ozone Transport Assessment Group (OTAG)-wide NO<sub>x</sub> cap on fuel combustors ≥ 250 MW.                      OTAG Level 2 NO<sub>x</sub> 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<sub>x</sub> waiver) NAAs.                      Phase II of the Ozone Transport Commission (OTC) NO<sub>x</sub> memorandum of understanding (MOU).                      0.15 lb/MMBtu OTAG-wide seasonal NO<sub>x</sub> 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 O<sub>3</sub> and carbon monoxide (CO) NAAs.                      Federal reformulated gasoline for O<sub>3</sub> 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<sub>x</sub> 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>

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 2010 emissions baseline; instead, the emission reductions and costs are assessed for future attainment of the current ozone standard. Appendix C discusses the methodology and results of this analysis.

Additionally, updated information regarding proposed Title I AIM Coatings and Consumer and Commercial Products rules and Title III 7 and 10 year MACT rules are incorporated in the 2010 CAA-control emissions.

Ozone air quality modeling analyses show that NO<sub>x</sub> 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<sub>x</sub> control measures far in advance of the 2010 air quality assessment undertaken for this RIA. Because these control measures will be applied for the purpose of attaining the current standard, they are included in the analytical baseline of this RIA.

The 2010 baseline reflects the application of regional NO<sub>x</sub> reductions that are intended to approximate the reductions EPA would propose based upon OTAG recommendations. The regional NO<sub>x</sub> controls applied for this analysis include: 1) OTAG-wide 0.15 lb/MMBtu NO<sub>x</sub> emission limit on utilities and on non-utility boilers  $\geq$  250 MW; 2) OTAG Level 2 NO<sub>x</sub> 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. Of the States for which OTAG did not currently recommend controls, only Louisiana, Oklahoma and Texas have areas that are projected to be

nonattainment for one or more of the standards evaluated. Because regional controls were not recommended for these States, the RIA may underpredict costs and benefits for these areas.

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 these reductions in the baseline is appropriate. This analysis, however, does not prejudge the outcome of negotiations with the automobile industry.

#### **4.3.7 2010 Baseline Emissions Results and Discussion**

Table 4.6 summarizes national 2010 CAA emissions by major sector. Appendix A.4 presents 2010 emissions by source category and major sector. Total emissions of VOC, NO<sub>x</sub>, SO<sub>2</sub>, and SOA are estimated to decrease from 1990 levels; however, emissions of PM<sub>10</sub> and PM<sub>2.5</sub> 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 (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. 2010 Title I and III area source controls are projected to achieve 5.7 million tons of VOC emission reductions per year.

National NO<sub>x</sub> emission reductions for Title I industrial point source controls are estimated to total 1.6 million tons per year: CAA-mandated controls and the NO<sub>x</sub> cap account for approximately 500,00 tons and 100,000 tons of NO<sub>x</sub> reductions respectively and OTAG-

wide Level 2 NO<sub>x</sub> controls contribute an additional 1 million tons per year of NO<sub>x</sub> reductions (U.S. EPA, 1997j). Title I area source NO<sub>x</sub> controls account for reductions of 1.4 million tons of NO<sub>x</sub> per year. Title I mandated controls, Title IV Acid Rain NO<sub>x</sub> requirements, and the OTAG-wide NO<sub>x</sub> cap result in an estimated 3 million tons of summertime NO<sub>x</sub> reductions from the utility sector (U.S. EPA, 1997a).

Title II mobile source VOC and NO<sub>x</sub> 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<sub>x</sub> nationally in 2010 (U.S. EPA, 1997j).

The Title IV Acid Rain Program accounts for an 8 million ton reduction in utility SO<sub>2</sub> emissions from 2010 no-control levels (U.S. EPA, 1997a).

#### **4.3.8 Key Uncertainties Associated with 2010 Baseline Emissions**

Table 4.8 summarizes the key uncertainties associated with the 2010 baseline 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 baseline. 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 4.6 Summary of National 2010 CAA Emissions Estimates by Major Sector**

<b>Major Sector</b>	<b>VOC (1000 tpy)</b>	<b>NO<sub>x</sub> (1000 tpy)</b>	<b>SO<sub>2</sub> (1000 tpy)</b>	<b>PM<sub>10</sub> (1000 tpy)</b>	<b>PM<sub>2.5</sub> (1000 tpy)</b>	<b>SOA (1000 tpy)</b>
Utility	50	3,755	9,746	277	111	0
Industrial Point Area	2,164	1,958	5,990	1,170	745	25
Nonroad	7,533	2,932	1,518	41,051	8,931	73
Motor Vehicle	1,888	2,063	236	336	292	24
	3,946	5,574	409	204	141	27
<b>Anthropogenic Subtotal</b>	<b>15,581</b>	<b>16,282</b>	<b>17,899</b>	<b>43,038</b>	<b>10,220</b>	<b>150</b>
Biogenics	25,988					3,325
Natural Sources	248	89	1	5,429	995	
<b>TOTAL</b>	<b>41,817</b>	<b>16,371</b>	<b>17,900</b>	<b>48,467</b>	<b>11,215</b>	<b>3,475</b>

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 PM and Ozone Air Quality Modeling Sections 4.4 and 4.5 and Chapters 6 and 7.

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 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**

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			✓ ✓
2010 Control Assumptions: - 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.	✓ ✓	✓	✓ ✓

#### 4.4 ESTIMATION OF BASELINE PM AIR QUALITY CONCENTRATIONS IN 2010

The methodology for estimation of baseline PM air quality concentrations for this assessment builds upon the previous method used in the December 1996 PM NAAQS 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<sub>10</sub> and PM<sub>2.5</sub> at county-level receptors. The following updates to data inputs, methodological refinements, and sensitivity analyses are implemented for this assessment:

- Updated Phase II CRDM air quality modeling results are employed;
- The source-receptor matrix is calibrated using 1993 -1995 Aerometric Information Retrieval System (AIRS) monitoring data for all 711 counties monitored for PM<sub>10</sub> in the 48 contiguous states during this 3-year period;
- The number of monitored counties covered in the nonattainment county analysis is increased (i.e., 504 counties vs. 470 in the December 1996 analysis);
- Sensitivity analyses of the following are conducted:
  - number of counties covered in the baseline PM nonattainment county analysis
  - fugitive dust adjustment factor;
- Analysis of PM air quality as it relates to regional haze visibility improvement goals is included.

##### 4.4.1 Overview of Phase II PM Air Quality Modeling

This section provides a general overview of the Phase II PM air quality modeling analysis. More detailed information follows in subsequent sections. The December 1996 PM RIA assessment employed the Phase I source-receptor (S-R) matrix as produced by the CRDM. The Phase I modeling results are thought to be deficient in that they likely underestimated the impacts of secondary particle precursor emissions. For Phase II, the Lagrangian Regional Model is used to guide the refinement of the CRDM to correct for this misestimation (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 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 are evaluated for violation of alternative standards. Figure 4.3 illustrates the development of 2010 baseline PM air quality concentrations.

## 4.4.2 Elements of PM Air Quality Modeling

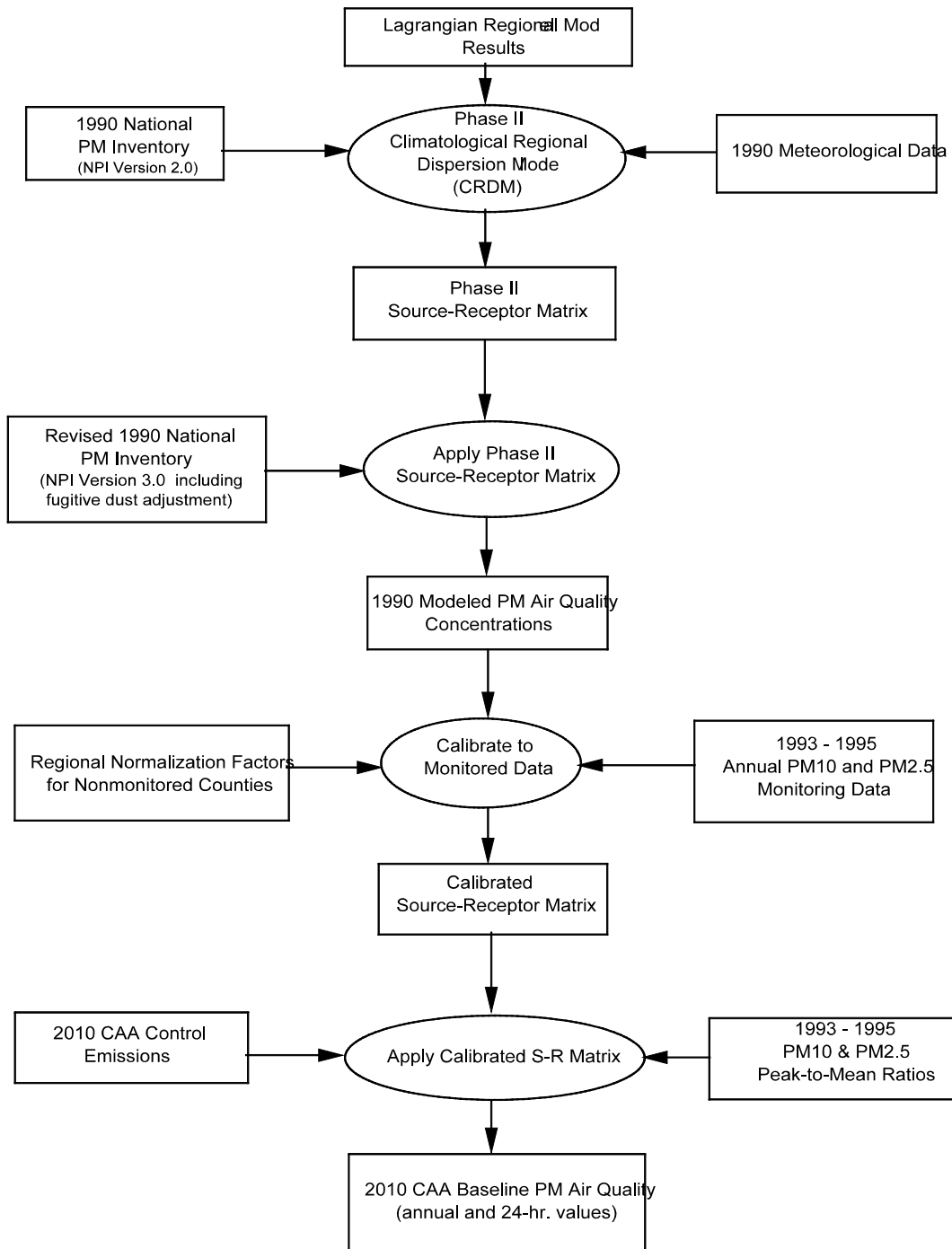
### 4.4.2.1 Lagrangian Regional Model

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 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 account of 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). CRDM

Figure 4.3  
Development of 2010 Baseline PM Air Quality



incorporates terms for wet and dry deposition and chemical conversion of SO<sub>2</sub> and NO<sub>x</sub>, 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 is 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 2.0 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.

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  $PM_{10}$  and  $PM_{2.5}$  (i.e., wind erosion and wild fires) are also included. Emissions of  $SO_2$ ,  $NO_x$ , 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  $PM_{10}$  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  $SO_2$ ,  $NO_x$ , and ammonia are multiplied by the ratios of the molecular weights of sulfate/ $SO_2$ , nitrate/nitrogen dioxide and ammonium/ammonia to obtain concentrations of sulfate, nitrate and ammonium.<sup>1</sup> 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:

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1 Ratio of molecular weights: Sulfate/ $SO_2$ = 1.5; nitrate/nitrogen dioxide = 1.35; ammonium/ammonia = 1.06.



- 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.<sup>1</sup>

#### 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 three-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  $PM_{10}$ , modeled and measured concentrations of sulfate, nitrate and organics are comparable (Latimer, 1996).

Additionally, some preliminary air quality modeling has been conducted using the Regional Acid Deposition Model-Regional Particulate Model (RADM-RPM) for the Eastern U.S. using 1990 emissions and meteorology (U.S. EPA, 1997b). This is a Eulerian gridded model incorporating more comprehensive physics and chemistry to enable better characterization of secondarily-formed pollutants than Lagrangian-based methods. In general, the CRDM results show a similar East-West trend in sulfate and nitrate concentrations within the same modeling

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1 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.

region. Also, the CRDM-predicted annual average concentrations of sulfate are within the range of RADM-RPM base-case predictions. Relative to RADM-RPM base case results, CRDM appears to overpredict nitrate concentrations in the Mid-west and underpredict nitrate concentrations in the Mid-Atlantic states.

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<sub>x</sub> and SO<sub>x</sub> in North America (U.S. EPA, 1996l). 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 can not 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<sub>2.5</sub> concentrations range from 1 - 4 ug/m<sup>3</sup> in the West and from 2 - 5 ug/m<sup>3</sup> 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. This issue is discussed in Chapter 6 for PM residual nonattainment areas.

Although the bulk of primary PM emissions are from anthropogenic and natural fugitive dust sources<sup>1</sup>, available speciated monitoring data indicate that fugitive dust contributes substantially less to total PM<sub>2.5</sub> levels relative to other particle species such as sulfates and nitrates. The CRDM-predicted average fugitive dust contribution to PM<sub>2.5</sub> mass is 31% in the East and 32% in the West (Pechan, 1997b). Speciated monitoring data show that minerals (i.e., crustal material) comprise approximately 5 percent of PM<sub>2.5</sub> mass in the East and approximately 15 percent of PM<sub>2.5</sub> 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<sub>10</sub> and PM<sub>2.5</sub> emissions used in this analysis are 40% and 73% greater, respectively, than the most recent National Emissions Trends Inventory estimates<sup>2</sup> (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<sub>2.5</sub> and actual ambient data. Two multiplicative factors are examined: 0.25 and 0.10. The 0.25 multiplicative adjustment results in a fugitive dust contribution to modeled ambient PM<sub>2.5</sub> concentrations of 10 - 17%, while the 0.10 multiplicative factor results in a 4 - 8% contribution.<sup>3</sup> Given that the 0.25 multiplicative factor appears to bring the modeled fugitive

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- 1 Natural and anthropogenic fugitive dust emissions account for 93% of PM<sub>10</sub> emissions and 76% of PM<sub>2.5</sub> emissions in the 1990 base year inventory (NPI version 3).
  - 2 Natural and anthropogenic fugitive dust emissions account for 86% of PM<sub>10</sub> emissions and 59% of PM<sub>2.5</sub> emissions in the most recent 1990 National Emission Trends Inventory.
  - 3 See map on p. 6-5 for delineation of cost modeling regions. Using 0.25 multiplicative factor, fugitive dust as percentage of PM<sub>2.5</sub> mass for: Central U.S. = 17.2%; Eastern U.S.= 10.4%; Western U.S.= 10.6%. Using 0.10 multiplicative factor, fugitive dust as percentage of PM<sub>2.5</sub> mass for: Central U.S. = 7.8%; Eastern U.S.= 4.5%; Western U.S.= 4.6%. By comparison, without using a multiplicative factor, fugitive dust as a percentage of PM<sub>2.5</sub> mass for: Central U.S. = 44.6%; Eastern U.S. = 30.9%; Western U.S. = 31.5%. As discussed previously, the overestimation of fugitive dust emissions from farming operations could easily

dust contribution to  $PM_{2.5}$  mass more within the range of values reported from speciated monitoring data, the main PM analysis of costs and benefits uses the 0.25 multiplicative factor to adjust fugitive emissions. However, a sensitivity analysis is conducted using the 0.10 multiplicative factor. The impact of the fugitive dust adjustment factor on PM nonattainment county counts is discussed in section 4.4.3. Appendix D provides more detailed information on this sensitivity analysis.

#### **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. This recent emissions inventory update concluded after completion of Phase II CRDM modeling. In order to account for this emission inventory refinement as well as the fugitive dust adjustment as discussed above, 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 overprediction or underprediction 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 value (U.S. EPA, 1997k). Tier 1 monitored counties cover the 504 counties with at least 50% data completeness and therefore have the highest level of certainty associated with the

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account for the increased fugitive dust contribution in the Central U.S., where farming operations are concentrated.

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 three years during the period 1993 -1995. Tier 3 monitored counties cover 107 additional counties with missing monitoring data for one or two of the three years 1993 - 1995. In total, Tiers 1, 2 and 3 cover 711 counties currently monitored for  $PM_{10}$  in the 48 contiguous states.<sup>1</sup> 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 region<sup>2</sup>.

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 three 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  $PM_{10}$ .<sup>3</sup>

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 SAS<sup>TM</sup> 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

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- 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).

annual average  $PM_{2.5}$  air quality data are calibrated to the spatially-averaged annual  $PM_{2.5}$  value<sup>1</sup> from the derived  $PM_{2.5}$  dataset. Additionally, the proposed form of the standard allows for averaging over three years of air quality data. These derived, annual  $PM_{2.5}$  data represent the annual average value over a three-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.

#### **4.4.2.6 Application of Calibrated Phase II S-R Matrix to 2010 CAA Control Emissions**

The calibrated Phase II S-R matrix is next applied to the 2010 CAA control emissions to predict baseline PM annual air quality at the county level. This baseline air quality reflects the fugitive dust emissions adjustment of 0.25.

#### **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.<sup>2</sup> The first peak-to-mean ratio is the three-year average 4th 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 4th 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 three-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 three-year average 98th percentile 24-hour peak  $PM_{2.5}$  value to the spatially averaged annual arithmetic mean  $PM_{2.5}$

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

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

value. This ratio is applied to the annual mean  $PM_{2.5}$  value to predict the three-year average 98th percentile 24-hour peak  $PM_{2.5}$  value (U.S. EPA, 1996e).

#### 4.4.3 PM Nonattainment Counties by Alternative

The model-predicted  $PM_{10}$  and  $PM_{2.5}$  air quality data for the 2010 CAA-control baseline is used to determine county air quality status. The rounding convention proposed for the PM NAAQS is used in the identification of counties predicted to have PM levels in 2010 greater than the standards examined.<sup>1</sup> Table 4.9 presents estimates of PM nonattainment counties by region. These results also reflect application of the 0.25 fugitive dust adjustment factor as discussed in Section 4.4.2.3. For the main analysis, nonattainment counties are identified from the Tier 1 set of 504 counties monitored during 1993 - 1995 for reasons discussed in Section 4.2 and because there is relatively more certainty associated with predicted air quality in these counties. Predicted PM concentrations are the most certain for the Tier 1 counties since the estimates are calibrated using 50% complete AIRS data as described in Section 4.4.2.5. This set represents approximately 70% of the counties within the 48 contiguous states monitored for  $PM_{10}$  during 1993 -1995, covering approximately 150 million people.

A sensitivity analysis is conducted for the 15/50 alternative to examine the extent of PM nonattainment when the Tier 1 county scope assumption is relaxed. In this sensitivity assessment, the set of counties from which nonattainment counties is identified is extended to include Tiers 2 and 3. This assumption increases monitored county coverage to all 711 counties monitored for  $PM_{10}$  in the contiguous 48 states during 1993 - 1995. It should be noted that the Tier 2 and 3 air quality estimates are less certain relative to Tier 1 estimates. The number of estimated nonattainment counties increases by 10 counties for the current  $PM_{10}$  standard (total of 53) and by 23 for the proposed  $PM_{2.5}$  standard (15/50) (total of 108). Appendix D presents the detailed results of this sensitivity analysis.

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1 Rounding convention:  $PM_{2.5}$  annual standard - rounded to the nearest 0.1;  $PM_{2.5}$  daily standard - rounded to the nearest 1;  $PM_{10}$  annual - rounded to the nearest 1;  $PM_{10}$  daily - rounded to the nearest 10.

A sensitivity analysis also is conducted for the 15/50 alternative to examine the extent of PM nonattainment when the 0.10 fugitive dust adjustment factor is employed. By reducing fugitive dust emissions by 90%, the number of initial nonattainment counties decreases by 11 counties for the proposed PM<sub>2.5</sub> standard (15/50) and stays the same for the current PM<sub>10</sub> standard. The 0.10 adjustment factor has implications for cost and residual nonattainment as discussed in Appendix D.

#### 4.4.4 Uncertainties in PM Air Quality Modeling

The methodology used to project PM concentrations in 2010 from 1990 emissions and ambient concentration data introduces several sources of uncertainty to the control strategy-cost and benefits analyses. Table 4.6 presents potential sources of uncertainty and associated biases in estimating 2010 initial PM nonattainment counties. “Positive bias” indicates that estimated 2010 nonattainment counties may be overestimated; “negative bias” indicates that estimated 2010 nonattainment counties may be underestimated; “bias unclear” indicates that the direction of impact from a given potential source of uncertainty on 2010 nonattainment counties 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 PM 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% for PM<sub>10</sub> and 73% for PM<sub>2.5</sub> relative to the more recent National Emissions Trends 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  $PM_{10}$  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  $\pm 7$  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  $PM_{10}$  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).

**Table 4.9 Predicted Counties in Initial Nonattainment of PM Standards in 2010  
Using 0.25 Fugitive Dust Adjustment Factor (Tier 1 counties only)**

Region <sup>1</sup>	Total Tier 1 Counties in Region <sup>2</sup>	Number of Counties Predicted in Initial Nonattainment of PM Alternative				
		PM <sub>10</sub> 50/150 - 1Ex) (current)	PM <sub>10</sub> <sup>3</sup> 50/150- 99th (selected)	PM <sub>2.5</sub> <sup>4</sup> 16/65-98th	PM <sub>2.5</sub> <sup>4</sup> 15/65-98th (selected)	PM <sub>2.5</sub> <sup>4</sup> 15/50- 98th (proposal)
Midwest/Northeast	218	6	2	38	56	58
Southeast	86	1	0	8	16	16
South Central	59	4	1	5	7	8
Rocky Mountain	64	12	1	8	11	18
West	49	15	6	11	12	16
Northwest	28	7	1	0	0	6
<b>Total Counties in Nonattainment</b>		45	11	70	102	122

- 
- 1 See map on p. 6-5 for delineation of control strategy modeling regions.
  - 2 Total number of monitored counties modeled in analysis = 504
  - 3 This alternative is analyzed incremental to 2010 baseline (i.e., prior to application of the National PM Strategy).
  - 4 These alternatives are analyzed incremental to the current PM<sub>10</sub> standard and are assessed prior to application of the National PM Strategy.

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<sub>2.5</sub> mass. CRDM may overestimate or underestimate other fine particle species when evaluating county-level model predictions relative to PM<sub>2.5</sub> ambient data. For example, in some PM residual nonattainment counties, the predicted biogenic organic contribution to PM<sub>2.5</sub> 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 2010 PM air quality predictions.

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 Regional Acid Deposition Model (RADM) results are used to complement the CRDM results. 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.

As indicated in the sensitivity analysis in Section 4.4.3, the Tier 1 geographic scope assumption underestimates to a small degree the number of predicted PM nonattainment areas relative to identifying potential nonattainment areas from across Tiers 1, 2 and 3 counties.

**Table 4.6 Uncertainties and Possible Biases in Estimating 2010 Nonattainment Counties**

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> - 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)	

## **4.5 ESTIMATION OF BASELINE OZONE AIR QUALITY CONCENTRATIONS IN 2010**

The methodology for estimating baseline ozone air quality concentrations for this assessment builds upon previous work conducted for the December 1996 Ozone NAAQS RIA (U.S. EPA, 1996g). Monitoring data for 1990 and ROM 2007 air quality estimates are used to develop 2010 baseline air quality and identify potential nonattainment areas of alternative ozone standards. Updates to data inputs and methodological refinements have been incorporated where feasible. Major updates and refinements to the December 1996 ozone air quality analysis are listed below.

- A more informed picture of the future ozone nonattainment situation is provided based on comparison of model-predicted nonattainment with:
  - 1993 - 1995 monitored air quality data;
  - Air quality modeling from comparable emission reduction scenarios using ROM and Urban Airshed Model-Variable scale (UAM-V);
  - State Implementation Plan air quality modeling information;
- Model-predicted nonattainment counties are based on counties having ozone monitors in 1990;
- The concept of marginal nonattainment areas is eliminated;
- The concept of downwind transport areas is incorporated into the baseline ozone nonattainment area analysis.

### **4.5.1 Overview of Development of 2010 Baseline Ozone Air Quality**

To assess national annual costs and benefits of alternative ozone standards in the absence of temporally and spatially comprehensive air quality modeling tools is a challenging task. Most ozone air quality models are run to examine peak ozone concentrations for specific ozone episodes. Rarely are models run for an entire ozone season. Additionally, available ozone air

quality modeling is limited in its geographic scope. The Eastern U.S. is covered by regional-scale models such as the ROM or the UAM-V; however, geographic coverage of available models outside of the Eastern U.S. is limited. Therefore, the development of baseline ozone air quality data relies upon a full year of ozone monitoring data and available seasonal air quality modeling results to create a national picture of ozone air quality concentrations across a full year.

Figure 4.4 illustrates the steps followed to develop 2010 baseline air quality. ROM air quality modeling information for 2007 is used in combination with 1990 historical ozone air quality monitoring data to develop 2007 ozone air quality for the 48 contiguous states. The 2007 predicted air quality is then adjusted to account for 2010 emissions inventory differences and additional ozone modeling and monitoring information (i.e., 1993 - 1995 AIRS monitoring data, ROM and UAM-V air quality modeling data) to yield 2010 baseline ozone air quality data. Because this future air quality is based on counties with monitoring data in 1990, the centroid model is used to develop air quality for nonmonitored counties through geographic interpolation. This data is input to the benefits analysis. The 2010 baseline ozone air quality data for monitored counties is used to identify ozone nonattainment areas. This information is input to the control strategy and cost analysis. The following sections describe in more detail the various components of the analysis as illustrated in Figure 4.4.

## **4.5.2 Elements of Ozone Air Quality Modeling**

### **4.5.2.1 2007 ROM Air Quality Modeling**

A series of ROM analyses are conducted for the ozone NAAQS proposal to serve as rough planning tools for the development of policies to implement a new ozone NAAQS as well as for estimation of costs and benefits. Covering the eastern 37 states, ROM air quality modeling results are available for the following scenarios: (1) 1990 basecase; (2) 2007 CAA-mandated control; (3) NO<sub>x</sub> and VOC across-the-board reductions (i.e., matrix runs) from this 2007 CAA scenario; and (4) 2007 regional control strategy (U.S. EPA, 1996b). Because of the

limited geographic scope of the ROM modeling domain and the need for ozone air quality predictions for the entire continental U.S. in order to assess national costs and benefits of alternative ozone standards, a methodology is developed to extrapolate ROM predictions to all counties in the U.S.

Given the limited availability of meteorological data for input to the air quality model, the ROM simulations selected 1987 meteorological conditions to predict hourly ozone concentrations for the June through August period. It is desirable to employ for air quality modeling purposes meteorological data that are representative of typical ozone-forming conditions. According to a method discussed by Cox and Chu (1996), 1987 is not a particularly severe year in the Northeast nor in the Gulf regions. In the South and the Midwest, 1987 does stand out as a rather conducive year for high ozone, though not as severe as 1988. When viewed in the context of the 10 year period 1986 - 1995, overall the year 1987 is not an unusually conducive year for high ozone across the Eastern U.S. Thus, 1987 is considered a representative meteorological year for ROM modeling purposes.

#### **4.5.2.2 Development of 1990 Ozone Air Quality Data**

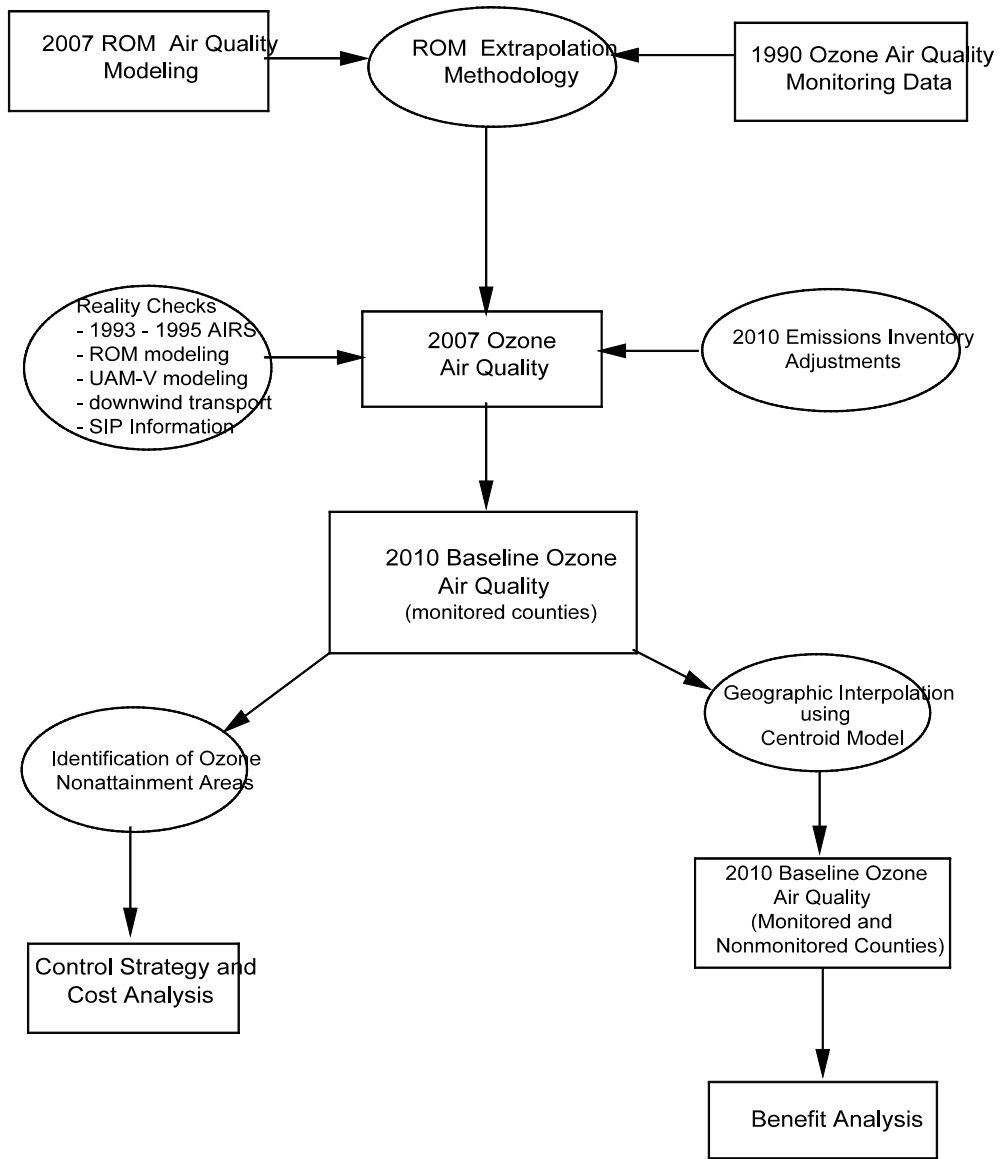
A dataset of empirical ozone concentration data was developed from AIRS. Hourly ozone concentration values meeting data inclusion criteria are obtained for the year 1990 for the contiguous 48 states. Given that the baseline emissions inventory from which 2007 emissions projections were made and the basecase ROM modeling both used 1990 emissions, the year 1990 was selected as a representative baseline year for ambient air quality. This 1990 air quality data set was corrected for duplicate monitor site records, obsolescence of monitor data and missing values (MathTech, 1997).

Although the form of the proposed standard is expressed as the average 3rd max concentration over a three year period and this analysis uses only one year of ozone monitoring data, an examination of the data shows that at the national level, 1990 compares well with the 1993 - 1995 period. An evaluation of 1990 annual 3rd max 8-hour design values relative to 1993



- 1995 average 3rd max design values was conducted. This assessment indicates that across the U.S. the difference between the 1990 annual 3rd maximum concentrations and the 1993-1995 average annual 3rd maximum concentrations is less than or equal to 0.015 ppm 90 percent of the time (U.S. EPA, 1997e). In spite of area-specific differences in 8-hour 3rd maximum values between 1990 and 1993-1995, ozone air quality data for the year 1990 is considered to be comparable at the national level to 8-hour average 3rd max design values for 1993-1995.

Figure 4.4. Development of 2010 Baseline Ozone Air Quality



#### 4.5.2.3 Temporal Extrapolation of 1990 Monitored Air Quality to 2007/2010 based on Regional Oxidant Model Results

This analysis next develops equations to predict expected ozone concentration values for the year 2007 for monitored counties based on available air quality modeling conducted under alternative future year emissions assumptions. The temporal extrapolations lead to future year baseline ozone concentrations that incorporate anticipated air quality improvements due to implementation of current CAA requirements. The 2007 air quality predictions for monitored counties are used to identify and define nonattainment areas for the control strategy and cost analysis. The year 2007 was used as the year of analysis for the December 1996 assessment (U.S. EPA, 1996g). The current analysis examines the ozone nonattainment situation in the year 2010. Adjustments are made to the 2007 nonattainment area air quality to account for the different analytical year. The adjustment methodology is discussed at the end of this section.

ROM ozone air quality predictions for the 1990 basecase, 2007 CAA-mandated control, and 2007 regional control strategy scenarios are used in two regression equations to determine the statistical relationships between 1990 and 2007 ozone air quality concentrations under two alternative emission scenarios. The first emissions scenario, the 2007 CAA control scenario, simulates the net effects of growth and application of control measures currently required by the CAA on ozone concentrations in 2007 throughout the modeling domain. The second scenario, the 2007 regional control strategy, augments the CAA control scenario with application of a NO<sub>x</sub> cap limiting emissions from utility boilers and other boilers  $\geq$  250 MW to 0.15 lb/MMBtu and initiation of a national low emission vehicle (NLEV) requirement beginning with the 1999 model year to the entire modeling domain. It should be noted that biogenic VOC emissions are modeled for all scenarios as an uncontrollable component of VOC emissions. Thus, biogenic VOCs are factored into the responsiveness of simulated ozone concentrations to changes in anthropogenic ozone precursors.

The equations used to predict average expected changes in ozone concentrations between 1990 and 2007 are generated through Ordinary Least Squares (OLS) regression of 1990 ROM

basecase ozone concentration predictions and a number of explanatory variables against 2007 ROM predictions for the two emissions scenarios (MathTech, 1997). As noted earlier, ROM air quality results are available for the Eastern U.S. However, air quality concentrations are needed for the entire country to assess national benefits and costs. Through the inclusion of other explanatory variables, the regression equations control for factors that may differ between the east and west and could therefore explain variations in concentration values between 1990 and 2007. The specifics of these regression equations are outlined in Appendix A.5.

The results for the CAA-control scenario indicate that, all else equal, 2007 hourly ozone concentrations can be expected to decrease relative to 1990 hourly ozone concentrations. The results for the regional control strategy scenario suggest that, all else equal, 2007 hourly ozone concentrations also can be expected to decrease relative to 1990 hourly ozone concentrations. Evaluating the mean 1-hour ozone concentration in the regression function indicates that the regional control strategy results in a 7% decrease in mean hourly ozone concentrations relative to 1990 mean air quality concentrations. In comparison, the CAA-control scenario results in a 3% decrease in mean hourly ozone concentrations relative to the 1990 modeled predictions (MathTech, 1997).

The regression analyses show that the projected concentration values for 2007 are primarily affected by 1990 concentration values. Given that the goal of this method is to create a dataset of predicted hourly ozone data for each monitor in 2007, the results of the regression analysis are next applied to each monitor by multiplying each hourly 1990 monitored value by the appropriate coefficient. Adjustments are made at the county level by computing the quantitative impacts of the remaining terms in the regression equation. For the East, the results under the 2007 regional control strategy are applied given that the NO<sub>x</sub> cap and NLEV are assumed to be in place by 2010 in the emissions baseline for the current analysis. For the Western U.S., the results of the regression analysis for the 2007 CAA-control scenario are applied since a comparable NO<sub>x</sub> cap is not assumed in the 2010 emissions baseline for these areas. The NO<sub>x</sub> cap constitutes the bulk of the NO<sub>x</sub> emissions under the regional control scenario as the NLEV program in the Eastern 37 states is assumed to be fully implemented

sometime beyond 2010. Thus application of the 2007 CAA-control case for the West is more appropriate.

#### **4.5.2.4 Identification of Ozone Nonattainment Areas**

The predicted 2007 air quality for counties with ozone monitoring data in 1990 is next used to determine nonattainment status of individual counties. The air quality distribution for each monitored county is reviewed to identify the concentration value that triggers nonattainment for a specific form of the ozone standard. For example, the 3rd highest daily maximum 8-hour value is identified to determine whether that value exceeds an alternative standard level. This value for each monitor is defined as the “standard measure” for that monitor and standard. The highest standard measure for each standard alternative among the monitors in a given county is used as the county “design value”. The design value location (i.e., monitor) may vary from one standard alternative to another in a given county. The design value for each county is evaluated against the concentration level triggering nonattainment to identify counties that do not meet each standard alternative. The rounding convention associated with the proposed ozone standard is factored into the concentration level triggering noncompliance for each standard alternative.<sup>1</sup>

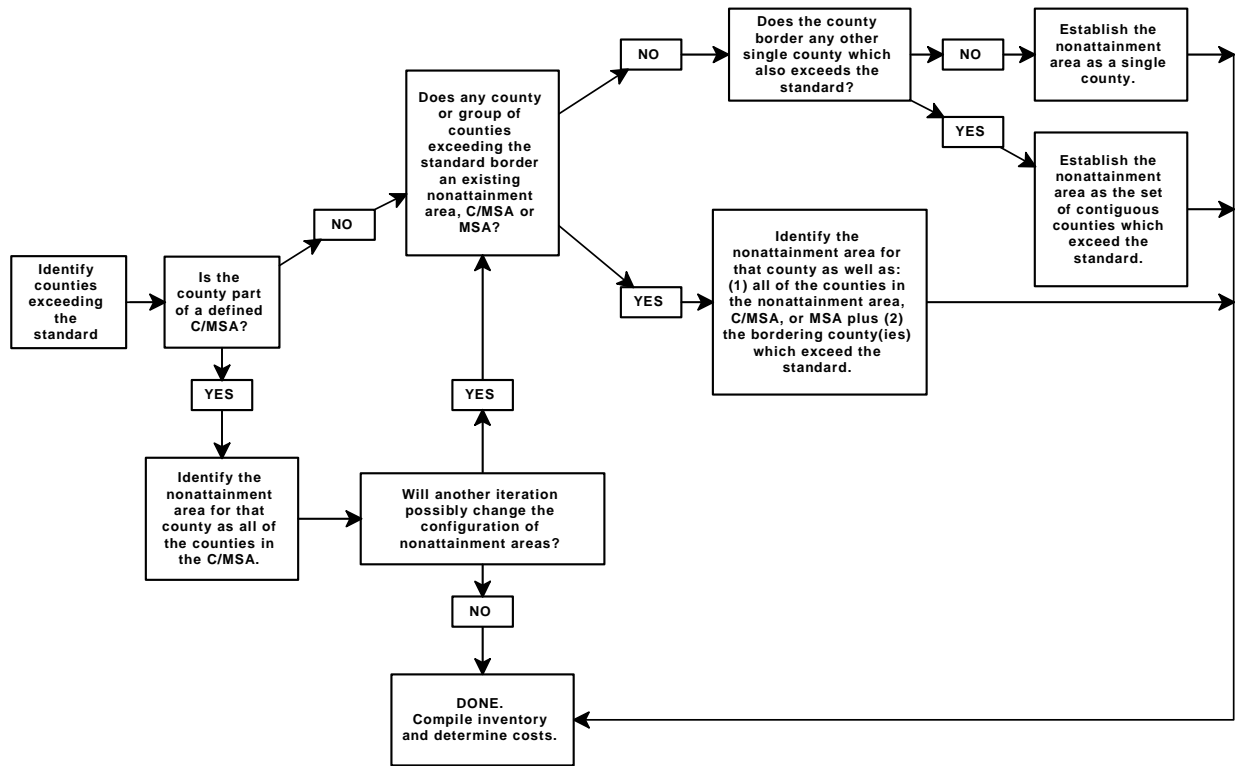
A series of steps are then followed to define nonattainment areas for each standard alternative based on the county design value. Nonattainment areas may be a single county or a group of counties in a Consolidated Metropolitan Statistical Area or Metropolitan Statistical Area (C/MSA). The general principle used here in identification of nonattainment areas is that if the air quality of an area violates the ozone standard or if sources in that area contribute to violations in a nearby area, the area is considered nonattainment. This is not to prejudge how States may make future decisions in implementing the new standards. Ozone monitors generally are placed in areas with a high probability of recording standard violations, typically in counties

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<sup>1</sup> The Federal Register Notice for the proposed ozone NAAQS (FR Vol. 61, No. 241, December 13, 1996) states that the rounding convention associated with the proposed standard is to round to the nearest 0.001 ppm. The current rounding convention is to round up digits equal to or greater than 5.

downwind of urban areas. Therefore, when these counties record violations of the standard, the upwind area(s) contributing the emissions should be included in the nonattainment area definition. The following schematic in Figure 4.5 describes the decision rules used to identify and define nonattainment areas (U.S. EPA, 1997d).

**Figure 4.5 Process for Identification and Definition of Ozone Nonattainment Areas**



**4.5.2.5 Adjustments to 2007 Ozone Air Quality to Develop 2010 Air Quality**

2010 Emissions Inventory Adjustments

As noted previously, the current analysis examines ozone air quality concentrations in the year 2010 assuming implementation of CAA-mandated controls and a regional control strategy in the East and CAA-mandated controls in the West. Because new air quality modeling is not conducted for this alternative analytical year, a method is developed to adjust 2007 baseline air quality in Eastern nonattainment areas to reflect changes in emissions between 2007 and 2010.

This adjustment is performed by comparing the 2010 NO<sub>x</sub> and VOC emissions for each

nonattainment area to the 2007 ROM-predicted NOx and VOC emission reduction targets (U.S. EPA, 1997f) needed for attainment of the most stringent ozone alternative considered in the analysis, the .08 ppm/8 hour/3rd max concentration. The change in VOC and/NOx emissions between 2007 and 2010 are counted towards achievement of the VOC and NOx emission reduction targets for the .08 ppm/8 hour/3rd max estimated for a given area as described below (U.S. EPA, 1997g):

$$\text{Air Quality Adjustment} = \frac{(\text{NOx}_{2007} - \text{NOx}_{2010}) + (\text{VOC}_{2007} - \text{VOC}_{2010})}{\text{NOx emission reduction target}_{2007} + \text{VOC emission reduction target}_{2007}}$$

- where:
- NOx<sub>2007</sub> = NOx emissions in 2007 (tpd)
  - NOx<sub>2010</sub> = NOx emissions in 2010 (tpd)
  - VOC<sub>2007</sub> = VOC emissions in 2007 (tpd)
  - VOC<sub>2010</sub> = VOC emissions in 2010 (tpd)
  - NOx/VOC emission reduction target = amount of NOx or VOC that needs to be reduced to achieve attainment in 2007 (tpd)

The air quality adjustment is then applied to the nonattainment area design value monitor for the 0.08 ppm/8 hour/3rd max alternative to determine the percent rollback to be applied at all monitors in the nonattainment area. This alternative is used because it is the most stringent of the ozone alternatives analyzed and adjustment to air quality for this alternative only preserves a consistent air quality distribution across all alternatives.

Once the necessary baseline air quality adjustments have been made to capture nonattainment area emission inventory differences between 2007 and 2010, the development of air quality values in monitored counties for 2010 is complete. Thus, for the remaining discussion, the baseline air quality is referred to as 2010 baseline ozone air quality. This adjusted air quality is used to evaluate for a second time nonattainment status of monitored



counties.

### Additional Information Applied to 2010 Baseline Ozone Air Quality

Given uncertainties in the method for predicting future year ozone concentrations, additional ozone air quality monitoring and modeling information is utilized to better characterize the 2010 ozone nonattainment picture.

#### *1993 - 1995 Ambient Ozone Air Quality Monitoring Data*

Ozone design values based on 1993 - 1995 AIRS data corresponding to the appropriate standard form are compared to the model-predicted design values. Those areas for which the 1993 - 1995 ozone design value is less than or equal to the level specified in the standard alternative are considered attainment for that alternative (U.S. EPA, 1997g). It is assumed for these areas that CAA-mandated controls will be sufficient to attain the specific standard alternative. There are three areas excluded from the current standard analyses based on this comparison. There are no areas excluded for the selected standard (0.08/4th max) analysis based on this comparison.

#### *ROM and UAM-V Air Quality Modeling Results*

Air quality modeling results for comparable emission reduction scenarios from ROM and UAM-V modeling for the Ozone Transport Assessment Group (OTAG) are also examined (U.S. EPA, 1997g). Those areas for which air quality modeling predicts will be in attainment of alternative standards in 2007 are considered attainment for this analysis based on this comparison. There are three areas excluded from the current standard analysis and 4 areas excluded from the selected standard (0.08/4th max) analysis based on this comparison.

#### *Downwind Nonattainment Areas Identified through Air Quality Modeling*

A number of small, individual counties are predicted to violate alternative standards in 2010. Upon closer inspection and examination of available ROM air quality results, these counties are determined to be downwind transport areas. Many of these counties are rural counties that have low NO<sub>x</sub> and VOC emissions. The predicted violations are a result of upwind contributions of ozone precursor emissions. Thus for control strategy and cost analysis purposes, control measures are not applied in these counties as it is assumed that upwind NO<sub>x</sub> and VOC reductions will mitigate the ozone problem in these downwind areas. For the benefit analysis, these downwind transport areas are assumed to be “attached” to the associated upwind nonattainment area. Thus, when calculating partial attainment air quality, air quality rollbacks for the upwind nonattainment area are applied to the downwind transport area in order to capture the air quality impacts of the upwind controls (U.S. EPA, 1997f).

#### **4.5.3 Ozone Nonattainment Areas by Alternative**

The model-predicted ozone air quality data for the 2010 CAA-control baseline is used to determine county air quality status. Nonattainment areas are identified from the set of counties monitored for ozone in 1990 as described in Section 4.5.2.4. Areas predicted to be in initial nonattainment of alternative ozone standards in 2010 for the East (i.e., 37 eastern States) versus the West are listed in Table 4.11. These are projections based on estimates and assumptions. Ultimate nonattainment area designations will be based on actual ambient monitoring data.

**Table 4.11 Predicted Nonattainment Areas for Alternative Ozone Standards in 2010**

Region	Number of Areas Predicted in Nonattainment of Ozone Alternatives			
	0.12ppm/1xx (current)	0.08ppm/3rd max (proposed)	0.08/4th max (selected)	0.08/5th max
East	5	20	12	8
West	4	8	7	7
<b>TOTAL</b>	<b>9</b>	<b>28</b>	<b>19</b>	<b>15</b>

#### 4.5.4 Geographic Interpolation of Baseline Ozone Air Quality Using the Centroid Model for Ozone Standard Benefit Analysis

In order to assess national ozone benefits of implementation of alternative ozone standards, hourly ozone concentrations across the entire country are required. Because there are counties both within and outside of nonattainment areas for which no monitoring data exists, the centroid model is used to predict hourly ozone air quality concentrations in those nonmonitored counties. Additionally, given that some counties may have more than one monitor, the centroid approach can be used to assign a single hourly value to the monitored county centroid for each hour throughout the year. The centroid model is an interpolation method which permits data from monitors proximate to a county centroid to be interpolated to the centroid location (MathTech, 1997). This analysis uses the geographic centroid available from the Bureau of the Census (BOC, 1992) rather than the population centroid. “Proxy” monitors are assumed to be located at each geographic county centroid throughout the U.S. The centroid model is used to calculate hourly ozone concentrations for each of the “proxy” monitors for 2010.

#### 4.5.5 Key Uncertainties Associated with 2010 Baseline Ozone Air Quality

There are many potential sources of uncertainty in the development of 2010 baseline ozone air quality. Although it is not possible to quantify the magnitude of the uncertainty, a qualitative discussion of uncertainties can be provided. In general, we believe that the national baseline ozone air quality results on net are not biased in either direction. Underestimates for individual nonattainment areas are balanced out by overestimates for other nonattainment areas.

Given that the methodologies used in the assessment of national costs and benefits are not sufficient to capture unique characteristics of each individual nonattainment area, area-specific baseline air quality results have a higher probability of bias (U.S. EPA, 1997l).

Table 4.12 presents potential sources of uncertainty and associated biases in estimating national 2010 baseline air quality. As described in Sections 4.3.3 and 4.3.8, there are uncertainties related to development of the 1990 emissions inventory and projection of those emissions to 2010. Biogenic VOCs may be underestimated relative to the more recent BEIS2 estimates, but potential biases in anthropogenic NO<sub>x</sub> and VOC emissions are unknown.

The emissions projections are input to the ROM model to produce future year ozone air quality predictions. The ROM is a regional-scale air quality model that is used in this analysis to estimate area-specific air quality. By definition, urban-scale characteristics of individual nonattainment areas are not captured in the ROM modeling. This approach increases the level of uncertainty in the national analysis and may produce positive or negative bias for any specific area. However, it is unclear if there is an overall bias in air quality at the national level.

Additionally, the ROM modeling relied on 1987 meteorology. Despite geographic variability in the severity of the meteorological data, overall, 1987 is not an unusually conducive year for high ozone for the period 1986 - 1995. Although there is uncertainty in predicting future meteorological conditions, reliance on 1987 data is not believed to bias the ozone air quality estimates. Finally, evaluation of ROM modeling has indicated that ROM 1990 base case predictions are higher relative to ozone monitoring data for some locations (U.S. EPA, 1996b). However, it is unknown whether or not ROM overpredicts for the future year scenarios. Because the prediction of future year air quality through the ROM extrapolation approach is primarily driven by the 1990 ozone concentration values, it is unclear if 2010 baseline ozone air quality is biased given ROM overprediction in the 1990 base case.

There are a number of potential sources of uncertainty associated with the ROM extrapolation methodology. As discussed previously, the extrapolation method is used to

develop air quality data for areas not covered by available air quality modeling. The extrapolation method employs one year of data for the year 1990. As discussed in section 4.5.2.2, there appear to be no biases introduced to the ozone analysis from these two potential sources of uncertainty. There is no reason to believe that the regression equation used to factor in growth and emissions control between the base case and projection years is biased (MathTech, 1997). Because of the lack of air quality modeling, extrapolation of air quality modeling results from the East to the West is necessary. This clearly brings uncertainty to the baseline ozone air quality concentrations for the West, although as discussed in Section 4.5.2.3, this method is largely driven by base case 1990 ambient monitoring values. It is unclear if there is any bias to this extrapolation procedure.

An air quality adjustment procedure is used to account for CAA-control emissions inventory changes between 2007 and 2010. 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. Thus there may be a small overestimate of ozone air quality in nonmonitored counties outside of nonattainment areas.

**Table 4.12 Uncertainties and Possible Biases in Estimating National  
2010 Baseline Ozone Air Quality**

Potential Source of Uncertainty	Positive Bias? (Overestimate)	Negative Bias? (Underestimate)	Bias Unclear
Development of 1990 emissions inventories and 2010 projections		✓ (biogenic VOC)	✓
ROM Modeling - Use 1987 meteorology - Use of regional model to estimate city-specific air quality - ROM tendency to overpredict			✓ ✓ ✓
ROM Extrapolation Methodology - Use 1 year of monitoring data - Use 1990 monitored air quality data - Regression - ROM extrapolation from East to West			✓ ✓ ✓ ✓
Emissions Inventory Adjustments	✓ (small)		
Centroid Model to predict ozone concentrations in nonmonitored counties	✓ (small)		

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