Chapter 4: Air Quality Impacts

Chapter Synopsis

This chapter details the three-step process we employed to estimate the air quality impacts of our emission control strategies. First we used the Community-Scale Air Quality (CMAQ) model to estimate the reductions in ambient concentration of PM2.5 resulting from our illustrative attainment strategy. Next, where our modeled attainment strategy did not result in attainment with the revised daily standard of 35μg/m^3 or the alternative more stringent annual standard of 14 μg/m^3 we conducted a supplemental control analysis for particular areas by examining additional emission controls on carbonaceous particles. As a final step, we made a final determination of attainment and non-attainment among those areas which were not able to attain the revised or alternative more stringent standard after applying additional controls on carbonaceous particles. For these areas we analyzed the CMAQ-projected design values within the context of the available empirical modeling and monitoring data to determine whether these areas attained the standard for the purposes of this analysis. Finally, in areas determined to be non-attainment after our full modeled and empirical assessments, we discuss how air quality might be affected by full attainment.

4.1 Modeled $PM_{2.5}$ Air Quality Estimates

4.1.1 Air Quality Modeling Overview

A national scale air quality modeling analysis was performed to estimate future year annual and daily $PM_{2.5}$ concentrations as well as visibility degradation (i.e., regional haze). These projections were used as inputs to the calculation of expected benefits from the alternative NAAQS considered in this assessment. The 2001-based CMAQ modeling platform was used as the tool for air quality modeling of future baseline emissions and control scenarios designed to attain specific daily and annual standards. In addition to the CMAQ model, the modeling platform includes the emissions, meteorology, and initial and boundary condition data which are inputs to this model. The CMAQ model is a three-dimensional grid-based Eulerian air quality model designed to estimate the formation and fate of oxidant precursors, primary and secondary particulate matter concentrations and deposition over regional and urban spatial scales (e.g., over the contiguous U.S.) (EPA, 1999; Byun and Schere, 2006; Dennis et al., 1996). Consideration of the different processes (e.g. transport and deposition) that affect primary (directly emitted) and secondary (formed by atmospheric processes) PM at the regional scale in different locations is fundamental to understanding and assessing the effects of pollution control measures that affect PM, ozone and deposition of pollutants to the surface.

The CMAQ model was peer-reviewed in 2003 for EPA as reported in "*Peer Review of CMAQ Model*" (Amar et al., 2004). The latest version of CMAQ (Version 4.5) was employed for this PM NAAQS RIA modeling analysis. This version reflects updates in a number of areas to improve the underlying science and address comments from the peer-review including (1) use of a state-of-the-science inorganic nitrate partitioning module (ISORROPIA) and updated gaseous, heterogeneous chemistry in the calculation of nitrate formation, (2) a state-of-the-science

secondary organic aerosol (SOA) module that includes a more comprehensive gas-particle partitioning algorithm from both anthropogenic and biogenic SOA, (3) an in-cloud sulfate chemistry module that accounts for the nonlinear sensitivity of sulfate formation to varying pH, and (4) an updated CB-IV gas-phase chemistry mechanism and aqueous chemistry mechanism that provide a comprehensive simulation of aerosol precursor oxidants.[1]

4.1.2 Model Domain and Configuration

As shown in Figure 4-1, the CMAQ modeling domain encompasses all of the lower 48 States and portions of Canada and Mexico (Figure 4-1). The domain extends from 126 degrees to 66 degrees west longitude and from 24 degrees north latitude to 52 degrees north latitude. The horizontal grid cells are approximately 36 km by 36 km. The modeling domain contains 14 vertical layers with the top of the modeling domain at about 16,200 meters, or 100 mb.

Figure 4-1. Map of the CMAQ Modeling Domain Used for PM NAAQS RIA.

[1] Please see the Community Modeling and Analysis System (CMAS) Center Web site for complete details on CMAQ version 4.5: http://www.cmascenter.org/

4.1.3 Model Inputs

The key inputs to the CMAQ model include emissions from anthropogenic and biogenic sources, meteorological data, and initial and boundary conditions. The CMAQ meteorological input files were derived from a simulation of the Pennsylvania State University / National Center for Atmospheric Research Mesoscale Model (Grell, Dudhia, and Stauffer, 1994) for the entire year of 2001. This model, commonly referred to as MM5, is a limited-area, nonhydrostatic, terrain-following system that solves for the full set of physical and thermodynamic equations which govern atmospheric motions. For this analysis, version 3.6.1 of MM5 was used. The horizontal domain consisted of a single 36 x 36 km grid with 165 by 129 cells, selected to cover the CMAQ modeling domain with some buffer to avoid boundary effects. The meteorological outputs from MM5 were processed to create model-ready inputs for CMAQ using the Meteorology-Chemistry Interface Processor (MCIP) version 3.1: horizontal wind components (i.e., speed and direction), temperature, moisture, vertical diffusion rates, and rainfall rates for each grid cell in each vertical layer (EPA, 1999).

The lateral boundary and initial species concentrations were obtained from a three-dimensional global atmospheric chemistry model, the GEOS-CHEM model (Yantosca, 2004). The global GEOS-CHEM model simulates atmospheric chemical and physical processes driven by assimilated meteorological observations from the NASA's Goddard Earth Observing System (GEOS). This model was run for 2001 with a grid resolution of 2 degree x 2.5 degree (latitude-longitude) and 20 vertical layers. The predictions were used to provide one-way dynamic boundary conditions at 3-hour intervals and the initial concentration field for the CMAQ simulations.

A complete description of the development and processing of model-ready meteorological inputs and initial and boundary condition inputs used for this analysis are discussed in the CAIR TSD (EPA, 2005). In addition, the development of the gridded, hourly model-ready emissions inputs used for the 2001 base year and each of the future year base cases and control scenarios are summarized below in this chapter.

4.1.4 Evaluation of Air Quality Modeling System

EPA performed an extensive evaluation of our CMAQ air quality modeling system as part of the support analyses for CAIR[2]. This evaluation has been updated to consider model performance using the revised base year emissions inventories, as described above in Chapter 2. The updated operational model performance evaluation for $PM_{2.5}$ and its related speciated components (e.g., sulfate, nitrate, elemental carbon, organic carbon, etc.) was conducted using the 2001 data in order to estimate the ability of the modeling system to replicate base year concentrations. The details of the $PM_{2.5}$ performance evaluation are provided in Appendix O. In summary, model performance statistics were calculated for pairs of observed/predicted concentrations. Statistics were generated for the following geographic groupings: (1) the entire modeling domain, (2) the Eastern U.S. and (3) the Western U.S. As in the evaluation for CAIR modeling, the "acceptability" of model performance for the PM NAAQS modeling was judged by comparing our results to those found in recent regional $PM_{2.5}$ model applications for other, non-EPA

[2] CMAQ Model Evaluation Report, March 2005 (CAIR Docket OAR-2005-00532149).

studies[3]. As decribed in Appendix X,, overall, the performance for this application is within the range or better than these other applications.

4.1.5 Model Simulation Scenarios

As part of our analysis the CMAQ modeling system was used to calculate daily and annual $PM_{2.5}$ concentrations and visibility estimates for each of the following seven emissions scenarios:

- 2001 base year
- 2015 base case projection with CAIR/CAMR/CAVR
- 2015 15/65 (projection to 2015 with controls estimated to attain an annual standard of 15 $\mu g/m^3$ and daily standard of 65 $\mu g/m^3$)
- 2020 base case projection with CAIR/CAMR/CAVR
- 2020 15/65 (projection to 2020 with controls estimated to attain an annual standard of 15 $\mu g/m^3$ and daily standard of 65 $\mu g/m^3$)
- 2020 15/35 (projection to 2020 with controls to estimated to attain an annual standard of 15 $\mu g/m^3$ and daily standard of 35 $\mu g/m^3$)
- 2020 14/35 (projection to with controls estimated to attain an annual standard of 14 $\mu g/m^3$ and daily standard of 35 $\mu g/m^3$)

Note that the 2020 15/65 scenario is the future baseline used for evaluating the benefits of the 15/35 and 14/35 alternative NAAQS. The growth assumptions and emissions controls for each of these scenarios are described elsewhere in the RIA.

We use the predictions from the model in a relative sense by combining the 2001 base-year predictions with predictions from each future-year scenario and speciated ambient air quality observations to determine $PM_{2.5}$ concentrations and visibility for each of the 2015 and 2020 scenarios. After completing this process, we then calculated daily and seasonal PM air quality metrics as inputs to the health and welfare impact functions of the benefits analysis. The following sections provide a more detailed discussion of our air quality projection method and a summary of the results.

4.1.6 Projection Methods for Air Quality Concentrations

To forecast future year annual average and daily 98th percentile $PM_{2.5}$ concentrations we used air quality modeling results from the $PM_{2.5}$ NAAQS CMAQ model runs.

In general, the procedures for projecting both the annual and daily $PM_{2.5}$ design values are based on utilization of model predictions in a relative sense. In this manner, the 2001 base year model predictions and the 2015 (or 2020) future-year model predictions are coupled with ambient data to forecast future concentrations. This approach is consistent with the EPA draft guidance document for modeling $PM_{2.5}$ (EPA, 2001).

[3] These other modeling studies represent a wide range of modeling analyses which cover various models, model configurations, domains, years and/or episodes, chemical mechanisms, and aerosol modules.

Projection Methodology for Annual Average Design Values

The procedures used to project the annual design values are generally consistent with the projection techniques used in the CAIR. The projected annual design values were calculated using the Speciated Modeled Attainment Test (SMAT) approach. This approach is used to ensure that the $PM_{2.5}$ concentrations are closely related to the observed ambient data. The SMAT procedure combines absolute concentrations of ambient data with the relative change in PM species from the model.

The SMAT uses a Federal Reference Method (FRM) mass construction methodology that results in reduced nitrates (relative to the amount measured by routine speciation networks), higher mass associated with sulfates (reflecting water included in FRM measurements), and a measure of organic carbonaceous mass that is derived from the difference between measured $PM_{2.5}$ and its noncarbon components. This characterization of $PM_{2.5}$ mass also reflects crustal material and other minor constituents. The resulting characterization provides a complete mass balance. It does not have any unknown mass that is sometimes presented as the difference between measured $PM_{2.5}$ mass and the characterized chemical components derived from routine speciation measurements. However, the assumption that all mass difference is organic carbon has not been validated in many areas of the US. The SMAT methodology uses the following $PM_{2.5}$ species components: sulfates, nitrates, ammonium, organic carbon mass, elemental carbon, crustal, water, and blank mass (a fixed value of $0.5ug/m^3$).

More complete details of the SMAT procedures used in the CAIR analysis can be found in the report "Procedures for Estimating Future $PM_{2.5}$ Values for the CAIR Final Rule by Application of the (Revised) Speciated Modeled Attainment Test (SMAT)" (EPA, 2004). For the PM NAAQS analysis, several datasets and techniques were updated. The changes and updates include:

1. Revised database of $PM_{2.5}$ speciation data which includes data from 2002 and 2003.
2. Revised interpolations of $PM_{2.5}$ species data using updated techniques.
3. An updated equation to calculate particle bound water.
4. Revised treatment of ambient ammonium data.

Documentation of these updates and changes can be found in (EPA, 2006).

Below are the steps we followed for projecting future $PM_{2.5}$ concentrations. These steps were performed to estimate future case concentrations at each FRM monitoring site. The starting point for these projections is a 5 year weighted average design value for each site. The weighted average is calculated as the average of the 1999–2001, 2000–2002, and 2001–2003 design values at each monitoring site. By averaging 1999–2001, 2000–2002, and 2001–2003, the value from 2001 is weighted three times, whereas, values for 2000 and 2002 are each weighted twice, and 1999 and 2003 are each weighted once. This approach has the desired benefits of (1) weighting the $PM_{2.5}$ values towards the middle year of the five-year period (2001), which is the base year for our emissions projections, and (2) smoothing out the effects of year-to-year variability in emissions and meteorology that occurs over the full five-year period. This approach provides a robust estimate of current air quality for use as a basis for future year projections.

Step 1: Calculate quarterly mean ambient concentrations for each of the major components of $PM_{2.5}$ (i.e., sulfate, nitrate, ammonium, elemental carbon, organic carbon, water, and crustal material) using the component species concentrations estimated for each FRM site.

The component species concentrations were estimated using an average of 2002 and 2003 ambient data from speciation monitors. The speciation data was interpolated to provide estimates for all FRM sites across the country. The interpolated component concentration information was used to calculate species fractions at each FRM site. The estimated fractional composition of each species (by quarter) was then multiplied by the 5 year weighted average 1999–2003 FRM quarterly mean concentrations at each site (e.g., 20% sulfate multiplied by 15.0 $\mu g/m^3$ of $PM_{2.5}$ equals 3 $\mu g/m^3$ sulfate). The end result is a quarterly concentration for each of the $PM_{2.5}$ species at each FRM site.

Step 2: Calculate quarterly average Relative Reduction Factors (RRFs) for sulfate, nitrate, elemental carbon, organic carbon, and crustal material. The species-specific RRFs for the location of each FRM are the ratio of the 2015 (or 2020) future year cases to the 2001 base year quarterly average model predicted species concentrations. The species-specific quarterly RRFs are then multiplied by the corresponding 1999–2003 quarterly species concentration from Step 1. The result is the future case quarterly average concentration for each of these species for each future year model run.

Step 3: Calculate future case quarterly average concentrations for ammonium and particle-bound water. The future case concentrations for ammonium are calculated using the future case sulfate and nitrate concentrations determined from Step 2 along with the degree of neutralization of sulfate (held constant from the base year). Concentrations of particle-bound water are calculated using an empirical equation derived from the AIM model using the concentrations of sulfate, nitrate, and ammonium as inputs.

Step 4: Calculate the mean of the four quarterly average future case concentrations to estimate future annual average concentration for each component species. The annual average concentrations of the components are added together to obtain the future annual average concentration for $PM_{2.5}$.

Step 5: For counties with only one monitoring site, the projected value at that site is the future case value for that county. For counties with more than one monitor, the highest future year value in the county is selected as the concentration for that county.

Change in Annual Average $PM_{2.5}$ for the Benefits Calculations

For the purposes of projecting future $PM_{2.5}$ concentrations for input to the benefits calculations, we applied the SMAT procedure using the base-year 2001 modeling scenario and each of the future-year scenarios. In our application of SMAT we used temporally scaled speciated $PM_{2.5}$ monitor data from 2002 as the set of base-year measured concentrations. Temporal scaling is based on ratios of model-predicted future case $PM_{2.5}$ species concentrations to the corresponding

model-predicted 2001 concentrations.[4] Output files from this process include both quarterly and annual mean $PM_{2.5}$ mass concentrations.

The SMAT procedures for calculating PM benefits are the same as documented above for projecting future nonattainment counties for the annual NAAQS with the following exceptions:

1. The benefits analysis uses interpolated $PM_{2.5}$ data[5] (FRM and IMPROVE) that cover all of the grid cells in the modeling domain (covering the entire country), whereas the nonattainment analysis is performed at each ambient monitoring site using measured FRM $PM_{2.5}$ data (only the species data are interpolated).
2. The benefits analysis is anchored by the interpolated $PM_{2.5}$ data from the single year of 2002, whereas the nonattainment analysis uses a 5-year weighted average (1999–2003) of $PM_{2.5}$ design values at each monitoring site.

Projection Methodology for 24-Hour Average Design Values

The daily design values are based on applying a projection method similar to that used for annual design values. Monitoring data for the years 1999 to 2003 are used as the basis for the projection of daily design values. Since the 24-hour NAAQS is based on annual 98th percentile values, we want to use ambient data and model data that represent the high concentrations at each site. As such, we have focused the 24-hour analysis on ambient data from the highest 25% of measured days[6] (by $PM_{2.5}$ concentration) in each quarter at each site. We are also deriving the modeled RRFs from the top 25% of modeled days for each quarter.

There are several steps in the projection for 24-hour concentrations for each of the base years of monitoring data:

Step 1: The first step in projecting the daily design value is to identify the maximum daily average $PM_{2.5}$ concentration in each quarter that is less than or equal to the annual 98th percentile value over the entire year. This results in data for each year (1999–2003) for each site which contains one quarter with the 98th percentile value and three quarters with the maximum values from each quarter which are less than or equal to the 98th percentile value.

Step 2: These quarterly $PM_{2.5}$ concentrations are then separated into their component species by multiplying the quarterly maximum daily concentration at each site by the estimated fractional composition of $PM_{2.5}$ species, by quarter, based on the observed species fractions for the top 25% days from speciation monitors in 2002 and 2003 (using the same methodology as the quarterly average fractional species data used in the annual average calculations from above).

[4] Monitoring data from 2002 was used to develop the species specific information because there was not sufficient PM2.5 speciation data for 2001 or previous years.

[5] Interpolation of the PM2.5 data is necessary for the benefits analysis because PM2.5 concentrations are needed for every grid cell. But for the design value calculations at the monitoring sites, interpolation of the measured PM2.5 is not needed.

[6] Many of the monitoring sites have a relatively infrequent measurement cycle (once every 6 days). Therefore, the top 25% of measured days from each quarter for those sites is ~3days. We believe that this is consistent with the high end of the distribution of days that represent the 98th percentile concentrations. Sites with more frequent measurement schedules will have more days in the mean top 25% of days.

Step 3: The component species are then projected by multiplying each species concentration by the quarterly relative reduction factors for each species derived from the 2015 (or 2020) and 2001 $PM_{2.5}$ air quality modeling (using quarterly RRFs derived from the top 25% modeled days in each quarter). The methodology is the same as used in the annual average calculations.

Step 4: The projected species components are then summed to obtain a $PM_{2.5}$ concentration for each quarter that represents a potential daily design value. This procedure is repeated for each of the years of monitoring data (1999–2003). The highest daily value for each year at each monitor is considered to be the estimated 98^{th} percentile value for that year.

Step 5: The estimated 98^{th} percentile values for each of the 5 years are averaged over 3 year intervals (1999–2001, 2000–2002, 2001–2003), and then averaged over the three interval averages. This creates a 5 year weighted average for each monitor. The projected daily design value for a county is then calculated as the maximum 5 year weighted average design value across all monitors within a county.

Annual and daily average county level design values were then compared to the potential alternative annual and daily standards and mapped.

4.1.7 Air Quality Modeling Results for $PM_{2.5}$

Annual average and daily average 98^{th} percentile $PM_{2.5}$ concentrations were estimated for each FRM site by applying the SMAT techniques described above to the CMAQ-predicted $PM_{2.5}$ species concentrations for each scenario modeled (i.e., 2015 baseline, 2020 baseline, 2020 15/65, 2020 14/35, and 2020 15/35). The projected annual and daily $PM_{2.5}$ concentrations are provided in Appendix M for all counties with an FRM site included in this analysis. In Table 4-1 we provide the highest projected design values for the 2020 base case scenario. Note that this table and subsequent tables with projected annual and daily values for the other scenarios modeled contain data for those counties that exceed a 14 $\mu g/m^3$ annual or 35 $\mu g/m^3$ daily NAAQS. This covers the range of annual and daily values which are the subjects of this analysis. Again, the data for all counties for all scenarios are provided in Appendix M.

The projected base and control-case design values below represent the initial step in our attainment analysis. Section 4.2 below describes how we analyzed these design values in the context of other available empirical data to make a final determination of attainment and non-attainment for certain areas. Note that section 4.1.6 above describes the methodology we followed to derive the modeled base case and control case daily design values in the tables that follow.

Table 4-1. Projected Annual and Daily $PM_{2.5}$ Design Values ($\mu g/m^3$): 2020 Base Case

State	County	2020 Base Annual ($\mu g/m^3$)	2020 Base Daily ($\mu g/m^3$)
California	Riverside Co	27.5	73.9
California	San Bernardino Co	24.6	65.8
California	Los Angeles Co	23.9	62.7
California	Kern Co	20.8	77.9
California	Tulare Co	20.6	73.6
California	Orange Co	20.2	40.7
California	Fresno Co	19.6	70.4
Michigan	Wayne Co	17.3	39.0
California	Kings Co	16.8	67.6
California	Stanislaus Co	16.2	59.2
Pennsylvania	Allegheny Co	16.2	52.7
California	San Joaquin Co	16.0	52.0
Alabama	Jefferson Co	15.7	36.3
California	San Diego Co	15.7	40.1
California	Merced Co	15.6	53.1
Ohio	Scioto Co	15.4	33.8
Georgia	Fulton Co	15.3	31.5
Illinois	Cook Co	15.3	36.5
Ohio	Cuyahoga Co	15.2	39.7
Illinois	Madison Co	15.1	35.3
Montana	Lincoln Co	14.9	42.2
California	Imperial Co	14.8	44.9
Illinois	St. Clair Co	14.5	30.2
Ohio	Hamilton Co	14.1	33.6
California	Ventura Co	14.0	38.7
Ohio	Jefferson Co	14.0	33.8
Indiana	Lake Co	13.3	40.4
California	Alameda Co	13.2	58.7
California	Butte Co	13.0	48.6
Maryland	Baltimore City	12.9	35.2
Oregon	Lane Co	12.8	53.0
California	Contra Costa Co	12.5	61.1
Idaho	Shoshone Co	12.4	36.0
Utah	Cache Co	12.3	51.4
Utah	Salt Lake Co	12.2	47.6
California	Sacramento Co	12.1	48.3
Pennsylvania	York Co	12.1	35.5
California	Santa Clara Co	12.0	52.3
Pennsylvania	Berks Co	12.0	35.3
California	Solano Co	11.7	57.3
Washington	Pierce Co	11.6	44.9
California	San Francisco Co	11.4	52.4
Washington	Snohomish Co	11.4	40.5
California	Placer Co	11.2	36.5

State	County	2020 Base Annual ($\mu g/m^3$)	2020 Base Daily ($\mu g/m^3$)
California	Sutter Co	10.9	37.9
Oregon	Jackson Co	10.8	37.2
California	San Mateo Co	10.5	41.6
Idaho	Power Co	10.4	36.4
Oregon	Klamath Co	10.0	38.7
California	Sonoma Co	9.8	38.2
California	San Luis Obispo Co	9.4	35.6
Idaho	Bannock Co	9.1	40.0
Utah	Utah Co	9.1	35.3
Utah	Weber Co	8.9	35.3
Utah	Box Elder Co	8.5	38.4
California	Inyo Co	6.0	37.7

Modeling Attainment of Current 15/65 NAAQS

The projected 2015 base case $PM_{2.5}$ concentrations were used in the analysis to determine which locations are expected to remain nonattainment post-existing programs and therefore need additional local controls for attainment of the current 15/65 NAAQS. In brief, procedures for determining the additional "local" controls need for each area to attain include (1) application of the Response Surface Model to estimate the emissions reduction targets needed for attainment of 15/65 and (2) identification of specific controls which achieve the emissions reduction targets. These controls were applied to the 2020 base case to form the 2020 15/65 scenario. Details on these procedures are provided in Chapter 2. Table 4-2 shows the amount of reduction in $PM_{2.5}$ provided by the controls in the 2020 15/65 scenario, compared to the 2020 base case for those counties that exceed a 14 $\mu g/m^3$ annual or 35 $\mu g/m^3$ daily NAAQS.

Table 4-2. Modeled Impact of 15/65 Controls on Annual and Daily $PM_{2.5}$ Design Values ($\mu g/m^3$): 2020

		Annual			Daily		
State	**County**	*2020 Base ($\mu g/m^3$)*	*2020 15/65 ($\mu g/m^3$)*	*Impact of 15/65 controls in Annual DV ($\mu g/m^3$)*	*2020 Base ($\mu g/m^3$)*	*2020 15/65 ($\mu g/m^3$)*	*Impact of 15/65 controls on Daily DV ($\mu g/m^3$)*
California	Riverside Co	27.5	22.7	-4.8	73.9	63.2	-10.7
California	San Bernardino Co	24.6	21.4	-3.2	65.8	58.1	-7.7
California	Los Angeles Co	23.9	21.6	-2.3	62.7	58.1	-4.6
California	Kern Co	20.8	18.6	-2.2	77.9	68.0	-9.9
California	Tulare Co	20.6	18.9	-1.7	73.6	65.4	-8.2
California	Orange Co	20.2	18.2	-2.0	40.7	35.6	-5.1
California	Fresno Co	19.6	17.3	-2.3	70.4	59.6	-10.8
Michigan	Wayne Co	17.3	16.9	-0.4	39.0	38.4	-0.6
California	Kings Co	16.8	15.6	-1.2	67.6	61.0	-6.6
California	Stanislaus Co	16.2	14.5	-1.7	59.2	51.5	-7.7
Pennsylvania	Allegheny Co	16.2	15.8	-0.4	52.7	51.5	-1.2
California	San Joaquin Co	16.0	14.4	-1.6	52.0	45.3	-6.7
Alabama	Jefferson Co	15.7	15.1	-0.6	36.3	34.2	-2.1
California	San Diego Co	15.7	13.7	-2.0	40.1	34.6	-5.5
California	Merced Co	15.6	14.4	-1.2	53.1	47.7	-5.4
Ohio	Scioto Co	15.4	15.1	-0.3	33.8	33.3	-0.5
Georgia	Fulton Co	15.3	14.9	-0.4	31.5	30.7	-0.8
Illinois	Cook Co	15.3	14.5	-0.8	36.5	35.3	-1.2
Ohio	Cuyahoga Co	15.2	14.7	-0.5	39.7	39.1	-0.6
Illinois	Madison Co	15.1	14.6	-0.5	35.3	34.4	-0.9
Montana	Lincoln Co	14.9	14.8	-0.1	42.2	41.8	-0.4
California	Imperial Co	14.8	14.4	-0.4	44.9	43.0	-1.9
Illinois	St. Clair Co	14.5	14.1	-0.4	30.2	29.4	-0.8
Ohio	Hamilton Co	14.1	13.7	-0.4	33.6	33.0	-0.6
California	Ventura Co	14.0	12.0	-2.0	38.7	33.4	-5.3
Indiana	Lake Co	13.3	12.4	-0.9	40.4	36.9	-3.5
California	Alameda Co	13.2	11.7	-1.5	58.7	50.7	-8.0
California	Butte Co	13.0	12.7	-0.3	48.6	46.3	-2.3
Oregon	Lane Co	12.8	12.7	-0.1	53.0	52.5	-0.5
California	Contra Costa Co	12.5	11.1	-1.4	61.1	52.6	-8.5
Idaho	Shoshone Co	12.4	12.3	-0.1	36.0	35.9	-0.1
Utah	Cache Co	12.3	12.3	0.0	51.4	51.3	-0.1
Utah	Salt Lake Co	12.2	12.2	0.0	47.6	47.5	-0.1
California	Sacramento Co	12.1	10.9	-1.2	48.3	42.0	-6.3
Pennsylvania	York Co	12.1	12.0	-0.1	35.5	35.4	-0.1
California	Santa Clara Co	12.0	11.3	-0.7	52.3	48.2	-4.1
California	Solano Co	11.7	10.2	-1.5	57.3	48.3	-9.0
Washington	Pierce Co	11.6	11.5	-0.1	44.9	44.7	-0.2
California	San Francisco Co	11.4	9.6	-1.8	52.4	42.4	-10.0
Washington	Snohomish Co	11.4	11.4	0.0	40.5	40.2	-0.3
California	Placer Co	11.2	9.8	-1.4	36.5	30.6	-5.9
California	Sutter Co	10.9	10.5	-0.4	37.9	35.5	-2.4

		Annual			Daily		
State	County	*2020 Base ($\mu g/m^3$)*	*2020 15/65 ($\mu g/m^3$)*	*Impact of 15/65 controls in Annual DV ($\mu g/m^3$)*	*2020 Base ($\mu g/m^3$)*	*2020 15/65 ($\mu g/m^3$)*	*Impact of 15/65 controls on Daily DV ($\mu g/m^3$)*
Oregon	Jackson Co	10.8	10.8	0.0	37.2	37.1	-0.1
California	San Mateo Co	10.5	9.6	-0.9	41.6	36.5	-5.1
Idaho	Power Co	10.4	10.4	0.0	36.4	36.3	-0.1
Oregon	Klamath Co	10.0	9.9	-0.1	38.7	38.5	-0.2
California	Sonoma Co	9.8	9.4	-0.4	38.2	35.3	-2.9
California	San Luis Obispo Co	9.4	8.6	-0.8	35.6	31.6	-4.0
Idaho	Bannock Co	9.1	9.1	0.0	40.0	39.9	-0.1
Utah	Box Elder Co	8.5	8.5	0.0	38.4	38.3	-0.1
California	Inyo Co	6.0	5.9	-0.1	37.7	36.0	-1.7

Modeling Attainment of the Alternative 15/35 and 14/35 NAAQS

As indicated above, the 2020 15/65 scenario serves as our regulatory base case for analyzing the benefits of the revised and alternative more stringent NAAQS. Table 4-3 shows the reductions in $PM_{2.5}$ expected from the emissions controls in the 2020 15/35 scenario. These $PM_{2.5}$ reductions are incremental to the 2020 15/65 base case concentrations. Results are provided for those counties that are projected to be nonattainment for 15/35 in the 2020 15/65 baseline scenario.

Table 4-3. Modeled Impact of 15/35 Controls on Annual and Daily $PM_{2.5}$ Design Values ($\mu g/m^3$): 2020

		Annual			Daily		
State	County	*2020 15/65 ($\mu g/m^3$)*	*2020 15/35 ($\mu g/m^3$)*	*Impact of 15/35 controls ($\mu g/m^3$)*	*2020 15/65 ($\mu g/m^3$)*	*2020 15/35 ($\mu g/m^3$)*	*Impact of 15/35 controls ($\mu g/m^3$)*
California	Riverside Co	22.7	22.3	-0.4	63.2	61.1	-2.1
California	Los Angeles Co	21.6	21.3	-0.3	58.1	56.8	-1.3
California	San Bernardino Co	21.4	21.1	-0.3	58.1	56.7	-1.4
California	Tulare Co	18.9	18.5	-0.4	65.4	64.2	-1.2
California	Kern Co	18.6	18.2	-0.4	68.0	66.5	-1.5
California	Orange Co	18.2	17.9	-0.3	35.6	35.0	-0.6
California	Fresno Co	17.3	16.9	-0.4	59.6	58.2	-1.4
Michigan	Wayne Co	16.9	16.8	-0.1	38.4	38.1	-0.3
California	Kings Co	15.6	15.2	-0.4	61.0	59.5	-1.5
Alabama	Jefferson Co	15.1	15.1	0.0	34.2	34.1	-0.1
Ohio	Scioto Co	15.1	15.0	-0.1	33.3	33.2	-0.1
Georgia	Fulton Co	14.9	14.9	0.0	30.7	30.7	0.0
Illinois	Madison Co	14.6	14.6	0.0	34.4	34.3	-0.1
Illinois	Cook Co	14.5	14.5	0.0	35.3	35.3	0.0
Montana	Lincoln Co	14.8	14.5	-0.3	41.8	41.3	-0.5
Ohio	Cuyahoga Co	14.7	14.4	-0.3	39.1	38.3	-0.8

		Annual			Daily		
State	County	*2020 15/65 ($\mu g/m^3$)*	*2020 15/35 ($\mu g/m^3$)*	*Impact of 15/35 controls ($\mu g/m^3$)*	*2020 15/65 ($\mu g/m^3$)*	*2020 15/35 ($\mu g/m^3$)*	*Impact of 15/35 controls ($\mu g/m^3$)*
Pennsylvania	Allegheny Co	15.8	14.2	-1.6	51.5	46.9	-4.6
California	San Joaquin Co	14.4	14.1	-0.3	45.3	44.0	-1.3
California	Stanislaus Co	14.5	14.1	-0.4	51.5	49.9	-1.6
California	Merced Co	14.4	14.0	-0.4	47.7	46.3	-1.4
Illinois	St. Clair Co	14.1	14.0	-0.1	29.4	29.3	-0.1
California	Imperial Co	14.4	13.8	-0.6	43.0	41.5	-1.5
Indiana	Lake Co	12.4	12.4	0.0	36.9	36.8	-0.1
Idaho	Shoshone Co	12.3	12.2	-0.1	35.9	35.6	-0.3
Utah	Cache Co	12.3	12.0	-0.3	51.3	50.0	-1.3
California	Butte Co	12.7	11.8	-0.9	46.3	42.2	-4.1
Oregon	Lane Co	12.7	11.7	-1.0	52.5	47.9	-4.6
California	Alameda Co	11.7	11.4	-0.3	50.7	49.5	-1.2
Utah	Salt Lake Co	12.2	11.3	-0.9	47.5	42.9	-4.6
California	Santa Clara Co	11.3	11.2	-0.1	48.2	47.1	-1.1
California	Contra Costa Co	11.1	10.9	-0.2	52.6	51.5	-1.1
California	Sacramento Co	10.9	10.5	-0.4	42.0	40.0	-2.0
Washington	Snohomish Co	11.4	10.4	-1.0	40.2	37.0	-3.2
Idaho	Power Co	10.4	10.1	-0.3	36.3	35.1	-1.2
California	Solano Co	10.2	9.9	-0.3	48.3	46.6	-1.7
Washington	Pierce Co	11.5	9.9	-1.6	44.7	38.0	-6.7
California	Sutter Co	10.5	9.6	-0.9	35.5	32.0	-3.5
California	San Francisco Co	9.6	9.4	-0.2	42.4	41.5	-0.9
California	San Mateo Co	9.6	9.4	-0.2	36.5	35.7	-0.8
Oregon	Jackson Co	10.8	9.1	-1.7	37.1	32.6	-4.5
Oregon	Klamath Co	9.9	8.9	-1.0	38.5	35.0	-3.5
Idaho	Bannock Co	9.1	8.8	-0.3	39.9	38.7	-1.2
Utah	Box Elder Co	8.5	8.3	-0.2	38.3	36.9	-1.4
California	Inyo Co	5.9	5.8	-0.1	36.0	35.4	-0.6

The interpolation procedure used to generate the national sets of daily design values was formulated to account for the potentially steep gradients in air pollution that occur around urbanized areas. In this procedure, urban areas that do not have sufficiently close speciation monitors may be assigned ambient species profiles based on rural monitoring networks that do not represent the effects on the species profile of local sources within the urban area. This may result in projected design values in the urban area that are not as responsive to local controls as might be expected. Section 4.1.10 below provides information on adjustments to these CMAQ modeled results to better reflect the responsiveness to local controls in Bannock County, ID (Pocatello), Cache County, UT (Logan), Pierce County, WA (Tacoma), and Snohomish County, WA (Seattle).

Table 4-4 shows the reductions in $PM_{2.5}$ expected from emissions controls in the 2020 14/35 scenario. These $PM_{2.5}$ reductions are incremental to the 2020 15/65 regulatory base case

concentrations. Results are provided for those counties that are projected to be nonattainment for 14/35 in the 2020 15/65 baseline scenario.

Table 4-4. Modeled impact of 2020 14/35 controls on annual and daily $PM_{2.5}$ design values ($\mu g/m^3$)

		Annual			Daily		
State	**County**	*2020 15/65 ($\mu g/m^3$)*	*2020 14/35 ($\mu g/m^3$)*	*Impact of 14/35 controls ($\mu g/m^3$)*	*2020 15/65 ($\mu g/m^3$)*	*2020 14/35 ($\mu g/m^3$)*	*Impact of 14/35 controls ($\mu g/m^3$)*
California	Riverside Co	22.7	22.3	-0.4	63.2	61.1	-2.1
California	Los Angeles Co	21.6	21.3	-0.3	58.1	56.8	-1.3
California	San Bernardino Co	21.4	21.1	-0.3	58.1	56.7	-1.4
California	Tulare Co	18.9	18.6	-0.3	65.4	64.3	-1.1
California	Kern Co	18.6	18.2	-0.4	68.0	66.6	-1.4
California	Orange Co	18.2	17.9	-0.3	35.6	35.0	-0.6
California	Fresno Co	17.3	17.0	-0.3	59.6	58.3	-1.3
Michigan	Wayne Co	16.9	16.4	-0.5	38.4	37.5	-0.9
Pennsylvania	Allegheny Co	15.8	14.1	-1.7	51.5	46.7	-4.8
California	Kings Co	15.6	15.2	-0.4	61.0	59.6	-1.4
Alabama	Jefferson Co	15.1	14.5	-0.6	34.2	33.0	-1.2
Ohio	Scioto Co	15.1	14.5	-0.6	33.3	32.4	-0.9
Georgia	Fulton Co	14.9	14.2	-0.7	30.7	29.6	-1.1
Montana	Lincoln Co	14.8	14.6	-0.2	41.8	41.3	-0.5
Ohio	Cuyahoga Co	14.7	14.1	-0.6	39.1	38.0	-1.1
Illinois	Madison Co	14.6	14.0	-0.6	34.4	33.2	-1.2
California	Stanislaus Co	14.5	14.1	-0.4	51.5	49.9	-1.6
Illinois	Cook Co	14.5	14.2	-0.3	35.3	34.7	-0.6
California	Imperial Co	14.4	13.8	-0.6	43.0	41.5	-1.5
California	Merced Co	14.4	14.0	-0.4	47.7	46.3	-1.4
California	San Joaquin Co	14.4	14.1	-0.3	45.3	44.0	-1.3
Illinois	St. Clair Co	14.1	13.4	-0.7	29.4	28.2	-1.2
California	Butte Co	12.7	11.7	-1.0	46.3	42.1	-4.2
Oregon	Lane Co	12.7	11.7	-1.0	52.5	48.0	-4.5
Indiana	Lake Co	12.4	12.2	-0.2	36.9	36.5	-0.4
Idaho	Shoshone Co	12.3	12.2	-0.1	35.9	35.6	-0.3
Utah	Cache Co	12.3	12.0	-0.3	51.3	50.0	-1.3
Utah	Salt Lake Co	12.2	11.3	-0.9	47.5	42.9	-4.6
California	Alameda Co	11.7	11.5	-0.2	50.7	49.6	-1.1
Washington	Pierce Co	11.5	10.0	-1.5	44.7	38.0	-6.7
Washington	Snohomish Co	11.4	10.4	-1.0	40.2	37.0	-3.2
California	Santa Clara Co	11.3	11.2	-0.1	48.2	47.1	-1.1
California	Contra Costa Co	11.1	10.9	-0.2	52.6	51.5	-1.1
California	Sacramento Co	10.9	10.5	-0.4	42.0	39.9	-2.1
Oregon	Jackson Co	10.8	9.1	-1.7	37.1	32.6	-4.5
California	Sutter Co	10.5	9.6	-0.9	35.5	32.0	-3.5
Idaho	Power Co	10.4	10.1	-0.3	36.3	35.1	-1.2
California	Solano Co	10.2	9.9	-0.3	48.3	46.6	-1.7

		Annual			Daily		
State	**County**	*2020 15/65 ($\mu g/m^3$)*	*2020 14/35 ($\mu g/m^3$)*	*Impact of 14/35 controls ($\mu g/m^3$)*	*2020 15/65 ($\mu g/m^3$)*	*2020 14/35 ($\mu g/m^3$)*	*Impact of 14/35 controls ($\mu g/m^3$)*
Oregon	Klamath Co	9.9	8.9	-1.0	38.5	35.0	-3.5
California	San Francisco Co	9.6	9.4	-0.2	42.4	41.5	-0.9
California	San Mateo Co	9.6	9.4	-0.2	36.5	35.7	-0.8
Idaho	Bannock Co	9.1	8.8	-0.3	39.9	38.7	-1.2
Utah	Box Elder Co	8.5	8.3	-0.2	38.3	36.9	-1.4
California	Inyo Co	5.9	5.8	-0.1	36.0	35.4	-0.6

4.1.8 Population-Weighted Air Quality Results

As a means of better describing the relationship between air quality changes and population exposure, below we provide population-weighted air quality results. Population-weighted air quality is simply the product of the projected PM2.5 air quality change and the population at each model grid cell. Weighting the air quality change in this way can help illuminate the extent to which the projected air quality improvement is occurring in locations where people are actually exposed. Table 4-5 summarizes the total and incremental population-weighted change in annual average $PM_{2.5}$ concentrations between each control scenario. The first row illustrates how the population-weighted air quality for each air quality modeling case declines across attainment scenarios as both the projected air quality improves and the number of individuals exposed decreases. The subsequent rows summarize the incremental change between the base and each of the attainment scenarios.

Table 4-5. Population-Weighted Impacts on Annual Average $PM_{2.5}$

Air Quality Metric	*2020 Baseline*	*2020 15/65 Attainment Scenario*	*2020 15/35 Attainment Scenario*	*2020 14/35 Attainment Scenario*
Population Weighted Average Concentration	10.372	10.003	9.894	9.713
Population Weighted Change from Base	---	0.369	0.478	0.659
Incremental Population-Weighted Change 15/65 to 15/35	---	---	0.109	---
Incremental Population-Weighted Change 15/65 to 14/35	---	---	---	0.290
Incremental Population-Weighted Change 15/35 to 14/35	---	---	---	0.181

4.1.9 Visibility Degradation Estimates

The PM2.5 modeling platform described above was also used to calculate changes in visibility degradation. The estimate of visibility benefits was based on the projected improvement in annual average visibility at Class I areas. There are 156 Federally mandated Class I areas which, under the Regional Haze Rule, are required to achieve natural background visibility levels by 2064. These Class I areas are mostly national parks, national monuments, and wilderness areas. There are currently 110 Interagency Monitoring of Protected Visual Environments (IMPROVE) monitoring sites (representing all 156 Class I areas) collecting ambient PM2.5 data at Class I areas, but only 81 of these sites have complete data for 2001. For this analysis, we quantified visibility improvement at the 116 Class I areas which have complete IMPROVE ambient data for 2001 or are represented by IMPROVE monitors with complete data.[7]

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

Visual range is a measure of visibility that is inversely related to the extinction coefficient. Visual range can be defined as the maximum distance at which one can identify a black object against the horizon sky. Visual range (in units of kilometers) can be calculated from b_{ext} using the formula: Visual Range (km) = $3912/b_{ext}$ (b_{ext} units are inverse megameters [Mm^{-1}])

The future year visibility impairment was calculated using a methodology which applies modeling results in a relative sense similar to the Speciated Modeled Attainment Test (SMAT). In calculating visibility impairment, the extinction coefficient is made up of individual component species (sulfate, nitrate, organics, etc). The predicted change in visibility is calculated as the percent change in the extinction coefficient for each of the PM species (on a daily average basis). The individual daily species extinction coefficients are summed to get a daily total extinction value. The daily extinction coefficients are converted to visual range and then averaged across all days. In this way, we can calculate annual average extinction and visual range at each IMPROVE site. Subtracting the annual average control case visual range from the base case visual range gives a projected improvement in visual range (in km) at each Class I area. This serves as the visibility input for the benefits analysis (See Chapter 5).

For visibility calculations, we are continuing to use the IMPROVE program species definitions and visibility formulas which are recommended in the draft modeling guidance. Each

[7] There are 81 IMPROVE sites with complete data for 2001. Many of these sites collect data that is "representative" of other nearby unmonitored Class I areas. There are a total of 116 Class I areas that are represented by the 81 sites. The matching of sites to monitors is taken from "Guidance for Tracking Progress Under the Regional Haze Rule".

IMPROVE site has measurements of PM2.5 species and therefore we do not need to estimate the species fractions in the same way that we did for FRM sites (using interpolation techniques and other assumptions concerning volatilization of species).

4.1.10 Adjustments to Modeled Daily Design Values for 15/35 Control Scenario

This subsection describes the approach taken to address the previously identified deficiency with specific interpolated species fractions at monitors where controls are unexpectedly ineffective by applying a more appropriate species profile that is from a similar urban area in close proximity to the area of concern (while not being close enough to be included in the interpolation). An indicator that the species profile may be non-representative is an excessively high percent of organic carbon. A high percent organic carbon at a site may be of concern because the SMAT method assigns organic carbon by a difference method where the sum of all other interpolated PM species is compared with the total FRM PM2.5 mass at the design value monitor and the difference between the two is assumed to be organic carbon. When interpolated species values are derived from speciation sites with very different PM composition, the differences in total mass tend to be larger, and thus the amount assigned to the organic carbon fraction will be large.

Based on the organic carbon fraction and the emissions profiles of the monitor locations, we identifed 4 monitor locations where a species profile adjustment would be appropriate: Bannock County, ID (Pocatello), Cache County, UT (Logan), Pierce County, WA (Tacoma), and Snohomish County, WA (Seattle). For the Bannock County, ID site, we determined that there were no speciation monitors located within 50 km of the FRM monitor. The two most likely candidate urban sites for speciation profiles include sites in Boise City, Idaho (Ada County) and in Davis County, UT (suburb of Salt Lake City). Using the speciation profiles for these counties results in a large reduction in the fraction associated with organic carbon, and a higher fraction of ammonium nitrates and sulfates. Depending on the specific speciation site selected, there are slight differences in the alternative profiles, however, the overall impact on design values is similar. Using the speciation profile from Ada County, the adjusted daily design value for the 2020 15/35 attainment strategy is 35.5 $\mu g/m^3$. Using the Davis County, UT species profile, the adjusted daily design value is 34.7 $\mu g/m^3$. As such, using either of the alternative speciation profiles, Bannock County attains the daily standard.

Cache County has no co-located speciation monitor available. However, there were three speciation monitors near Salt Lake City located within 85 km of the Cache County FRM monitor. Using the average of the speciation profiles from these 3 monitors resulted in a large reduction in the fraction attributed to organic carbon, and increases in the fractions associated with ammonium nitrate and ammonium sulfate. Some experimental monitoring conducted by Utah State University suggests that even this alternative speciation profile may be understating the contribution of nitrates in wintertime months, when nitrate may contribute over 70 percent of total mass. Using the alternative speciation profile results in an estimated daily design value in Cache County for the 2020 15/35 attainment strategy of 44.6 $\mu g/m^3$. Thus, even with the alternative species profile, Cache County does not attain with available controls. However, the design value is now much closer to the design value of 40.7 $\mu g/m^3$ in Salt Lake City. In both of the above cases (Bannock County and Cache County) prior to the use of alternative speciation profiles, organic carbon was estimated to account for 90 percent or more of the total mass.

Based on the alternative speciation profiles, organic carbon may in reality account for less than 25 percent PM2.5 in wintertime months when peak daily concentrations are likely to occur.

In Washington, the two monitor locations near Seattle and Tacoma that were relatively unresponsive to emission controls were also characterized by unusually high organic carbon fractions. The monitor in Pierce County (Tacoma) had over 70 percent estimated organic carbon, while the monitor in Snohomish County (Seattle suburb) had over 85 percent estimated organic carbon. Using only the closest speciation monitors for each of these sites resulted in relatively large reductions in the estimated percent organic carbon at each monitor. For the Pierce County monitor, we used a speciation monitor located in Seattle, approximately 45 km from the FRM site. This resulted in a decrease in the percent organic carbon at the monitor to 50 percent, and increases in percent elemental carbon to 15 percent, with smaller increases in crustal, nitrates, and sulfates. This resulted in an adjusted daily design value for the 2020 15/35 attainment strategy of 34.2 $\mu g/m^3$, thus resulting in attainment at this monitor. For the Snohomish County monitor, we used a speciation monitor located close to Seattle, approximately 35km from the FRM site. This resulted in a decrease in the percent organic carbon at the monitor to 65 percent, with increases in percent elemental carbon to 11 percent, with smaller increases in crustal, nitrates, and sulfates. This resulted in an adjusted daily design value for the 2020 15/35 attainment strategy of 34.2 $\mu g/m^3$, thus resulting in attainment at this monitor.

The adjusted design values are provided below in Table 4-6. These adjusted daily design values form the starting point for the next step in the nonattainment determination process, which continues in Section 4.2.

Table 4-6. Adjusted Daily Design Values for 15/35 Control Scenario

Location	*Adjusted 15/35 Daily DV*
Bannock County, ID	35.5
Cache County, UT	44.6
Pierce County, WA	34.2
Snohomish County, WA	34.2

4.1.11 Characterization of Air Quality Modeling and Limitations to the Analysis

While EPA's regional scale air quality modeling system has been extensively peer reviewed and represents the state of the science in terms of the formation and fate of $PM_{2.5}$ in the atmosphere, a number of factors affect the conclusions that can be reached about the effectiveness, costs, and benefits of alternative control strategies:

- Overall, the air quality model performs well in predicting monthly to seasonal concentrations, similar to other state-of-the-science air quality model applications for

$PM_{2.5}$.[8] The model is less well suited to predicting 24-hour values. Thus, there is less certainty in analyses involving 24-hour model predictions than those involving longer-term averages (i.e., month, quarter, annual) concentrations.

- In general, model performance is better for the Eastern U.S. than for the West. The air quality model performs well in predicting the formation of sulfates, which are the dominant species in the East. Ambient monitoring data indicate high levels of PM in the West, especially in California, are dominated by nitrate and organics. While the modeling system performs well for nitrate in the East, large under predictions are noted in the West. In both the East and West, carbonaceous aerosols are the most challenging species for the modeling system to predict in terms of evaluation against ambient data. There is considerable uncertainty and lack of understanding of formation, fate, and properties of organic particles.[9] It is estimated that only 10 to 20 percent of the PM organic compounds have been quantified using existing methodologies. Work is underway at EPA and elsewhere to improve our understanding or secondary organic aerosols and our ability to characterize these compounds and their precursors in air quality models. In view of these limitations and uncertainties, current air quality models, including CMAQ, may understate the reduction in secondary organic PM from controls on particle-forming VOCs, including aromatic compounds and higher carbon alkanes and olefins.

- A number of uncertainties arise from use of baseline data from EPA's National Emissions Inventory, especially in terms of the overall magnitude of emissions of primary particles from stationary and mobile sources, spatial allocation of area and other source categories, and the relative split of emissions into $PM_{2.5}$ species. Of particular concern is the apparent disparity between estimated contributions of mobile source emissions with receptor modeling results based on ambient air quality data. While the results of the source receptor modeling studies themselves contain significant uncertainties (particularly in dealing with secondary organic aerosols, or SOAs), it is probable that the mobile source emission inventory of directly emitted PM2.5 is biased low. The most uncertain portion of the current mobile source inventory for direct PM2.5 is probably that from gasoline vehicles and nonroad equipment. While it is likely that updated emissions estimates from these sources will be higher than those used in our analysis, it is not certain the extent to which existing emissions control programs will reduce these emissions.

- Additional uncertainty is introduced as a result of our limited understanding concerning the collective impact on future-year emission estimates from economic growth estimates, increases in technological efficiencies, and limited information on the effectiveness of control programs.

[8] U.S. Environmental Protection Agency, March 2005. Updated CMAQ Model Performance Evaluation for the 2001 Annual Simulation, Appendix C. Office of Air Quality Planning and Standard, Research Triangle Park, NC. (Docket No. OAR-2005-0053-2149).
[9] Particulate Matter Science for Policy Makers, a NARSTO Assessment. McMurry, P. M.F. Shepherd, and J.S. Vickery, 2004.

- The set-up of the CMAQ modeling system used for this assessment was configured with a 36 kilometer receptor grid, which spreads point and mobile source emissions that may be concentrated in particular locations across the entire area of each grid. This serves to obscure local-scale air quality improvements that result from urban-area controls. To the extent that this occurs, our estimates may underestimate the effectiveness of local or urban-area controls as compared to broad scale regional controls. We performed a sensitivity modeling analysis with CMAQ in which we modeled our 2015 base case at 12 km resolution for a modeling domain covering the Eastern U.S. The results of this analysis are provided in Appendix N.

4.2 Supplemental Carbonaceous Particle Emission Controls Analysis

Because we based our selection of controls on the expected impact on $PM_{2.5}$ (which we estimated by using the RSM-derived µg/ton estimates described in Chapter 3), in some locations the CMAQ-modeled impact on $PM_{2.5}$ at the violating monitor was less than expected. In these cases our control strategies did not result in full attainment of the standards, even though additional cost-effective carbonaceous particle controls were still available in our database of AirControlNET and developmental emission controls.[10] To demonstrate the costs and benefits of reaching full attainment in these areas, we identified remaining cost-effective carbonaceous particle emission controls in each of the projected residual nonattainment areas. We then determined whether those supplemental controls would likely be sufficient to simulate full attainment with the revised and more stringent alternative standards. If we estimated these controls to be sufficient, then we included the costs of those controls were in our overall full attainment cost estimate (see Chapter 6). Note that this method does not apply to the projected non-attainment areas of Salt Lake City and many counties in California, where we exhausted emission controls in our CMAQ analysis; for these areas we estimated full attainment cost by using an a cost-extrapolation methodology that we describe in Chapter 6.

Supplemental Analysis to Simulate Attainment with Revised Daily Standard of 35 µg/m^3

After modeling the air quality impacts of our illustrative attainment strategy for the revised 15/35 standards, we determined that two locations, Eugene OR and Cleveland, OH, did not simulate attainment with the revised daily standard of 35 µg/m3. However, our emission controls database indicated that there were still carbonaceous particle controls available to apply. We calculated the average $PM_{2.5}$ impact per ton of reducing elemental and organic carbonaceous particles in each location, and then estimated the amount of additional elemental and organic carbonaceous particle emissions reductions that would be necessary to reach attainment. If the total amount of tons available was less than the amount needed, then we added the costs to the full attainment cost estimate and continued with the weight of evidence assessment discussed in Section 4.4 below. After applying supplemental controls, we found that neither Cleveland nor Eugene was able to attain the 15/35 revised standards. For a discussion of the emissions reductions and engineering costs associated with the application of these controls, see Chapter 6.

[10] For a description of the emission controls available in the AirControlNET database, and a discussion of the developmental emission controls, see Chapter 3.

Supplemental Analysis to Simulate Attainment with Alternative More Stringent Annual Standard of 14 $\mu g/m^3$

After modeling the air quality impacts of our illustrative attainment strategy for the 14/35 standards, we determined that Birmingham, AL, Chicago, IL, and Cleveland, OH had not simulated attainment. However, our emission controls database indicated that there were still carbonaceous particle emission controls available to apply. We calculated the average $PM_{2.5}$ impact per ton of reducing elemental and organic carbonaceous particles in each location, and then estimated the amount of additional elemental and organic carbonaceous particle emissions reductions that would be necessary to reach attainment. We then used that impact per ton estimate to determine the number of tons of carbonaceous particles would be necessary to control to simulate attainment the residual increment to attainment (the modeled design value after application of the illustrative control scenario minus 14.05). Finally, we calculated the total remaining tons of emissions that could be reduced with known controls. If the total controllable tons was greater than or equal to the amount of tons needed to reach full attainment, then we added the costs of control to the overall full attainment cost. If the total amount of tons available was less than the amount needed, then we added the costs to the full attainment cost estimate and continued with the weight of evidence assessment discussed in Section 4.4 below. After applying supplemental controls, we found that Birmingham, Chicago and Cleveland were able to attain the more stringent alternative standards of 14/35. For a discussion of the emissions reductions and engineering costs associated with the application of these controls, see Chapter 6.

Calculating Monetized Human Health Benefits of Achieving the Residual Air Quality Increment Through Supplemental Controls

It is extremely difficult to accurately estimate the benefits of fully attaining a set of ambient $PM_{2.5}$ standards when using the supplemental controls approach. This difficulty is due to the complex nature of the atmospheric chemistry and fate and transport mechanisms that connect precursor emissions with ambient concentrations of $PM_{2.5}$. In the absence of air quality modeling associated with specific sets of emissions controls, it is not certain how ambient $PM_{2.5}$ levels throughout the U.S. would be affected by programs to bring residual nonattainment areas into attainment. If broad scale programs to reduce transport of precursor emissions were enacted, then ambient PM2.5 levels throughout a region would be reduced. On the other hand, if extremely local reductions in emissions affecting a single nonattaining monitor were enacted, then air quality improvements would be very localized, with little impact on regional ambient $PM_{2.5}$ levels. When modeling benefits, we have assumed that these areas would apply emission controls using the latter method.

In order to provide at least a lower bound estimate of the benefits associated with fully attaining the revised and alternative standards, we used a simple rollback approach. This approach makes the bounding assumption that ambient $PM_{2.5}$ concentrations can be reduced only at monitors that are above the standards, regardless of the proximity of neighboring monitors. In essence, the monitor values are simply rolled back so that no monitor in the U.S. is above the standard being analyzed. From a benefits perspective, this leads to a likely downward bias in the estimates, because populations are assumed to be exposed to a distance weighted average of surrounding

monitors, so their exposure to the reductions at a single nonattaining monitor will be weighted less if there are other, attaining monitors in close proximity.

Below we provide a summary of the mechanics of these calculations:

Step 1: Rollback annual design values from modeled levels to 15 $\mu g/m^3$ to simulate attainment of the 1997 standards.

Step 2: Estimate the improvement in the daily standard that results from meeting the annual standard. This estimated impact on the daily standard is based on relationships between annual and daily design values from existing air quality modeling results. For example, in Los Angeles, the daily design value is typically 2.6 times the annual design value. Assuming this relationship will continue to hold in the future, for every 1 $\mu g/m^3$ reduction in the annual design value there would be approximately a 2.6 $\mu g/m^3$ reduction in the daily design value. This relationship was derived for each nonattainment monitor.

Step 3: Rollback daily design values from the estimated values resulting from Step 2 to the revised daily standard of 35 $\mu g/m^3$.

Step 4: Estimate the impact of meeting the revised 35 $\mu g/m^3$ standard on annual design values. Similar to the calculations in Step 2, we used the relationship between annual and daily design values to estimate how annual design values would be affected by reducing the daily design values. Following the example above, for every 1 $\mu g/m^3$ reduction in the daily design value, the annual design value would be reduced by 0.38 $\mu g/m^3$.

Step 5: Rollback annual design values from the estimated values resulting from Step 4 to the alternative more stringent annual standard of 14 $\mu g/m^3$.

Step 6: Combine rolled-back annual design value data from Step 1 with modeled design value data from the 15/65 baseline CMAQ modeling for attaining monitors and interpolate the annual design values to CMAQ grid cell domain to provide the baseline air quality inputs for the benefits analysis (details of the spatial interpolation method are provided in Appendix H).

Step 7: Combine rolled back annual design value data from Step 4 with modeled design value data from the 15/35 CMAQ modeling for attaining monitors and interpolate to CMAQ grid cell domain to provide air quality inputs for the benefits analysis for the 15/35 standards.

Step 8: Combine rolled back annual design value data from Step 5 with modeled design value data from the 14/35 CMAQ modeling for attaining monitors and interpolate to CMAQ grid cell domain to provide air quality inputs for the benefits analysis for the 15/35 standards.

For a discussion and presentation of modeled and full attainment benefits, see Chapter 5.

4.3 Illustrative Attainment Determinations

In this section we make a final determination of attainment for those areas whose projected design values, based on the air quality modeling analysis, exceed the revised or more stringent alternative standards, and for which supplemental controls did not simulate full attainment. To make this determination we combine the projected design values from the air quality modeling with urban-area specific data, including: an analysis of the projected violating monitor, dispersion modeling, a characterization of emissions inventory uncertainties, modeling uncertainties and updated design values. In this way we assess whether the balance of empirical data suggests that each projected nonattainment county will or will not attain the revised and more stringent alternative standards. In the subsections below we outline the data we drew upon to make these attainment determinations and then analyze each of the six areas that the air quality modeling analysis projects to violate one or more standards. These areas include: Detroit, MI, Pittsburgh, PA, Cleveland, OH, Salt Lake City UT, Eugene, OR and Libby MT. We separately present an analysis of projected non-attainment areas in California at the end of this chapter.

Table 4-7 below summarizes the projected annual and daily design values for each of these six urban areas. The design values in these tables reflect the application of any supplemental carbonaceous particle emission controls, and thus vary from the CMAQ-projected design values found in the preceding tables:

Table 4-7. Areas Projected to Not Attain the Revised or Alternative More Stringent $PM_{2.5}$ Standards

			2020 Basecase Design Values		2020 Control Case: Annual Design Values		2020 Control Case: Daily Design Values	
State	County	Violating Monitor	*Annual*	*Daily*	*15/35*	*14/35*	*15/35*	*14/35*
Ohio	Cuyahoga	390350038	15.2	39.7	14.4	14.0	36.6	35.4
Michigan	Wayne	261630033	17.3	39.0	16.8	16.4	38.1	37.5
Pennsylvania	Allegheny	420030064	16.2	52.7	14.12	14.0	46.9	46.7
Montana	Lincoln	300530018	14.9	42.2	14.5	14.0	41.3	41.3
	Box Elder	490030003	8.5	38.4	8.3	8.3	36.9	36.9
Utah	Cache	490050004	12.3	51.4	12.0	12.0	44.6	44.6
	Salt Lake	490350003	12.2	47.6	11.3	11.3	42.9	42.9
Oregon	Lane	410392013	12.8	53.0	11.7	11.7	48.0	48.0

4.3.1 Data Sources

Our attainment determination considered a variety of data sources, each of which we describe below. Because not all of these data were available for, or germane to, each urban area we did not include all data sources in each attainment determination.

Detailed Monitor and Emissions Analyses

EPA sought to better understand the local-scale characteristics of those monitors that, based on 1993 to 2003 measured data, are projected to violate the 1997 standards or the revised or more stringent alternative standards. To develop this information, EPA conducted four general types of evaluations, where we: 1) using aerial photographs, identified the proximate areas of the monitoring sites in order to explore the potential impacts of local sources; 2) recalculated baseline design values; 3) re-evaluated modeled speciation profiles; and 4) gleaned pertinent information on the specific geographic areas and associated monitoring sites from online sources and/or from EPA regional office staff. EPA evaluated thirteen different geographic areas, encompassing approximately 20 priority monitoring sites, with one or more of these methods.

More detail on the four evaluative techniques we employed is presented below; these are followed by summaries of the pertinent findings.

1. *Examinations utilizing geographic information systems (GIS) and aerial photographs of the local areas around an area's priority monitors (those projected to violate the revised or more stringent alternative standards) to explore the potential impacts of local sources.* These studies employed gathering data on the priority monitors, and mapping these data along with the locations of point sources as provided in the emission data set representing the 2015 base case, which incorporates all known controls from the base year inventory of 2001. Aerial photos were used to capture the area surrounding the priority monitors. Some aerial views were evaluated across different time periods, as available, to ascertain the possibility that activity, and thus source profiles, may have changed over time and may not accurately represent the area. A common issue noted in this review relates to the precision of the inventory point source coordinates (latitude and longitudes). The precision of the point source locations is accurate to only 2 decimal places. This equates to a precision of about half of a kilometer if rounded, and 1 kilometer if truncated. It thus becomes difficult to match sources in the inventory with sources shown in the aerial photographs. Therefore, it is not known to the extent to which sources are underrepresented or located in different areas from the photos. A frequent observation of the aerial photo review was that there are some emission source types that are not well characterized in the emission inventory. For example, emissions from railroads or depots are based on national level emissions that are allocated to grid cells using railway miles and railway activity. Areas with heavy rail use or rail depots could have significant local impacts that are nearly impossible to model accurately in a national-level analysis.
2. *Recalculation of initial baseline design values.* The design values that were originally calculated could overestimate the actual aggregated regulatory values due to our treatment of data "flagged" for exceptional events. Under current EPA guidance and practice, only data flagged for events that have been approved ('concurred' with) by the appropriate EPA regional office (RO) are excluded when making comparisons to the NAAQS. The flagging process, as a whole, includes: flagging of data by the State monitoring agency in an appropriate timeframe; submission (by the State agency) of documentation proving the event occurred and its causal role in a NAAQS exceedence; subsequent review of the documentation by the RO; and eventual

acceptation or rejection of the assertion by the RO. States typically flag about 85% more $PM_{2.5}$ data than are documented.. This discrepancy usually exists because States often only submit documentation for flagged data points that could make a difference in an attainment/nonattainment designation. Because the annual NAAQS is controlling in most areas, it could be several years before it could be determined if flagged data points make that difference. Thus, flagged values for which documentation was not submitted could actually be legitimate, but irrelevant to current NAAQS levels. This phenomenon must be taken into account in the evaluation of future nonattainment scenarios given different ambient air standards. Also, in certain situations some States flag data for their own purposes, such as for internal trends analyses. These cases do not always have supporting documentation. It takes resources to compile supporting documentation in a cohesive manner, and the States often do not expend these resources unless a nonattainment designation is imminent. Based on the flagging and documentation of several large regional exceptional events (e.g., the Quebec fire of 2002) it is speculated that most "flagged but not documented" events are potentially valid. Furthermore, most documented events are generally eventually approved. Thus, this exercise entailed treating all flagged events as documented and approved events. In some cases, this recalculation lowered the baseline model DV such that it would not result in future modeled nonattainment.

3. *Comparison of species profiles used in the projection of future design values to alternative, potentially more representative profiles.* The species profiles used for projecting future design values were based on limited 2002 Speciation Trends Network (including State speciation sites, or "STN+") data. More robust (i.e., multi-year) estimates of speciation profiles are now available for some of the priority monitoring sites. Also, some newer speciation monitors closer to the priority sites now have data. These newer data are useful for determining a more representative estimates of the speciation profiles in the vicinity of priority sites. A lack of representative profiles for the priority sites increases the potential for underpredicting the species emitted by local sources (e.g., crustal material), further limiting our ability to show the impact of potential control strategies on these sources. If an area was deemed to be in residual nonattainment of the annual standard, then the speciation profile review focused on the aggregate annualized profile. If an area was deemed to be in residual nonattainment of the 24-hour standard, then the speciation profile review focused on the profile(s) of the quarter(s) with the highest concentrations (that is, the one(s) where the 98^{th} percentile was expected).

4. *Gleaning of information from online sources and/or EPA Regional Office staff.* Internet queries were conducted with search engines such as Excite and Google to garner relevant information about the geographic areas and monitoring locations with respect to particle pollution. This information included studies of air quality trends and characterization by universities and state and local air quality organizations. Staff in EPA Regional Offices were contacted to summarize the particle problem in these areas, provide site-specific characterizations, ascertain the identity of possible sources, and/or verify various postulations.

Readers interested in reviewing the complete monitor and emissions analysis should consult the technical support document located in the docket.

Local-Scale Dispersion Modeling (AERMOD)

EPA used local-scale air quality modeling to examine the spatial variability of direct $PM_{2.5}$ concentrations associated with emissions of primary $PM_{2.5}$ within each urban area, and to estimate the contribution of primary $PM_{2.5}$ emissions from local sources in the urban area to ambient $PM_{2.5}$ concentrations at Federal Reference Method (FRM) monitoring sites. In addition, attribution of the modeled concentrations to specific emission source groups in the urban area such as electric generating facilities, industrial facilities, residential wood burning, commercial cooking, mobile sources and others (see Appendix B for a complete list) allowed for an investigation into the impact of controls of primary $PM_{2.5}$ emissions from local sources on attainment. This assessment complements the regional-scale modeling analyses through its ability to provide concentrations at a higher spatial resolution and an estimate of the impact of local sources of primary $PM_{2.5}$. We focused this assessment on five urban areas: Birmingham, Seattle, Detroit, Pittsburgh and Chicago. Each of these areas has different characteristics in terms of the mixture of emissions sources, meteorology, and associated $PM_{2.5}$ air quality issues. This assessment focused on future incremental impacts of direct $PM_{2.5}$ sources within these areas after implementation of the regulatory base case.

Based on 2001 meteorology data and the 2015 regulatory base case emissions inventory used in the CMAQ analysis, the AERMOD modeling system was applied to each urban area to provide concentration estimates of directly emitted $PM_{2.5}$ by species across a specified network of receptors within each urban area. AERMOD provides a more refined geographic view of local $PM_{2.5}$ concentrations compared to the coarse view provided by the 36 kilometer resolution of used for our CMAQ modeling. Appendix B provides detailed results for each urban area for both annual and daily concentrations. These results indicate high annual concentration gradients for primary PM2.5 over distances much less than the 36 or 12 kilometer resolution typically used in photochemical grid modeling for the study area. Furthermore, local sources of primary $PM_{2.5}$ are significant contributors to these concentration gradients. These sources vary in their importance by monitor location and include industrial sources (iron and steel manufacturing, coke ovens, pulp and paper mills), human activities like residential wood/waste burning, and onroad and nonroad sources.[11]

Updated Design Value Data

Our 2020 base case design values were determined using data which includes ambient design values calculated with 1999–2003 monitoring data. Because the projections of future design values are sensitive to the design values used in the base years, it may be insightful in some projected non-attainment areas to assess whether or not more current design value differ greatly from what was used in our projections. For example, an area that we project to not attain the revised standards by a small margin might be expected to attain, or might be closer to attainment,

[11] Note that while we modeled nonroad mobile sources, the inventories for locomotives are not yet detailed enough to allow us to fully capture the air quality impacts associated with controlling this source.

if we used much lower design values as the starting point for our projections. For this reason, we have examined more current design value data to improve our characterization of the potential for future improvement in air quality in these areas.

Source Apportionment Studies

Source apportionment analyses such as receptor modeling are useful in both qualifying and quantifying potential fine particulate regional and local source impacts on a receptor's ambient concentrations. Receptor modeling techniques utilize measured ambient species' concentrations to estimate the contribution that regional and local sources have at a given receptor which, in this case, is an ambient monitoring location. Currently, two established receptor models are being widely used for source apportionment: the Chemical Mass Balance (CMB) and Positive Matrix Factorization (PMF). Both have been used to characterize fine particulate source contributions to ambient $PM_{2,5}$levels. For one projected non-attainment area below we consider the source apportionment data to better characterize the impact of our control strategies on the monitor projected to not attain the 1997 standards, the proposed revised standards and the alternative revised standards.

4.2.2 *Area Specific Analyses*

The subsections that follow detail each of the six urban-area analyses we performed. As noted above, these urban areas include Cleveland, Detroit and Pittsburgh in the East and Salt Lake City, Libby, and Eugene in the West.

4.2.2.2 Cleveland

Projected Design Values. Under the base case, the Cuyahoga county monitor violates the revised daily standard. In our control case we were unable to simulate attainment with the revised daily standard of 35 $\mu g/m^3$ under our 15/35 control scenario. However, we were able to meet the revised daily standard of 35 $\mu g/m^3$ under our 14/35 control scenario, indicating that the addition of regional emission reductions were effective in bringing this area into attainment with a tighter daily standard.

Table 4-8. Projected Design Values for Priority Site in Cuyahoga County, Ohio

		2020 Basecase Design Values		2020 Control Case: Annual Design Values		2020 Control Case: Daily Design Values	
County	Violating Monitor	*Annual*	*Daily*	*15/35*	*14/35*	*15/35*	*14/35*
Cuyahoga	390350038	15.2	39.7	14.4	14.0	36.6	35.4

Monitoring and Emissions Analysis. Monitoring site 390350038 is the priority monitor for Cleveland and has a projected 2020 base DV of 19.3 $\mu g/m^3$ based on 1999-2003 monitoring data. The next highest DV in the area is 1.3 $\mu g/m^3$ lower (18.0) but is less than a mile away. As with

the priority Cleveland monitor and its closest counterpart, this fact suggests that local emission sources account for the increment. Based on a review of aerial photographs, the Cleveland priority monitor appears to have numerous potential local $PM_{2.5}$ influences consisting of heavy transportation and industrial sources. However, the 2015 base inventory shows no point sources in the immediate area or even in the 1 kilometer radius and few emission sources with the 3 kilometer radius. Several steel manufacturing operations are present in the inventory within the 3 kilometer radius but their emission estimates are atypically low. Hence, the industrial areas are probably not properly characterized in the inventory. The monitor is located in a major transportation corridor, containing an interstate, railroads and ports (on the Cuyahoga River). There are several railroad lines within a kilometer of the monitor; a dense set of railroad lines lie approximately 500 meters away. The monitor is approximately 75 meters from Interstate 490, and 130 meters from a cloverleaf intersection. Port terminals along the Cuyahoga are about 700 – 1300 meters from the monitor.

Updated Design Values. More current design value data in Figures 4-3 and 4-4 below for the Cleveland priority monitor (site 390350038) suggests a slight upward trend in the daily design value and a slight downward trend in the annual design value. Had the analysis used more current design value data to project future baseline air quality in Cleveland, it is possible that our estimates of the baseline daily values might be higher and the baseline annual values might be somewhat lower.

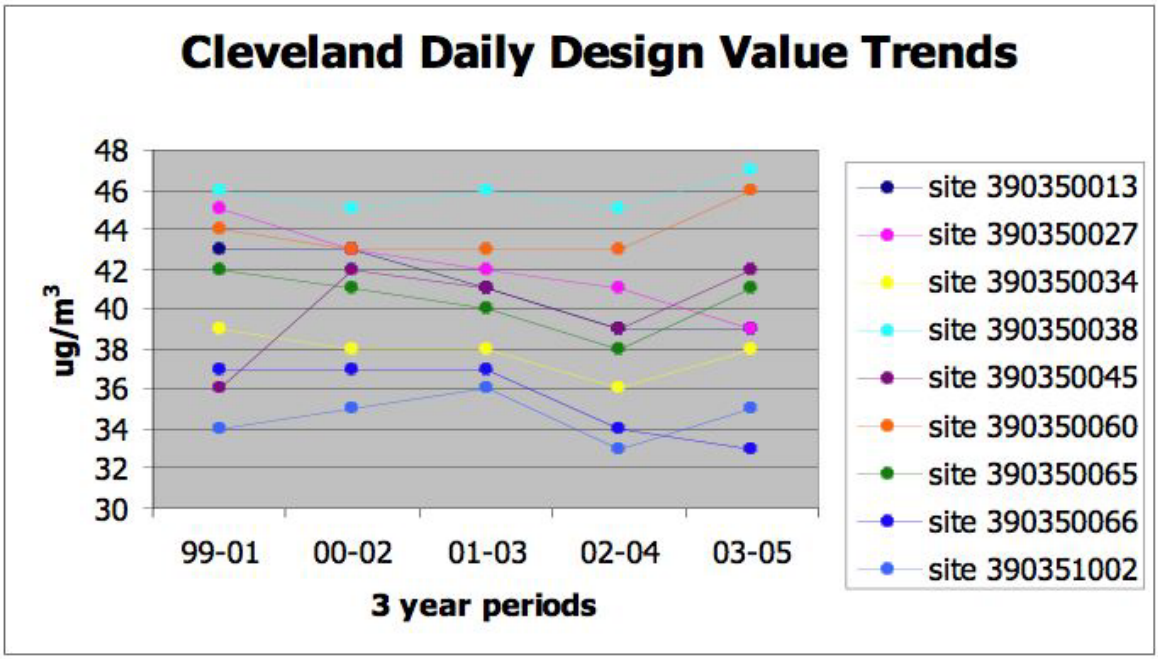

Figure 4-3. Daily Design Value Trend for Monitors in Cleveland Metropolitan Area

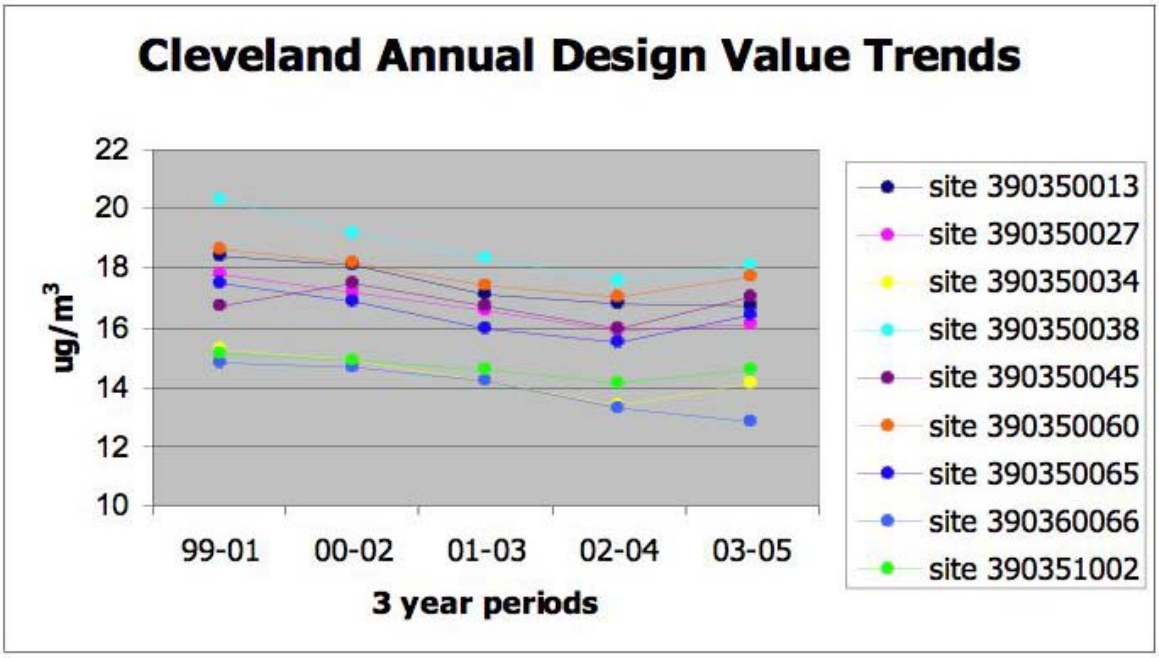

Figure 4-4. Annual Design Value Trend for Monitors in Cleveland Metropolitan Area

Conclusions. The monitoring and emissions analysis indicates that there are a sufficient number of sources located in close proximity to the monitor that are likely contributing to high annual and daily design values. Due to uncertainties in our emissions inventories, we may not have fully captured the impact of controlling these sources in our air quality modeling. Moreover, due in part to the relatively coarse-scale of our modeling grid cells our analysis was most likely not able to fully capture the near-field effects of controlling these sources. This suggests that an emission reduction strategy that applies controls to sources in close proximity to the priority monitor would be expected to further reduce future design values.

The updated design value data suggests a declining trend in the annual design value but an increasing trend in the daily value at the priority monitor. Thus, using these updated design values in our air quality modeling would be unlikely to have produced 2015 and 2020 base case air quality estimates that significantly differ from our current projections.

Considering the balance of the empirical evidence above, we believe that for the purposes of this illustrative attainment analysis that our projected design values do not properly characterize the future air quality at the priority monitor in Cleveland and that the controls we simulated were more effective than we modeled. Thus, for the purposes of this analysis, we are presuming that Cleveland does attain the new and more stringent alternative standards in 2020.

4.2.2.2 Detroit

Projected Design Values. Under the base case, the Wayne county monitor violates the revised and more stringent alternative standards. The Wayne county monitor also violates each both of these standards under the three control cases.

Table 4-9. Projected Design Values for Wayne County, Michigan

County	Violating Monitor	2020 Basecase Design Values		2020 Control Case: Annual Design Values		2020 Control Case: Daily Design Values	
		Annual	*Daily*	*15/35*	*14/35*	*15/35*	*14/35*
Wayne	261630033	17.3	39.0	16.8	16.4	38.1	37.5

Monitoring and Emissions Analysis. There are two priority monitors in the Detroit area, site 261630033 with a starting DV of 19.5 $\mu g/m^3$ and site 261630015 with a starting DV of 17.4 $\mu g/m^3$ based on 1999-2003 design value data. Other $PM_{2.5}$ monitors located elsewhere in the Detroit MSA indicate a much lower design value. Available speciation data from years not used in the attainment analysis shows that the interpolated model data for this location has significantly lower metals/crustal material than actually is present. The speciation profile we used for the site 261630015 was obtained by interpolation of measurements at other sites. That data had about 4% of the PM mass as crustal material. However, updated speciation data, from a collocated monitor at site 261630015, shows the crustal fraction to be closer to 14%. This indicates that local, directly emitted PM, have a greater influence on this site, compared to what we used in our analysis.. In addition, a review of aerial photographs of the vicinity of site 261630015 from different years, indicates that construction and/or demolition activity occurred in the immediate vicinity of the site during the model base timeframe. This would also affect the magnitude of $PM_{2.5}$ and the speciation for this site in a way that we could not account for in our analysis.

Our analysis of emissions data indicates that both priority sites in Detroit are likely to be highly influenced by nearby emissions sources located within 3 km of the site. Many of these sources may not have been characterized with the precision needed for a local scale assessment for these locations. As noted in the general analyses method descriptions, the point source locations in our inventory are specificed to 2 decimal places. This equates to a precision of about half of a kilometer, if rounded and 1 km if truncated. Also as previously noted, emissions for railroads and switching yards are not specified to the exact location of individual rail lines and yards. Site 261630033 is extremely close to a large number of parallel railroad lines (4 parallel lines adjacent and maybe 50 meters away from monitor). Furthermore, there appears to be point sources at the railroad which may correspond to nearby sources that are in our inventory.

AERMOD Analysis. Figure 4-6 shows the spatial distribution of $PM_{2.5}$ for Detroit resulting from AERMOD modeling of primary $PM_{2.5}$ emissions from local sources. These modeling results indicate high annual concentration gradients of primary $PM_{2.5}$ within typical photochemical modeling grid resolutions. Thus, spatial gradients exist within the study area for primary $PM_{2.5}$ with a variety of local sources such as metal manufacturing, commercial cooking,

and onroad and nonroad vehicles being significant contributors depending upon the location of the monitor. The local sources of direct PM2.5 contribute roughly 25 percent of the projected concentrations of total $PM_{2.5}$ at monitoring site 261630033. Based on application of the 15/65 control set in Detroit, AERMOD predicted reductions in annual direct PM2.5 that were roughly 2.5 times higher than that predicted by CMAQ, i.e., a reduction in predicted direct PM2.5 concentrations by 0.68 µg/m^3 versus 0.26 µg/m^3. The models produced similar reductions in direct PM2.5 concentrations for the 15/35 control set, i.e., a reduction in predicted direct PM2.5 concentrations by 0.046 µg/m^3 versus 0.057 µg/m^3. For the 14/35 control set, the AERMOD predicted reductions were again higher than the CMAQ predictions like the 15/65 control set. The difference in results here are due to the nature of the controls so that when controls are applied to stationary point sources there will be greater differences while controls applied to more dispersed sources like area and mobile will result in more similar results.

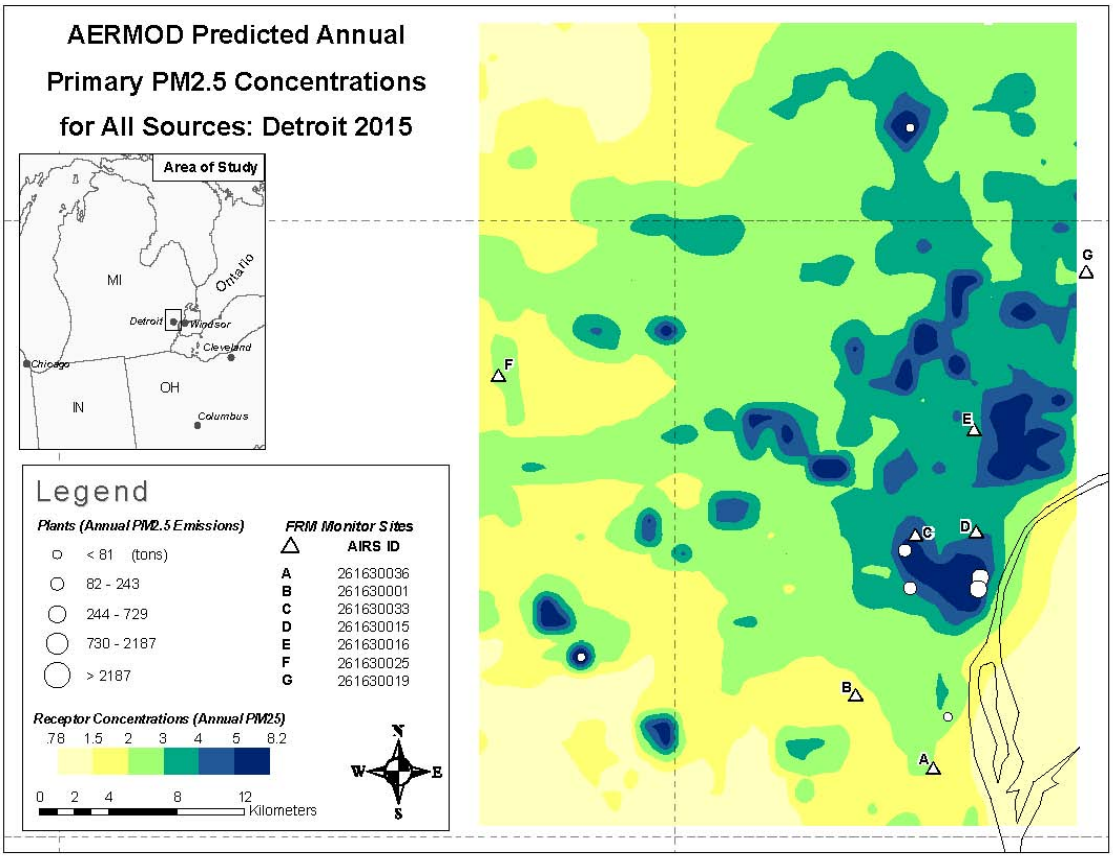

Figure 4-6. Spatial Gradient in Detroit, MI of AERMOD Predicted Annual Primary $PM_{2.5}$ Concentrations (ug/m^3) for All Sources: 2015
Note: Dashed lines reflect the 36km grid cells from regional-scale modeling with CMAQ model.

Source Apportionment Analysis. Table 4-10 summarizes the methods used for three studies within the Detroit Metropolitan Area.

Table 4-10: Summary of Methods Used for Three Studies within the Detroit Metropolitan Area

Study	*Ambient Data Collected*	*Type of Analysis Performed*
Rizzo, M. "A Source Apportionment Analysis of the Dearborn Speciation Trends Network Site." USEPA OAQPS. 2005.	Speciation Trends Network data collected at Dearborn site in Detroit, MI between May 2002 and August 2004 (106 samples)	Used PMF to perform receptor modeling and HYSPLIT for wind trajectory analysis of the receptor modeling results. Compared Dearborn location to four other sites within the Midwestern United States
Center for Air Resources Engineering and Science. Final Report of the Project: Analyses of Midwest PM-Related Measurements. Clarkson University. 2005.	Speciation Trends Network data collected at Dearborn between May 2002 and December 2003 (89 samples); Allen Park between December 2000 and December 2003 (320 observations)	Used PMF to perform receptor modeling; Receptor modeling results were analyzed using meteorological data
Hafner, H., Brown, S., McCarthy, M. Data Analyses for Detroit, Michigan, Air Toxics Data Collected in 2001. Prepared for Lake Michigan Air Directors Consortium. Final Report STI-903553-2557-FR. 2004.	Carbonyl, VOC, Speciated $PM_{2.5}$ (Speciation Trends Network), Metals (TSP), SVOCs and PAHs collected at three Detroit sites (Allen Park, East 7 Mile and 696/Lodge) during 2001	Used PMF to perform receptor modeling; Source contributions represent total contribution from the sum of $PM_{2.5}$, VOC, SVOC

Tables 4-11 through 4-13 show the source apportionment results for the studies listed in Table 4-8.

Table 4-11: PMF Results for Two Sites in Detroit MI and Compared to Other Sites within the Midwestern United States

Source	Contribution (Percent of Total $PM_{2.5}$ in parentheses) ($\mu g/m^3$)				
	Detroit, MI (Dearborn)	*Detroit, MI (Allen Park)*	*Chicago, IL*	*Indianapolis, IN*	*Mayville, WI*
Soil	1.4 (7%)		0.6 (4%)	0.3 (2%)	0.4 (3%)
Industrial (Utility and Petroleum Refineries)	1.7 (8%)	0.7 (4%)	0.2 (1%)	0.7 (4%)	0.5 (4%)
Road Salt	0.8 (4%)	0.4 (2%)	0.5 (3%)		
Fe/Mn (Qualified Diesel)	1.3 (6%)	0.2 (1.1%)	0.1 (0.6%)	0.2 (1%)	1.5 (12%)
Vehicles	5.3 (25%)	5.9 (35%)	4.1 (26%)	5.9 (32%)	2.1 (17%)
Nitrates	3.7 (18%)	3.5 (21%)	3.3 (21%)	2.9 (16%)	3.2 (26%)
Sulfates	4.6 (22%)	5.0 (30%)	5.4 (35%)	6.8 (37%)	3.9 (31%)
Steel (Metals Processing)	1.1 (5%)	0.3 (2%)	0.4 (3%)	1.3 (7%)	
Vegetative Burning	0.9 (4%)	0.9 (5%)	0.9 (6%)	0.2 (1%)	0.9 (7%)
Copper		0.1 (0.6%)	0.1 (0.6%)		
Total $PM_{2.5}$	20.8	16.9	15.5	18.4	12.4

Source: Rizzo, M. 2005. "A Source Apportionment Analysis of the Dearborn Speciation Trends Network Site." USEPA OAQPS.

Table 4-12: Average Source Contributions and Percent of Total Fine Particulate for Two Sites in the Detroit Metropolitan Area

Source	*Average Contribution (Percent of Total $PM_{2.5}$ in parentheses) ($\mu g/m^3$)*	
	Allen Park (Site 261630001)	*Dearborn (Site 261630033)*
Secondary Sulfate	5.1 (30.5%)	8.0 (35.9%)
Secondary Nitrate	3.4 (20.1%)	3.98 (17.9%)
Soil	0.98 (5.9%)	2.23 (10.1%)
Aged Sea and Road Salt	0.46 (2.7%)	0.46 (2.1%)
Iron & Steel	0.84 (5.1%)	2.32 (10.5%)
Spark-ignition Vehicles	3.7 (22.1%)	4.07 (18.4%)
Diesel Vehicles	0.84 (5.1%)	1.13 (5.1%)
Biomass Burning	0.37 (2.2 %)	
Mixed Industrial	0.41 (2.5%)	

Source: Center for Air Resources Engineering and Science. 2005. Final Report of the Project: Analyses of Midwest PM-Related Measurements. Clarkson University.

Table 4-13: Total $PM_{2.5}$, VOC and SVOC Contributions at Five Sites within the Detroit Metropolitan Area

Source	*Average Contribution (Percent of Total $PM_{2.5}$, VOC, SVOC in parentheses) ($\mu g/m^3$)*		
	Allen Park (Site 261630001)	*696/Lodge (Site 261250010)*	*East 7 Mile (Site 261630019)*
Motor Vehicle	1.33 (6%)	1.74 (7%)	1.73 (11%)
Secondary Sulfates/Nitrates	9.63 (36%)	8.70 (36%)	5.40 (35%)
Coal, smelter	2.02 (9%)		
Industrial, oil	2.87 (14%)	0.23 (1%)	
Secondary VOCs	4.18 (19%)	4.88 (21%)	6.44 (41%)
Industrial	2.30 (12%)	3.38 (14%)	1.21 (8%)
Diesel (trains and trucks)	1.15 (6%)	2.04 (9%)	
Background organic carbon/wood burning		2.83 (12%)	
Industrial PAH			0.12 (1%)
Soil			0.56 (4%)

Source: Hafner, H., Brown, S., McCarthy, M. 2004. Data Analyses for Detroit, Michigan, Air Toxics Data Collected in 2001. Prepared for Lake Michigan Air Directors Consortium. Final Report STI-903553-2557-FR.

Common sources seen across all three studies include secondary sulfates and nitrates, diesel emissions, gasoline vehicle emissions, road salt, soil and biomass (vegetative) burning. Secondary sulfates and nitrates consistently account for approximately 40 to 50% of the total fine particulate at the sites in Detroit. Furthermore, the relative similarity in contribution of secondary particles across sites in the Midwest suggests the regional influence of secondarily formed particulate matter. While a large portion of the ambient $PM_{2.5}$ consists of regional sources, local emissions from gasoline and diesel vehicles can contribute a combined total of approximately 25 to 30% of the total fine particulate. This leaves other local point sources potentially contributing approximately 20% of the remaining $PM_{2.5}$ mass. For Detroit, these source categories include road salt which is highly seasonal, soil which has a similar source signature to cement kilns, metals processing facilities, biomass burning and other mixed industrial sources such as local area power generation facilities.

Updated Monitoring Data. Figures 4-7 and 4-8 below illustrate the trend in daily and annual design values for monitors in the Detroit area between 1999 and 2005. The daily and annual design value trends between 1999 and 2003 for sites 261630033 and 261630015—the two violating monitors in Detroit—are upward sloping and slightly declining, respectively. Between 2001 and 2005, these two sites indicate declining annual design values. These trends suggest that Detroit might be closer to attainment of the 1997 standards for the 2020 base case than we projected in our analysis.

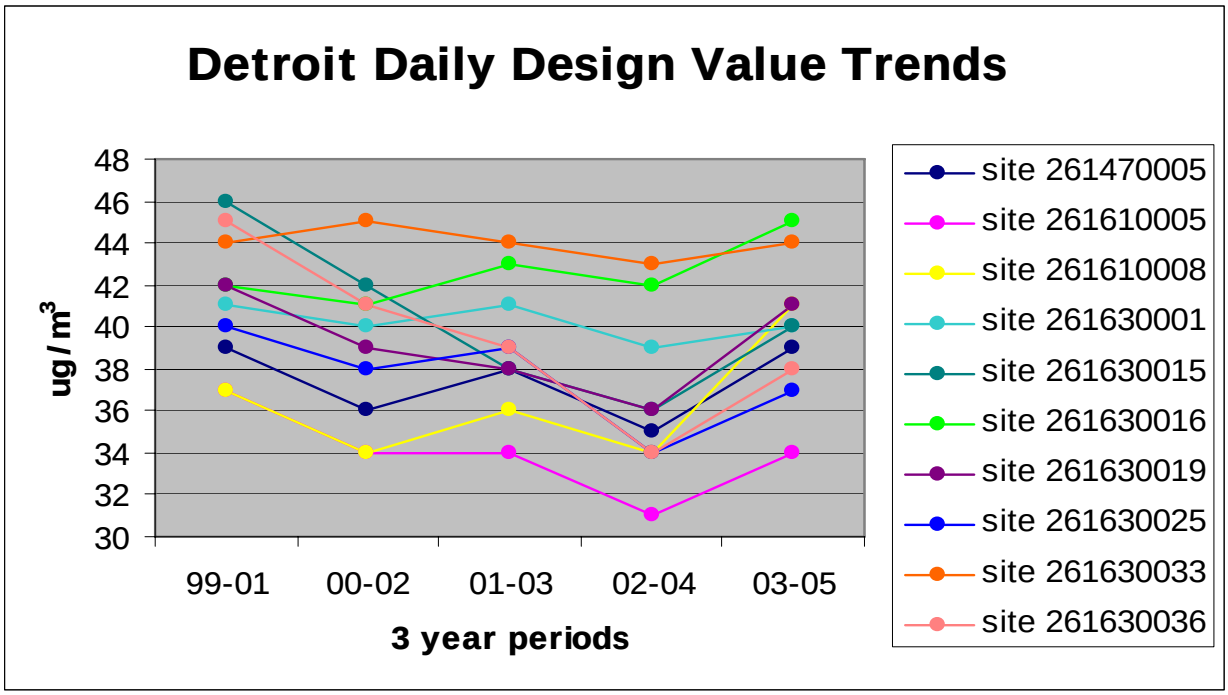

Figure 4-7. Daily Design Value Trend for Monitors in Detroit Metropolitan Area

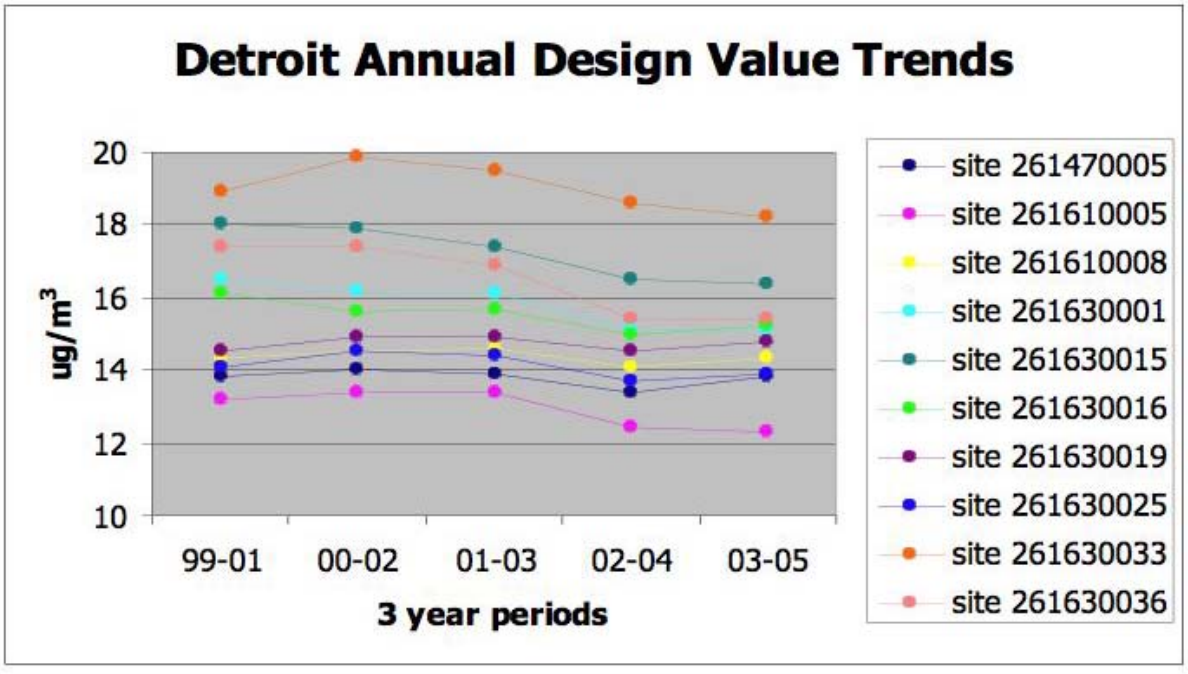

Figure 4-8. Annual Design Value Trend for Monitors in Detroit Metropolitan Area

Conclusions. The monitoring and emissions analysis identifies sources near the priority monitoring sites that may not be well characterized for a local air quality assessment. Thus, we may not have fully captured the benefits of controls in our projected design value analysis. The AERMOD local-scale modeling indicates that controlling local sources of direct $PM_{2.5}$ would have a substantial impact on the design value at the violating monitor—impacts that our projected design values likely do not fully reflect due to the coarse resolution of our CMAQ modeling and uncertainties in the location and characterization of emissions sources. The source apportionment studies highlight the importance of mobile sources and indicates that we may not have fully captured the air quality benefits associated with controlling these sources. Finally, the updated design value data suggests that the air quality trend is improving. Taken together, these data argue that for the purpose of this illustrative analysis, we are presuming that Detroit attains the selected revised and alternative revised standards for the purposes of this analysis.

4.2.2.3 Pittsburgh

Projected Design Values. The air quality modeling analysis projects Allegheny County to violate the annual 1997 standard as well as the daily revised standard and the more stringent alternative revised standards in 2020 under our base case emissions. For our control cases, the analysis projects this area to exceed the annual and daily 1997 standards as well as the revised and more stringent alternative daily standard.

Table 4-14. Projected Design Values for Allegheny County, Pennsylvania

County	Priority Monitor	2020 Basecase Design Values		2020 Control Case: Annual Design Values		2020 Control Case: Daily Design Values	
		Annual	*Daily*	*15/35*	*14/35*	*15/35*	*14/35*
Allegheny	420030064	16.2	52.7	14.12	14.0	46.9	46.7

Monitoring and Emissions Analysis. Monitoring site 420030064 was the monitoring site in the Pittsburgh area that remained nonattainment of both annual and alternative daily standard NAAQS levels. This monitoring site is situated close to several large industrial facilities, including Clairton Coke Works and U.S. Steel Irvin Plant. Pollution roses indicate that most of the highest $PM_{2.5}$ concentrations result when the wind blows from the southeast where the Clairton facilities are located. The speciation profile used in our projection analysis for this site consists of approximately 27% sulfate, 6% nitrate, 10% ammonium, 8% water, 41% organic carbon mass (OCM), 4% elemental carbon (EC), and 4% metals /crustal materials (MCM). Updated speciation data available at the monitor site indicate the following speciation: 29% sulfate, 3% nitrate, 11% ammonium, 9% water, 33% OCM, 11% EC, and 3% MCM. The fractions of sulfate, ammonium, MCM, and total carbon (sum of OCM and EC) are fairly consistent. However, it appears that (1) nitrate was overestimated initially and (2) the OCM/EC split was not representative for this site in that there is considerably more EC than we initially assumed. From a daily standard perspective, more than just one quarter merited attention; most high values occur in either quarter 2 or quarter 3 depending on the definition of ‘high’. Quarter 2

has more values over 65 µg/m^3 (from 1999–2005) but quarter 3 has more values over 35 µg/m^3. Although comparisons of initial versus revised profiles for these two quarters show some inconsistencies (e.g., sulfate appears overestimated in initial analysis in quarter 3 but looks reasonable for quarter 2), both quarters clearly show that EC was significantly underestimated initially (by a factor of about 4).

AERMOD Analysis. Figure 4-9 shows the spatial distribution of direct $PM_{2.5}$ for Pittsburgh resulting from AERMOD modeling of primary $PM_{2.5}$ emissions from a limited set of local sources. These modeling results indicate high annual concentration gradients of primary $PM_{2.5}$ within typical photochemical modeling grid resolutions. Thus, spatial gradients exist within the study area for primary $PM_{2.5}$ with a variety of local sources such as metal manufacturing, coal combustion, and mining being significant contributors to direct PM2.5 at monitoring site 420030064. The modeled local sources of direct PM2.5 emitted roughly 5,700 tons resulting in a total contribution of 1.75 ug/m^3 to the total annual concentrations of PM2.5 at monitoring site 420030064. AERMOD results reflecting July 23rd show a total contribution of 7.89 ug/m^3 from these sources to the daily annual concentrations of PM2.5 at this monitor. Given the limited number of local sources modeled through AERMOD, the modeling results are not comparable to those obtained from CMAQ which included all regional and local sources of direct PM2.5 contributing to this monitoring site.

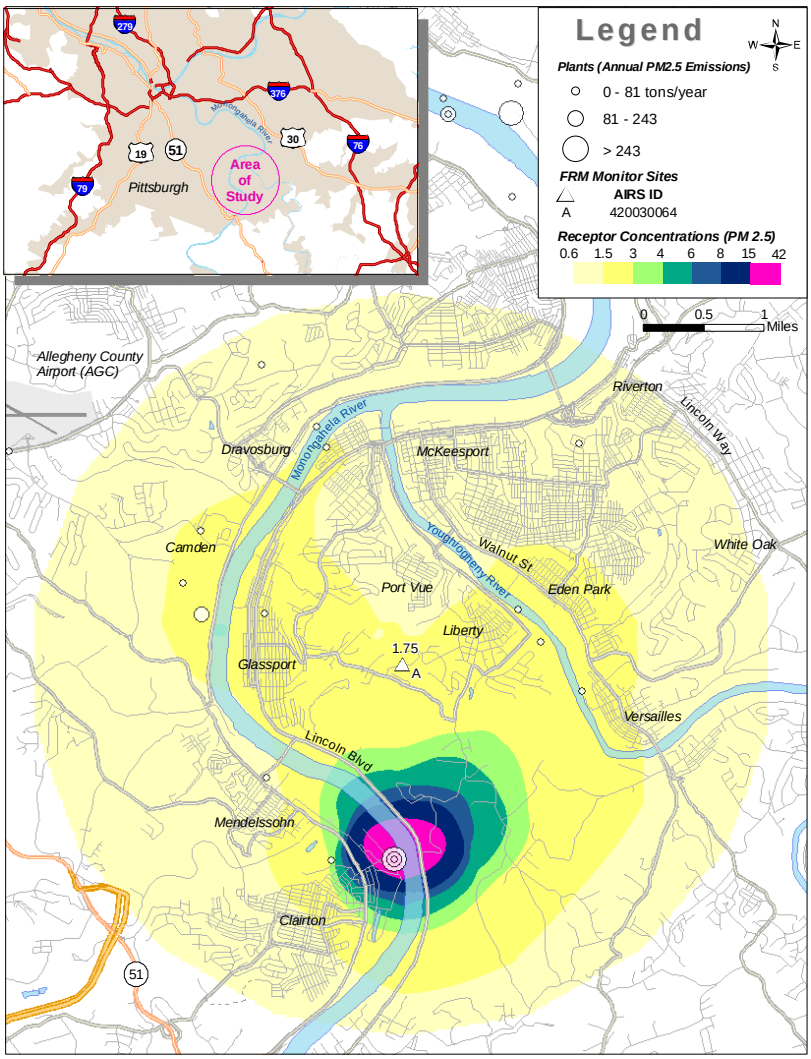

Figure 4-9. Spatial Gradient in Pittsburgh, PA of AERMOD Predicted Annual Primary $PM_{2.5}$ Concentrations (ug/m^3) for Selected Sources: 2015

Updated Design Values. The six-year annual and daily design value trend illustrated in figures 4-10 and 4-11 below for the priority monitor 420030064 indicates a fairly flat trend for the annual design value and a slightly increasing trend for the daily design value. Had we used more current design value data, our 2020 base-case estimates of the daily design value might have been somewhat higher than we projected.

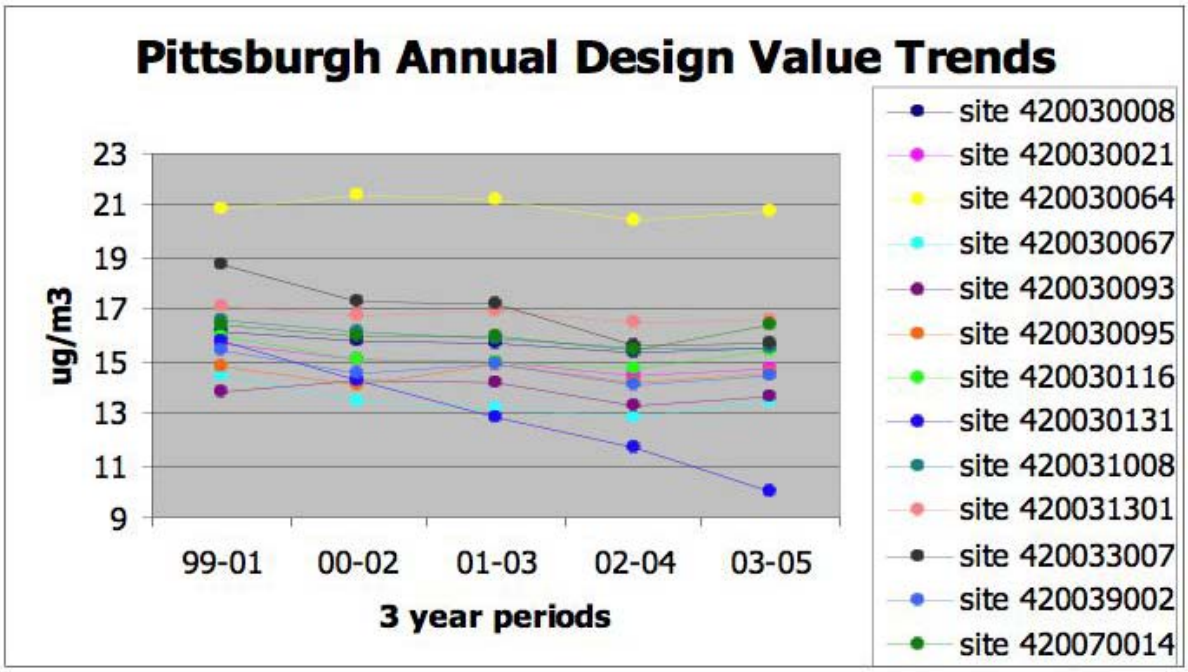

Figure 4-10. Annual Design Value Trend for Monitors in Pittsburgh Metropolitan Area

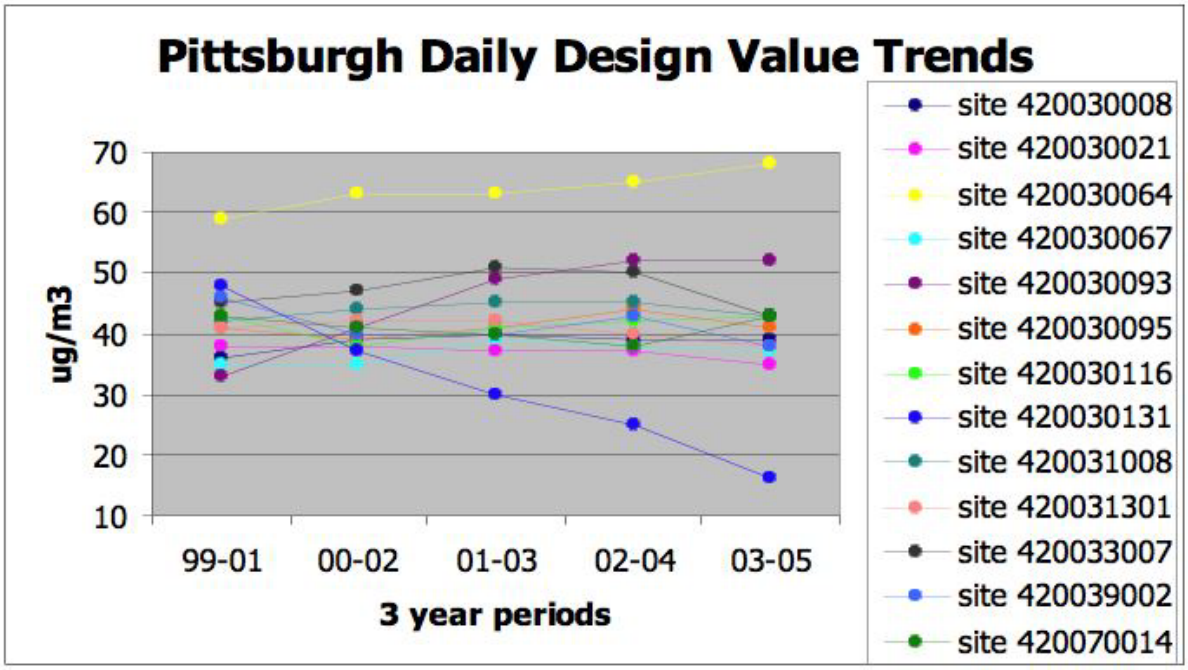

Figure 4-11. Daily Design Value Trend for Monitors in Pittsburgh Metropolitan Area

Conclusions. The non-attainment problem at site 420030064 in Alleghany County is principally associated with meeting the selected revised daily standard. The site is projected to exceed this standard by approximately 10 μg/m^3. The AERMOD local-scale modeling suggests that there is a significant spatial gradient in $PM_{2.5}$ concentrations surrounding several facilities. Consequently, controlling the emissions at these facilities may substantially improve the ability of the county to attain the selected daily standard. However, we cannot make a determination that Pittsburgh would attain with our modeled controls.

4.2.2.4 Libby, Montana

Projected Design Values. Lincoln County (Libby, MT) is projected to attain the 1997 standards in 2020 in both our base and control cases. Lincoln does not reach simulated attainment with the proposed revised daily standard or the alternative revised annual standard in 2020 after the application of emission controls.

Table 4-15. Projected Design Values for Lincoln County, Montana

		2020 Basecase Design Values		2020 Control Case: Annual Design Values		2020 Control Case: Daily Design Values	
County	Violating Monitor	*Annual*	*Daily*	*15/35*	*14/35*	*15/35*	*14/35*
Lincoln	300530018	14.9	42.2	14.5	14.0	41.3	41.3

Monitoring and Emissions Analysis. Libby is a small, isolated northwestern Montana town with no industry that produces a significant level of emissions. The town is in a deep valley and has very cold, long winters. Because of the topography of the area and northern geographic location, this area is susceptible to strong wintertime temperature inversions with low wind speeds that result in poor atmospheric dispersion. Thus, pollutants can become trapped below the inversion, producing high short-term concentrations.

Emissions from woodstoves used during the winter are a large source of directly emitted $PM_{2.5}$ in Libby. Woodstoves are used heavily as there is no natural gas supply into the area and there is an abundance of firewood. The combination of short-term wintertime inversion events and the ubiquity of wood stove emissions results in high daily concentrations of $PM_{2.5}$. In fact, source attribution analyses identify residential woodsmoke as the source of 82% of the wintertime $PM_{2.5}$. Currently, there is an extensive woodstove changeout program being implemented in Libby that is expected to mitigate these contributions.[12]

Almost all high $PM_{2.5}$ values (greater than 35 μg/m^3) occur during the winter months (November through March). The speciation profile for the high quarter (quarter 1) had over 95% of the mass identified as OCM. More robust collocated profiles for the top 25% of quarter 1 shows the OCM component to be closer to 85% with EC being the majority of the difference (i.e., EC was underestimated in the model profile). Summertime wildfire $PM_{2.5}$ impacts are not uncommon in parts of Montana, but this location only has had an average of one day a year flagged for forest fire events.

[12] http://www.lincolncountymt.us/woodstovechangeout/

Wildfire and prescribed burning emissions represent a substantial proportion of total PM2.5 emissions in Lincoln County. EPA estimates annual wildfire and prescribed burning emissions to be approximately 550 tons of $PM_{2.5}$, or about 70% of the total PM2.5 emissions for this county. Because these emissions originate from wildfires and prescribed burning, they are largely stochastic and uncontrollable; therefore, they have complicated our attempts to simulate attainment with the daily design value for this county. Moreover, the manner in which EPA temporally and spatially allocates these emissions is subject to substantial uncertainties that are likely to have implications for our attainment analysis. First, EPA modeled the fires using an average of 5 years of data for monthly allocation, which smoothes peak fire years from any given state. This approach results in EPA's allocation of emissions to winter months (when the 98th percentile design value in Lincoln County occurs) even though the fire emissions in those months are small and more likely should have been zero. Because the fire emissions are not zero in these months, emissions controls on other sources have less percent reduction needed for showing attainment in these counties through modeling. Second, when allocating these emissions to each month, the processing approach assumes that these emissions occur every day of the month at the same rate; this does not represent real wildfire or prescribed burning events that typically are shorter in duration, e.g., a single day to one week. Third, the spatial assignment of fire emissions allocates emissions to forested areas in the state, since the information on where the fires actually occurred was not available in a form we could use for this work.

The combined affect of these uncertainties is to potentially over-state the daily design value. EPA is adjusting these assumptions as it implements its updated 2002 National Emissions Inventory.

Updated Design Values. The six-year design value trend for Lincoln County indicates a slight downward trend in the annual and daily design value for the priority monitor, site 300530018. Thus, had we projected future air quality off of more current 2001-2005 design values our 2020 base case design values would likely be somewhat lower than we projected by using 1999-2003 design values.

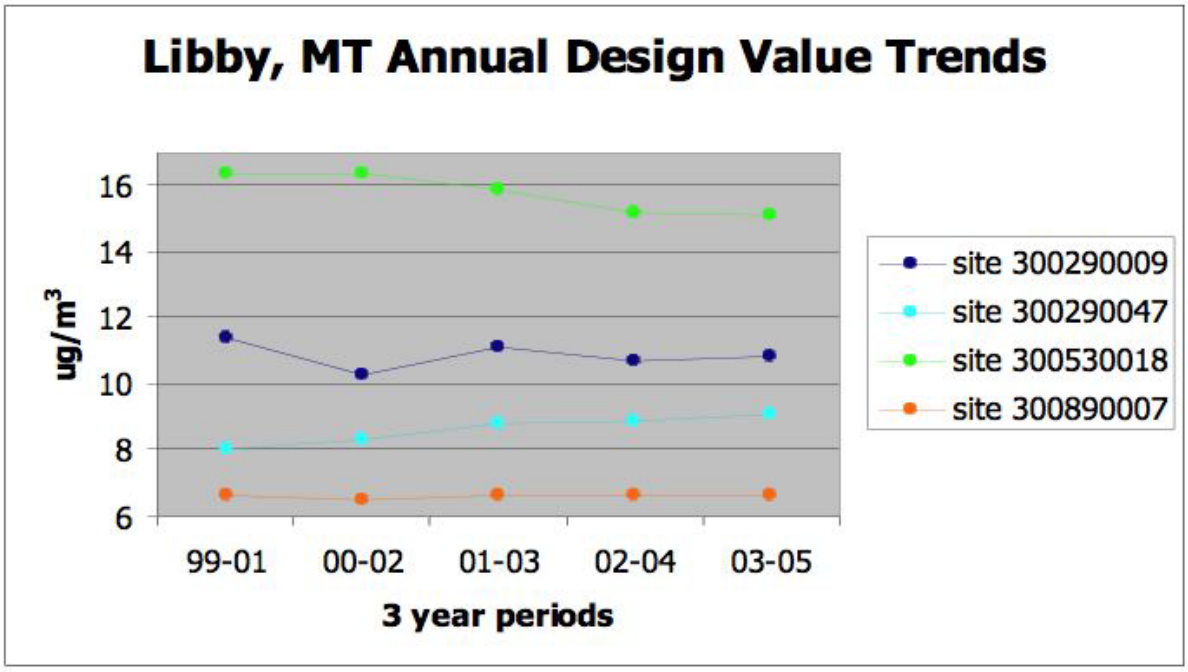

Figure 4-12. Annual Design Value Trend for Monitors in Libby Metropolitan Area

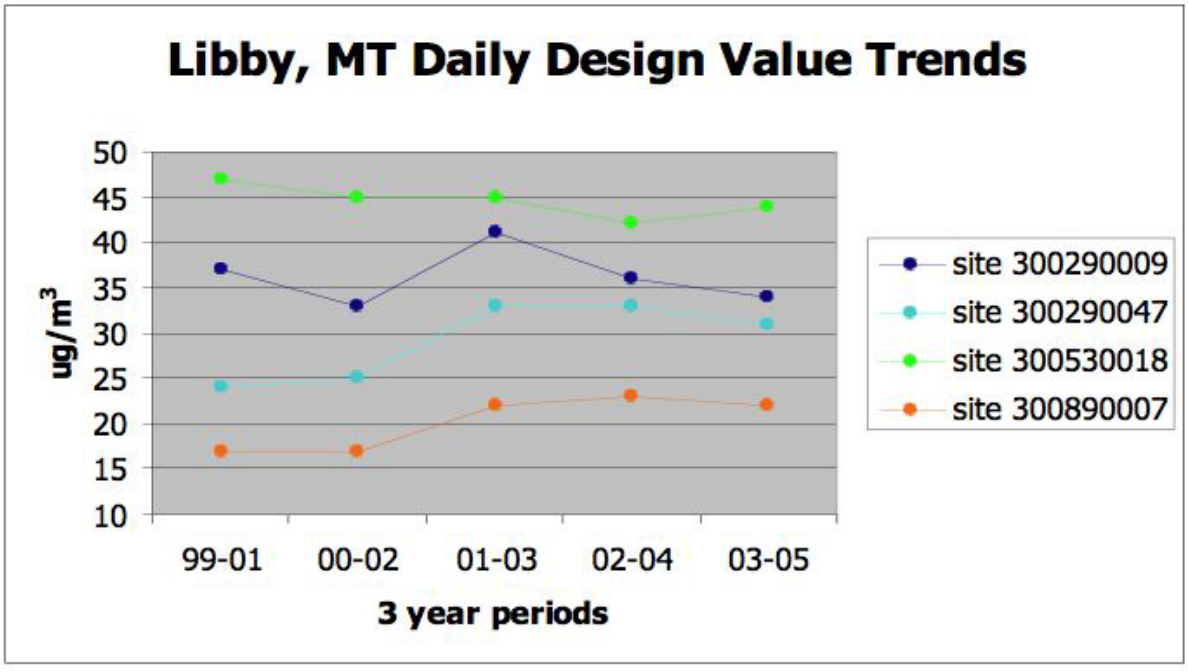

Figure 4-13. Daily Design Value Trend for Monitors in Libby Metropolitan Area

Conclusions. Wintertime inversions coupled with high emissions of PM from woodstoves are key to the nonattainment problem in Libby, MT. Uncertainties in the our analysis, including the tendency to obscure near-field effects, likely understate the effectiveness of our emission

controls (particularly the effectiveness of the wood stove change-out program). The temporal allocation of wildfire emissions is also likely to have overstated the daily design value projections. Finally, the moderately improving trend in design values suggests that we may have slightly over-estimated 2020 annual and daily design values. The balance of the empirical evidence suggests that for the purposes of this illustrative analysis, we presume that Libby will be able to attain the proposed revised standards.

4.2.2.5 Salt Lake City

Projected Design Values. Box Elder, Cache and Salt Lake Counties are projected to attain the 1997 standards in the base and control cases. These three counties do not attain the proposed revised daily standard after applying emission controls.

Table 4-16. Projected Design Values for Salt Lake City, Utah

County	Violating Monitors	2020 Basecase Design Values		2020 Control Case: Annual Design Values		2020 Control Case: Daily Design Values	
		Annual	*Daily*	*15/35*	*14/35*	*15/35*	*14/35*
Box Elder	490030003	8.5	38.4	8.3	8.3	36.9	36.9
Cache	490050004	12.3	51.4	12.0	12.0	44.6	44.6
Salt Lake	490350003	12.2	47.6	11.3	11.3	42.9	42.9

Monitor and Emissions Analysis. There are four $PM_{2.5}$ monitoring sites in Salt Lake county that have similar, high (model) 24-hour design values: site 490350003 has a DV of 57 µg/m3; site 490350012 has a DV of 55 µg/m3; site 490353006 also has a DV of 55 µg/m3; and site 490353007 has a DV of 53 µg/m3. All of the monitoring sites are located in the 500 square mile Salt Lake Valley. This valley is surrounded in every direction except the northwest by steep mountains that at some points rise 7,100 ft from the valley floor's base elevation. It lies nearly encircled by the Wasatch Mountains on the east, the Oquirrh Mountains on the west, the Traverse Mountains to the south, and the Great Salt Lake on the northwest. As with Libby, MT, wintertime temperature inversions contribute significantly to the high $PM_{2.5}$ levels. Over 98% of the site-day exceedances of the 35 µg/m3 level (from 1999 through 2005) occurred during the four month November through February. Speciation monitoring is conducted at site 490353006. A comparison of the modeled profile at that site location for the highest quarter (quarter 1) to the updated actual (collocated) profile for the top 25% days of that quarter revealed that nitrate was underestimated in the initial model runs. The model profile had 27% nitrate and the comparison profile has 32% nitrate. Similar results were obtained in comparisons of modeled data at the other site locations with the speciation site's updated data. Those comparisons also identified an apparent overestimation of the OCM fraction in the model runs (of up to 15%).

Updated Design Values. The three monitors in and around Salt Lake City projected to violate the proposed revised standard (sites 490350003, 49005004, and 490030003), see a flat or slightly upward trend in the annual design value and a downward trend in the daily design value. This

improved trend in daily design value trend suggests that were to have projected daily design values off of these later data that our base case might reflect lower projected daily design values.

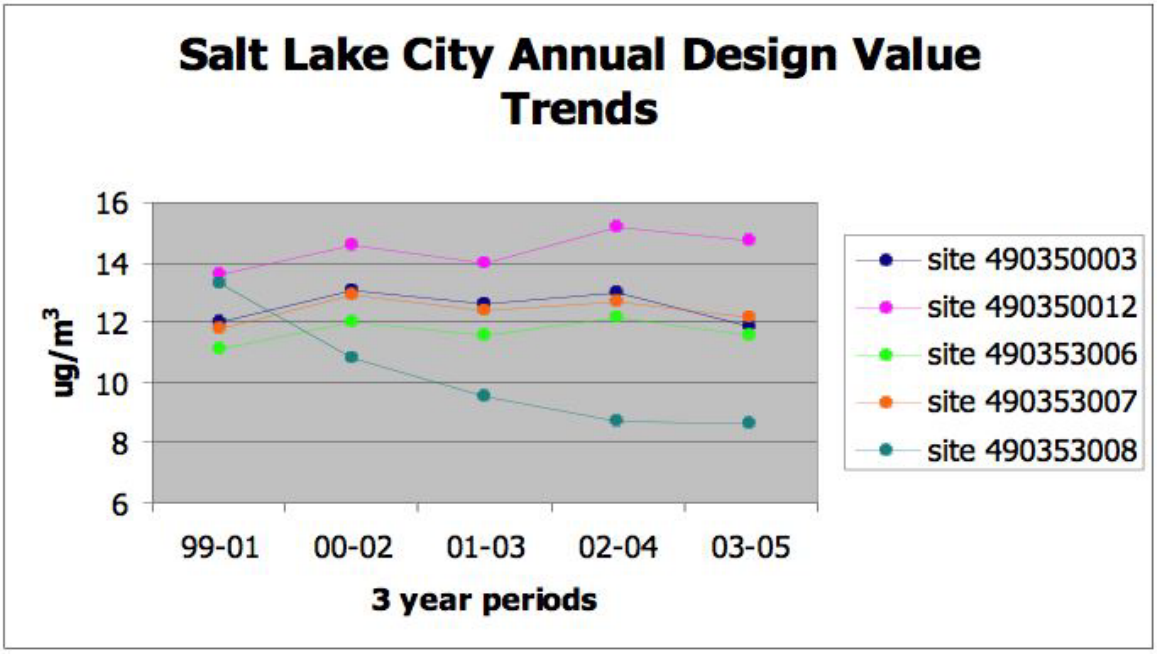

Figure 4-14. Annual Design Value Trend for Monitors in Salt Lake City Metropolitan Area

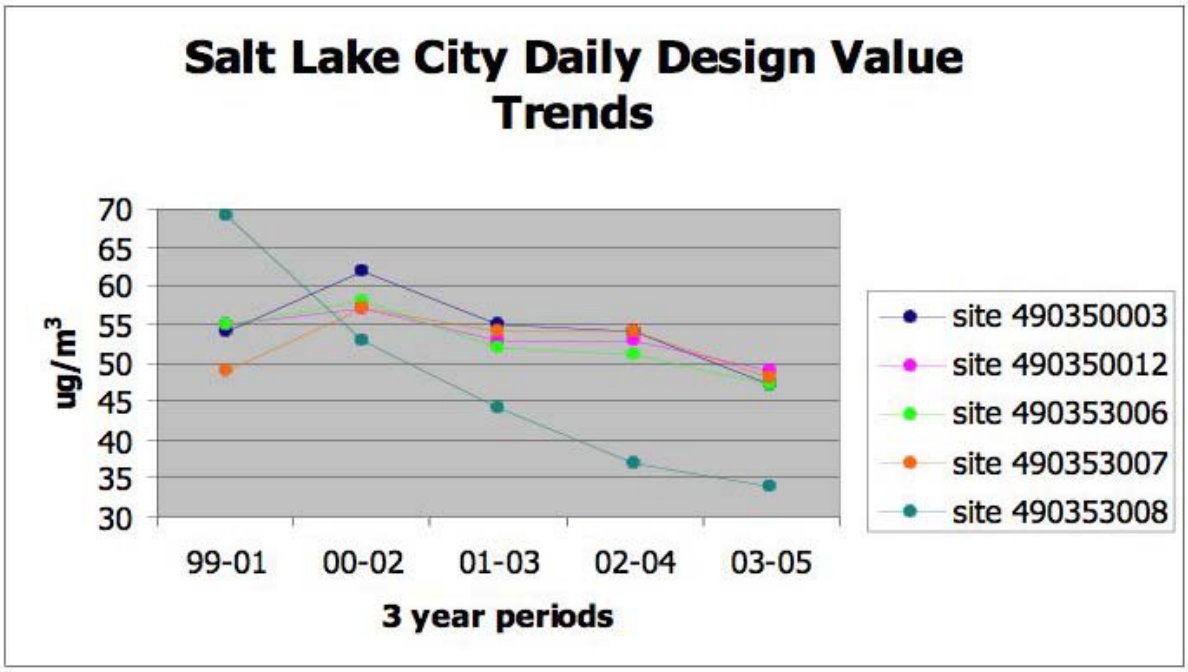

Figure 4-15. Daily Design Value Trend for Monitors in Salt Lake City Metropolitan Area

Conclusions. Wintertime inversions in the Salt Lake Valley contribute to elevated daily design values among the three monitors projected to not attain the proposed revised daily standard. Updated design value data suggests a significant downward trend in the daily design value. While Salt Lake experiences a seasonal air quality problem, we did not model the seasonal application of emission controls, and thus may not have fully captured the air quality improvements associated with our control strategy. Moreover, the relatively coarse-scale air quality modeling may not have adequately reflected the near-field effects of our control strategy. However, the magnitude by which Cache and Salt Lake counties are projected to not attain the proposed revised standard (as much as 15 $\mu g/m^3$) suggests that the area would remain out of attainment after implementing the emission controls we identified for this area in chapter 3. The weight of the empirical evidence suggest that Salt Lake City would not be able to attain the selected standard by 2020 with the emission controls that we have identified.

4.2.2.6 Eugene, Oregon

Projected Design Values. The Lane county monitor is projected to attain the revised and alternative revised annual standard. However, the county does not attain the revised daily standard after the simulated application of emission controls.

Table 4-17. Projected Design Values for Lane County, Oregon

County	Violating Monitors	2020 Basecase Design Values		2020 Control Case: Annual Design Values		2020 Control Case: Daily Design Values	
		Annual	*Daily*	*15/35*	*14/35*	*15/35*	*14/35*
Lane	410392013	12.8	53.0	11.7	11.7	48.0	48.0

Monitoring and Emissions Data. Monitoring site 410392013 is located in Oakridge city, which is southeast of the larger urban areas of Eugene and Springfield. Oakridge is located in a small narrow valley surrounded by steep mountains of the Cascade range. As with Salt Lake City and Libby, the major source of particle pollution in Oakridge, specifically very high concentrations during wintertime, is woodsmoke emissions trapped by temperature inversions. A woodstove changeout program is imminent. There are some local emission sources which may exacerbate the $PM_{2.5}$ problem. The Oakridge site is about 200 meters from highway 58 and about 400 meters from Union Pacific railroad line. Although no nearby speciation data are available (the nearest site is over 125 miles away), a review of the modeled Oakridge profile information was conducted using a surrogate speciation site. Libby, MT (site 300530018) was deemed a similar site due to topography and wood smoke impacts. Based on a comparison of the modeled (interpolated) Oakridge site profile for the high quarter (quarter 1) with actual data from Libby, the following supposition was made. The modeled speciation profile probably overestimated organic carbon and significantly underestimated elemental carbon.

Wildfire and prescribed burning emissions represent a substantial proportion of total PM2.5 emissions in Eugene County. EPA estimates annual wildfire and prescribed burning emissions to be approximately 3,300 tons of $PM_{2.5}$, or about 50% of the total PM2.5 emissions for this county. Because these emissions originate from wildfires and prescribed burning, they are largely stochastic and uncontrollable; therefore, they have complicated our attempts to simulate attainment with the daily design value for this county. Moreover, the manner in which EPA temporally and spatially allocates these emissions is subject to substantial uncertainties that are likely to have implications for our attainment analysis. First, EPA modeled the fires using an average of 5 years of data for monthly allocation, which smoothes peak fire years from any given state. This approach results in EPA's allocation of emissions to winter months (when the 98th percentile design value in Eugene County occurs). Even though the fire emissions in those months are small, they should most likely have been zero. Because the fire emissions are not zero in these months, emissions controls on other sources have less percent reduction needed for showing attainment in these counties through modeling. Second, when allocating these emissions to each month, the processing approach assumes that these emissions occur every day of the month at the same rate; this does not represent real wildfire or prescribed burning events that typically last 1 day to 1 week. Third, the spatial assignment of fire emissions allocates emissions to forested areas in the state, since the information on where the fires actually occurred was not available in a form we could use for this work.

The combined affect of these uncertainties is to potentially over-state the daily design value.

Updated Design Values. The daily and annual design value trends for the priority Eugene monitor (site 410392013) are fairly constant between 1999 to 2005, as illustrated by figures 4-16 and 4-17. Thus, the use of more current 2002-2005 design value measurements to project future air quality would be unlikely to have produced estimates that were significantly different from our existing estimates.

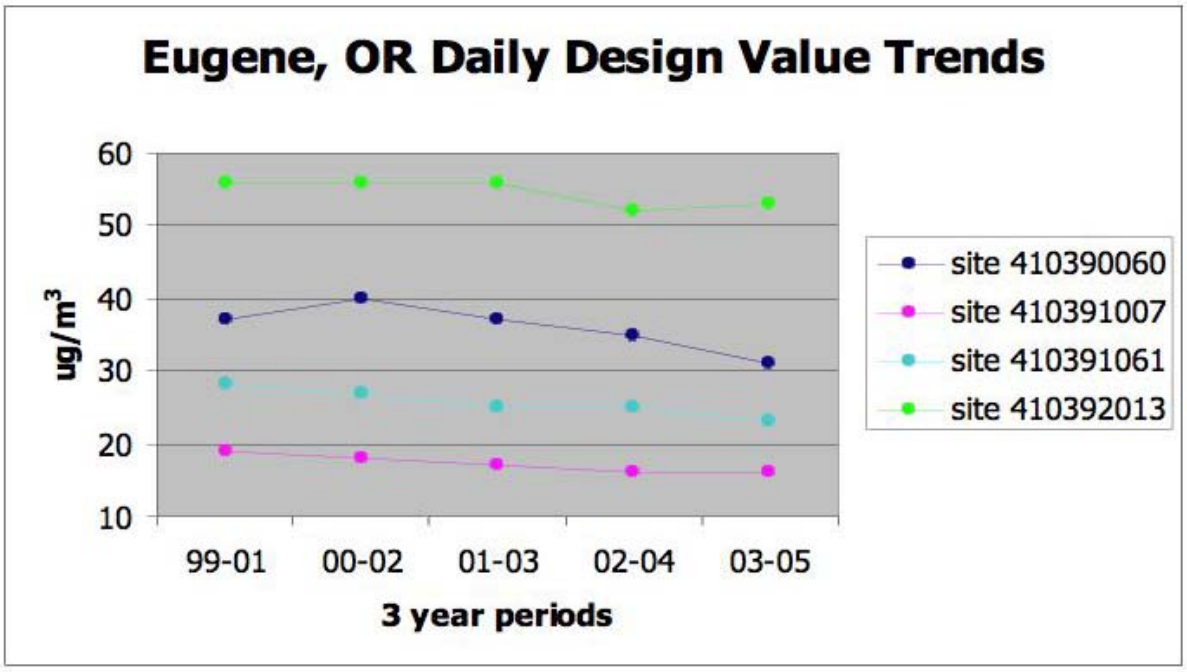

Figure 4-16. Daily Design Value Trend for Monitors in Eugene Metropolitan Area

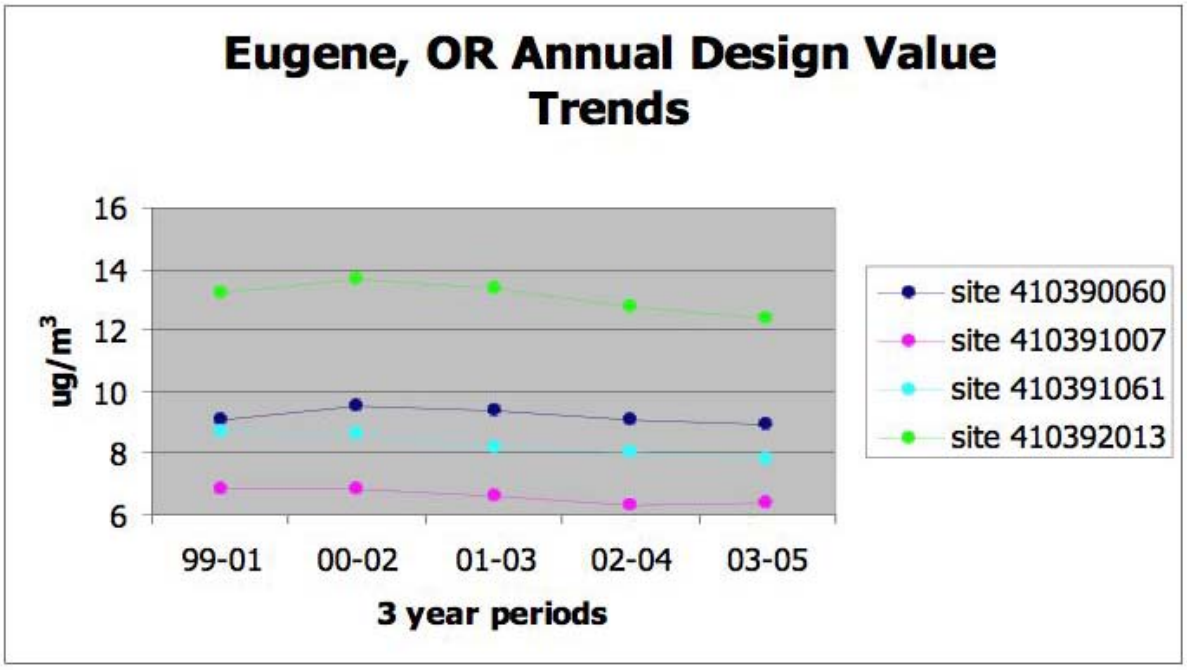

Figure 4-17. Annual Design Value Trend for Monitors in Eugene Metropolitan Area

Conclusions. The $PM_{2.5}$ problem in this county is primarily short-term in nature. Wood smoke emissions, trapped by wintertime inversions, significantly contribute to the projected non-attainment of the selected daily standard. The temporal allocation of wildfire emissions is also likely to have overstated the projected daily design value. The balance of empirical data suggests that for the purposes of this illustrative analysis, we presume that Eugene will attain the revised daily standard in 2020.

Table 4-18: Attainment Determinations for Selected Urban Areas

Urban Area and Standard Alternative	*Annual or Daily Controlling?*	*Projected Nonattainment Increment*	*Final Attainment Determination*
15/35			
Libby, MT	Daily	6 μg/m^3	Attains revised standard
Salt Lake City	Daily	10 μg/m^3	Does not attain revised standard
Eugene, OR	Daily	13 μg/m^3	Attains revised standard
Detroit	Annual	1.75 μg/m^3	Attains revised standard
Pittsburgh	Daily	12 μg/m^3	Does not attain revised standard
Cleveland	Daily	3 μg/m^3	Attains revised standard
14/35			
Detroit	Annual	2.25 μg/m^3	Attains alternative revised standard
Pittsburgh	Daily	12 μg/m^3	Attains alternative revised annual standard. Does not attain revised daily standard.

Within this illustrative attainment analysis, each of these urban areas located outside of California—with the exception of Salt Lake City and Pittsburgh—would attain the revised and more stringent alternative revised standards. As described above, Salt Lake City is a special case due both to its unique topography that exacerbates wintertime inversions, and the magnitude of its projected non-attainment with the revised daily standard. To estimate full attainment cost for this urban area, we have developed extrapolated cost estimates described in Chapter 6.

Implications for the estimation of benefits and costs in these six areas

The determination of attainment and non-attainment for these urban areas has certain implications for our final estimates of full attainment costs and benefits. As we describe above, the empirical data support a determination that certain projected non-attainment areas will likely attain the revised and more stringent alternative standards. As such, we believe that the emission controls that we have applied are sufficient to reach attainment, even if our air quality modeling does not reflect this result. Thus, our cost estimates derived from AirControlNET and supplemental controls in Chapter 6 reflect the cost of a control strategy that reaches simulated attainment with the revised and alternative revised standards for those areas that we note in table 4-18 above. As we describe above, when making an attainment determination for a given area, we adjusted the design value to be equal to the revised standard or more stringent alternative standard. Thus, we use this adjusted design value when performing the benefits assessment in these areas.

4.3 Special Analyses for California

It is well-recognized that California faces a set of unique and exceptionally difficult challenges in meeting national air quality standards, including those for fine particulates. The projected design values above indicate that several California counties will not attain the revised or alternative more stringent standards. California poses a unique $PM_{2.5}$ nonattainment challenge in this RIA due both to the magnitude of this projected nonattainment and the number of California-specific limitations in our data and tools. Both this chapter and the controls analysis in chapter 3 describe four factors that tend to inhibit our ability to simulate attainment in all California counties:

1. We exhausted our emission controls database, which prevented us from controlling all emission sources that contribute to nonattainment.
2. Key uncertainties exist with regard to both emissions inventories and air quality modeling in the West, which may understate the effectiveness of certain controls.
3. The relatively broad spatial resolution of our air quality modeling (36 km) means that emission reductions from local sources are not accurately "captured" by the relevant nonattaining monitors, resulting in possible understatements of local control efficiencies.[13]
4. The magnitude of projected non-attainment is larger than any other state, making the task of simulating attainment much more challenging than elsewhere in the nation.

Even as we recognize the limitations to our models and the magnitude of the state's challenge, we are able to make a number of analytical observations on the nature of California's PM problem. This section describes these limitations and observations in greater depth before providing updated design values for projected non-attainment counties and characterizing the impact that California's emerging emission reduction programs may have on future attainment.

[13] For further discussion of the CMAQ air quality model grid scale and its implications for our controls analysis, see discussion earlier in this chapter.

4.3.1 Understanding the California Nonattainment Problem

Projected Non-Attainment

The scope and magnitude of the $PM_{2.5}$ problem is unique in California. As Chapter 3 describes, our control strategy applied all cost-effective and available direct $PM_{2.5}$, NOx and NH_3 emission controls in the state.[14] As Table 4-17 below shows, our control-case modeling projects twelve counties to violate one or both of the 1997 annual and daily standards in 2020. Our modeling also projects another ten counties to violate the proposed revised daily standard and two counties to violate the alternative revised annual standard. The projected non-attainment is evenly distributed between counties located in the north and south parts of the state. See Chapter 2 for a map illustrating the geographic distribution of projected non-attainment in the baseline with the revised and more stringent alternative standards.

[14] We did not apply NH_3 controls in the San Joaquin Valley because modeling indicates that these controls would not be effective because the area is NO_x limited.

Table 4-17. Projected Design Values for California Counties Projected to Violate the Revised or Alternative Revised Standards.

County Name	Violating Monitor	2020 Base Case Design Values		2020 Control Case: Annual Design Values		2020 Control Case: Daily Design Values	
		Annual	*Daily*	*15/35*	*14/35*	*15/35*	*14/35*
Violates 35 µg Daily Std. Only							
Inyo	060271003	6.0	37.7	5.8	5.8	35.4	35.4
Sonoma	060970003	9.9	38.2	9.2	9.2	34.1	34.1
San Mateo	060811001	10.5	41.6	9.4	9.4	35.7	35.7
San Francisco	060750005	11.4	52.5	9.5	9.5	41.5	41.5
Solano	060950004	11.7	57.3	9.9	9.9	46.6	46.6
Santa Clara	060852003	12.0	52.3	11.2	11.2	47.1	47.1
Sacramento	060670010	12.1	48.3	10.5	10.5	40.0	39.9
Contra Costa	060130002	12.5	61.1	10.9	10.9	51.5	51.5
Butte	060070002	13.0	48.6	11.8	11.8	42.2	42.1
Alameda	060010007	13.2	58.7	11.5	11.5	49.5	49.6
Violates 14 µg Annual Std. and 35 Daily Std.							
Ventura	061112002	14.0	38.7	11.8	11.8	32.7	32.7
Imperial	060250005	14.8	44.9	13.8	13.8	41.5	41.
Violates 15 µg Annual Std. and 35 Daily Std.							
Merced	060472510	15.6	53.1	14.0	14.0	46.3	46.3
San Diego	060731002	15.7	40.1	13.5	13.5	34.0	34.0
San Joaquin	060771002	16.0	52.0	14.1	14.1	44.0	44.0
Stanislaus	060990005	16.2	59.3	14.1	14.1	49.9	49.9
Kings	060310004	16.8	67.6	15.2	15.2	59.5	59.6
Fresno	060190008	19.6	70.4	17.0	17.0	58.2	58.3
Orange	060590007	20.2	40.7	17.9	17.9	35.0	35.0
Tulare	061072002	20.6	73.6	18.5	18.6	64.3	64.3
Kern	060290010	20.8	77.9	18.2	18.2	66.5	66.6
Los Angeles	060371601	23.9	62.7	21.3	21.3	56.8	56.8
San Bernardino	060710025	24.6	65.8	21.1	21.1	56.7	56.8
Riverside	060658001	27.5	73.9	22.3	22.3	61.1	61.1

Emission Inventory and Air Quality Modeling Uncertainties

As described earlier in this chapter, there are some uncertainties associated with the mobile source inventory and specifically, emissions of organic carbon. Several recent source apportionment studies indicate that it is possible that EPA's mobile source inventories understate these emissions. To the extent that EPA emission inventories underestimate these emissions, then the emission control strategies that we applied in California would be less effective in simulating attainment of the revised or alternative more stringent standards.

As described above, CMAQ air quality model performance is generally less robust in the West as compared to the East. CMAQ performs well in predicting the chemistry formation of sulfate and nitrate in the Eastern U.S., where sulfate species are a larger component, and nitrates a smaller component, of PM2.5. However, in the West, and particularly California where nitrate and organics dominate, the modeling system tends to under-predict nitrate. Thus, CMAQ may understate the reductions achieved through application of certain NOx controls. We also used a 36-kilometer grid resolution, which may have the effect of obscuring the air quality effects associated with local-scale emission reductions.

These limitations are especially important for our ability to model attainment in California. Our control strategies for California are heavily weighted toward reductions in both $PM_{2.5}$ and NOx, and CMAQ's ability to reflect accurately NOx reductions in the West is limited. Finally, due to the density of emission sources in California and the large number of monitors projected to violate the 1997 and proposed revised standards, the 36 kilometer grid cell resolution is a limitation which can underestimate the effectiveness of local or urban-area controls. For all these reasons, our modeling of future air quality scenarios and impacts in California is associated with a higher degree of uncertainty than is similar analysis for other parts of the U.S.

4.3.2 Characterizing the Impact of California's Emission Reduction Programs on Future Nonattainment

As mentioned above, California will have to implement an aggressive strategy of both known and innovative control measures to reduce emissions of direct PM and PM precursors to meet the 1997 or the selected revised standards. Later sections in this analysis (see Chapter 6) make reference to the potential benefits and costs of attaining the standards, but the question of *how* California might reach attainment still remains. Our analytical limitations, along with the scope of California's nonattainment problem, prevent us from modeling pathways to full attainment—as we do for other nonattaining areas of the country—but we can summarize some of California's likely strategies and describe how they promise to help the state reach attainment for the 1997 and selected revised standards.

As of this RIA's writing, the areas of California that are likely to face nonattainment issues are in the early stages of analytical modeling to determine the target reductions in PM and its precursors; these are the approximate amounts that are likely to be necessary to reach attainment with the current standard (15 annual/65 daily). While these efforts are focused on meeting the standards already in place, the fact that California has its own, lower standards for ambient $PM_{2.5}$

(12 annual) allows us to characterize the state's control strategies in the context of meeting the revised or more stringent alternative NAAQS.

The California Air Resources Board (CARB) has conducted initial rollback analyses for two areas that are likely to be in nonattainment with future PM standards, the South Coast and the San Joaquin Valley nonattainment areas. These analyses present *preliminary* ideas on the scale of the precursor reductions that would be needed. The estimated range of necessary NOx, SOx, and VOC reductions in both areas is between 45-50% measured from 2005 levels, or between 20-30% measured from 2014 emissions levels (that is, reductions beyond those achieved from fleet turnover to more stringent mobile source standards). No numbers are available for direct PM contributions. It must be emphasized that these numbers present bounding estimates for the State as it considers types of controls and extent of various reduction contributions to make; they are not finalized targets.

Such preliminary figures are informative in that they describe the approximate size of the reductions that are likely necessary, but a great deal of analysis remains to be done with regard to designing an implementation program. Still, CARB and various air districts in the state have already devoted substantial time to understanding and addressing ambient PM emissions, and it is possible to get a sense of what future attainment pathways might look like based on the work that has already been done.

For example, both the South Coast and the San Joaquin Valley are likely to see reductions of NOx and VOCs as a result of the following representative control strategies:

(1) The Goods Movement Action Plan Emission Reduction Plan measures;
(2) Incentive programs to accelerate fleet turnover or retrofit;
(3) New State and Federal mobile emission standards;
(4) State and local regulations mandating retrofit of mobile sources (especially light duty vehicles, heavy duty diesel vehicles, construction equipment, and, in the case of the San Joaquin valley, farm equipment);
(5) Electrification of small combustion sources;
(6) Possibly, some improvements in energy efficiencies associated with the State's climate change action plan.

Other control strategies are also possible, including regulations that tighten limits in existing rules for stationary/area sources as well as development of new rules.

CARB recently approved an "Emission Reduction Plan for Ports and Goods Movement in California," as part of its effort to ensure an environmentally friendly system of goods movement within the state.[15] "Goods movement" encompasses activities including international trade, port activities, logistical services, and short- and long-haul transportation of materials and finished goods. As a policy approach, the goods movement Emission Reduction Plan (ERP) helps focus emissions abatement efforts on areas that have been identified as current and projected significant contributors to air emissions of multiple pollutants, including particulates. The ERP encompasses existing measures and regulations as well as a slate of new or in-progress control

[15] More information can be found at http://www.arb.ca.gov/planning/gmerp/gmerp.htm

strategies, including those that affect marine ships, commercial harbor craft, cargo handling equipment, trucks, locomotives, and some other areas.

We highlight the ERP here to draw attention to the fact that California is already conducting analyses on policies that are designed to achieve emission reductions of magnitudes similar to those that will likely be necessary to reach attainment with various PM standards. For example, if all the elements of the ERP are fully implemented, by 2020 NOx emissions will be reduced by 63% over 2001 levels, SOx by 78%, and diesel PM by 79%.

At this point it is impossible to fully and accurately characterize the impact of these programs on future air quality attainment/nonattainment status in California's various areas. We can, however, make a number of basic observations with regard to potential attainment pathways.

a) *Mobile source emissions will be aggressively targeted.* Given the large contribution of NOx, VOCs, and direct PM (from diesel-powered vehicles) in California, it is evident that any attainment strategy will focus extensively on reducing emissions from the mobile source sector. California has already taken a leadership role in efforts to address port-related emissions, for example.

b) *Costs will be significant.* Given the magnitude and nature of California's PM situation, it is clear that the costs of reducing emissions to move closer to the standard will be significant. In section 6.2 of Chapter 6 we provide an estimate for the cost of California reaching full attainment with the revised and more stringent alternative standards. While there is a significant amount of uncertainty associated with this cost estimate—as explained in Chapter 6—it is apparent that the cumulative cost of reaching attainment would be sizeable. While California has not conducted a formal costing exercise with regard to meeting the PM standards, the costs associated with emission reduction programs, such as the Goods Movement ERP, are of a similar magnitude to what one might expect. For example, CARB estimates the cumulative cost of implementing the Goods Movement ERP strategies by 2020 to be between $6-10 billion in present value dollars.

c) *New and advanced technologies are likely to play a role.* Historical experience has shown that the obligation to meet national air quality standards has created incentives and pressures for technological advances that aid in improving air quality, and it can be anticipated that similar dynamics will exist as California moves to meet the standards. To address the particularly difficult issues the state faces with regard to the PM standards, substantial technological advance is needed, particularly with regard to mobile sector technologies. California has a number of initiatives in place that encourage such advances, ranging from more "conventional" approaches employed in the Goods Movement ERP, to more far-reaching strategies focused on vehicles powered by hydrogen fuel cells.[16] It is difficult to pinpoint the exact catalyst for such change, and in the case of California, there are potentially multiple reasons the State would seek to encourage technological change in the transportation and/or energy sectors. Once again,

[16] See http://hydrogenhighway.ca.gov/ for more information on California's pilot programs involving hydrogen technologies.

it should be stressed that the costs that might be incurred if technological shifts in the mobile sector occurred at a scale large enough to substantially improve air quality would be significant. At the same time, technological change brings with it positive externalities that may serve to reduce overall attainment costs on a nationwide level.

4.3.2 Updated Design Values

There is a clear trend toward decreasing design values over the past six years among California monitors. The figures below illustrate this trend for monitors that in 1999-2001 exceeded either the existing 15 ug/m3 or more stringent alternative 14 ug/m3 annual standard, or the revised 35 ug/m3 daily standard. While we captured some of this improving trend when we projected future air quality off of 1999-2003 design value data, more current data would likely have yielded lower projected 2015 and 2020 baseline design values.

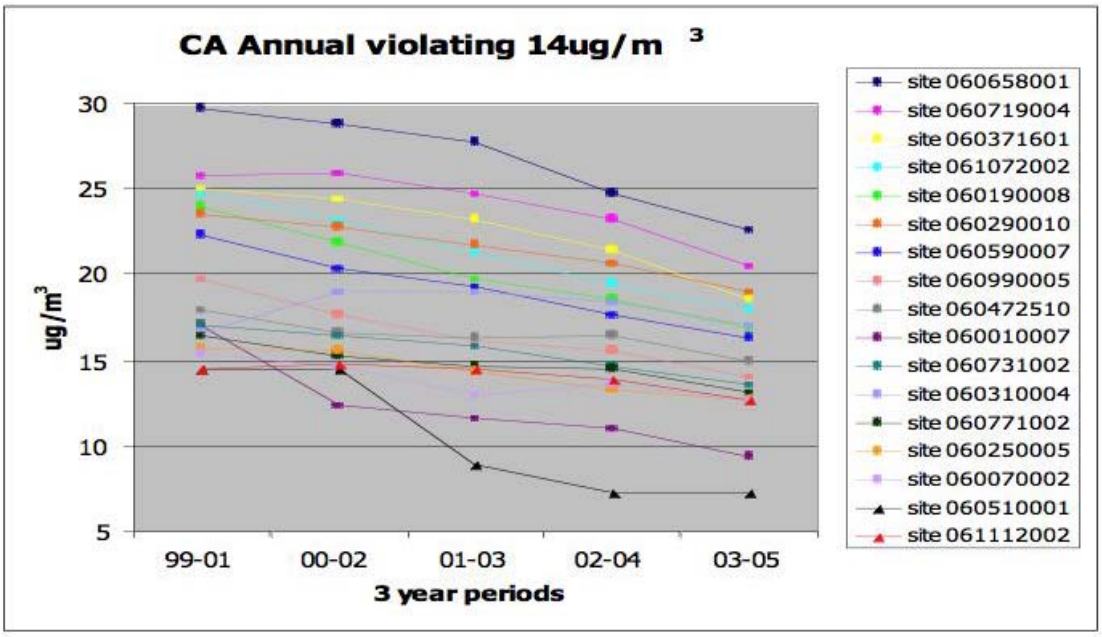

Figure 4-18. Trend in Annual Design Values Among Monitors Currently Violating either 1997 Annual Standard or More Stringent Alternative Annual Standard

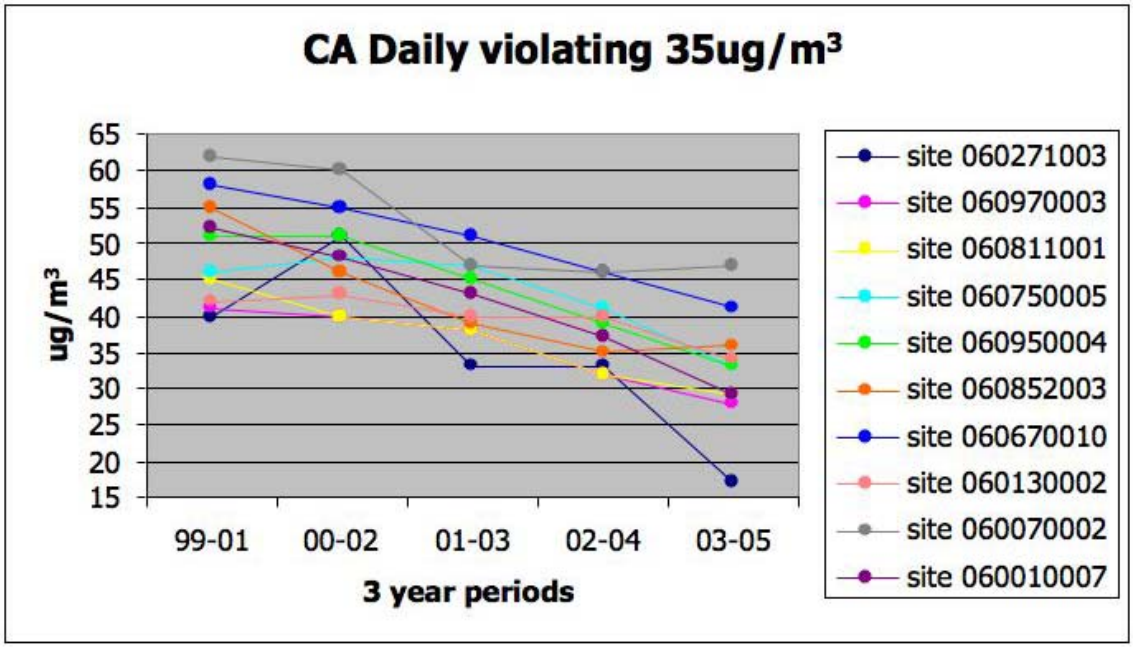

Figure 4-19. Trend in Daily Design Values Among Monitors Currently Violating Revised Daily Standard

4.3.5 Conclusions

As described above, California exhibits a number of unique attributes that made simulating attainment with the revised and more stringent alternative standards especially challenging. California-specific emission inventory and air quality modeling uncertainties made the emission controls analysis more difficult than it was for other projected non-attainment areas. However, the implementation of an ambitious emission control strategy that focuses on an array of emission sources is likely to achieve a substantial improvement in future air quality. An examination of the design value data over the past six years indicates that the overall trend in design values is trending downward—suggesting that many areas may be able to attain the revised daily standard by 2020.

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