

**Draft Regulatory Impact Analysis:
Control of Emissions of Air Pollution from
Locomotive Engines and
Marine Compression-Ignition Engines
Less than 30 Liters per Cylinder**

**Chapter 2
Air Quality and Resulting Health and Welfare
Effects of Air Pollutions from Mobile Sources**

Assessment and Standards Division
Office of Transportation and Air Quality
U.S. Environmental Protection Agency

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CHAPTER 2: Air Quality and Resulting Health and Welfare Effects of Air Pollution from Mobile Sources

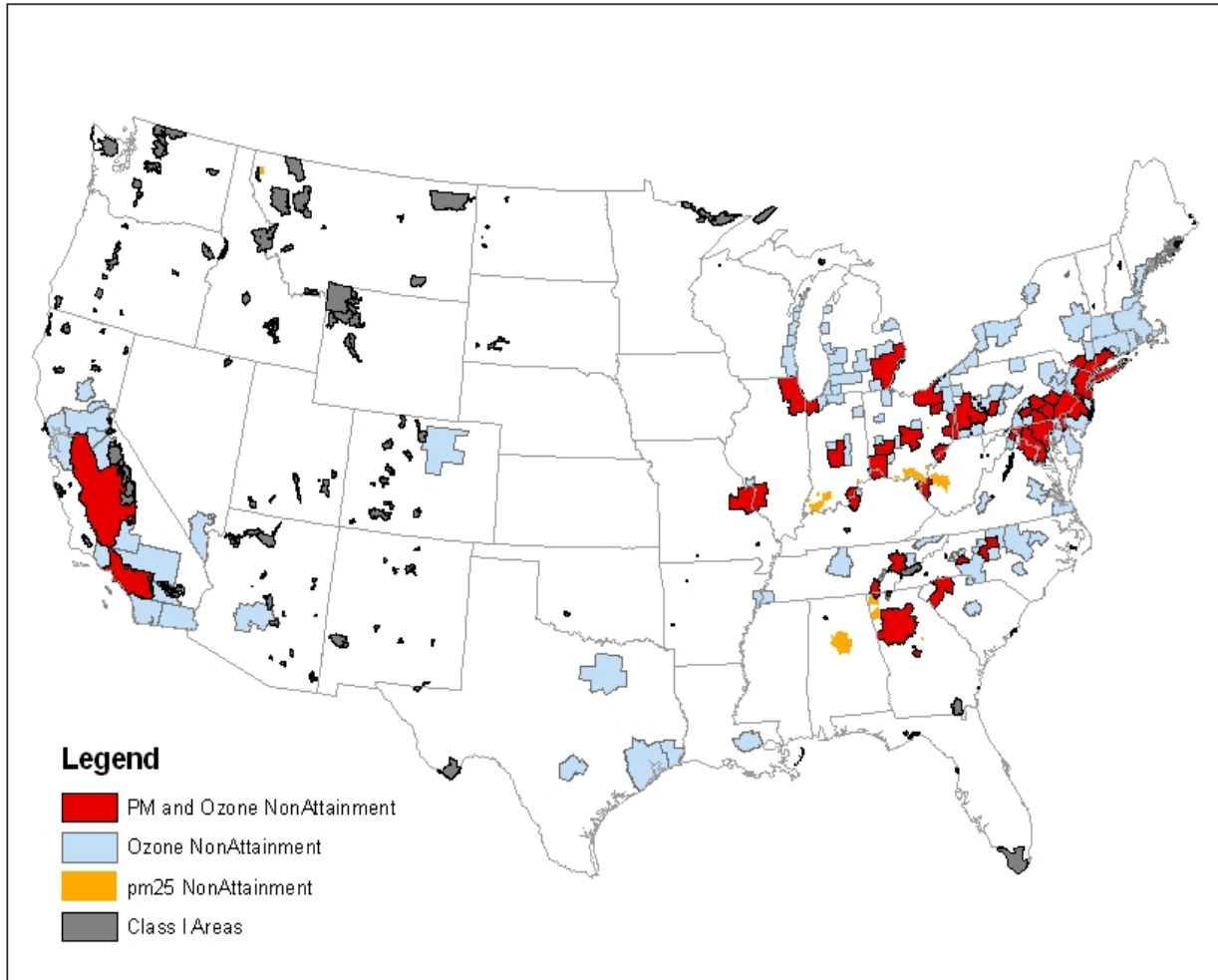
Locomotive and marine diesel engines subject to today's proposal generate significant emissions of particulate matter (PM) and nitrogen oxides (NO_x) that contribute to nonattainment of the National Ambient Air Quality Standards (NAAQS) for PM_{2.5} and ozone. These engines also emit hazardous air pollutants or air toxics which are associated with serious adverse health effects. Emissions from locomotive and marine diesel engines also cause harm to public welfare and contribute to visibility impairment and other harmful environmental impacts across the US. Therefore, EPA is proposing to adopt new standards to control these emissions.

The health and environmental effects associated with these emissions are a classic example of a negative externality (an activity that imposes uncompensated costs on others). With a negative externality, an activity's social cost (the cost borne to society imposed as a result of the activity taking place) exceeds its private cost (the cost to those directly engaged in the activity). In this case, as described in this chapter, emissions from locomotives and marine diesel engines and vessels impose public health and environmental costs on society. However, these added costs to society are not reflected in the costs of those using these engines and equipment. The market system itself cannot correct this externality because firms in the market are rewarded for minimizing their production costs, including the costs of pollution control. In addition, firms that may take steps to use equipment that reduces air pollution may find themselves at a competitive disadvantage compared to firms that do not. To correct this market failure and reduce the negative externality from these emissions, it is necessary to give producers the market signals for the social costs generated from the emissions. The standards EPA is proposing will accomplish this by mandating that locomotives and marine diesel engines reduce their emissions to a technologically feasible limit. In other words, with this proposed rule the costs of the transportation services produced by these engines and equipment will reflect social costs more efficiently.

Today millions of Americans continue to live in areas with unhealthy air quality that may endanger public health and welfare (i.e., levels not requisite to protect the public health with an adequate margin of safety). With regard to PM_{2.5} nonattainment, EPA recently finalized PM_{2.5} nonattainment designations (70 FR 943, Jan 5, 2005) and as of October 2006 there are 88 million people living in 39 areas (which include all or part of 208 counties) that either do not meet the PM_{2.5} NAAQS or contribute to violations in other counties. These numbers do not include the people living in areas where there is a significant future risk of failing to maintain or achieve the PM_{2.5} NAAQS. Currently, ozone concentrations exceeding the level of the 8-hour ozone NAAQS occur over wide geographic areas, including most of the nation's major population centers. As of October 2006 there are approximately 157 million people living in 116 areas (461 full or partial counties) designated as not in attainment with the 8-hour ozone NAAQS. These numbers do not include the people living in areas where there is a future risk of failing to maintain or achieve the 8-hour ozone NAAQS. Figure 2-1 illustrates the widespread nature of these problems highlighting counties which are currently designated in nonattainment for the 8-hour ozone, PM_{2.5}

NAAQS, or for both pollutants. It also shows the location of mandatory class I federal areas for visibility.

Figure 2.1-1 Air Quality Problems are Widespread (October 2006)



Emissions from locomotive and marine diesel engines account for substantial portions of today's ambient $PM_{2.5}$ and NO_x levels [20 percent of total mobile source NO_x emissions and 25 percent of total mobile source diesel $PM_{2.5}$ emissions]. Over time, the relative contribution of these engines to air quality problems will increase unless EPA takes action to reduce their pollution levels. By 2030 locomotive and marine diesel engines could constitute more than 65 percent of mobile source diesel $PM_{2.5}$ emissions and 35 percent of mobile source NO_x emissions.

Under today's proposed comprehensive standards annual NO_x emissions would be reduced by more than 765,000 tons and annual $PM_{2.5}$ emissions by about 28,000 tons in

2030. We estimate that the reduced PM_{2.5} levels would produce nationwide air quality improvements. According to air quality modeling performed in conjunction with this proposed rule, if finalized, all current PM_{2.5} nonattainment areas would experience a resulting decrease in their 2020 and 2030 PM_{2.5} design values (DV). In addition, all 116 monitored mandatory class I federal areas would also experience improved visibility. For the current 39 PM_{2.5} nonattainment areas (annual DVs greater than 15µg/m³) the average population weighted modeled future-year annual PM_{2.5} DVs would on *average* decrease by 0.06 µg/m³ in 2020 and by 0.14 µg/m³ in 2030. The *maximum* decrease for future-year annual PM_{2.5} DVs in these nonattainment areas would be 0.35µg/m³ in 2020 and 0.90µg/m³ in 2030.

This rule would also result in ozone benefits in 2030 for 114 of the current 116 ozone nonattainment areas. According to air quality modeling performed for this rulemaking, the proposed locomotive and marine diesel engine emissions controls are expected to provide nationwide improvements in ozone levels. On a population-weighted basis, the average modeled future-year 8-hour ozone design values would decrease by 0.29 ppb in 2020 and 0.80 ppb in 2030. Within projected ozone nonattainment areas, the average decrease would be somewhat higher: -0.30 ppb in 2020 and - 0.88 ppb in 2030.^A The *maximum* decrease for future-year DVs over the U.S. would be -1.10 ppb in 2020 and -2.90 ppb in 2030

While EPA has already adopted many emission control programs that are expected to reduce both ambient ozone and PM levels, including the Clean Air Interstate Rule (CAIR) (70 FR 25162, May 12, 2005), the Clean Air Nonroad Diesel rule (69 FR 38957, June 29, 2004), the additional PM_{2.5} and NO_x emissions reductions resulting from this locomotive and marine diesel engine rule would be important to states' efforts in attaining and maintaining the Ozone and PM_{2.5} NAAQS near term and in the decades to come.

2.1 Particulate Matter

In this section we review the health and welfare effects of PM_{2.5}. We also describe air quality monitoring and modeling data that indicate many areas across the country continue to be exposed to high levels of ambient PM_{2.5}. Emissions of hydrocarbons (HCs) and NO_x from the engines subject to this proposed rule contribute to these PM concentrations. Information on air quality was gathered from a variety of sources, including monitored PM concentrations, air quality modeling done for recent EPA rulemakings and other state and local air quality information.

^A This is in spite of the fact that NO_x reductions can at certain times in some areas cause ozone levels to increase. Such "disbenefits" are predicted in our modeling, but these results make clear that the overall effect of the proposed rule is positive. The two nonattainment areas that show slight increases in 2030 as a result of the rule are Los Angeles / South Coast Air Basin (0.1 ppb) and Norfolk-Virginia Beach-Newport News (0.8 ppb)

2.1.1 Science of PM Formation

Particulate matter (PM) represents a broad class of chemically and physically diverse substances. It can be principally characterized as discrete particles that exist in the condensed (liquid or solid) phase spanning several orders of magnitude in size. PM is further described by breaking it down into size fractions. PM₁₀ refers to particles generally less than or equal to 10 micrometers (µm). PM_{2.5} refers to fine particles, those particles generally less than or equal to 2.5 µm in diameter. Inhalable (or “thoracic”) coarse particles refer to those particles generally greater than 2.5 µm but less than or equal to 10 µm in diameter. Ultrafine PM refers to particles less than 100 nanometers (0.1 µm). Larger particles tend to be removed by the respiratory clearance mechanisms, whereas smaller particles are deposited deeper in the lungs.

Particles span many sizes and shapes and consist of hundreds of different chemicals. Particles are emitted directly from sources and are also formed through atmospheric chemical reactions; the former are often referred to as “primary” particles, and the latter as “secondary” particles. In addition, there are also physical, non-chemical reaction mechanisms that contribute to secondary particles. Particle pollution also varies by time of year and location and is affected by several weather-related factors, such as temperature, clouds, humidity, and wind. A further layer of complexity comes from particles’ ability to shift between solid/liquid and gaseous phases, which is influenced by concentration and meteorology, especially temperature.

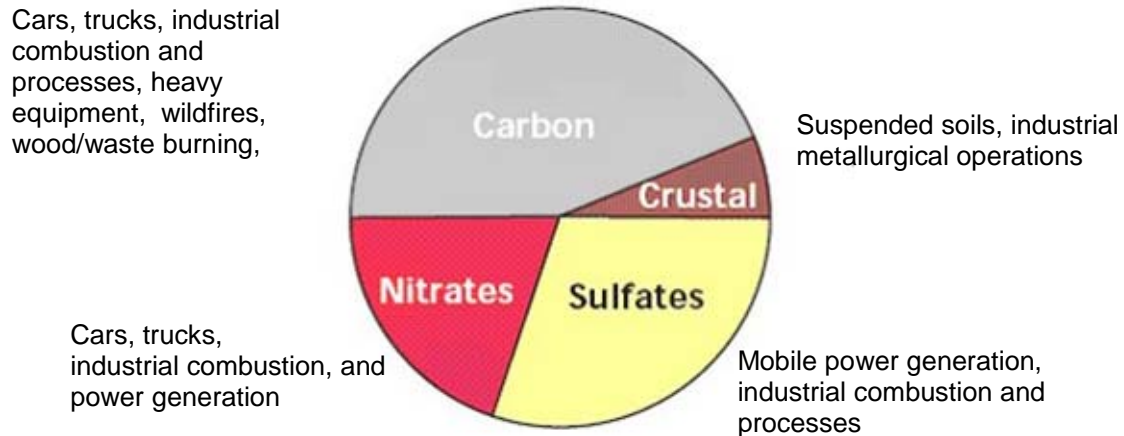
Particles are made up of different chemical components. The major chemical components include carbonaceous materials (carbon soot and organic compounds), and inorganic compounds including, sulfate and nitrate compounds that usually include ammonium, and a mix of substances often apportioned to crustal materials such as soil and ash (Figure 2-2). The different components that make up particle pollution come from specific sources and are often formed in the atmosphere. As mentioned above, particulate matter includes both “primary” PM, which is directly emitted into the air, and “secondary” PM. Primary PM consists of carbonaceous materials (soot and accompanying organics)—emitted from cars, trucks, heavy equipment, forest fires, some industrial processes and burning waste—and both combustion and process related fine metals and larger crustal material from unpaved roads, stone crushing, construction sites, and metallurgical operations. Secondary PM forms in the atmosphere from gases. Some of these reactions require sunlight and/or water vapor. Secondary PM includes:

Sulfates formed from sulfur dioxide emissions from power plants and industrial facilities;

Nitrates formed from nitrogen oxide emissions from cars, trucks, industrial facilities, and power plants; and

Organic carbon formed from reactive organic gas emissions from cars, trucks, industrial facilities, forest fires, and biogenic sources such as trees.

Figure 2-2 National Average of Source Contribution to Fine Particle Levels



Source: The Particulate Matter Report, USEPA 454-R-04-002, Fall 2004. Carbon reflects both organic carbon and elemental carbon. Organic carbon accounts for emissions from a wide range of sources including locomotive and marine diesel engines as well as automobiles, biogenic, gas-powered off-road vehicles, and wildfires. Elemental carbon is formed from both diesel and gasoline powered sources.

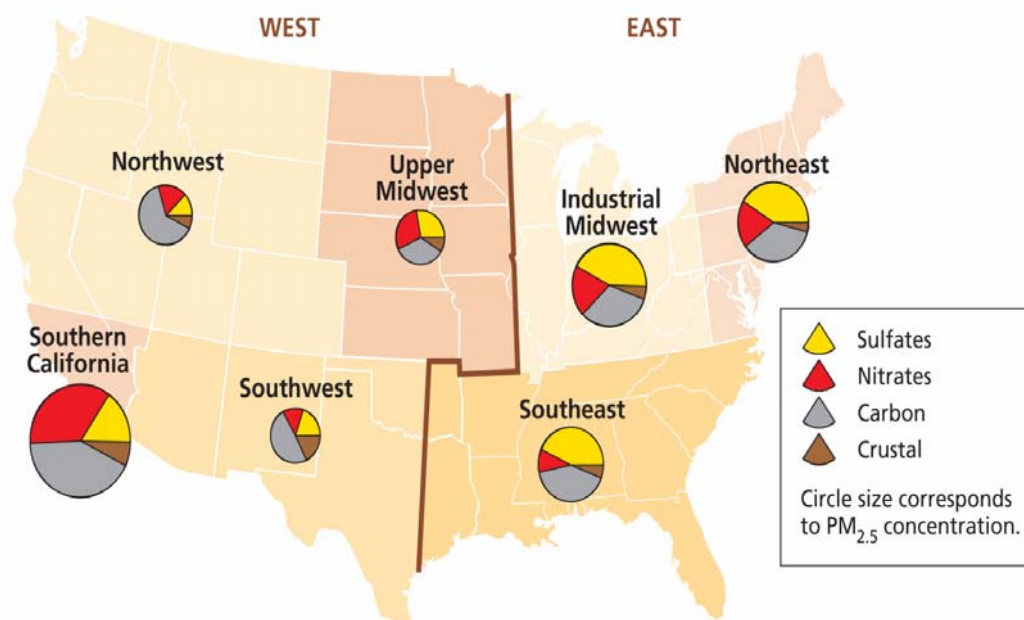
2.1.1.1 Composition of PM_{2.5} in Selected Urban Areas

Note that fine particles can be transported long distances by wind and weather and can be found in the air thousands of miles from where they formed. The relative contribution of various chemical components to PM_{2.5} varies by region of the country, as illustrated in Figure 2-3. Data on PM_{2.5} composition are available from the EPA Speciation Trends Network and the IMPROVE Network, covering both urban and rural areas in numerous regions of the U. S.

These data show that carbonaceous PM_{2.5} makes up the major component for PM_{2.5} in both urban and rural areas in the Western U.S. Carbonaceous PM_{2.5} includes both elemental and organic carbon. Nitrates formed from NO_x also play a major role in the western U.S., especially in the California area where nitrates are responsible for about a quarter of the ambient PM_{2.5} concentrations. Sulfate plays a lesser role in these regions by mass, but it remains important to visibility impairment discussed below. For the Eastern and mid U.S., these data show that both sulfates and carbonaceous PM_{2.5} are major contributors to ambient PM_{2.5} in both urban and rural areas. In some eastern areas, carbonaceous PM_{2.5} is responsible for up to half of ambient PM_{2.5} concentrations. Sulfate is also a major

contributor to ambient $PM_{2.5}$ in the Eastern U.S. and in some areas sulfate makes greater contribution than carbonaceous $PM_{2.5}$.

Figure 2-3 Average $PM_{2.5}$ Composition in Urban areas by Region, 2003



2.1.1.2 Regional and Local Source Contributions to Formation of $PM_{2.5}$

Both local and regional sources contribute to particle pollution. Figure 2-4 shows how much of the $PM_{2.5}$ mass can be attributed to local versus regional sources for 13 selected urban areas. The urban excess is estimated by subtracting the measured $PM_{2.5}$ species at a regional monitor location^B (assumed to be representative of regional background) from those measured at an urban location.

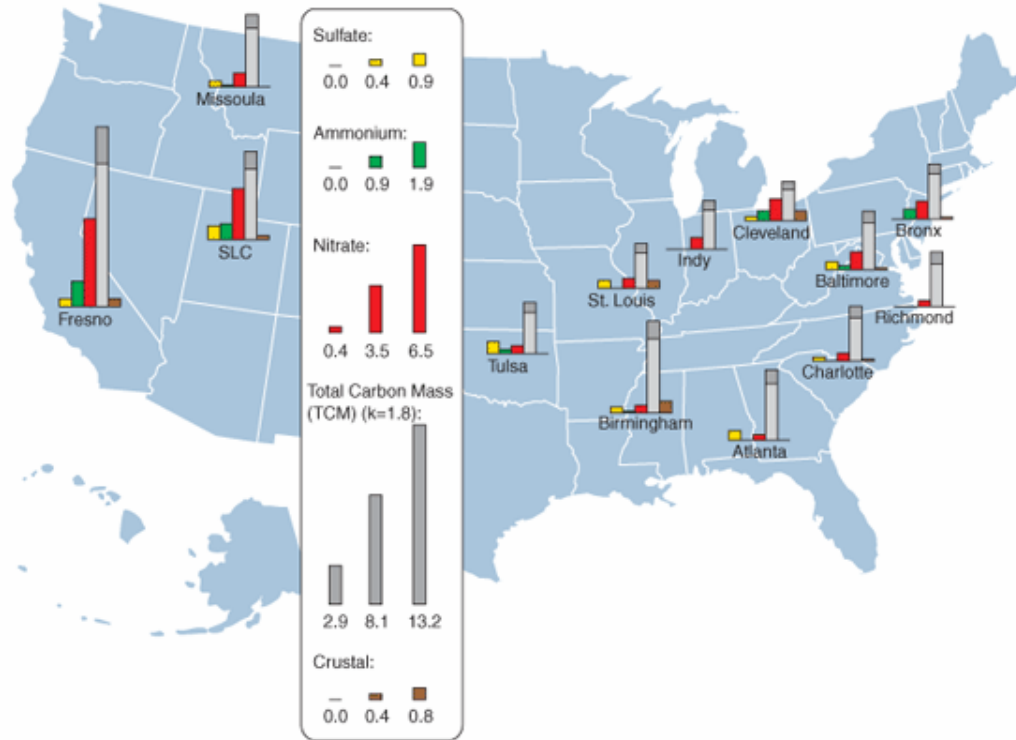
As shown in Figure 2-4, we observe a large urban excess across the U.S. for most $PM_{2.5}$ species but especially for total carbon mass. All of these locations have consistently high urban excess for total carbon mass with Fresno, CA and Birmingham, AL having the

^B Regional concentrations are derived from the rural IMPROVE monitoring network Interagency Monitoring of Protected Visual Environments. See <http://vista.cira.colostate.edu/improve>.

largest observed measures. Larger urban excess of nitrates is seen in the western U.S. with Fresno, CA and Salt Lake City, UT significantly higher than all other areas across the nation. These results indicate that local sources of these pollutants are indeed contributing to the PM_{2.5} air quality problem in these areas.

Urban and nearby rural PM_{2.5} concentrations suggest substantial regional contributions to fine particles in the East. The measured PM_{2.5} concentration is not necessarily the maximum for each urban area. As expected for a predominately regional pollutant, only a modest urban excess is observed for sulfates

Figure 2-4. Estimated "Urban Excess" of 13 Urban Areas by PM_{2.5} Species Component



Note: Total Carbon Mass (TCM) is the sum of Organic Carbon (OC) and Elemental Carbon (EC). In this graph, the light grey is OC and the dark grey is EC. See: Turpin, B. and H-J, Lim, 2001: Species contributions to PM_{2.5} mass concentrations: Revisiting common assumptions for estimating organic mass, Atmospheric Environment, 35, 602-610.

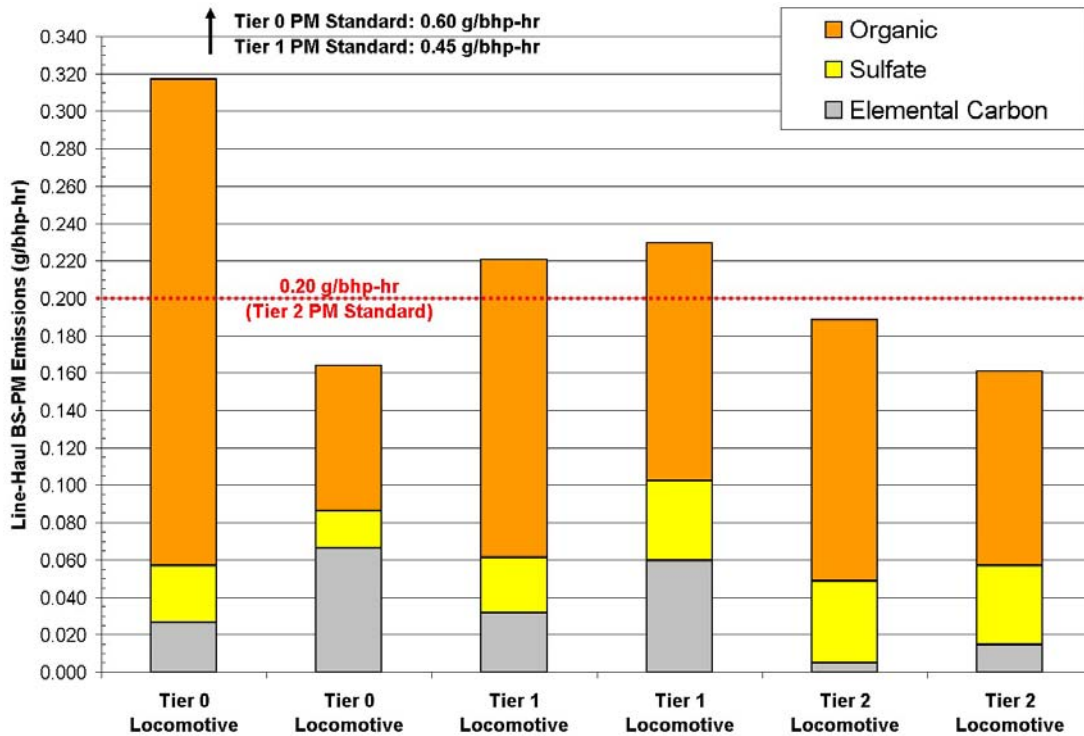
In the East, regional pollution contributes more than half of total PM_{2.5} concentrations. Rural background PM_{2.5} concentrations are high in the East and are somewhat uniform over large geographic areas. These regional concentrations come from emission sources such as power plants, natural sources, and urban pollution and can be transported hundreds of miles and reflect to some extent the denser clustering of urban areas

in the East as compared to the West. The local and regional contributions for the major chemical components that make up urban PM_{2.5} are sulfates, carbon, and nitrates.

2.1.1.3 Composition of PM_{2.5} in Locomotive and Marine Diesel Engines

Locomotive and Marine Diesel engines contribute significantly to ambient PM_{2.5} levels, largely through emissions of carbonaceous PM_{2.5}. As discussed in the previous section, carbonaceous PM_{2.5} is a major portion of ambient PM_{2.5}, especially in populous urban areas. For the medium speed diesel engine commonly used in locomotive and Category 2 marine applications, the majority of the total carbon PM is organic carbon. Locomotive and marine diesels also emit high levels of NO_x which react in the atmosphere to form secondary PM_{2.5} (namely ammonium nitrate). Locomotive and marine diesel engines also emit SO₂ and HC which form secondary PM_{2.5} (namely sulfates and organic carbonaceous PM_{2.5}). Figure 2-5 shows the relative contribution of elemental and organic carbon to PM emissions for six Tier 0, Tier 1, and Tier 2 locomotives (three locomotive engines were 2-stroke while 3 locomotive engines were 4- stroke). This recent data, while limited to six locomotives, suggest that locomotives, regardless of when it was built, tend to emit a very high level of organic carbon PM precisely the type of carbon that appears to be responsible for a high percentage of the urban excess PM_{2.5} species across the US.

Figure 2-5: PM emissions for 6 locomotives tested using 3000 ppm sulfur nonroad diesel fuel.



The proposed locomotive and marine engine standards would reduce emissions of carbonaceous PM. NO_x emissions, a prerequisite for formation of secondary nitrate aerosols, would also be reduced. The proposed standards would also reduce VOC emissions. The emission inventories are discussed in detail in Chapter 3 for primary PM_{2.5} emissions from these sources. This proposed rule would also reduce secondary PM produced from these engines emissions.

As discussed in Sections 2.2 diesel PM also contains small quantities of numerous mutagenic and carcinogenic compounds associated with the particles (and also organic gases). In addition, while toxic trace metals emitted by locomotive and marine diesel engines represent a very small portion of the national emissions of metals (less than one percent) and a small portion of diesel PM (generally much less than one percent of diesel PM), we note that several trace metals of potential toxicological significance and persistence in the environment are emitted by diesel engines. These trace metals include chromium, manganese, mercury and nickel. In addition, small amounts of dioxins have been measured in highway engine diesel exhaust, some of which may partition into the particulate phase; dioxins are a major health concern but diesel engines are a minor contributor to overall dioxin emissions. Diesel engines also emit polycyclic organic matter (POM), including polycyclic aromatic hydrocarbons (PAH), which can be present in both gas and particle phases of diesel exhaust. Many PAH compounds are classified by EPA as probable human carcinogens.

2.1.2 Health Effects of PM Pollution

As stated in the EPA Particulate Matter Air Quality Criteria Document (PM AQCD), available scientific findings “demonstrate well that human health outcomes are associated with ambient PM.”^C We are relying on the data and conclusions in the PM AQCD and PM staff paper, which reflects EPA’s analysis of policy-relevant science from the PM AQCD, regarding the health effects associated with particulate matter.^{1,2} We also present additional recent studies published after the cut-off date for the PM AQCD.^{D3} Taken together this information supports the conclusion that PM-related emissions such as those controlled in this action are associated with adverse health effects. Information on PM-related mortality and morbidity is presented first, followed by information on near-roadway exposure studies, marine ports and rail yard exposure studies.

^C Personal exposure includes contributions from many different types of particles, from many sources, and in many different environments. Total personal exposure to PM includes both ambient and nonambient components; and both components may contribute to adverse health effects.

^D These additional studies are included in the 2006 Provisional Assessment of Recent Studies on Health Effects of Particulate Matter Exposure. The provisional assessment did not and could not (given a very short timeframe) undergo the extensive critical review by EPA, CASAC, and the public, as did the PM AQCD. The provisional assessment found that the “new” studies expand the scientific information and provide important insights on the relationship between PM exposure and health effects of PM. The provisional assessment also found that “new” studies generally strengthen the evidence that acute and chronic exposure to fine particles and acute exposure to thoracic coarse particles are associated with health effects.

2.1.2.1 Short-term Exposure Mortality and Morbidity Studies

As discussed in the PM AQCD, short-term exposure to PM_{2.5} is associated with mortality from cardiopulmonary diseases (PM AQCD, p. 8-305), hospitalization and emergency department visits for cardiopulmonary diseases (PM AQCD, p. 9-93), increased respiratory symptoms (PM AQCD, p. 9-46), decreased lung function (PM AQCD Table 8-34) and physiological changes or biomarkers for cardiac changes (PM AQCD, Section 8.3.1.3.4). In addition, the PM AQCD describes a limited body of new evidence from epidemiologic studies for potential relationships between short term exposure to PM and health endpoints such as low birth weight, preterm birth, and neonatal and infant mortality. (PM AQCD, Section 8.3.4).

Among the studies of effects from short-term exposure to PM_{2.5}, several studies specifically address the contribution of mobile sources to short-term PM_{2.5} effects on daily mortality. These studies indicate that there are statistically significant associations between mortality and PM related to mobile source emissions (PM AQCD, p.8-85). The analyses incorporate source apportionment tools into daily mortality studies and are briefly mentioned here. Analyses incorporating source apportionment by factor analysis with daily time-series studies of daily death indicated a relationship between mobile source PM_{2.5} and mortality.^{4,5} Another recent study in 14 U.S. cities examined the effect of PM₁₀ exposures on daily hospital admissions for cardiovascular disease. They found that the effect of PM₁₀ was significantly greater in areas with a larger proportion of PM₁₀ coming from motor vehicles, indicating that PM₁₀ from these sources may have a greater effect on the toxicity of ambient PM₁₀ when compared with other sources.⁶ These studies provide evidence that PM-related emissions, specifically from mobile sources, are associated with adverse health effects.

In terms of morbidity, short-term studies have shown associations between ambient PM_{2.5} and cardiovascular and respiratory hospital admissions (PM AQCD, p. 9-93), decreased lung function (PM AQCD Table 8-34), and physiological cardiac changes (PM AQCD, Section 8.3.1.3.4).

2.1.2.2 Long-term Exposure Mortality and Morbidity Studies

Long-term exposure to elevated ambient PM_{2.5} is associated with mortality from cardiopulmonary diseases and lung cancer (PM AQCD, p. 8-307), and effects on the respiratory system such as decreased lung function or the development of chronic respiratory disease (PM AQCD, pp. 8-313, 8-314). Of specific importance to this proposal, the PM AQCD also notes that the PM components of gasoline and diesel engine exhaust represent one class of hypothesized likely important contributors to the observed ambient PM-related increases in lung cancer incidence and mortality (PM AQCD, p. 8-318).

The PM AQCD and PM Staff Paper emphasize the results of two long-term studies, the Six Cities and American Cancer Society (ACS) prospective cohort studies, based on several factors – the inclusion of measured PM data, the fact that the study populations were similar to the general population, and the fact that these studies have undergone extensive reanalysis (PM AQCD, p. 8-306, Staff Paper, p.3-18).^{7,8,9} These studies indicate that there are significant associations for all-cause, cardiopulmonary, and lung cancer mortality with

long-term exposure to PM_{2.5}. One analysis of a subset of the ACS cohort data, which was published after the PM AQCD was finalized but in time for the 2006 Provisional Assessment, found a larger association than had previously been reported between long-term PM_{2.5} exposure and mortality in the Los Angeles area using a new exposure estimation method that accounted for variations in concentration within the city.¹⁰

As discussed in the PM AQCD, the morbidity studies that combine the features of cross-sectional and cohort studies provide the best evidence for chronic exposure effects. Long-term studies evaluating the effect of ambient PM on children's development have shown some evidence indicating effects of PM_{2.5} and/or PM₁₀ on reduced lung function growth (PM AQCD, Section 8.3.3.2.3). In another recent publication included in the 2006 Provisional Assessment, investigators in southern California reported the results of a cross-sectional study of outdoor PM_{2.5} and measures of atherosclerosis in the Los Angeles basin.¹¹ The study found significant associations between ambient residential PM_{2.5} and carotid intima-media thickness (CIMT), an indicator of subclinical atherosclerosis, an underlying factor in cardiovascular disease.

2.1.2.3 Roadway-Related Exposure and Health Studies

A recent body of studies reinforces the findings of these PM morbidity and mortality effects by looking at traffic-related exposures, PM measured along roadways, or time spent in traffic and adverse health effects. While many of these studies did not measure PM specifically, they include potential exhaust exposures which include mobile source PM because they employ indices such as roadway proximity or traffic volumes. One study with specific relevance to PM_{2.5} health effects is a study that was done in North Carolina looking at concentrations of PM_{2.5} inside police cars and corresponding physiological changes in the police personnel driving the cars. The authors report significant elevations in markers of cardiac risk associated with concentrations of PM_{2.5} inside police cars on North Carolina state highways.¹² A number of studies of traffic-related pollution have shown associations between fine particles and adverse respiratory outcomes in children who live near major roadways.^{13,14,15}

2.1.2.4 Marine Ports and Rail Yard Studies

Recently, new studies from the State of California provides evidence that PM_{2.5} emissions within marine ports and rail yards contribute significantly to elevated ambient concentrations near these sources¹⁶ and that a substantial number of people experience exposure to fresh locomotive and marine diesel engine emissions, raising potential health concerns. Additional information on near roadway, marine port, and rail yard emissions and potential health effects can be found in Section 2.3.1.4 of this draft RIA.

2.1.3 Attainment and Maintenance of the PM_{2.5} NAAQS

EPA has recently amended the NAAQS for PM_{2.5} (71 FR 61144, October 17, 2006). The final rule, signed on September 21, 2006 and published on October 17, 2006, addressed revisions to the primary and secondary NAAQS for PM to provide increased protection of public health and welfare, respectively. The primary PM_{2.5} NAAQS include a short-term

(24-hour) and a long-term (annual) standard. The level of the 24-hour PM_{2.5} NAAQS has been revised from 65 µg/m³ to 35 µg/m³ to provide increased protection against health effects associated with short-term exposures to fine particles. The current form of the 24-hour PM_{2.5} standard was retained (e.g., based on the 98th percentile concentration averaged over three years). The level of the annual PM_{2.5} NAAQS was retained at 15 µg/m³, continuing protection against health effects associated with long-term exposures. The current form of the annual PM_{2.5} standard was retained as an annual arithmetic mean averaged over three years, however, the following two aspects of the spatial averaging criteria were narrowed: (1) the annual mean concentration at each site shall be within 10 percent of the spatially averaged annual mean, and (2) the daily values for each monitoring site pair shall yield a correlation coefficient of at least 0.9 for each calendar quarter.

With regard to the secondary PM_{2.5} standards, EPA has revised these standards to be identical in all respects to the revised primary standards. Specifically, EPA has revised the current 24-hour PM_{2.5} secondary standard by making it identical to the revised 24-hour PM_{2.5} primary standard and retained the annual PM_{2.5} secondary standard. This suite of secondary PM_{2.5} standards is intended to provide protection against PM-related public welfare effects, including visibility impairment, effects on vegetation and ecosystems, and material damage and soiling.

The proposed emission reductions from this rule would assist PM_{2.5} nonattainment areas in reaching the standard by each area's respective attainment date and assist PM_{2.5} maintenance areas in maintaining the PM_{2.5} standards in the future. The emission reductions will also help continue to lower ambient PM levels and resulting health impacts into the future. In this section we present information on current and future PM_{2.5} levels.

2.1.3.1 Current PM_{2.5} Air Quality

A nonattainment area is defined in the Clean Air Act (CAA) as an area that is violating an ambient standard or is contributing to a nearby area that is violating the standard. In 2005, EPA designated 39 nonattainment areas for the 1997 PM_{2.5} NAAQS based on air quality design values (using 2001-2003 or 2002-2004 measurements) and a number of other factors.^E(70 FR 943, January 5, 2005; 70 FR 19844, April 14, 2005). These areas are comprised of 208 full or partial counties with a total population exceeding 88 million. The 1997 PM_{2.5} nonattainment counties, areas and populations, as of October 2006, are listed in Appendix 2A to this RIA. The 1997 PM_{2.5} NAAQS was recently revised and the 2006 PM_{2.5} NAAQS became effective on December 18, 2006. Nonattainment areas will be designated with respect to the 2006 PM_{2.5} NAAQS in early 2010.

As can be seen in Figure 2-1 ambient PM_{2.5} levels exceeding the 1997 PM_{2.5} NAAQS are widespread throughout the country. States with PM_{2.5} nonattainment areas will be required to take action to bring those areas into compliance in the future. Most PM_{2.5} nonattainment areas will be required to attain the 1997 PM_{2.5} NAAQS in the 2010 to 2015

^E The full details involved in calculating a PM_{2.5} design value are given in Appendix N of 40 CFR Part 50.

time frame and then be required to maintain the 1997 PM_{2.5} NAAQS thereafter.^F The attainment dates associated with the potential nonattainment areas based on the 2006 PM_{2.5} NAAQS would likely be in the 2015 to 2020 timeframe. The emission standards being proposed in this action would become effective between 2008 and 2017. The expected PM_{2.5} and PM_{2.5} precursor inventory reductions from the standards being proposed in this action will be needed by states to attain or maintain the PM_{2.5} NAAQS.

Table 2-1 provides an estimate of the counties violating the 2006 PM_{2.5} NAAQS based on 2003-05 air quality data. The areas designated as nonattainment for the 2006 PM_{2.5} NAAQS will be based on three years of air quality data from later years. Also, the county numbers in the summary table include only the counties with monitors violating the 2006 PM_{2.5} NAAQS. The monitored county violations may be an underestimate of the number of counties and populations that will eventually be included in areas with multiple counties designated nonattainment. Currently more than 106 million people live in counties where monitors show violation of the 2006 standards.

Table 2-1 Counties violating the 2006 PM_{2.5} NAAQS based on 2003-2005 Air Quality Data

Fine Particle Standards: Current Nonattainment Areas and Other Violated Counties		
	Number of Counties	Population
1997 PM _{2.5} Standards: 39 areas currently designated	208	88,394,000
2006 PM _{2.5} Standards: Counties with violating monitors	49	18,198,676
Total	257	106,592,676

2.1.3.2 Current and Projected Composition of Urban PM_{2.5} for Selected Areas

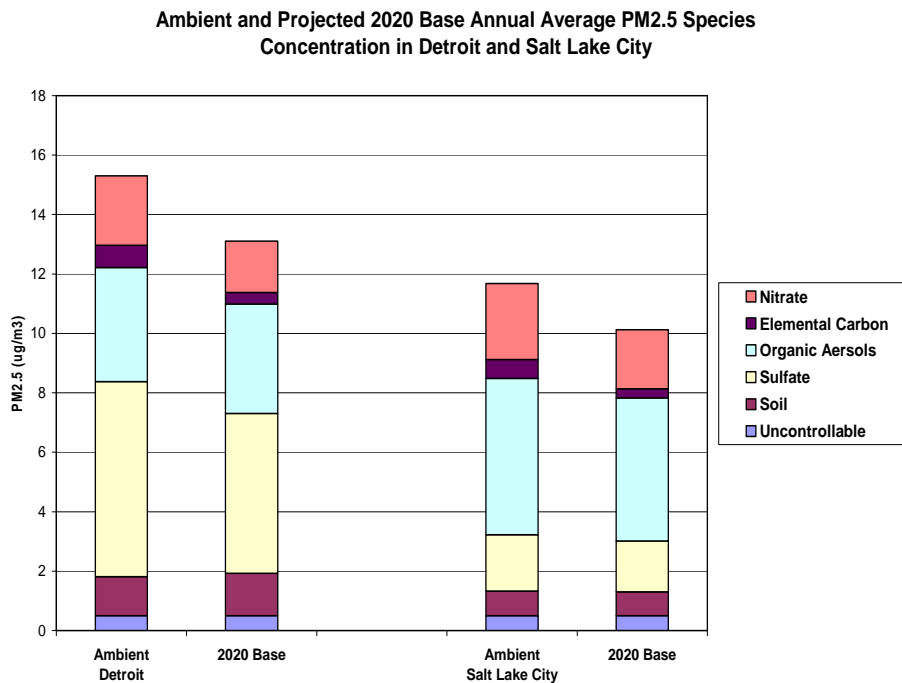
Based on CMAQ modeling for the new PM NAAQS standard, a local perspective of PM_{2.5} levels and composition was developed by EPA to elaborate further on the nature of the PM_{2.5} air quality problem after implementation of the CAIR/CAMR/CAVR rules, the national mobile rules for light and heavy-duty vehicles and nonroad mobile sources, and current state programs that were on the books as of early 2005.¹⁷ As an illustrative example, the PM NAAQS RIA developed a localized analysis of current ambient and future-year speciation for two cities, one in the East (Detroit) and one in the West (Salt Lake City).¹⁸

Figure 2-6 shows projected PM_{2.5} component species concentrations (i.e., sulfate, nitrate, elemental carbon, organic aerosols, crustal, and uncontrollable PM_{2.5}) for current ambient data (5 year weighted average, 1999–2003) and a 2020 regulatory base case with the

^F The EPA finalized PM_{2.5} attainment and nonattainment areas in April 2005. The EPA finalized the PM Implementation rule in Nov. 5, 2005, 70 FR 65984).

addition of the controls mentioned in the previous paragraph. Note that organic aerosols include directly emitted organic carbon and organic carbon particles formed in the atmosphere from anthropogenic sources and biogenic sources. Uncontrollable PM_{2.5} is based upon a 0.5 µg/m³ PM_{2.5} blank mass correction used in the Speciated Modeled Attainment Test (SMAT) approach, in which a number of adjustments and additions were made to the measured species data to provide for consistency with the chemical components retained on the FRM Teflon filter.¹⁹ The analysis provided here specifically looks at one area in the East (Detroit), and one in the West (Salt Lake City).

Figure 2-6. Base Case and Projected PM_{2.5} Component Species Concentrations in Detroit and Salt Lake City



Note: The ambient and projected 2020 base case annual design values above are averages taken across multiple urban area monitors. Thus, while the average 2020 Detroit base case design value reflected above is lower than the projected base case design values at certain Detroit monitors.

Notably, organic aerosols constitute a large fraction of the overall remaining PM_{2.5} mass in Detroit and Salt Lake City. Sulfate is a considerable part of the total PM_{2.5} mass in both cities and is the largest contributor to PM_{2.5} mass in Detroit. Nitrate is a relatively small source of PM_{2.5} for Detroit but nitrate is the second largest contributor to the remaining PM_{2.5} problem in Salt Lake City; the exception is that on higher days, nitrate represents the largest contributor in Salt Lake City. The relatively large contribution of sulfate to PM_{2.5} mass in

Detroit is characteristic of the urban air pollution mixture in the East, while the nitrate contribution to PM_{2.5} mass in Salt Lake City is characteristic of that found in the West.

2.1.4 Source Apportionment Studies of PM_{2.5}

Determining sources of fine particulate matter is complicated in part because the concentrations of various components are influenced by both primary emissions and secondary atmospheric reactions. As described earlier, when attempting to characterize the sources affecting PM_{2.5} concentrations, it is important to note that both regional and local sources impact ambient levels. In the eastern US, regional fine particles are often dominated by secondary particles including sulfates, organics (primary and secondary) and nitrates. These are particles which form through atmospheric reactions of emitted sulfur dioxide, oxides of nitrogen and ammonia, and are transported over long distances. Conversely, local contributions to fine particles are likely dominated by directly emitted particulate matter from sources such as gasoline and diesel mobile sources, including locomotive and marine diesel engines²⁰, industrial facilities (e.g., iron and steel manufacturing, coke ovens, or pulp mills), and residential wood and waste burning.

Development of effective and efficient emission control strategies to lower PM_{2.5} ambient concentrations can be aided by determining the relationship between the various types of emissions sources and elevated levels of PM_{2.5} at ambient monitoring sites. Source apportionment analyses such as receptor modeling are useful in this regard by both qualifying and quantifying potential fine particulate regional and local source impacts on a receptor's ambient concentrations. The goal is to apportion the mass concentrations into components attributable to the most significant sources. Receptor modeling techniques are observation-based models which utilize measured ambient concentrations of PM_{2.5} species to quantify the contribution that regional and local sources have at a given receptor which, in this case, is an ambient monitoring location.²¹ These techniques are very useful in characterizing fine particulate source contributions to ambient PM_{2.5} levels; however, there are inherent limitations including but not limited to the adequacy (e.g., vintage and representativeness) of existing source profiles in identifying source groups or specific sources, availability and completeness of ambient datasets to fully inform these techniques, and current scientific understanding and measured data to relate tracer elements to specific sources, production processes, or activities. Additionally, commingling of similar species from different sources in one "factor" can make it difficult to relate the "factor" to a particular source.

A literature compilation summarizing source apportionment studies was conducted as part of a research and preparation program for the CAIR (EPA, 2005) rule, which was focused on PM_{2.5} transport.²² Literature selected in this compilation represented key source apportionment research, focusing primarily on recent individual source apportionment studies in the eastern U.S. The sources identified are grouped into seven categories: secondary sulfates, mobile, secondary nitrates, biomass burning, industrial, crustal and salt, and other/not identified. Some of these studies are based on older ambient databases and more recent ambient data have shown improvement and reduced levels of ambient PM_{2.5} concentrations across the U.S., especially in the East, which affects the quantitative conclusions one may draw from these studies. Notably, the relative fraction of sulfates has

continued to decrease with the implementation of the acid rain program and removal of sulfur from motor vehicle fuels. More routine monitoring for specific tracer compounds that are unique to individual sources can lead to better separation of blended “factors” such as secondary commingled sulfates and organic aerosols which are more attributed to emissions from vehicles and vegetation. Western studies have focused on sources impacting both high population areas such as Seattle, Denver, the San Joaquin Valley, Los Angeles, San Francisco as well as national parks.^{23,24,25,26,27,28,29,30,31,32} More routine monitoring for specific tracer compounds that are unique to individual sources can lead to better separation of blended “factors” such as secondary commingled sulfates and organic aerosols which are more attributed to emissions from vehicles and vegetation.

As mentioned previously, the sources of PM_{2.5} can be categorized as either direct emissions or contributing to secondary formation. The results of the studies showed that approximately 20 to 60% of the fine particle mass comes from secondarily formed nitrates and sulfates depending on the area of the country, with nitrates predominantly affecting the West, sulfates in the East and a mixture of the two in the Industrial Midwest.

The precursors of these particles are generally gaseous pollutants such as sulfur dioxide or oxides of nitrogen, which react with ammonia in the atmosphere to form ammonium salts. Dominant sources of SO₂ include power generation facilities, which, are also sources of NO_x along with mobile sources including locomotive and marine diesel engines. The result of recent and future reductions in precursor emissions from electrical generation utilities and mobile sources, however, will lead to a reduction in precursor contributions which would aid in limiting the production of secondary sulfates and nitrates. Also, reductions in gasoline and diesel fuel sulfur will reduce mobile source SO₂ emissions.

In addition, secondary organic carbon aerosols (SOA) also make a large contribution to the overall total PM_{2.5} concentration in both the Eastern and Western United States. For many of the receptor modeling studies, the majority of organic carbon is attributed to mobile source emissions (including both gasoline and diesel). While vehicles emit organic carbon particulate, the various organic gases also emitted by these sources react in the atmosphere to form SOA which shows a correlation to the other secondarily formed aerosols due to common atmospheric reactions. As section 2.1.1.3 of this RIA discusses, based on current data, locomotives and larger marine diesel engines which have similar engine characterizations emit a relatively large amount of organic PM. Other common sources of the organic gases which form SOA include vegetation, vehicles, and industrial VOC and SVOC emissions. However, due to some limits on data and a lack of specific molecular markers, current receptor modeling techniques have some difficulty attributing mass to SOA. Therefore, currently available source apportionment studies may be attributing an unknown amount of SOA in ambient PM to direct emissions of mobile sources; concurrently, some secondary organic aerosol found in ambient samples may, as mentioned above, be coming from mobile sources and not be fully reflected in these assessments. Research is underway to improve estimates of the contribution of SOA to total fine particulate mass.

While gaseous precursors of PM_{2.5} are important contributors, urban primary sources still influence peak local concentrations that exceed the NAAQS, even if their overall contributions are smaller. The mixture of industrial source contributions to mass vary across

the nation and include emissions from heavy manufacturing such as metal processing (e.g., steel production, coke ovens, foundries), petroleum refining, and cement manufacturing, among others. Other sources of primary PM_{2.5} are more seasonal in nature. One such source is biomass burning, which usually contributes more during the winter months when households burn wood for heat, but also contributes episodically during summer as a result of forest fires. Other seasonal sources of primary PM include soil, sea salt and road salting operations that occur in winter months. The extent of these primary source contributions to local PM_{2.5} problems varies across the U.S. and can even vary within an urban area. The key for individual areas is to understand the nature of the problem (i.e., determining the relationship between various types of emissions sources and elevated levels of PM_{2.5} at ambient monitoring) in order to develop effective and efficient emission control strategies to reduce PM_{2.5} ambient concentrations through local control program scenarios

2.1.5 Risk of Future Violations

States with PM_{2.5} nonattainment areas will be required to take action to bring those areas into compliance in the future. Based on the final rule designating and classifying 1997 PM_{2.5} nonattainment areas, most of these areas will be required to attain the 1997 PM_{2.5} NAAQS in the 2009 to 2014 time frame and then be required to maintain the PM_{2.5} NAAQS thereafter.

As mentioned in Section 2.1.3, the 1997 PM_{2.5} NAAQS was recently revised (71 FR 61144, October 17, 2006) and the 2006 NAAQS, effective on December 18, 2006, revised the level of the 24-hour PM_{2.5} standard to 35 µg/m³ from the old standards of 65 µg/m³ and retained the level of the annual PM_{2.5} standard at 15 µg/m³.³³ The nonattainment areas will be designated with respect to the 2006 PM NAAQS in early 2010. The attainment dates associated with the potential new PM_{2.5} nonattainment areas would likely be in the 2015 to 2020 timeframe. The emission standards being proposed in this action will become effective between 2008 and 2017 and it is anticipated that the expected PM_{2.5} inventory reductions from the standards being proposed will be useful to states seeking to attain or maintain both the 1997 PM_{2.5} NAAQS as well as the 2006 PM standards.

Even with the implementation of all current state and federal regulations, including the CAIR Rule, the NO_x SIP call, nonroad and on-road diesel rules and the Tier 2 rule, there are projected to be U.S. counties violating the PM_{2.5} NAAQS well into the future. EPA modeling conducted as part of the final PM NAAQS rule projects that in 2015, with all current controls in effect, up to 52 counties, with a population of 53 million people, may not attain some combination of the annual standard of 15 µg/m³ and the daily standard of 35 µg/m³, and that even in 2020 up to 48 counties with a population of 54 million people may still not be able to attain either the annual, daily, or both the annual and daily PM_{2.5} standards.³⁴ This does not account for additional areas that have air quality measurements within 10 percent of the 2006 PM_{2.5} standard. These areas, although not violating the standards, would also benefit from the emissions reductions being proposed, ensuring long term maintenance of the PM NAAQS. For example, in 2015, an additional 27 million people are projected to live in 54 counties that have air quality measurements within 10 percent of the 2006 PM NAAQS. In 2020, 25 million people, in 50 counties, will continue to have air quality measurements within 10 percent of the revised standards. The expected PM_{2.5}

reductions from this proposed in this action will be needed by states to both attain and maintain the PM_{2.5} NAAQS.³⁵

States and state organizations have told EPA that they will need the reductions proposed in this proposed rule in order to be able to attain or maintain the 1997 PM_{2.5} standards as well as necessary to attain the 2006 PM_{2.5} NAAQS.³⁶

In conjunction with this rulemaking, we performed a series of PM_{2.5} air quality modeling simulations for the continental U.S. The model simulations were performed for five emissions scenarios:

- (1) 2001/2002 baseline projection,
- (2) 2020 baseline projection,
- (3) 2020 projection with locomotive/marine diesel engine controls,
- (4) 2030 baseline projection, and
- (5) 2030 projection with locomotive/marine diesel engine controls.

Further discussion of this modeling, including evaluations of model performance relative to predicted future air quality, occur in section 2.1.5.2 of this RIA and also in the AQ Modeling TSD.

The model outputs from the 2001/2002, 2020 and 2030 baselines, combined with current air quality data, were used to identify areas expected to exceed the PM_{2.5} NAAQS in 2020 and 2030. These areas became candidates for being determined to be residual exceedance areas which would require additional emission reductions to attain and maintain the PM_{2.5} NAAQS. The impacts of the locomotive/marine diesel engine controls were determined by comparing the model results in the future year control runs against the baseline simulations of the same year. This modeling supports the conclusion that there are a substantial number of counties across the US projected to experience PM_{2.5} concentrations at or above the PM_{2.5} NAAQS in 2020 and 2030. Emission reductions from locomotive and marine diesel engines will be helpful for these counties in attaining and maintaining the PM_{2.5} NAAQS.

2.1.5.1 Air Quality Modeling Results for PM_{2.5}

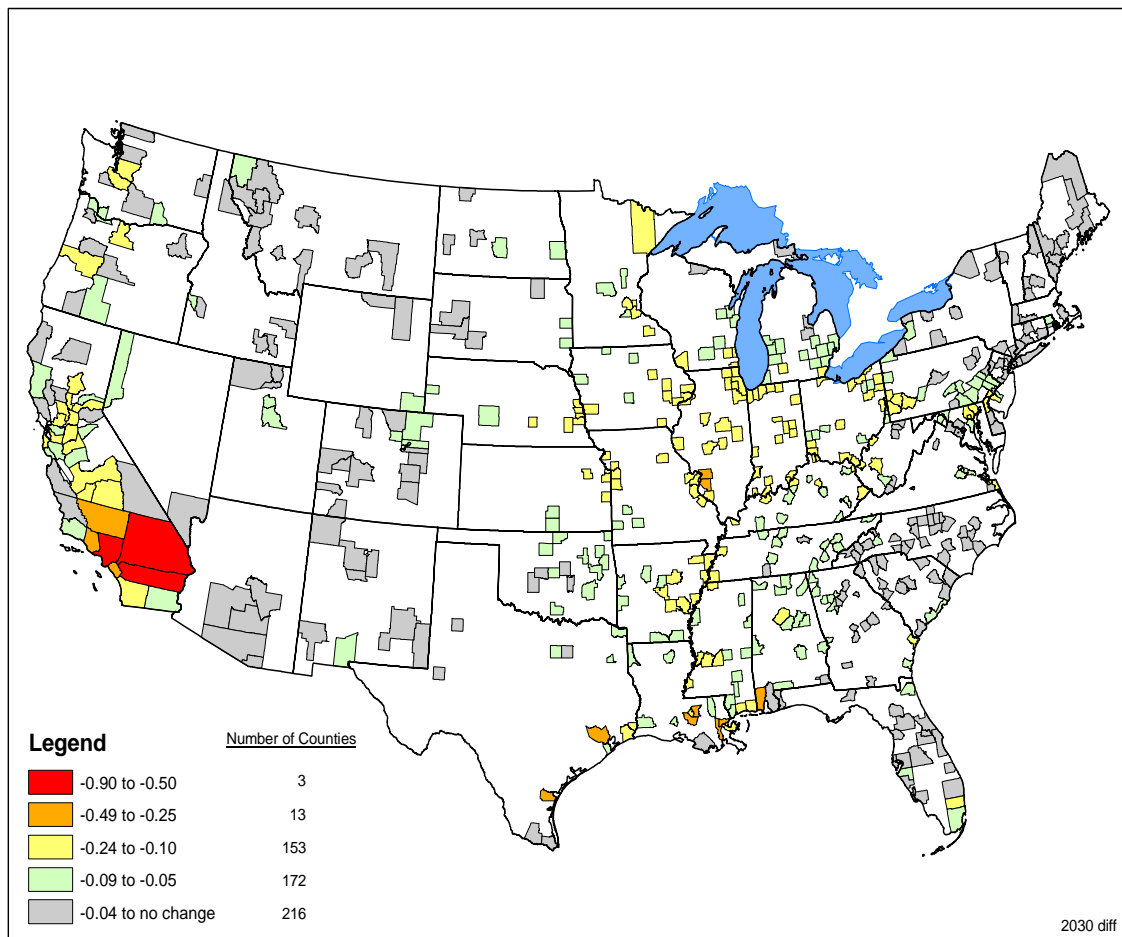
According to air quality modeling performed for this rulemaking, the proposed locomotive and marine diesel engine standards are expected to provide nationwide improvements in PM_{2.5} levels. On a population-weighted basis, the average modeled future-year annual PM_{2.5} design value for all counties is expected to decrease by 0.06 µg/m³ in 2020 and 0.13 µg/m³ in 2030. In counties predicted to have annual design values greater than 15 µg/m³ the average decrease would be somewhat higher: 0.16 µg/m³ in 2020 and 0.36 µg/m³ in 2030. In addition, those counties that are within 10 percent of the annual PM_{2.5} design value would see their average DV decrease by 0.06 µg/m³ in 2020 and 0.23 µg/m³ in 2030. The maximum decrease for future-year annual PM_{2.5} design values in 2020 would be

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$0.35\mu\text{g}/\text{m}^3$ and $0.90\mu\text{g}/\text{m}^3$ in 2030. Note that for the current 39 $\text{PM}_{2.5}$ nonattainment areas the average population weighted modeled future-year annual $\text{PM}_{2.5}$ design values would on average decrease by $0.06\mu\text{g}/\text{m}^3$ in 2020 and by $0.14\mu\text{g}/\text{m}^3$ in 2030.

The geographic impact of the proposed locomotive and marine diesel engine controls in 2030 on annual $\text{PM}_{2.5}$ design values (DV) in counties across the US, can be seen in Figure 2-7. A complete set of maps illustrating the geographic impact of various alternatives explored as part of this rulemaking are available in Air Quality Modeling TSD for this rulemaking.

Figure 2-7 Impact of Proposed Locomotive/Marine controls on annual $\text{PM}_{2.5}$ Design Values (DV) in 2030



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Figure 2-7 illustrates that the greatest emission reductions in 2030 are projected to occur in Southern California where three counties would experience reductions in their PM_{2.5} design values of -0.50 to -0.90 µg/m³. The next level of emission reductions would occur among 13 counties geographically dispersed along the Gulf Coast, near St. Louis, and again Southern California. An additional 325 counties spread across the US would see a decrease in PM_{2.5} DV ranging from -0.05 to -0.24 µg/m³.

Table 2-2 lists the counties with 2020 and 2030 projected annual PM_{2.5} design values that violate the annual standard or are within 10 percent of it. Counties are marked with a “V” in the table if their projected design values are greater than or equal to 15.05 µg/m³. Counties are marked with an “X” in the table if their projected design values are greater than or equal to 13.55 µg/m³, but less than 15.05 µg/m³. These are counties that are not projected to violate the standard, but to be close to it, so the proposed rule will help assure that these counties continue to meet the standard. The current design values of these counties are also listed. Recall that we project future design values only for counties that have current design values, so this list is limited to those counties with ambient monitoring data sufficient to calculate current 3-year design values.

Table 2-2 Counties with 2020 and 2030 Projected Annual PM_{2.5} Design Values in Violation or within 10 percent of the Annual PM_{2.5} Standard. In the Base and Control cases.

State	County	1999 – 2003 Average Design Value (µg/m ³)	2020		2030		2000 Population
			Base	Control	Base	Control	
AL	Jefferson	19.05	V	V	V	V	662,046
CA	Fresno	21.85	V	V	V	V	799,406
CA	Imperial	15.22	X	X	X	X	142,360
CA	Kern	22.74	V	V	V	V	661,644
CA	Kings	18.52	V	V	V	V	129,460
CA	Los Angeles	24.21	V	V	V	V	9,519,334
CA	Merced	16.73	V	V	V	V	210,553
CA	Orange	20.39	V	V	V	V	2,846,288
CA	Riverside	28.82	V	V	V	V	1,545,386
CA	San Bernardino	25.27	V	V	V	V	1,709,433
CA	San Diego	16.44	X	X	V	X	2,813,831
CA	San Joaquin	15.46	V	V	V	V	563,597

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CA	Stanislaus	17.87	V	V	V	V	446,996
CA	Tulare	23.06	V	V	V	V	368,020
GA	Bibb	16.42	X	X	X	X	153,887
GA	Clayton	17.51	X	X	X	X	236,516
GA	Floyd	16.67	X	X	X	X	90,565
GA	Fulton	19.51	V	V	V	V	816,005
IL	Cook	18.00	V	V	V	V	5,376,739
IL	Madison	17.40	V	X	V	V	258,940
IL	St. Clair	16.87	X	X	X	X	256,081
KY	Jefferson	17.07	X		X	X	693,603
MI	Wayne	19.62	V	V	V	V	2,061,161
MT	Lincoln	16.24	X	X	X	X	18,837
NY	New York	16.67	X	X	X	X	1,537,194
OH	Cuyahoga	19.25	V	V	V	V	1,393,977
OH	Hamilton	18.55	X	X	X	X	845,302
OH	Jefferson	18.36	X	X	X	X	73,894
OH	Scioto	19.53	V	V	V	V	79,195
PA	Allegheny	21.17	V	V	V	V	1,281,665
PA	Philadelphia	16.39			X	X	1,517,549
TX	Harris	14.13			X	X	3,400,577
WV	Cabell	17.22			X		96,784
WV	Kanawha	17.75	X	X	X	X	200,072

2.1.5.2 PM Air Quality Modeling and Methods

2.1.5.2.1 Air Quality Modeling Overview

A national scale air quality modeling analysis was performed to estimate future year annual and daily PM_{2.5} concentrations and visibility. These projections were used as inputs to the calculation of expected benefits from the locomotive and marine emissions controls considered in this assessment. The 2001-based CMAQ modeling platform was used as the

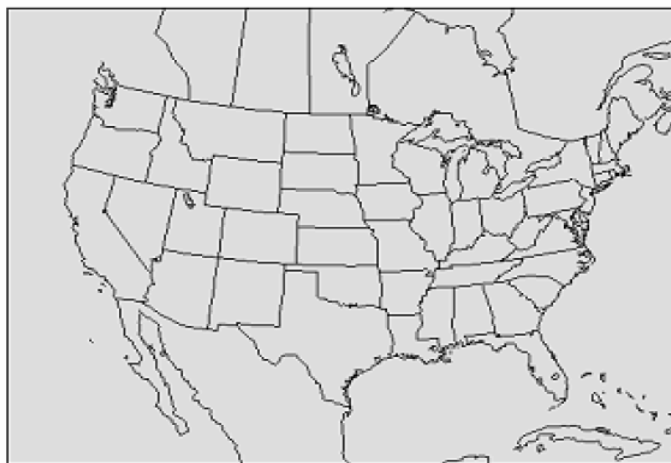
tool for the air quality modeling of future baseline emissions and control scenarios. In addition to the CMAQ model, the modeling platform includes the emissions, meteorology, and initial and boundary condition data which are inputs to this model. The CMAQ model is a three-dimensional grid-based Eulerian 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.).^{37 38 39} Consideration of the different processes 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”.⁴⁰ The latest version of CMAQ (Version 4.5) was employed for this 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.⁴¹

2.1.5.2.2 Model Domain and Configuration

As shown in Figure 2-8 the CMAQ modeling domain encompasses all of the lower 48 States and portions of Canada and Mexico (Figure 2.1-6). 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 2-8. Map of the CMAQ modeling domain.



2.1.5.2.3 *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 Model⁴² 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 maximize the coverage of the ETA model analysis region and completely 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 to derive the specific inputs to CMAQ: horizontal wind components (i.e., speed and direction), temperature, moisture, vertical diffusion rates, and rainfall rates for each grid cell in each vertical layer.⁴³

The lateral boundary and initial species concentrations are provided by a three-dimensional global atmospheric chemistry model, the GEOS-CHEM model.⁴⁴ 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 three-hour intervals and an 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.⁴⁵ 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 above in Chapter 2.

2.1.5.2.4 *CMAQ Evaluation*

An 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 CMAQ modeling system to replicate base year concentrations. In summary, model performance statistics were calculated for observed/predicted pairs of daily/monthly/seasonal/annual concentrations. Statistics were generated for the following geographic groupings: domain wide, Eastern vs. Western (divided along the 100th meridian), and each Regional Planning Organization (RPO) region.⁴⁶ The "acceptability" of model performance was judged by comparing our results to those found in recent regional PM_{2.5} model applications for other, non-EPA studies⁴⁷. Overall, the performance for this application is within the range or better than these other applications. A detailed summary of the 2001 CMAQ model performance evaluation is available within the PM NAAQS RIA, Appendix O.

2.1.5.2.5 *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 eleven emissions scenarios:

2001 base year

2020 base line projection 2020 with projection of impact of primary locomotive/marine control case, low control option, high control option, and a locomotive only control case

2030 base line projection

2030 with projection of impact of primary locomotive/marine control case, low control option, high control option, and a locomotive-only control case

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 2020 and 2030 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.

2.1.5.2.6 *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 Clean Air Interstate Rule (CAIR). The projected annual design values were calculated using the Speciated Modeled Attainment Test (SMAT) approach. The SMAT uses an 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 non-carbon 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 0.5 µg/m³).

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)".⁴⁸ For this latest 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 "Procedures for Estimating Future PM_{2.5} Values for the PM NAAQS Final Rule by Application of the Speciated Modeled Attainment Test (SMAT)" (EPA, 2006).⁴⁹ 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 percent sulfate multiplied by 15.0 µg/m³ of PM_{2.5} equals 3 µg/m³ 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.

2.1.6 Environmental Effects of PM Pollution

In this section we discuss public welfare effects of PM and its precursors including visibility impairment, atmospheric deposition, and materials damage and soiling.

2.1.6.1 Visibility Impairment

Visibility can be defined as the degree to which the atmosphere is transparent to visible light.⁵⁰ Visibility impairment manifests in two principal ways: as local visibility impairment and as regional haze.⁵¹ Local visibility impairment may take the form of a localized plume, a band or layer of discoloration appearing well above the terrain as a result from complex local meteorological conditions. Alternatively, local visibility impairment may manifest as an urban haze, sometimes referred to as a “brown cloud.” This urban haze is largely caused by emissions from multiple sources in the urban areas and is not typically attributable to only one nearby source or to long-range transport. The second type of visibility impairment, regional haze, usually results from multiple pollution sources spread over a large geographic region. Regional haze can impair visibility over large regions and across states.

Visibility is important because it has direct significance to people’s enjoyment of daily activities in all parts of the country. Individuals value good visibility for the well-being it provides them directly, where they live and work, and in places where they enjoy recreational opportunities. Visibility is also highly valued in significant natural areas such as national parks and wilderness areas, and special emphasis is given to protecting visibility in these areas. For more information on visibility see the PM AQCD as well as the 2005 PM Staff Paper.^{52,53}

Fine particles are the major cause of reduced visibility in parts of the United States. To address the welfare effects of PM on visibility, EPA set secondary PM_{2.5} standards which would work in conjunction with the establishment of a regional haze program. The secondary (welfare-based) PM_{2.5} NAAQS was established as equal to the suite

of primary (health-based) NAAQS. Furthermore, Section 169 of the Act provides additional authority to remedy existing visibility impairment and prevent future visibility impairment in the 156 national parks, forests and wilderness areas labeled as mandatory class I federal areas (62 FR 38680-81, July 18, 1997). These areas are defined in Section 162 of the Act as those national parks exceeding 6,000 acres, wilderness areas and memorial parks exceeding 5,000 acres, and all international parks which were in existence on August 7, 1977. In July 1999 the regional haze rule (64 FR 35714) was put in place to protect the visibility in mandatory class I federal areas. A list of the mandatory class I federal areas is included in Appendix 2D. Visibility can be said to be impaired in both PM_{2.5} nonattainment areas and mandatory class I federal areas.

Control of locomotive and marine diesel engine emissions will improve visibility across the nation. The PM and NO_x emissions from locomotive and marine diesel engines subject to this proposed rule either directly emit PM_{2.5} or contribute to formation of secondary PM-precursors and contribute to these visibility effects. This is evident in the PM_{2.5} visibility modeling completed for this rulemaking. In this section we present current information and projected estimates about both visibility impairment related to ambient PM_{2.5} levels across the country and visibility impairment in mandatory class I federal areas. We conclude that visibility will continue to be impaired in the future and the projected emission reductions from this proposed action will help improve visibility conditions across the country and in mandatory class I federal areas. More detailed discussions on visibility are contained in the EPA PM AQCD and the revised PM NAAQS rule RIA.^{54, 55}

2.1.6.1.1 Current Visibility Impairment

The need for reductions in the levels of PM_{2.5} is widespread. Currently, high ambient PM_{2.5} levels are measured throughout the country. Fine particles may remain suspended for days or weeks and travel hundreds to thousands of kilometers, and thus fine particles emitted or created in one county may contribute to ambient concentrations in a neighboring region.⁵⁶

As mentioned above the secondary PM_{2.5} standards were set as equal to the suite of primary PM_{2.5} standards. Recently designated PM_{2.5} nonattainment areas indicate that almost 90 million people live in 208 counties that are in nonattainment for the 1997 PM_{2.5} NAAQS, (see Appendix 2A for the complete list of current nonattainment areas). Thus, at least these populations (plus others who travel to these areas) would likely be experiencing visibility impairment.

As discussed in the Staff Paper (EPA 2004, section 6.2), in mandatory class I federal areas, visibility levels on the 20 percent haziest days in the West are about equal to levels on the 20 percent best days in the East. Despite improvement through the 1990's, visibility in the rural East remains significantly impaired, with an average visual range of approximately 20 km on the 20 percent haziest days (compared to the naturally occurring visual range in the eastern US of about 150 ±45km). In the rural West, the average visual range showed little change over this period, with an average visual range of approximately 100km on the 20 percent haziest days (compared to the naturally occurring visual range in the western US of about 230 ±40km).

In urban areas, visibility levels show far less difference between eastern and western regions. For example, the average visual ranges on the 20 percent haziest days in eastern and western urban areas are approximately 20 km and 27 km, respectively (Schmidt et al., 2005). Even more similarity is seen in considering 4-hour (12 to 4 pm.) average PM_{2.5} concentrations for which the average visual ranges on the 20 percent haziest days in eastern and western urban areas are approximately 26 km and 31 km, respectively (Schmidt et al., 2005).

2.1.6.1.2 Current Visibility Impairment at Mandatory Class I Federal Areas

Detailed information about current and historical visibility conditions in mandatory class I federal areas is summarized in the EPA Report to Congress and the 2002 EPA Trends Report.^{57,58} The conclusions draw upon the Interagency Monitoring of Protected Visual Environments (IMPROVE) network data. One of the objectives of the IMPROVE monitoring network program is to provide regional haze monitoring representing all mandatory class I federal areas where practical. The National Park Service report also describes the state of national park visibility conditions and discusses the need for improvement⁵⁹

The regional haze rule requires states to establish goals for each affected mandatory class I federal area that 1) improves visibility on the haziest days (20% most impaired days), 2) ensures no degradation occurs on the cleanest days (20% least impaired days), and 3) achieves natural background visibility levels by 2064. Although there have been general trends toward improved visibility, progress is still needed on the haziest days. Specifically, as discussed in the 2002 EPA Trends Report, without the effects of pollution a natural visual range in the United States is approximately 75 to 150 km in the East and 200 to 300 km in the West. In 2001, the mean visual range for the worst days was 29 km in the East and 98 km in the West.⁶⁰ Table 2-3 below provides the current visibility deciviews for each of the 116 monitored federal class 1 areas along with the natural background values for each area.

The level of visibility impairment in an area is based on the light-extinction coefficient and a unitless visibility index, called a “deciview”, which is used in the valuation of visibility. The deciview metric provides a scale for perceived visual changes over the entire range of conditions, from clear to hazy. Under many scenic conditions, the average person can generally perceive a change of one deciview. The higher the deciview value, the worse the visibility. Thus, an improvement in visibility is a decrease in deciview value.

2.1.6.1.3 Future Visibility Impairment

Additional emission reductions will be needed from a broad set of sources, including those proposed in this action, as part of the overall strategy to achieve the visibility goals of the Act and the regional haze program.

Modeling conducted for this proposed rule was used to project visibility conditions in 116 of the mandatory class I federal areas across the US in 2020 and 2030 as a result of the proposed locomotive and marine diesel standards. The results indicate that improvements in visibility would occur in all 116 mandatory class I federal areas, although all these areas

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would continue to have annual average deciview levels above background in both 2020 and 2030. Table 2-3 below indicates the current monitored deciview values, the natural background levels each area is attempting to reach, and also the projected deciview values in 2020 and 2030 with and without the proposed standards. In 2030, the greatest visibility improvement due to this proposed rule would occur at Agua Tibia (-0.24 deciview) located in San Diego County, California followed by San Georonio (-0.22 deciview) in San Bernadino County, California.

Table 2-3 Current and Future projected Visibility Conditions With and Without Proposed Locomotive and Marine Diesel Rule in Mandatory Class I Federal Areas (Annual Average Deciview)

Annual Results		DeciViews ^a					
Site name	state	1998-2002 Baseline Visibility (deciviews)	2020base case without controls	2020 base case with proposed controls	2030 base case without controls	2030 base case with proposed controls	Natural Background (deciviews)
Acadia	ME	22.7	12.84	12.83	12.88	12.87	11.5
Agua Tibia	CA	23.2	16.03	15.94	15.98	15.74	7.2
Anaconda - Pintler	MT	18.0	7.53	7.52	7.53	7.51	7.9
Arches	UT	12.3	8.19	8.18	8.22	8.19	7.3
Badlands	SD	12.0	11.42	11.39	11.38	11.32	7.0
Bandelier	NM	17.3	8.63	8.62	8.66	8.63	7.3
Big Bend	TX	13.2	12.16	12.15	12.17	12.15	7.0
Black Canyon of the Gunnison	CO	18.4	6.84	6.83	6.83	6.81	6.9
Desolation	CA	11.6	7.63	7.61	7.59	7.55	7.1
Bob Marshall	MT	14.2	9.25	9.24	9.24	9.21	7.4
Boundary Waters Canoe Area	MN	20.0	12.06	12.04	12.10	12.04	11.2
Bryce Canyon	UT	11.5	7.53	7.51	7.53	7.51	7.1
Bridger	WY	27.6	6.98	6.97	6.97	6.95	11.3
Brigantine	NJ	12.0	18.49	18.46	18.61	18.55	7.0
Cabinet Mountains	MT	13.8	8.55	8.53	8.57	8.52	7.4
Caney Creek	AR	25.9	17.52	17.47	17.52	17.43	11.3
Canyonlands	UT	12.0	8.06	8.06	8.09	8.08	7.0
Caribou	CA	25.9	7.64	7.62	7.60	7.55	11.4
Carlsbad Caverns	NM	14.8	11.74	11.73	11.74	11.71	7.3
Chassahowitzka	FL	17.6	18.54	18.52	18.62	18.58	7.0
Chiricahua NM	AZ	25.7	8.60	8.59	8.59	8.57	11.5
Chiricahua W	AZ	13.9	8.60	8.59	8.59	8.57	6.9
Craters of the Moon	ID	13.9	8.74	8.72	8.71	8.66	6.9
Dome Land	CA	14.7	11.89	11.87	11.73	11.66	7.1
Dolly Sods	WV	12.9	16.79	16.77	16.84	16.80	7.1
Eagles Nest	CO	27.6	6.26	6.25	6.26	6.24	11.3
Emigrant	CA	20.3	9.50	9.49	9.41	9.37	7.1
Everglades	FL	19.6	14.33	14.32	14.40	14.38	7.3

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Fitzpatrick	WY	11.3	6.98	6.97	6.97	6.95	7.1
Flat Tops	CO	17.6	6.32	6.31	6.33	6.31	7.1
Galiuro	AZ	20.3	8.58	8.57	8.58	8.55	11.2
Gates of the Mountains	MT	11.5	6.43	6.42	6.43	6.40	7.1
Gila	NM	11.3	8.20	8.19	8.20	8.18	7.1
Glacier	MT	13.9	12.38	12.32	12.40	12.29	6.9
Glacier Peak	WA	11.2	7.61	7.59	7.67	7.63	7.2
Grand Teton	WY	13.5	7.55	7.54	7.53	7.51	7.0
Great Gulf	NH	19.5	12.87	12.87	12.90	12.89	7.6
Great Sand Dunes	CO	14.0	8.52	8.51	8.51	8.50	7.8
Great Smoky Mountains	TN	12.1	18.16	18.12	18.19	18.11	7.1
Guadalupe Mountains	TX	23.2	11.76	11.74	11.76	11.72	11.3
Hells Canyon	OR	13.1	10.66	10.63	10.64	10.56	7.1
Isle Royale	MI	29.5	12.48	12.46	12.50	12.45	11.4
Jarbidge	NV	17.6	7.11	7.10	7.11	7.08	7.0
James River Face	VA	18.1	17.89	17.84	17.93	17.83	7.3
Joshua Tree	CA	21.1	12.35	12.30	12.34	12.20	11.2
Joyce Kilmer – Slickrock	NC	28.5	18.16	18.12	18.19	18.11	11.2
Kalmiopsis	OR	12.6	9.02	9.01	9.02	8.99	7.1
Kings Canyon	CA	19.5	16.46	16.44	16.36	16.30	7.1
Lava Beds	CA	29.5	8.21	8.18	8.18	8.12	11.5
La Garita	CO	14.8	7.19	7.18	7.19	7.18	7.7
Lassen Volcanic	CA	23.5	7.68	7.66	7.64	7.59	7.1
Linville Gorge	NC	11.6	16.84	16.80	16.87	16.80	7.1
Lostwood	ND	14.8	13.24	13.22	13.19	13.15	7.3
Lye Brook	VT	16.6	12.71	12.70	12.75	12.73	7.5
Mammoth Cave	KY	27.9	19.95	19.91	19.97	19.87	11.4
Marble Mountain	CA	19.6	9.13	9.11	9.09	9.04	7.3
Maroon Bells – Snowmass	CO	23.9	6.15	6.14	6.16	6.14	11.3
Mazatzal	AZ	30.2	9.38	9.37	9.43	9.40	11.5
Medicine Lake	MT	17.1	12.38	12.35	12.34	12.28	7.7
Mesa Verde	CO	11.3	8.16	8.15	8.18	8.16	7.1
Mingo	MO	13.1	19.15	19.09	19.15	19.02	6.9
Mission Mountains	MT	17.7	8.91	8.90	8.89	8.87	7.3
Mount Hood	OR	12.8	7.55	7.53	7.63	7.56	7.1
Mokelumne	CA	27.5	7.69	7.68	7.63	7.60	11.3
Moosehorn	ME	14.2	13.23	13.23	13.26	13.25	7.4
Mount Rainier	WA	12.9	10.31	10.28	10.37	10.30	7.1
Mount Jefferson	OR	21.4	8.21	8.20	8.25	8.20	11.4
Mount Washington	OR	14.0	8.31	8.29	8.36	8.32	7.8
Mount Zirkel	CO	15.7	7.70	7.69	7.72	7.70	7.8
North Cascades	WA	18.9	7.76	7.75	7.81	7.79	7.9
Okefenokee	GA	15.7	17.83	17.80	17.87	17.80	7.9
Otter Creek	WV	11.7	16.74	16.71	16.77	16.73	7.1
Pasayten	WA	14.0	7.67	7.65	7.67	7.62	7.8
Petrified Forest	AZ	26.4	8.54	8.50	8.55	8.48	11.5

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Pine Mountain	AZ	27.6	9.30	9.29	9.29	9.26	11.3
Presidential Range – Dry	NH	14.7	12.61	12.61	12.66	12.66	7.8
Rawah	CO	13.5	7.55	7.54	7.55	7.53	7.0
Red Rock Lakes	WY	13.1	7.53	7.52	7.51	7.49	6.9
Redwood	CA	23.2	9.49	9.46	9.46	9.38	11.3
Cape Romain	SC	11.7	17.14	17.10	17.28	17.17	7.1
Rocky Mountain	CO	12.1	8.36	8.34	8.37	8.33	7.1
Roosevelt Campobello	ME	16.5	13.35	13.34	13.37	13.37	7.8
Salt Creek	NM	14.1	12.12	12.09	12.07	12.02	7.1
San Geronio	CA	21.4	13.72	13.63	13.65	13.43	11.4
San Jacinto	CA	17.7	13.33	13.22	13.12	12.85	7.0
San Pedro Parks	NM	21.5	7.20	7.19	7.20	7.18	7.1
Sawtooth	ID	21.5	8.49	8.48	8.48	8.46	7.1
Scapegoat	MT	11.4	9.09	9.07	9.08	9.06	7.0
Selway - Bitterroot	MT	13.6	7.53	7.51	7.54	7.48	7.2
Seney	MI	14.2	13.22	13.20	13.27	13.21	7.3
Sequoia	CA	12.3	15.96	15.93	15.73	15.66	7.3
Shenandoah	VA	23.8	16.26	16.23	16.27	16.20	11.4
Sierra Ancha	AZ	23.5	9.50	9.49	9.50	9.47	7.1
Sipsey	AL	27.6	19.15	19.10	19.16	19.06	11.3
Alpine Lakes	WA	13.4	10.92	10.88	11.03	10.92	6.9
South Warner	CA	28.7	8.31	8.29	8.27	8.23	11.4
Eagle Cap	OR	16.6	11.25	11.21	11.24	11.14	7.3
Strawberry Mountain	OR	19.6	11.35	11.33	11.34	11.28	7.5
Swanquarter	NC	14.7	16.39	16.37	16.43	16.39	6.9
Sycamore Canyon	AZ	24.6	10.71	10.66	10.72	10.64	11.2
Teton	WY	16.1	7.71	7.70	7.70	7.68	7.0
Theodore Roosevelt	ND	12.1	11.96	11.89	11.91	11.79	7.1
Three Sisters	OR	17.6	8.31	8.29	8.36	8.32	7.3
Superstition	AZ	14.8	9.89	9.87	9.86	9.84	7.3
Thousand Lakes	CA	15.7	7.68	7.66	7.64	7.59	7.9
UL Bend	MT	14.7	9.16	9.15	9.13	9.10	7.2
Upper Buffalo	AR	25.5	16.89	16.85	16.88	16.79	11.3
Voyageurs	MN	18.4	11.25	11.23	11.25	11.21	11.1
Weminuche	CO	11.6	6.90	6.89	6.89	6.88	7.1
West Elk	CO	11.3	6.18	6.17	6.19	6.17	7.1
Wind Cave	SD	16.0	9.56	9.52	9.55	9.47	7.2
Wolf Island	GA	26.4	18.14	18.11	18.18	18.13	11.4
Yellowstone	WY	12.1	7.69	7.67	7.67	7.65	7.1
Yolla Bolly - Middle Eel	CA	17.1	9.31	9.30	9.28	9.23	7.4
Yosemite	CA	17.6	9.30	9.28	9.21	9.17	7.1
Zion	UT	13.5	8.92	8.89	8.95	8.90	7.0

a) The level of visibility impairment in an area is based on the light-extinction coefficient and a unitless visibility index, called a “deciview”, which is used in the valuation of visibility. The deciview metric provides a scale for perceived visual changes over the entire range of conditions, from clear to hazy. Under many scenic conditions, the average person can generally perceive a change of one deciview. The higher the deciview value, the worse the visibility. Thus, an improvement in visibility is a decrease in deciview value.

2.1.6.1.4 Visibility Modeling Methodology

The modeling platform described in Section 2.1.5 above was also used to project changes in visibility. The estimate of visibility benefits was based on the projected improvement in annual average visibility at mandatory class I federal 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 mandatory class I federal 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 mandatory class I federal areas) collecting ambient PM_{2.5} data at mandatory class I federal areas, but only 81 of these sites have complete data for 2001. For this analysis, we quantified visibility improvement at the 116 mandatory class I federal areas which have complete IMPROVE ambient data for 2001 or are represented by IMPROVE monitors with complete data.^G

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: $\text{Visual Range (km)} = 3912/b_{\text{ext}}$ (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

^G There are 81 IMPROVE sites with complete data for 2001. Many of these sites collect data that is “representative” of other nearby unmonitored mandatory class I federal areas. There are a total of 116 mandatory class I federal 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”.

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 mandatory class I federal area. This serves as the visibility input for the benefits analysis (See Chapter 6).

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 IMPROVE site has measurements of PM_{2.5} 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).

2.1.6.2 Other PM Related Welfare Effects

Particulate matter contributes to adverse effects on vegetation and ecosystems, and to soiling and materials damage. These welfare effects result predominately from exposure to excess amounts of specific chemical species, regardless of their source or predominant form (particle, gas or liquid). Reflecting this fact, the PM AQCD concludes that regardless of size fractions, particles containing nitrates and sulfates have the greatest potential for widespread environmental significance, while effects are also related to other chemical constituents found in ambient PM, such as trace metals and organics. (The Staff Paper notes that some of these other components are regulated under separate statutory authorities, e.g., section 112 of the CAA.) The following characterizations of the nature of these welfare effects are based on the information contained in the PM AQCD and Staff Paper.

2.1.6.2.1 *Effects on Vegetation and Ecosystems*

Potentially adverse PM-related effects on vegetation and ecosystems are principally associated with particulate nitrate and sulfate deposition. In characterizing such effects, it is important to recognize that nitrogen and sulfur are necessary and beneficial nutrients for most organisms that make up ecosystems, with optimal amounts of these nutrients varying across organisms, populations, communities, ecosystems and time scales. Therefore, it is impossible to generalize to all species in all circumstances as to the amount at which inputs of these nutrients or acidifying compounds become stressors. The Staff Paper recognizes the public welfare benefits from the use of nitrogen (N) and sulfur (S) nutrients in fertilizers in managed agricultural and commercial forest settings.

2.1.6.2.1.1 Vegetation Effects

At current ambient levels, risks to vegetation from short-term exposures to dry deposited particulate nitrate or sulfate are low. However, when found in acid or acidifying deposition, such particles do have the potential to cause direct leaf injury. Specifically, the responses of forest trees to acid precipitation (rain, snow) include accelerated weathering of leaf cuticular surfaces, increased permeability of leaf surfaces to toxic materials, water, and disease agents; increased leaching of nutrients from foliage; and altered reproductive processes—all which serve to weaken trees so that they are more susceptible to other stresses (e.g., extreme weather, pests, pathogens). Acid deposition with levels of acidity associated

with the leaf effects described above are currently found in some locations in the eastern US (EPA 2003). Even higher concentrations of acidity can be present in occult depositions (e.g., fog, mist or clouds) which more frequently impacts higher elevations. Thus, the risks of leaf injury occurring from acid deposition in some areas of the eastern U.S. is high. However, based on currently available information, the contribution of particulate sulfates and nitrates to the total acidity found at these locations is not clear.

2.1.6.2.1.2 Ecosystem Effects

The nitrogen and sulfur containing components of PM have been associated with a broad spectrum of ecosystems impacts that result from either the nutrients or acidifying characteristics of the deposited compounds.

Reactive nitrogen is the form of nitrogen that is available to support the growth of plants and microorganisms. Since the mid-1960's reactive nitrogen creation through natural processes has been overtaken by reactive nitrogen creation as a result of human processes, and is now accumulating in the environment on the local, regional and global scale. Some reactive nitrogen emission are transformed into ambient PM and deposited onto sensitive ecosystems. Some of the most significant detrimental effects associated with excess reactive nitrogen deposition are those associated with a syndrome known as "nitrogen saturations.": These effects include; (1) Decreased productivity, increased mortality, and/or shifts in plant community composition, often leading to decreased biodiversity in many natural habitats wherever atmospheric reactive nitrogen deposition increases significantly and critical thresholds are exceeded; (2) leaching of excess nitrate and associated base cations from soils into streams, lakes, and rivers, and mobilization of soil aluminum; and (3) alternation of ecosystem processes such as nutrient and energy cycles through changes in the functioning and species composition of beneficial soil organisms (Galloway and Cowling, 2002). Thus, through its effects on habitat suitability, genetic diversity, community dynamics and composition, nutrient status, energy and nutrient cycling, and frequency and intensity of natural disturbance regimes (fire), exceed reactive nitrogen deposition is have profound and adverse impact on essential ecological attributes associated with terrestrial ecosystems. In the US numerous forests now show severe symptoms of nitrogen saturation. For other forested locations, ongoing expansion in nearby urban areas will increase the potential for nitrogen saturation unless there are improved emissions controls.

Excess nutrient inputs into aquatic ecosystems (i.e. streams, rivers, lakes, estuaries or oceans) either form direct atmospheric deposition, surface runoff, or leaching from nitrogen saturated soils into ground or surface waters can contribute to conditions of severe water oxygen depletion; eutrophication and algae blooms; altered fish distributions, catches, and physiological states; loss of biodiversity; habitat degradation; and increases in the incidence of disease.

In the U.S., forests that are now showing severe symptoms of nitrogen saturation include: the northern hardwoods and mixed conifer forests in the Adirondack and Catskill Mountains of New York; the red spruce forests at Whitetop Mountain, Virginia, and Great Smoky Mountains National Park, North Carolina; mixed hardwood watersheds at Fernow Experimental Forest in West Virginia; American beech forests in Great Smoky Mountains

National Park, Tennessee; mixed conifer forests and chaparral watersheds in southern California and the southwestern Sierra Nevada in Central California; the alpine tundra/subalpine conifer forests of the Colorado Front Range; and red alder forests in the Cascade Mountains in Washington.

2.1.6.2.1.2.1 Eutrophication, Nitrification, and Fertilization

In recent decades, human activities have greatly accelerated nutrient impacts, such as nitrogen deposition in both aquatic and terrestrial systems. Nitrogen deposition in aquatic systems can cause excessive growth of algae and lead to degraded water quality and associated impairment of fresh water and estuarine resources for human uses.⁶¹ Nitrogen deposition on terrestrial systems can cause fertilization and lead to ecosystem stress and species shift.

Eutrophication is the accelerated production of organic matter, particularly algae, in a water body. This increased growth can cause numerous adverse ecological effects and economic impacts, including nuisance algal blooms, dieback of underwater plants due to reduced light penetration, and toxic plankton blooms. Algal and plankton blooms can also reduce the level of dissolved oxygen, which can adversely affect fish and shellfish populations.

Deposition of nitrogen contributes to elevated nitrogen levels in waterbodies. The NO_x reductions from today's promulgated standards will help reduce the airborne nitrogen deposition that contributes to eutrophication of watersheds, particularly in aquatic systems where atmospheric deposition of nitrogen represents a significant portion of total nitrogen loadings.

Severe and persistent eutrophication often directly impacts human activities. For example, losses in the nation's fishery resources may be directly caused by fish kills associated with low dissolved oxygen and toxic blooms. Declines in tourism occur when low dissolved oxygen causes noxious smells and floating mats of algal blooms create unfavorable aesthetic conditions. Risks to human health increase when the toxins from algal blooms accumulate in edible fish and shellfish, and when toxins become airborne, causing respiratory problems due to inhalation. According to the NOAA report, more than half of the nation's estuaries have moderate to high expressions of at least one of these symptoms – an indication that eutrophication is well developed in more than half of U.S. estuaries.⁶²⁵

In its Third Report to Congress on the Great Waters, EPA reported that atmospheric deposition contributes from 2 to 38 percent of the nitrogen load to certain coastal waters.⁶³ A review of peer reviewed literature in 1995 on the subject of air deposition suggests a typical contribution of 20 percent or higher.⁶⁴ Human-caused nitrogen loading to the Long Island Sound from the atmosphere was estimated at 14 percent by a collaboration of federal and state air and water agencies in 1997.⁶⁵ The National Exposure Research Laboratory, U.S. EPA, estimated based on prior studies that 20 to 35 percent of the nitrogen loading to the Chesapeake Bay is attributable to atmospheric deposition.⁶⁶ The mobile source portion of atmospheric NO_x contribution to the Chesapeake Bay was modeled at about 30 percent of total air deposition.¹⁰

In U.S. terrestrial systems, the nutrient whose supply most often sets the limit of possible plant based productivity at a given site is nitrogen. By increasing available nitrogen, overall ecosystem productivity may be expected to increase for a time, and then decline as nitrogen saturation is reached. However, because not all vegetation, organisms, or ecosystems react in the same manner to increased nitrogen fertilization, those plants or organisms that are predisposed to capitalize on any increases in nitrogen availability gain an advantage over those that are not as responsive to added nutrients, leading to a change in plant community composition and diversity. Changes to plant community composition and structure within an ecosystem are of concern because plants in large part determine the food supply and habitat types available for use by other organisms. Further, in terrestrial systems, plants serve as the integrators between above-ground and below-ground environments and influence nutrient, energy and water cycles. Because of these linkages, chronic excess nutrient nitrogen additions can lead to complex, dramatic, and severe ecosystem level responses such as changes in habitat suitability, genetic diversity, community dynamics and composition, nutrient status, energy and nutrient cycling, and frequency and intensity of natural disturbance regimes such as fire.

These types of effects have been observed both experimentally and in the field. For example, experimental additions of nitrogen to a Minnesota grassland dominated by native warm-season grasses produced a shift to low-diversity mixtures dominated by coolseason grasses over a 12 year period at all but the lowest rate of nitrogen addition.⁶⁷ Similarly, the coastal sage scrub (CSS) community in California has been declining in land area and in drought deciduous shrub density over the past 60 years, and is being replaced in many areas by the more nitrogen responsive Mediterranean annual grasses. Some 25 plant species are already extinct in California, most of them annual and perennial forbs that occurred in sites now experiencing conversion to annual grassland. As CSS converts more extensively to annual grassland dominated by invasive species, loss of additional rare species may be inevitable. Though invasive species are often identified as the main threat to rare species, it is more likely that invasive species combine with other factors, such as excess N deposition, to promote increased productivity of invasive species and resulting species shifts.

Deposition of nitrogen from the engines covered in this proposal contributes to elevated nitrogen levels in bodies of water and on land. The NO_x reductions proposed in this action will reduce the airborne nitrogen deposition that contributes to eutrophication of watersheds and nitrogen saturation on land.

2.1.6.2.1.2.2 Atmospheric Deposition

Wet and dry deposition of ambient particulate matter delivers a complex mixture of metals (e.g., mercury, zinc, lead, nickel, aluminum, and cadmium), organic compounds (e.g., POM, dioxins, and furans) and inorganic compounds (e.g., nitrate, sulfate) to terrestrial and aquatic ecosystems. The chemical form of the compounds deposited is impacted by a variety of factors including ambient conditions (e.g., temperature, humidity, oxidant levels) and the sources of the material. Chemical and physical transformations of the particulate compounds occur in the atmosphere as well as the media onto which they deposit. These transformations in turn influence the fate, bioavailability and potential toxicity of these compounds. Atmospheric deposition has been identified as a key component of the environmental and

human health hazard posed by several pollutants including mercury, dioxin and PCBs.⁶⁸

Adverse impacts on water quality can occur when atmospheric contaminants deposit to the water surface or when material deposited on the land enters a water body through runoff. Potential impacts of atmospheric deposition to water bodies include those related to both nutrient and toxic inputs. Adverse effects to human health and welfare can occur from the addition of excess particulate nitrate nutrient enrichment which contribute to toxic algae blooms and zones of depleted oxygen that can lead to fish kills, frequently in coastal waters. Particles contaminated with heavy metals or other toxins may lead to the ingestion of contaminated fish, ingestion of contaminated water, damage to the marine ecology, and limited recreational uses. Several studies have been conducted in U.S. coastal waters and in the Great Lakes Region in which the role of ambient PM deposition and runoff is investigated.^{69,70,71,72,73}

Adverse impacts on soil chemistry and plant life have been observed for areas heavily impacted by atmospheric deposition of nutrients, metals and acid species, resulting in species shifts, loss of biodiversity, forest decline and damage to forest productivity. Potential impacts also include adverse effects to human health through ingestion of contaminated vegetation or livestock (as in the case for dioxin deposition), reduction in crop yield, and limited use of land due to contamination.

In the following subsections, atmospheric deposition of heavy metals and particulate organic material is discussed.

2.1.6.2.1.2.2.1 Heavy Metals

Heavy metals, including cadmium, copper, lead, chromium, mercury, nickel and zinc, have the greatest potential for influencing forest growth (PM AQCD, p. 4-87).⁷⁴ Investigation of trace metals near roadways and industrial facilities indicate that a substantial burden of heavy metals can accumulate on vegetative surfaces. Copper, zinc, and nickel have been documented to cause direct toxicity to vegetation under field conditions (PM AQCD, p. 4-75). Little research has been conducted on the effects associated with mixtures of contaminants found in ambient PM. While metals typically exhibit low solubility, limiting their bioavailability and direct toxicity, chemical transformations of metal compounds occur in the environment, particularly in the presence of acidic or other oxidizing species. These chemical changes influence the mobility and toxicity of metals in the environment. Once taken up into plant tissue, a metal compound can undergo chemical changes, accumulate and be passed along to herbivores or can re-enter the soil and further cycle in the environment.

Although there has been no direct evidence of a physiological association between tree injury and heavy metal exposures, heavy metals have been implicated because of similarities between metal deposition patterns and forest decline (PM AQCD, p. 4-76).⁷⁵ Contamination of plant leaves by heavy metals can lead to elevated soil levels. Trace metals absorbed into the plant frequently bind to the leaf tissue, and then are lost when the leaf drops (PM AQCD, p. 4-75). As the fallen leaves decompose, the heavy metals are transferred into the soil.^{76,77}

The environmental sources and cycling of mercury are currently of particular concern due to the bioaccumulation and biomagnification of this metal in aquatic ecosystems and the potent toxic nature of mercury in the forms in which it is ingested by people and other animals. Mercury is unusual compared with other metals in that it largely partitions into the gas phase (in elemental form), and therefore has a longer residence time in the atmosphere than a metal found predominantly in the particle phase. This property enables mercury to travel far from the primary source before being deposited and accumulating in the aquatic ecosystem. The major source of mercury in the Great Lakes is from atmospheric deposition, accounting for approximately eighty percent of the mercury in Lake Michigan.^{78,79} Over fifty percent of the mercury in the Chesapeake Bay has been attributed to atmospheric deposition.⁸⁰ Overall, the National Science and Technology Council (NSTC, 1999) identifies atmospheric deposition as the primary source of mercury to aquatic systems. Forty-four states have issued health advisories for the consumption of fish contaminated by mercury; however, most of these advisories are issued in areas without a mercury point source.

Elevated levels of zinc and lead have been identified in streambed sediments, and these elevated levels have been correlated with population density and motor vehicle use.^{81,82} Zinc and nickel have also been identified in urban water and soils. In addition, platinum, palladium, and rhodium, metals found in the catalysts of modern motor vehicles, have been measured at elevated levels along roadsides.⁸³ Plant uptake of platinum has been observed at these locations.

2.1.6.2.1.2.2 Polycyclic Organic Matter

Polycyclic organic matter (POM) is a byproduct of incomplete combustion and consists of organic compounds with more than one benzene ring and a boiling point greater than or equal to 100 degrees centigrade.⁸⁴ Polycyclic aromatic hydrocarbons (PAHs) are a class of POM that contain compounds which are known or suspected carcinogens.

Major sources of PAHs include mobile sources. PAHs in the environment may be present as a gas or adsorbed onto airborne particulate matter. Since the majority of PAHs are adsorbed onto particles less than 1.0 μm in diameter, long range transport is possible. However, studies have shown that PAH compounds adsorbed onto diesel exhaust particulate and exposed to ozone have half lives of 0.5 to 1.0 hours.⁸⁵

Since PAHs are insoluble, the compounds generally are particle reactive and accumulate in sediments. Atmospheric deposition of particles is believed to be the major source of PAHs to the sediments of Lake Michigan.^{86,87} Analyses of PAH deposition to Chesapeake and Galveston Bay indicate that dry deposition and gas exchange from the atmosphere to the surface water predominate.^{88,89} Sediment concentrations of PAHs are high enough in some segments of Tampa Bay to pose an environmental health threat. EPA funded a study to better characterize the sources and loading rates for PAHs into Tampa Bay.⁹⁰ PAHs that enter a water body through gas exchange likely partition into organic rich particles and be biologically recycled, while dry deposition of aerosols containing PAHs tends to be more resistant to biological recycling.⁹¹ Thus, dry deposition is likely the main pathway for PAH concentrations in sediments while gas/water exchange at the surface may lead to PAH distribution into the food web, leading to increased health risk concerns.

Trends in PAH deposition levels are difficult to discern because of highly variable ambient air concentrations, lack of consistency in monitoring methods, and the significant influence of local sources on deposition levels.⁹² Van Metre et al. (2000) noted PAH concentrations in urban reservoir sediments have increased by 200-300% over the last forty years and correlates with increases in automobile use.⁹³

Cousins et al. (1999) estimates that greater than ninety percent of semi-volatile organic compound (SVOC) emissions in the United Kingdom deposit on soil.⁹⁴ An analysis of polycyclic aromatic hydrocarbon (PAH) concentrations near a Czechoslovakian roadway indicated that concentrations were thirty times greater.

2.1.6.2.1.2.2.3 Materials Damage and Soiling

The deposition of airborne particles can also reduce the aesthetic appeal of buildings and culturally important articles through soiling, and can contribute directly (or in conjunction with other pollutants) to structural damage by means of corrosion or erosion.⁹⁵ Particles affect materials principally by promoting and accelerating the corrosion of metals, by degrading paints, and by deteriorating building materials such as concrete and limestone. Particles contribute to these effects because of their electrolytic, hygroscopic, and acidic properties, and their ability to sorb corrosive gases (principally sulfur dioxide). The rate of metal corrosion depends on a number of factors, including the deposition rate and nature of the pollutant; the influence of the metal protective corrosion film; the amount of moisture present; variability in the electrochemical reactions; the presence and concentration of other surface electrolytes; and the orientation of the metal surface.

2.2 Ozone

In this section we review the health and welfare effects of ozone. We also describe the air quality monitoring and modeling data which indicate that people in many areas across the country continue to be exposed to high levels of ambient ozone and will continue to be into the future. Emissions of nitrogen oxides (NO_x) and volatile organic compounds (VOCs) from locomotive and marine diesel engines subject to this proposed rule have been shown to contribute to these ozone concentrations. Information on air quality was gathered from a variety of sources, including monitored ozone concentrations, air quality modeling forecasts conducted for this rulemaking, and other state and local air quality information.

The proposed emission reductions from this rule would assist 8-hour ozone nonattainment and maintenance areas in reaching the standard by each area's respective attainment date, and maintaining the 8-hour ozone standard in the future. The emission reductions will also help continue to lower ambient ozone levels and resulting health impacts.

2.2.1 Science of Ozone Formation

Ground-level ozone pollution is formed by the reaction of VOCs and nitrogen oxides (NO_x) in the atmosphere in the presence of heat and sunlight. These pollutants, often referred to as ozone precursors, are emitted by many types of pollution sources such as highway and nonroad vehicles, power plants, chemical plants, refineries, makers of consumer

and commercial products, and smaller area sources.

The science of ozone formation, transport, and accumulation is complex.⁹⁶ Ground-level ozone is produced and destroyed in a cyclical set of chemical reactions, many of which are sensitive to temperature and sunlight. When ambient temperatures and sunlight levels remain high for several days and the air is relatively stagnant, ozone and its precursors can build up and result in more ozone than typically would occur on a single high-temperature day. Ozone also can be transported into an area from pollution sources found hundreds of miles upwind, resulting in elevated ozone levels even in areas with low VOC or NO_x emissions.

The highest levels of ozone are produced when both VOC and NO_x emissions are present in significant quantities on clear summer days. Relatively small amounts of NO_x enable ozone to form rapidly when VOC levels are relatively high, but ozone production is quickly limited by removal of the NO_x. Under these conditions NO_x reductions are highly effective in reducing ozone while VOC reductions have little effect. Such conditions are called “NO_x-limited”. Because the contribution of VOC emissions from biogenic (natural) sources to local ambient ozone concentrations can be significant, even some areas where man-made VOC emissions are relatively low can be NO_x-limited.

When NO_x levels are relatively high and VOC levels relatively low, NO_x forms inorganic nitrates (i.e., particles) but relatively little ozone. Such conditions are called “VOC-limited.” Under these conditions, VOC reductions are effective in reducing ozone, but NO_x reductions can actually increase local ozone under certain circumstances. Even in VOC-limited urban areas, NO_x reductions are not expected to increase ozone levels if the NO_x reductions are sufficiently large.

Rural areas are usually NO_x-limited, due to the relatively large amounts of biogenic VOC emissions in many rural areas. Urban areas can be either VOC- or NO_x-limited, or a mixture of both, in which ozone levels exhibit moderate sensitivity to changes in either pollutant.

Ozone concentrations in an area also can be lowered by the reaction of nitric oxide with ozone, forming nitrogen dioxide (NO₂); as the air moves downwind and the cycle continues, the NO₂ forms additional ozone. The importance of this reaction depends, in part, on the relative concentrations of NO_x, VOC, and ozone, all of which change with time and location.

The current ozone National Ambient Air Quality Standards (NAAQS) has an 8-hour averaging time.^H The 8-hour ozone NAAQS, established by EPA in 1997, is based on well-documented science demonstrating that more people were experiencing adverse health effects at lower levels of exertion, over longer periods, and at lower ozone concentrations than addressed by the previous one-hour ozone NAAQS. The current ozone NAAQS

^H EPA’s review of the ozone NAAQS is underway and a proposal is scheduled for May 2007 with a final rule scheduled for February 2008.

addresses ozone exposures of concern for the general population and populations most at risk, including children active outdoors, outdoor workers, and individuals with pre-existing respiratory disease, such as asthma. The 8-hour ozone NAAQS is met at an ambient air quality monitoring site when the average of the annual fourth-highest daily maximum 8-hour average ozone concentration over three years is less than or equal to 0.084 ppm.

2.2.2 Health Effects of Ozone

Exposure to ambient ozone contributes to a wide range of adverse health effects¹. These health effects are well documented and are critically assessed in the EPA ozone air quality criteria document (ozone AQCD) and EPA staff paper.^{97,98} We are relying on the data and conclusions in the ozone AQCD and staff paper, regarding the health effects associated with ozone exposure.

Ozone-related health effects include lung function decrements, respiratory symptoms, aggravation of asthma, increased hospital and emergency room visits, increased asthma medication usage, inflammation of the lungs and a variety of other respiratory effects and cardiovascular effects. People who are more susceptible to effects associated with exposure to ozone include children, asthmatics and the elderly. There is also suggestive evidence that certain people may have greater genetic susceptibility. Those with greater exposures to ozone, for instance due to time spent outdoors (e.g. outdoor workers), are also of concern.

Based on a large number of scientific studies, EPA has identified several key health effects associated with exposure to levels of ozone found today in many areas of the country. Short-term (1 to 3 hours) and prolonged exposures (6 to 8 hours) to higher ambient ozone concentrations have been linked to lung function decrements, respiratory symptoms, increased hospital admissions and emergency room visits for respiratory problems.^{99, 100, 101, 102, 103, 104} Repeated exposure to ozone can increase susceptibility to respiratory infection and lung inflammation and can aggravate preexisting respiratory diseases, such as asthma.^{105, 106, 107, 108, 109} Repeated exposure to sufficient concentrations of ozone can also cause inflammation of the lung, impairment of lung defense mechanisms, and possibly irreversible changes in lung structure, which over time could lead to premature aging of the lungs and/or chronic respiratory illnesses, such as emphysema and chronic bronchitis.^{110, 111, 112, 113}

Children and adults who are outdoors and active during the summer months, such as construction workers and other outdoor workers, are among those most at risk of elevated ozone exposures.¹¹⁴ Children and outdoor workers tend to have higher ozone exposure because they typically are active outside, working, playing and exercising, during times of day and seasons (e.g. the summer) when ozone levels are highest.¹¹⁵ For example, summer camp studies in the Eastern United States and Southeastern Canada have reported significant reductions in lung function in children who are active outdoors.^{116, 117, 118, 119, 120, 121, 122, 123}

¹ Human exposure to ozone varies over time due to changes in ambient ozone concentration and because people move between locations which have notable different ozone concentrations. Also, the amount of ozone delivered to the lung is not only influenced by the ambient concentration but also by the individuals breathing route and rate.

Further, children are more at risk of experiencing health effects from ozone exposure than adults because their respiratory systems are still developing. These individuals (as well as people with respiratory illnesses such as asthma, especially asthmatic children) can experience reduced lung function and increased respiratory symptoms, such as chest pain and cough, when exposed to relatively low ozone levels during prolonged periods of moderate exertion.^{124, 125, 126, 127}

EPA typically quantifies ozone-related health impacts in its regulatory impact analyses (RIAs) when possible. In the analysis of past air quality regulations, ozone-related benefits have included morbidity endpoints and welfare effects such as damage to commercial crops. EPA has not recently included a separate and additive mortality effect for ozone, independent of the effect associated with fine particulate matter. For a number of reasons, including 1) advice from the Science Advisory Board (SAB) Health and Ecological Effects Subcommittee (HEES) that EPA consider the plausibility and viability of including an estimate of premature mortality associated with short-term ozone exposure in its benefits analyses and 2) conclusions regarding the scientific support for such relationships in EPA's 2006 Air Quality Criteria for Ozone and Related Photochemical Oxidants (the CD), EPA is in the process of determining how to appropriately characterize ozone-related mortality benefits within the context of benefits analyses for air quality regulations. As part of this process, we are seeking advice from the National Academy of Sciences (NAS) regarding how the ozone-mortality literature should be used to quantify the reduction in premature mortality due to diminished exposure to ozone, the amount of life expectancy to be added and the monetary value of this increased life expectancy in the context of health benefits analyses associated with regulatory assessments.

Since the NAS effort is not expected to conclude until 2008, the agency is currently deliberating how best to characterize ozone-related mortality benefits in its rulemaking analyses in the interim. For the analysis of the proposed locomotive and marine standards, we do not quantify an ozone mortality benefit. So that we do not provide an incomplete picture of all of the benefits associated with reductions in emissions of ozone precursors, we have chosen not to include an estimate of total ozone benefits in the proposed RIA. By omitting ozone benefits in this proposal, we acknowledge that this analysis underestimates the benefits associated with the proposed standards. For more information regarding the quantified benefits included in this analysis, please refer to Chapter 6.

2.2.3 Current 8-Hour Ozone Levels

The proposed locomotive and marine engine emission reductions will assist 8-hour ozone nonattainment areas in reaching the standard by each area's respective attainment date and assist in maintaining the 8-hour ozone standard in the future. In this section and the next section we present information on current and model-projected future 8-hour ozone levels.

A nonattainment area is defined in the CAA as an area that is violating a NAAQS or is contributing to a nearby area that is violating the NAAQS. EPA designated nonattainment areas for the 8-hour ozone NAAQS in June 2004. The final rule on Air Quality Designations and Classifications for the 8-hour Ozone NAAQS (69 FR 23858, April 30, 2004) lays out the

factors that EPA considered in making the 8-hour ozone nonattainment designations, including 2001-2003 measured data, air quality in adjacent areas, and other factors.^J

As of October 2006 there are approximately 157 million people living in 116 areas designated as not in attainment with the 8-hour ozone NAAQS. There are 461 full or partial counties that make up the 8-hour ozone nonattainment areas. These numbers do not include the people living in areas where there is a future risk of failing to maintain or achieve the 8-hour ozone NAAQS. Figure 2-1 illustrates the widespread nature of these current problems. Shown in this figure are counties designated as nonattainment for the 8-hour ozone NAAQS, PM_{2.5} nonattainment counties, and mandatory class I federal areas. The current 8-hour ozone nonattainment areas, nonattainment counties, and populations are listed in Appendix 2C to this draft RIA.

Counties designated as 8-hour ozone nonattainment were classified, on the basis of their one-hour ozone design value, as Subpart 1 or Subpart 2 (69 FR 23951, April 30, 2004). Areas classified as Subpart 2 were then further classified, on the basis of their 8-hour ozone design value, as marginal, moderate, serious, severe or extreme. The maximum attainment date assigned to an ozone nonattainment area is based on the area's classification.

States with 8-hour ozone nonattainment areas will be required to take action to bring those areas into compliance in the future. Based on the final rule designating and classifying 8-hour ozone nonattainment areas (69 FR 23951, April 30, 2004), most 8-hour ozone nonattainment areas will be required to attain the 8-hour ozone NAAQS in the 2007 to 2013 time frame and then be required to maintain the 8-hour ozone NAAQS thereafter.^K We expect many of the 8-hour ozone nonattainment areas will need to adopt additional emission reduction programs. The expected NO_x and VOC reductions from the standards proposed in this action would be useful to states as they seek to either attain or maintain the 8-hour ozone NAAQS.

Further insight into the need for reductions from this rule can be gained by evaluating counties at various levels above the level of the 8-hour ozone NAAQS. As shown in Table 2-4 below, of the 158 million people living in counties with 2001-2003 design value

^J An ozone design value is the concentration that determines whether a monitoring site meets the NAAQS for ozone. Because of the way they are defined, design values are determined based on three consecutive-year monitoring periods. For example, an 8-hour design value is the fourth highest daily maximum 8-hour average ozone concentration measured over a three-year period at a given monitor. The full details of these determinations (including accounting for missing values and other complexities) are given in Appendices H and I of 40 CFR Part 50. Due to the precision with which the standards are expressed (0.08 parts per million (ppm) for the 8-hour), a violation of the 8-hour standard is defined as a design value greater than or equal to 0.085 ppm or 85 parts per billion (ppb). For a county, the design value is the highest design value from among all the monitors with valid design values within that county. If a county does not contain an ozone monitor, it does not have a design value. However, readers should note that ozone design values generally represent air quality across a broad area and that absence of a design value does not imply that the county is in compliance with the ozone NAAQS. Therefore, our analysis may underestimate the number of counties with design values above the level of NAAQS.

^K The Los Angeles South Coast Air Basin 8-hour ozone nonattainment area will have to attain before June 15, 2021.

measurements above the 8-hour ozone NAAQS, almost 90 million live in counties with 2001-2003 8-hour ozone design values above 95 ppb.

Table 2-4 Population Living in Counties with 2001-2003 8-hour Ozone Design Values Shown

2001-2003 8-hour Ozone Design Value (ppb)	Number of Counties Within The Concentration Range	2000 Population Living in Counties Within The Concentration Range (Millions, 2000 Census Data)
>95	25	89.7
>90 <=95	47	40.0
>85 <= 90	54	29.6

EPA’s review of the ozone NAAQS is currently underway and a proposal is scheduled for June 2007 with a final rule scheduled for March 2008. If the ozone NAAQS is revised then new nonattainment areas could be designated. While EPA is not relying on it for purposes of justifying this proposal, the emission reductions from this proposed rulemaking would also be helpful to states if there is an ozone NAAQS revision.

2.2.4 Projected 8-Hour Ozone Levels

EPA has already adopted many emission control programs that are expected to reduce ambient ozone levels. These control programs include the Clean Air Interstate Rule (70 FR 25162, May 12, 2005), the Clean Air Nonroad Diesel rule (69 FR 38957, June 29, 2004), and the Heavy Duty Engine and Vehicle Standards and Highway Diesel Fuel Sulfur Control Requirements (66 FR 5002, Jan. 18, 2001). As a result of these programs, the number of areas that fail to meet the 8-hour ozone NAAQS in the future is expected to decrease.

The base case air quality modeling completed for this proposed rule predicts that without additional local, regional or national controls there will continue to be a need for reductions in 8-hour ozone concentrations in some areas in the future. The determination that an area is at risk of exceeding the 8-hour ozone standard in the future was made for all areas with current design values greater than or equal to 85 ppb (or within a 10 percent margin) and with modeling evidence that concentrations at and above this level will persist into the future. Those interested in greater detail should review the air quality modeling TSD.

With reductions from programs already in place (but excluding the emission reductions from this rule), the number of counties with projected 8-hour ozone design values at or above 85 ppb in 2020 is expected to be 31 counties where 35 million people are projected to live. In addition, in 2020, 89 counties where 60 million people are projected to live, will be within 10 percent of violating the 8-hour ozone NAAQS. Table 2- 5 below provides the full list of counties in 2020 projected to have design values at or above 85 ppb as well as the 89 counties within 10 percent of violating the NAAQS in 2020. By 2030 27

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current ozone nonattainment counties would still remain impacting 37 million people. Even in 2030, 75 million people, living in 108 counties would continue to be within 10 percent of the current 8-hour ozone standard.

Clearly the almost 300,000 tons of annual NO_x reductions in 2020 and the more than 7650,000 NO_x tons reduced in 2030 would be very important to these areas as they struggle to attain the 8-hour ozone standard or continue to maintain the standards. Table 2-5 below shows the current 8-hour ozone nonattainment areas which are projected to be in nonattainment in 2020 and 2030 as well as those current nonattainment areas, which will be in attainment but within 10 percent of not meeting the standard. The table also presents ozone design values and populations in 2020 and 2030.

Table 2-5 Counties with 2020 and 2030 projected Annual 8-hour Ozone Design Values in Violation or within 10 percent of the Annual Ozone Standard in the Base and Control Cases.

State	County	2001-2003 Average Ozone DV (ppb)	2020		2030		2020 population
			base	control	base	control	
AZ	Maricopa	85.0	X	X	X	X	4,609,780
CA	Amador	88.0	X	X	X	X	52,471
CA	Calaveras	92.3	X	X	X		58,261
CA	El Dorado	105.7	X	X	X	X	236,310
CA	Fresno	111.3	V	V	V	X	1,066,878
CA	Imperial	87.0	V	V	V	V	161,555
CA	Kern	112.0	X	X	X	X	876,131
CA	Kings	97.3	V	V	V	V	173,390
CA	Los Angeles	110.0	V	V	X	X	10,376,013
CA	Madera	90.7	V	V	V	V	173,940
CA	Mariposa	88.3	X	X	X	X	22,272
CA	Merced	101.3	V	V	X	X	277,863
CA	Nevada	97.7	V	V	V	V	131,831
CA	Orange	82.7	X	X	X	X	3,900,599
CA	Placer	100.3	X	X	X		451,620
CA	Riverside	108.7	V	V	X	X	2,252,510
CA	Sacramento	99.7	V	V	V	V	1,640,590
CA	San Bernardino	129.3	X	X	X	X	2,424,764
CA	San Diego	94.0	V	V	V	V	3,863,460
CA	Stanislaus	94.0	X	X	X	X	607,766
CA	Tehama	84.3	X	X			64,298
CA	Tulare	105.3	X	X			477,296
CA	Tuolumne	91.5	V	V	V	V	70,570
CA	Ventura	97.7	V	X	X	X	1,023,136
CO	Douglas	82.5	V	V	V	V	303,846
CO	Jefferson	83.7	X	X	X	X	655,782
CT	Fairfield	98.7	X	X	X	X	962,824
CT	Hartford	89.3	V	V	V	V	942,284
CT	Middlesex	98.0	X	X	X	X	177,500

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CT	New Haven	99.0	V	V	V	V	898,415
CT	New London	90.7	V	V	V	V	280,729
CT	Tolland	93.0	X	X	X	X	152,653
DC	Washington	94.3	X	X	X	X	554,330
DE	Kent	91.3	X	X	X	X	153,635
DE	New Castle	95.3	X		X		584,627
DE	Sussex	93.3	X	X	X	X	202,387
GA	De Kalb	95.3	X	X	X		801,817
GA	Fulton	99.0	X	X	X		929,278
IL	Cook	87.7	X	X	X	X	5,669,479
IN	Hamilton	93.3	X	X	X	X	279,537
IN	Lake	90.7	X	X	X		509,293
IN	Marion	90.0	X	V	X	X	935,610
IN	Porter	89.0	X				188,604
IN	Shelby	93.5	X				50,387
KY	Campbell	91.7	X	X			95,622
LA	East Baton Rou	87.3	X	X	X		522,399
LA	Iberville	86.7	X	X	X		33,130
MD	Anne Arundel	101.0	X		X		596,924
MD	Baltimore	93.0	X	X	V	X	855,464
MD	Cecil	102.7	X	X	X	X	109,425
MD	Harford	103.7	V	V	V	V	317,847
MD	Kent	99.0	V	V	V	V	21,407
MD	Montgomery	88.7	X	X	X	X	1,060,716
MD	Prince Georges	95.0	X	X	X	X	944,987
MA	Barnstable	94.7	X	X	X	X	283,735
MA	Bristol	92.7	X	X	X	X	605,591
MI	Allegan	92.0	X	X	X	X	141,851
MI	Macomb	91.0	X	X			894,095
MI	Muskegon	92.0	X	X	X	X	183,444
MI	Oakland	87.0	X	X	X	X	1,443,380
MI	Wayne	88.0	X	X	X	X	1,908,196
MO	St Louis	89.3	X	X	X	X	1,057,171
MO	St Louis City	87.0	X	X	X	X	303,712
NJ	Bergen	92.5	X	X	X		944,507
NJ	Camden	102.3	X	X	X	X	547,817
NJ	Cumberland	96.7	V	V	V	V	161,512
NJ	Gloucester	100.3	X	X	X	X	304,105
NJ	Hudson	88.0	V	V	V	V	694,357
NJ	Hunterdon	97.3	X	X	X	X	160,989
NJ	Mercer	102.3	X	X	X	X	392,236
NJ	Middlesex	100.7	V	V	V	V	934,654
NJ	Monmouth	95.7	V	V	V	V	741,640
NJ	Morris	97.7	V	X	V	X	548,694
NJ	Ocean	109.0	X	X	X	X	644,323
NY	Erie	96.0	V	V	V	V	959,145
NY	Jefferson	91.7	X	X	X	X	119,264
NY	Niagara	91.0	X	X	X		220,989
NY	Putnam	91.3	X	X	X	X	124,395
NY	Richmond	96.0	X	X	X	X	561,360
NY	Suffolk	98.5	X	X	X	X	1,598,742
NY	Westchester	92.0	V	V	V	V	1,027,798

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OH	Ashtabula	94.0	X	X	X	X	108,355
OH	Geauga	98.3	X	X	X	X	114,438
OH	Lake	92.7	X	X	X	X	250,353
PA	Allegheny	93.0	X	X	X	X	1,242,587
PA	Beaver	90.7	X	X	X	X	186,566
PA	Bucks	103.0	X	X	X	X	711,275
PA	Chester	96.5	V	V	V	V	528,797
PA	Delaware	93.7	X	X	X	X	548,283
PA	Lancaster	94.0	X	X	X	X	568,258
PA	Lehigh	93.3	X	X	X		351,875
PA	Montgomery	96.3	X	X	X		805,003
PA	Northampton	93.0	X	X	X	X	301,041
PA	Philadelphia	97.5	X	X	X		1,394,176
RI	Kent	95.3	V	V	V	V	183,833
RI	Providence	90.3	X	X	X	X	648,008
RI	Washington	93.3	X	X	X	X	156,286
TX	Brazoria	91.0	X	X	X	X	322,385
TX	Dallas	91.0	X	X	X	X	2,828,339
TX	Denton	99.0	X	X	X	X	715,168
TX	Galveston	92.0	X	X	X		318,966
TX	Gregg	88.3	X	X	X	X	132,922
TX	Harris	105.0	X	X	X	X	4,588,812
TX	Jefferson	90.5	V	V	V	V	272,075
TX	Tarrant	98.3	X	X	X	X	2,137,957
VA	Alexandria Cit	90.0	X	X	X	X	132,893
VA	Arlington	95.7	X	X	X	X	208,368
VA	Charles City	89.3	V	V	V	V	8,086
VA	Fairfax	96.3	X	X	X		1,281,265
VA	Hampton City	88.7	X	X	X	X	161,913
VA	Hanover	94.0	X	X	X	X	109,984
VA	Suffolk City	87.3	X	X	X	X	72,313
WI	Door	92.7	X	X		X	34,106
WI	Kenosha	98.7	X	X	X		184,825
WI	Kewaunee	90.0	V	V	V	V	21,040
WI	Manitowoc	90.0	X	X	X		85,187
WI	Milwaukee	91.3	X	X	X	X	927,845
WI	Ozaukee	95.3	X	X	X	X	110,294
WI	Racine	91.7	X	X	X	X	212,351
WI	Sheboygan	98.0	X	X	X	X	128,777

2.2.4.1 Ozone Modeling Results with proposed controls

This section summarizes the results of our modeling of ozone air quality impacts in the future due to the reductions in locomotive and marine diesel emissions proposed in this action. Specifically, we compare baseline scenarios to scenarios with the proposed controls. Our modeling indicates that the reductions from this proposed rule will contribute to reducing ambient ozone concentrations and potential exposures in future years.

Chapter 2: Air Quality and Resulting Health and Welfare Effects

According to air quality modeling performed for this rulemaking, the proposed locomotive and marine diesel engines standards are expected to provide nationwide improvements in ozone levels for the vast majority of areas. Specifically, this proposed rule would result in ozone benefits for all but two U.S. ozone nonattainment areas in both their 2020 and 2030 ozone design values. There are two areas with small (i.e., less than 1 ppb) increases in their annual 8-hour ozone design values due to the NO_x disbenefits which occurs in some VOC-limited ozone nonattainment areas. Briefly NO_x reductions can at certain times and in some areas cause ozone levels to increase slightly. Section 2.2.4.1.1 provides additional detail about NO_x disbenefits.

Despite of the localized areas that experience small increases, the overall effect of this proposed rule is positive with 454 (of 473) counties experiencing at least a 0.1 ppb decrease in both their 2020 and 2030 ozone design values. On a population-weighted basis, the average modeled future-year 8-hour ozone design values would decrease by 0.29 ppb in 2020 and 0.80 ppb in 2030. Within projected ozone nonattainment areas in 2030, the average decrease would be somewhat higher: -0.30 ppb in 2020 and - 0.88 ppb in 2030 while the *maximum* decrease for future-year design values would be -1.10 ppb in 2020 and -2.90 ppb in 2030.

Table 2-6 shows the average change in future year eight-hour ozone design values. Average changes are shown 1) for all counties with 2020 baseline design values, 2) for counties with baseline design values that exceeded the standard in 2001-2003 (“violating” counties), and 3) for counties that did not exceed the standard, but were within 10 percent of it in 2001-2003. This last category is intended to reflect counties that meet the standard, but will likely benefit from help in maintaining that status in the face of growth. The average and population-weighted average over all counties demonstrates a broad improvement in ozone air quality. The average across violating counties shows that the proposed rule will help bring these counties into attainment. Since some of the VOC and NO_x emission reductions expected from this proposed rule will go into effect during the period when areas will need to attain the 8-hour ozone NAAQS, the projected reductions in emissions are expected to assist States and local agencies in their effort to attain and maintain the 8-hour ozone standard. The average over counties within ten percent of the standard shows that the proposed rule will also help those counties to maintain the standard. All of these metrics show a decrease in 2020 and a larger decrease in 2030, indicating in four different ways the overall improvement in ozone air quality.

Table 2-6 Average change in projected future year 8-hour ozone design value

Average ^a	Number of US Counties	Change in 2020 design value ^b (ppb)	Change in 2030 design value ^b (ppb)
All	473	0.32	0.86
All, population-weighted	473	0.29	0.80
Violating counties ^c	277	0.33	0.88
Violating counties ^c , population-weighted	277	0.29	0.87

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Counties within 10 percent of the standard ^d	146	0.35	0.94
Counties within 10 percent of the standard ^d , population-weighted	146	0.32	1.02

- a) averages are over counties with 2020 modeled design values
- b) assuming the nominal modeled control scenario
- c) counties whose 2001 baseline design values exceeded the 8-hour ozone standard (≥ 85 ppb)
- d) counties whose 2001 baseline values were less than but within 10 percent of the 8-hour ozone standard.

The impact of the proposed reductions has also been analyzed with respect to those areas that have the highest projected design values. We project that there will be 27 US counties with design values at or above 85 ppb in 2030. After implementation of this proposed action, we project that 3 of these 27 counties will attain the standard. Further, 17 of the 27 counties will be at least 10 percent closer to a design value of less than 85 ppb, and on average all 27 counties will be 29 percent closer to a design value of less than 85 ppb.

The geographic impact of these emissions reductions in 2030 on annual ozone design values in counties across the US, can be seen in Figure 2-9.

Figure 2-9 Impact of Proposed Locomotive/Marine controls on annual Ozone Design Values (DV) in 2030

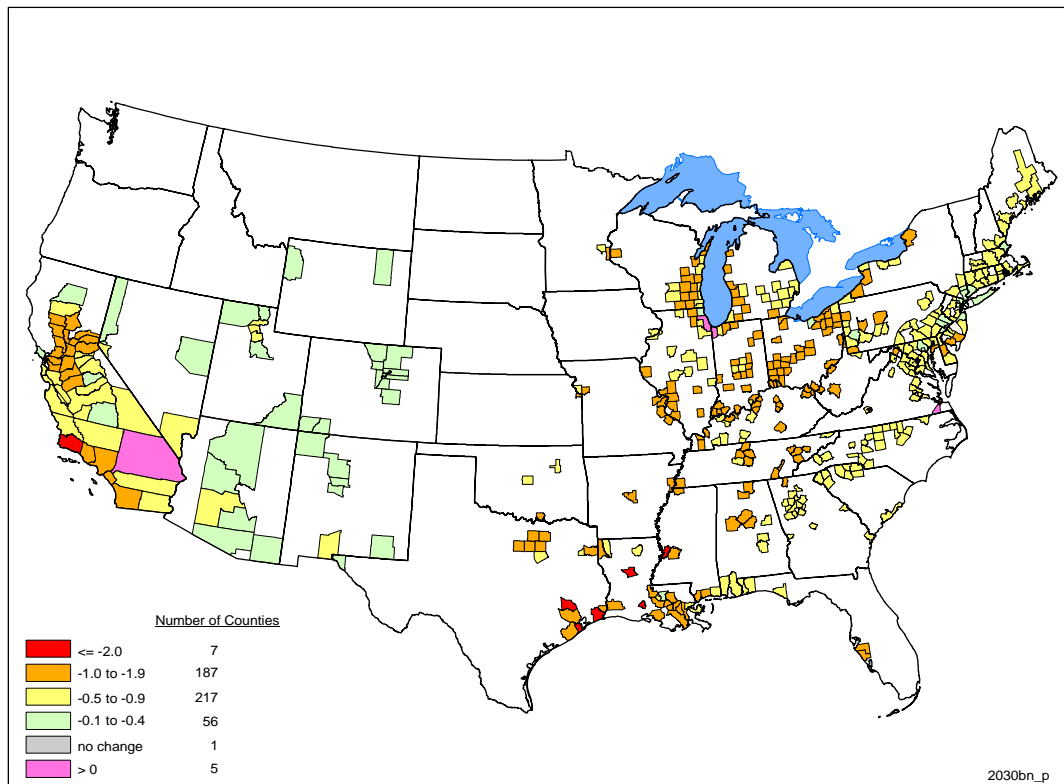


Figure 2.9 shows those US counties in 2030 which are projected to experience a change in their ozone design values as a result of this proposed rule. The most significant decreases, equal or greater than -2.0 ppb, would occur in 7 counties across the US including: Grant (-2.1ppb) and Lafayette (-2.0 ppb) Counties in Louisiana; Montgomery (-2.0 ppb), Galveston (-2.0ppb), and Jefferson (-2.0 ppb) Counties in Texas; Warren County (-2.9 ppb) in Mississippi; and Santa Barbara County (-2.7 ppb) in California. One hundred eighty-seven (187) counties would see annual ozone design value reductions from -1.0 to -1.9 ppb while an estimated 217 additional counties would see annual design value reductions from -0.5 to -0.9 ppb. Note that 5 counties including: Suffolk (+1.5 ppb) and Hampton (+ 0.8 ppb) Counties in Virginia; Cook County (+ 0.7 ppb) in Illinois; Lake County (+ 0.2 ppb) in Indiana; and San Bernardino County (+ 0.1 ppb) in California are projected to experience increased ozone design values because of the NO_x disbenefit that occurs under certain conditions.

It should be noted that the emission control scenarios used in the air quality and benefits modeling are slightly different than the emission control program being proposed. The differences reflect further refinements of the regulatory program since we performed the air quality modeling for this rule. Chapter 3 of this RIA describes the changes in the inputs and resulting emission inventories between the preliminary assumptions used for the air quality modeling and the final proposed regulatory scenario. These refinements to the proposed program would not significantly change the results summarized here or our conclusions drawn from this analysis.

2.2.4.1.1 Potentially Counterproductive Impacts on Ozone Concentrations from NO_x Emissions Reductions

While the proposed rule would reduce ozone levels generally and provide significant national ozone-related health benefits, this is not always the case at the local level. Due to the complex photochemistry of ozone production, NO_x emissions lead to both the formation and destruction of ozone, depending on the relative quantities of NO_x, VOC, and ozone catalysts such as the OH and HO₂ radicals. In areas dominated by fresh emissions of NO_x, ozone catalysts are removed via the production of nitric acid which slows the ozone formation rate. Because NO_x is generally depleted more rapidly than VOC, this effect is usually short-lived and the emitted NO_x can lead to ozone formation later and further downwind. The terms “NO_x disbenefits” or “ozone disbenefits” refer to the ozone increases that can result from NO_x emissions reductions in these localized areas. According to the NARSTO Ozone Assessment, these disbenefits are generally limited to small regions within specific urban cores and are surrounded by larger regions in which NO_x control is beneficial.

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In the context of ozone disbenefits, some have postulated that present-day weekend conditions serve as a demonstration of the effects of future NO_x reduction strategies because NO_x emissions decrease more than VOC emissions on weekends, due to a disproportionate decrease in the activity of heavy-duty diesel trucks and other diesel equipment. Recent research indicates that ambient ozone levels are higher in some metropolitan areas on

weekends than weekdays.^{129, 130} There are other hypotheses for the cause of the “weekend effect.”¹³¹ For instance, the role of ozone and ozone precursor carryover from previous days is difficult to evaluate because of limited ambient data, especially aloft. The role of the changed timing of emissions is difficult to evaluate because of limited ambient and emissions inventory information. It is also important to note that in many areas with “weekend effects” (e.g., Los Angeles and San Francisco) significant ozone reductions have been observed over the past 20 years for all days of the week, during a period in which both NO_x and VOC emissions have been greatly reduced.

EPA maintains that the best available approach for determining the value of a particular emissions reduction strategy is the net air quality change projected to result from the rule, evaluated on a nationwide basis and for all pollutants that are health and/or welfare concerns. The primary tool for assessing the net impacts of this rule are the air quality simulation models. Model scenarios of 2020 and 2030 with and without the proposed controls are compared to determine the expected changes in future pollutant levels resulting from the proposed rule. There are several factors related to the air quality modeling and inputs which should be considered regarding the disbenefit issue. First, our future year modeling conducted does not contain any local governmental actions beyond the controls proposed in this rule. It is possible that significant local controls of VOC and/or NO_x could modify the conclusions regarding ozone changes in some areas. Second, recent work by CARB has indicated that model limitations and uncertainties may lead to overestimates of ozone disbenefits attributed to NO_x emission reductions. While EPA maintains that the air quality simulations conducted for the rule represent state-of-the-science analyses, any changes to the underlying chemical mechanisms, grid resolution, and emissions/meteorological inputs could result in revised conclusions regarding the strength and frequency of ozone disbenefits.

A wide variety of ozone metrics were considered in the assessment of the proposed emissions reductions. Three of the most important assessments are: 1) the effect of the proposed rule on projected future-year ozone design values, 2) the effect of the proposed rule in assisting local areas in attainment and maintenance of the NAAQS, and 3) an economic assessment of the rule benefits based on existing health studies.

Based only on the reductions from today’s rule, our modeling predicts that in 2020 and 2030 periodic ozone disbenefit would occur in up to five counties: Suffolk and Hampton Counties in Virginia, Cook County in Illinois, Lake County in Indiana, and San Bernardino County in California. Despite these localized increases, the net ozone impact of the rule nationally is positive for the majority of the analysis metrics as described in section 2.2.4.1 above.

Historically, NO_x reductions have been very successful at reducing regional/national ozone levels. Consistent with that fact, the photochemical modeling completed for this rule indicates that the emissions reductions proposed today will significantly assist in the attainment and maintenance of the ozone NAAQS at the national level. Furthermore, NO_x reductions also result in reductions in PM and its associated health and welfare effects. This rule is one aspect of overall emissions reductions that States, local governments, and Tribes need to reach their clean air goals. It is expected that future local and national controls that decrease VOC, CO, and regional ozone will mitigate any localized disbenefit. EPA will

continue to rely on local attainment measures to ensure that the NAAQS are not violated in the future. Many organizations with an interest in improved air quality support the rule because they believe the resulting NO_x reductions would reduce both ozone and PM¹³². EPA believes that a balanced air quality management approach that includes NO_x emissions reductions from nonroad engines is needed as part of the Nation's progress toward clean air.

Another category of potential effects that may change in response to ozone reduction strategies results from the shielding provided by ozone against the harmful effects of ultraviolet radiation (UV-B) derived from the sun. The great majority of this shielding results from naturally occurring ozone in the stratosphere, but the 10 percent of total "column" ozone present in the troposphere also contributes.¹³³ A variable portion of this tropospheric fraction of UV-B shielding is derived from ground level ozone related to anthropogenic air pollution. Therefore, strategies that reduce ground level ozone could, in some small measure, increase exposure to UV-B from the sun.

While it is possible to provide quantitative estimates of benefits associated with globally based strategies to restore the far larger and more spatially uniform stratospheric ozone layer, the changes in UV-B exposures associated with ground level ozone reduction strategies are much more complicated and uncertain. Comparatively smaller changes in ground-level ozone (compared to the total ozone in the troposphere) and UV-B are not likely to measurably change long-term risks of adverse effects.

2.2.4.2 Ozone Air Quality Modeling Methodology

To model the ozone air quality benefits of this rule we also used the CMAQ model. CMAQ simulates the numerous physical and chemical processes involved in the formation, transport, and destruction of ozone. This model is commonly used in developing attainment demonstration State Implementation Plans as well as for estimating the ozone reductions expected to occur from a reduction in emitted pollutants. The model was applied for two separate domains: a) a 36 km continental U.S. domain as described in Section 2.1.5, and b) a smaller eastern U.S. grid with a grid resolution of 12 km.

For ozone modeling results over the western U.S. the 36 km modeling results were used, but only for those periods within the months from May to October. Over the eastern U.S. we utilized two periods of episodic modeling to generate the projections: June 15-30, 2001 and July 15–August 10, 2001. Model configurations for the finer-scale episodic modeling was identical to that described in Section 2.1.5.2 except for the use of finer-scale MM5 meteorological inputs and that the boundary conditions were taken from the appropriate 36 km continental U.S. simulations.

2.2.5 Environmental Effects of Ozone Pollution

There are a number of public welfare effects associated with the presence of ozone in the ambient air.¹³⁴ In this section we discuss the impact of ozone on plants, including trees, agronomic crops and urban ornamentals.

2.2.5.1 Impacts on Vegetation

The Air Quality Criteria Document for Ozone and related Photochemical Oxidants notes that “ozone affects vegetation throughout the United States, impairing crops, native vegetation, and ecosystems more than any other air pollutant. Like carbon dioxide (CO₂) and other gaseous substances, ozone enters plant tissues primarily through apertures (stomata) in leaves in a process called “uptake”.¹³⁵ Once sufficient levels of ozone, a highly reactive substance, (or its reaction products) reaches the interior of plant cells, it can inhibit or damage essential cellular components and functions, including enzyme activities, lipids, and cellular membranes, disrupting the plant's osmotic (i.e., water) balance and energy utilization patterns.^{136,137} This damage is commonly manifested as visible foliar injury such as chlorotic or necrotic spots, increased leaf senescence (accelerated leaf aging) and/or reduced photosynthesis. All these effects reduce a plant's capacity to form carbohydrates, which are the primary form of energy used by plants.¹³⁸ With fewer resources available, the plant reallocates existing resources away from root growth and storage, above ground growth or yield, and reproductive processes, toward leaf repair and maintenance. Studies have shown that plants stressed in these ways may exhibit a general loss of vigor, which can lead to secondary impacts that modify plants' responses to other environmental factors. Specifically, plants may become more sensitive to other air pollutants, more susceptible to disease, insect attack, harsh weather (e.g., drought, frost) and other environmental stresses. Furthermore, there is evidence that ozone can interfere with the formation of mycorrhiza, essential symbiotic fungi associated with the roots of most terrestrial plants, by reducing the amount of carbon available for transfer from the host to the symbiont.^{139,140}

Ozone can produce both acute and chronic injury in sensitive species depending on the concentration level and the duration of the exposure. Ozone effects also tend to accumulate over the growing season of the plant, so that even lower concentrations experienced for a longer duration have the potential to create chronic stress on sensitive vegetation. Not all plants, however, are equally sensitive to ozone. Much of the variation in sensitivity between individual plants or whole species is related to the plant's ability to regulate the extent of gas exchange via leaf stomata (e.g., avoidance of O₃ uptake through closure of stomata).^{141,142,143} Other resistance mechanisms may involve the intercellular production of detoxifying substances. Several biochemical substances capable of detoxifying ozone have been reported to occur in plants including the antioxidants ascorbate and glutathione. After injuries have occurred, plants may be capable of repairing the damage to a limited extent.¹⁴⁴

Because of the differing sensitivities among plants to ozone, ozone pollution can also exert a selective pressure that leads to changes in plant community composition. Given the range of plant sensitivities and the fact that numerous other environmental factors modify plant uptake and response to ozone, it is not possible to identify threshold values above which ozone is consistently toxic for all plants. The next few paragraphs present additional information on ozone damage to trees, ecosystems, agronomic crops and urban ornamentals.

Ozone also has been conclusively shown to cause discernible injury to forest trees.^{145,146} In terms of forest productivity and ecosystem diversity, ozone may be the pollutant with the greatest potential for regional-scale forest impacts. Studies have

demonstrated repeatedly that ozone concentrations commonly observed in polluted areas can have substantial impacts on plant function.^{147, 148}

Because plants are at the center of the food web in many ecosystems, changes to the plant community can affect associated organisms and ecosystems (including the suitability of habitats that support threatened or endangered species and below ground organisms living in the root zone). Ozone impacts at the community and ecosystem level vary widely depending upon numerous factors, including concentration and temporal variation of tropospheric ozone, species composition, soil properties and climatic factors.¹⁴⁹ In most instances, responses to chronic or recurrent exposure in forested ecosystems are subtle and not observable for many years. These injuries can cause stand-level forest decline in sensitive ecosystems.^{150,151,152} It is not yet possible to predict ecosystem responses to ozone with much certainty; however, considerable knowledge of potential ecosystem responses has been acquired through long-term observations in highly damaged forests in the United States.

Laboratory and field experiments have also shown reductions in yields for agronomic crops exposed to ozone, including vegetables (e.g., lettuce) and field crops (e.g., cotton and wheat). The most extensive field experiments, conducted under the National Crop Loss Assessment Network (NCLAN) examined 15 species and numerous cultivars. The NCLAN results show that “several economically important crop species are sensitive to ozone levels typical of those found in the United States.”¹⁵³ In addition, economic studies have shown reduced economic benefits as a result of predicted reductions in crop yields associated with observed ozone levels.^{154, 155, 156}

Urban ornamentals represent an additional vegetation category likely to experience some degree of negative effects associated with exposure to ambient ozone levels. It is estimated that more than \$20 billion (1990 dollars) are spent annually on landscaping using ornamentals, both by private property owners/tenants and by governmental units responsible for public areas.¹⁵⁷ This is therefore a potentially costly environmental effect. However, in the absence of adequate exposure-response functions and economic damage functions for the potential range of effects relevant to these types of vegetation, no direct quantitative analysis has been conducted.

2.3 Air Toxics

People experience elevated risk of cancer and other noncancer health effects from exposure to air toxics. Mobile sources are responsible for a significant portion of this risk. According to the National Air Toxic Assessment (NATA) for 1999, mobile sources were responsible for 44 percent of outdoor toxic emissions and almost 50 percent of the cancer risk. Benzene is the largest contributor to cancer risk of all 133 pollutants quantitatively assessed in the 1999 NATA. Mobile sources were responsible for 68 percent of benzene emissions in 1999. In response, EPA has proposed a series of mobile source and fuel controls that address this serious problem.^L Although the 1999 NATA did not quantify

^L U.S. EPA (2006). Control of Hazardous Air Pollutants From Mobile Sources. 71 FR 15804; March 29, 2006.

cancer risks associated with exposure to this diesel exhaust, EPA has concluded that diesel exhaust ranks with the other air toxic substances that the national-scale assessment suggests pose the greatest relative risk.

At the same time, nearly the entire U.S. population was exposed to an average level of air toxics that has the potential for adverse respiratory health effects (noncancer). This will continue to be the case in 2030, even though toxics levels will be lower. Mobile sources were responsible for 74 percent of the noncancer (respiratory) risk from outdoor air toxics in 1999. The majority of this risk was from acrolein, and formaldehyde also contributed to the risk of respiratory health effects. Mobile sources will continue to be responsible for the majority of noncancer risk from outdoor air toxics in 2030. Although not included in NATA's estimates of noncancer risk, PM from gasoline and diesel mobile sources contribute significantly to the health effects associated with ambient PM.

It should be noted that the NATA modeling framework has a number of limitations which prevent its use as the sole basis for setting regulatory standards. These limitations and uncertainties are discussed on the 1999 NATA website.¹⁵⁸ Even so, this modeling framework is very useful in identifying air toxic pollutants and sources of greatest concern, setting regulatory priorities, and informing the decision making process.

The following section provides an overview of air toxics which are associated with nonroad engines including locomotive and marine diesel engines and provides a discussion of the health risks associated with each air toxic.

2.3.1 Diesel Exhaust PM

Locomotive and marine diesel engine PM_{2.5} emissions include diesel exhaust (DE), a complex mixture comprised of carbon dioxide, oxygen, nitrogen, water vapor, carbon monoxide, nitrogen compounds, sulfur compounds and numerous low-molecular-weight hydrocarbons. A number of these gaseous hydrocarbon components are individually known to be toxic including aldehydes, benzene and 1,3-butadiene. The diesel particulate matter (DPM) present in diesel exhaust consists of fine particles (< 2.5µm), including a subgroup with a large number of ultrafine particles (< 0.1 µm). These particles have large surface area which makes them an excellent medium for adsorbing organics as well as their small size makes them highly respirable and able to reach the deep lung. Many of the organic compounds present on the particles and in the gases are individually known to have mutagenic and carcinogenic properties. Diesel exhaust varies significantly in chemical composition and particle sizes between different engine types (heavy-duty, light-duty), engine operating conditions (idle, accelerate, decelerate), and fuel formulations (high/low sulfur fuel).¹⁵⁹ Also, there are emission differences between on-road and nonroad engines because the nonroad engines are generally of older technology. This is especially true for locomotive and marine diesel engines.

After emission from the tailpipe, diesel exhaust undergoes dilution as well as chemical and physical changes in the atmosphere. The lifetime for some of the compounds present in diesel exhaust ranges from hours to days. Although the 1999 National-Scale Air Toxics Assessment (NATA) did not quantify cancer risks associated with exposure to this

pollutant, EPA has concluded that diesel exhaust ranks with the other air toxic substances that the national-scale assessment suggests pose the greatest relative risk. Following is a discussion of the health risks associated with diesel exhaust.

A number of health studies have been conducted regarding diesel exhaust including epidemiologic studies of lung cancer in groups of workers, and animal studies focusing on non-cancer effects specific to diesel exhaust. Diesel exhaust PM (including the associated organic compounds which are generally high molecular weight hydrocarbon types but not the more volatile gaseous hydrocarbon compounds) is generally used as a surrogate measure for diesel exhaust.

2.3.1.1 Potential Cancer Effects of Diesel Exhaust

In addition to its contribution to ambient PM inventories, diesel exhaust is of specific concern because it has been judged to pose a lung cancer hazard for humans as well as a hazard from noncancer respiratory effects such as pulmonary inflammation.

EPA's 2002 final "Health Assessment Document for Diesel Engine Exhaust" (the EPA Diesel HAD classified diesel exhaust as likely to be carcinogenic to humans by inhalation at environmental exposures, in accordance with the revised draft 1996/1999 EPA cancer guidelines.^{160,161} In accordance with earlier EPA guidelines, diesel exhaust would be similarly classified as a probable human carcinogen (Group B1).^{162,163} A number of other agencies (National Institute for Occupational Safety and Health, the International Agency for Research on Cancer, the World Health Organization, California EPA, and the US Department of Health and Human Services) have made similar classifications.^{164, 165,166,167,168} The Health Effects Institute has also made numerous studies and report on the potential carcinogenicity of diesel exhaust.^{169, 170, 171} Numerous animal and bioassay/genotoxic tests have been done on diesel exhaust.^{172, 173} Also, case-control and cohort studies have been conducted on railroad engine exposures^{174, 175,176} in addition to studies on truck workers.^{177, 178,179, 180} Also, there are numerous other epidemiologic studies including some studying mine workers and fire fighters.^{181, 182}

More specifically, the EPA Diesel HAD states that the conclusions of the document apply to diesel exhaust in use today including both onroad and nonroad engines. The EPA Diesel HAD acknowledges that the studies were done on engines with older technologies generally for onroad and that "there have been changes in the physical and chemical composition of some DE [diesel exhaust] emissions (onroad vehicle emissions) over time, though there is no definitive information to show that the emission changes portend significant toxicological changes." In any case, the diesel technology used for locomotive and marine diesel engines typically lags that used for nonroad engines which have been subject to PM standards since 1998, thus it is reasonable to assume that the hazards identified from older technologies may be largely applicable to locomotive and marine engines.

For the Diesel HAD, EPA reviewed 22 epidemiologic studies on the subject of the carcinogenicity of workers exposed to diesel exhaust in various occupations, finding increased lung cancer risk, although not always statistically significant, in 8 out of 10 cohort studies and 10 out of 12 case-control studies within several industries, including railroad

workers. Relative risk for lung cancer associated with exposure ranged from 1.2 to 1.5, although a few studies show relative risks as high as 2.6. Additionally, the Diesel HAD also relied on two independent meta-analyses, which examined 23 and 30 occupational studies respectively, which found statistically significant increases in smoking-adjusted relative lung cancer risk associated with diesel exhaust, of 1.33 to 1.47. These meta-analyses demonstrate the effect of pooling many studies and in this case show the positive relationship between diesel exhaust exposure and lung cancer across a variety of diesel exhaust-exposed occupations.^{183,184,185}

Retrospective health studies of railroad workers have played an important part in finding that diesel exhaust is a likely human carcinogen. Key evidence of the diesel exhaust exposure linkage to lung cancer comes from two retrospective case-control studies of railroad workers. The Garshick railroad study¹⁸⁶ looked at more than 55,000 railroad workers post-1959 which coincided with the widespread dieselization of the railroads. The study found that the risk of lung cancer increased with increasing duration of employment, and that the youngest workers had the highest risk of dying. The second railroad study authored by Swanson et al.¹⁸⁷ found statistically significant excess risks, when adjusted for age, smoking, and race, among railroad workers employed for more than 10 years and heavy truck drivers employed for more than 20 years. In addition, a 1988 industrial hygiene study documented the increased lung cancer risks associated with different railroad worker job classifications.¹⁸⁸ Thirty-nine job titles were originally identified and were then collapsed, for statistical analyses, into 5 categories including clerks, signal maintainers, engineers/firers, brakemen/conductors/hostlers, and shop workers. The study documented that those in closest contact with diesel exhaust exhibited the highest level of lung cancer risk. Train workers (engineers/firers etc.) had the highest risk, shop workers an intermediate level, and clerks the lowest lung cancer risk.

EPA generally derives cancer unit risk estimates to calculate population risk more precisely from exposure to carcinogens. In the simplest terms, the cancer unit risk is the increased risk associated with average lifetime exposure of $1 \mu\text{g}/\text{m}^3$. EPA concluded in the Diesel HAD that it is not possible currently to calculate a cancer unit risk for diesel exhaust due to a variety of factors that limit the current studies, such as a lack of standard exposure metric for diesel exhaust and the absence of quantitative exposure characterization in retrospective studies.

However, in the absence of a cancer unit risk, the EPA Diesel HAD sought to provide additional insight into the possible ranges of risk that might be present in the population. Such insights, while not confident or definitive, nevertheless contribute to an understanding of the possible public health significance of the lung cancer hazard. An exploratory analysis was used to characterize a possible risk range by comparing a typical environmental exposure level to a selected range of occupational exposure levels and then proportionally scaling the occupationally observed risks according to the exposure ratios to obtain an estimate of the possible environmental risk. If the occupational and environmental exposures are similar, the environmental risk would approach the risk seen in the occupational studies whereas a much higher occupational exposure indicates that the environmental risk is lower than the occupational risk. A comparison of environmental and occupational exposures

showed that for certain occupations the exposures are similar to environmental exposures while, for others, they differ by a factor of about 200 or more.

The first step in this process is to note that the occupational relative risk of 1.4, or a 40 percent from increased risk compared to the typical 5 percent lung cancer risk in the U.S. population, translates to an increased risk of 2 percent (or 10^{-2}) for these diesel exhaust exposed workers. The Diesel HAD derived a typical nationwide average environmental exposure level of $0.8 \mu\text{g}/\text{m}^3$ for diesel PM from on-highway sources for 1996. This estimate was based on national exposure modeling; the derivation of this exposure is discussed in detail in the EPA Diesel HAD. Diesel PM is a surrogate for diesel exhaust and, as mentioned above, has been classified as a carcinogen by some agencies.

The possible environmental risk range was estimated by taking the relative risks in the occupational setting, EPA selected 1.4 and converting this to absolute risk of 2% and then ratioing this risk by differences in the occupational versus environmental exposures of interest. A number of calculations are needed to accomplish this, and these can be seen in the EPA Diesel HAD. The outcome was that environmental risks from diesel exhaust using higher estimates of occupational exposure could range from a low of 10^{-4} to 10^{-5} or be as high as 10^{-3} if lower estimates of occupational exposure were used. Note that the environmental exposure of interest ($0.8 \mu\text{g}/\text{m}^3$) remains constant in this analysis, while the occupational exposure is a variable. The range of possible environmental risk is a reflection of the range of occupational exposures that could be associated with the relative and related absolute risk levels observed in the occupational studies.

While these risk estimates are exploratory and not intended to provide a definitive characterization of cancer risk, they are useful in gauging the possible range of risk based on reasonable judgment. It is important to note that the possible risks could also be higher or lower and a zero risk cannot be ruled out. Some individuals in the population may have a high tolerance to exposure from diesel exhaust and low cancer susceptibility. Also, one cannot rule out the possibility of a threshold of exposure below which there is no cancer risk, although no evidence is available on this point. As discussed in the Diesel HAD, there is a relatively small difference between some occupational studies where increased lung cancer risk is reported and concentrations sometimes seen in ambient settings.

EPA recently assessed air toxic emissions and their associated risk (the National-Scale Air Toxics Assessment or NATA for 1996 and 1999), and we concluded that diesel exhaust ranks with substances that the national-scale assessment suggests pose the greatest relative risk.^{189,190} This national assessment estimates average population inhalation exposures to diesel PM for nonroad as well as on-highway sources. These are the sum of ambient levels in various locations weighted by the amount of time people spend in each of the locations. The EPA Diesel HAD states that use of the 1996 NATA exposure estimates instead of the $0.8 \mu\text{g}/\text{m}^3$ estimate results in a similar risk perspective.

In summary, even though EPA does not have a specific carcinogenic potency with which to accurately estimate the carcinogenic impact of diesel exhaust, the likely hazard to humans together with the potential for significant environmental risks leads us to conclude

that diesel exhaust emissions from locomotive and marine engines present public health issues of concern to this proposal.

2.3.1.2 Other Health Effects of Diesel Exhaust

Noncancer health effects of acute and chronic exposure to diesel exhaust emissions are also of concern to the Agency. The Diesel HAD established an inhalation Reference Concentration (RfC) specifically based on animal studies of diesel exhaust. An RfC is defined by EPA as “an estimate of a continuous inhalation exposure to the human population, including sensitive subgroups, with uncertainty spanning perhaps an order of magnitude, which is likely to be without appreciable risks of deleterious noncancer effects during a lifetime.” EPA derived the RfC from consideration of four well-conducted chronic rat inhalation studies showing adverse pulmonary effects.^{191, 192, 193, 194} The diesel RfC is based on a “no observable adverse effect” level of $144 \mu\text{g}/\text{m}^3$ that is further reduced by applying uncertainty factors of 3 for interspecies extrapolation and 10 for human variations in sensitivity. The resulting RfC derived in the Diesel HAD is $5 \mu\text{g}/\text{m}^3$ for diesel exhaust as measured by diesel PM. This RfC does not consider allergenic effects such as those associated with asthma or immunologic effects. There is growing evidence that diesel exhaust can exacerbate these effects, but the exposure-response data is presently lacking to derive an RfC. The EPA Diesel HAD states, “With DPM [diesel particulate matter] being a ubiquitous component of ambient PM, there is an uncertainty about the adequacy of the existing DE [diesel exhaust] noncancer database to identify all of the pertinent DE-caused noncancer health hazards” (p. 9-19).

While there have been relatively few human studies associated specifically with the noncancer impact of diesel PM alone, diesel PM is frequently part of the ambient particles studied in numerous epidemiologic studies. Conclusions that health effects associated with ambient PM in general is relevant to diesel PM is supported by studies that specifically associate observable human noncancer health effects with exposure to diesel PM. As described in the Diesel HAD, these studies include some of the same health effects reported for ambient PM, such as respiratory symptoms (cough, labored breathing, chest tightness, wheezing), and chronic respiratory disease (cough, phlegm, chronic bronchitis and suggestive evidence for decreases in pulmonary function). Symptoms of immunological effects such as wheezing and increased allergenicity are also seen. Studies in rodents, especially rats, show the potential for human inflammatory effects in the lung and consequential lung tissue damage from chronic diesel exhaust inhalation exposure. The Diesel HAD notes that acute or short-term exposure to diesel exhaust can cause acute irritation (e.g., eye, throat, bronchial), neurophysiological symptoms (e.g., lightheadedness, nausea), and respiratory symptoms (cough, phlegm). There is also evidence for an immunologic effect such as the exacerbation of allergenic responses to known allergens and asthma-like symptoms.^{195,196,197} The Diesel HAD lists numerous other studies as well. Also, as discussed in more detail previously, in addition to its contribution to ambient PM inventories, diesel PM is of special concern because it has been associated with an increased risk of lung cancer.

Diesel exhaust has been shown to cause serious noncancer effects in occupational exposure studies. One recent study¹⁹⁸ of a small group of railroad workers and electricians

found that exposure to diesel exhaust resulted in neurobehavioral impairments in one or more areas including reaction time, balance, blink reflex latency, verbal recall, and color vision confusion indices. Pulmonary function tests also showed that 10 of the 16 workers had airway obstruction and another group of 10 of 16 workers had chronic bronchitis, chest pain, tightness, and hyperactive airways. Finally, a variety of studies have been published subsequent to the completion of the Diesel HAD. One such study, published in 2006¹⁹⁹ found that railroad engineers and conductors with diesel exhaust exposure from operating trains had an increased incidence of chronic obstructive pulmonary disease (COPD) mortality. The odds of COPD mortality increased with years on the job so that those who had worked more than 16 years as an engineer or conductor after 1959 had an increased risk of 1.61 (95% confidence interval, 1.12 - 2.30). EPA is assessing the significance of this study within the context of the broader literature.

The Diesel HAD also briefly summarizes health effects associated with ambient PM and discusses the EPA's annual NAAQS of $15 \mu\text{g}/\text{m}^3$. There is a much more extensive body of human data showing a wide spectrum of adverse health effects associated with exposure to ambient PM, of which diesel exhaust is an important component. The $\text{PM}_{2.5}$ NAAQS is designed to provide protection from the non-cancer and premature mortality effects of $\text{PM}_{2.5}$ as a whole, of which diesel PM is a constituent.

Also, as mentioned earlier in the health effects discussion for $\text{PM}_{2.5}$, there are a number of other health effects associated with PM in general, and mobile source exhaust including diesels in particular, that provide additional evidence for the need for significant emission reductions from locomotive and marine diesel sources.

As indicated earlier, a number of recent studies have associated living near roadways with adverse health effects. Two of the studies cited earlier will be mentioned again here as examples of the type of work that has been done. A Dutch study (discussed earlier by G. Hoek and others) of a population of people 55-69 years old found that there was an elevated risk of heart and lung related mortality among populations living near high traffic roads. In a review discussed earlier of studies (by R. Delfino) of the respiratory health of people living near roadways, another publication indicated that the risk of asthma and related respiratory disease appeared elevated in people living near heavy traffic. These studies offer evidence that people exposed most directly to emissions from mobile sources including those from diesels face an elevated risk of illness or death.

All of these health effects plus the designation of diesel exhaust as a likely human carcinogen provide ample health justification for control.

2.3.1.3 Diesel Exhaust PM Ambient Levels

Because diesel PM is part of overall ambient PM and cannot be easily distinguished from overall PM, we do not have direct measurements of diesel PM in the ambient air. Diesel PM concentrations are estimated instead using one of three approaches: 1) ambient air quality modeling based on diesel PM emission inventories; 2) using elemental carbon concentrations in monitored data as surrogates; or 3) using the chemical mass balance (CMB) model in conjunction with ambient PM measurements. (Also, in addition to CMB,

UNMIX/PMF have also been used). Estimates using these three approaches are described below. In addition, estimates developed using the first two approaches above are subjected to a statistical comparison to evaluate overall reasonableness of estimated concentrations from ambient air quality modeling. It is important to note that, while there are inconsistencies in some of these studies on the relative importance of gasoline and diesel PM, the studies which are discussed in the Diesel HAD all show that diesel PM is a significant contributor to overall ambient PM. Some of the studies differentiate nonroad from on-highway diesel PM.

2.3.1.3.1 Toxics Modeling and Methods

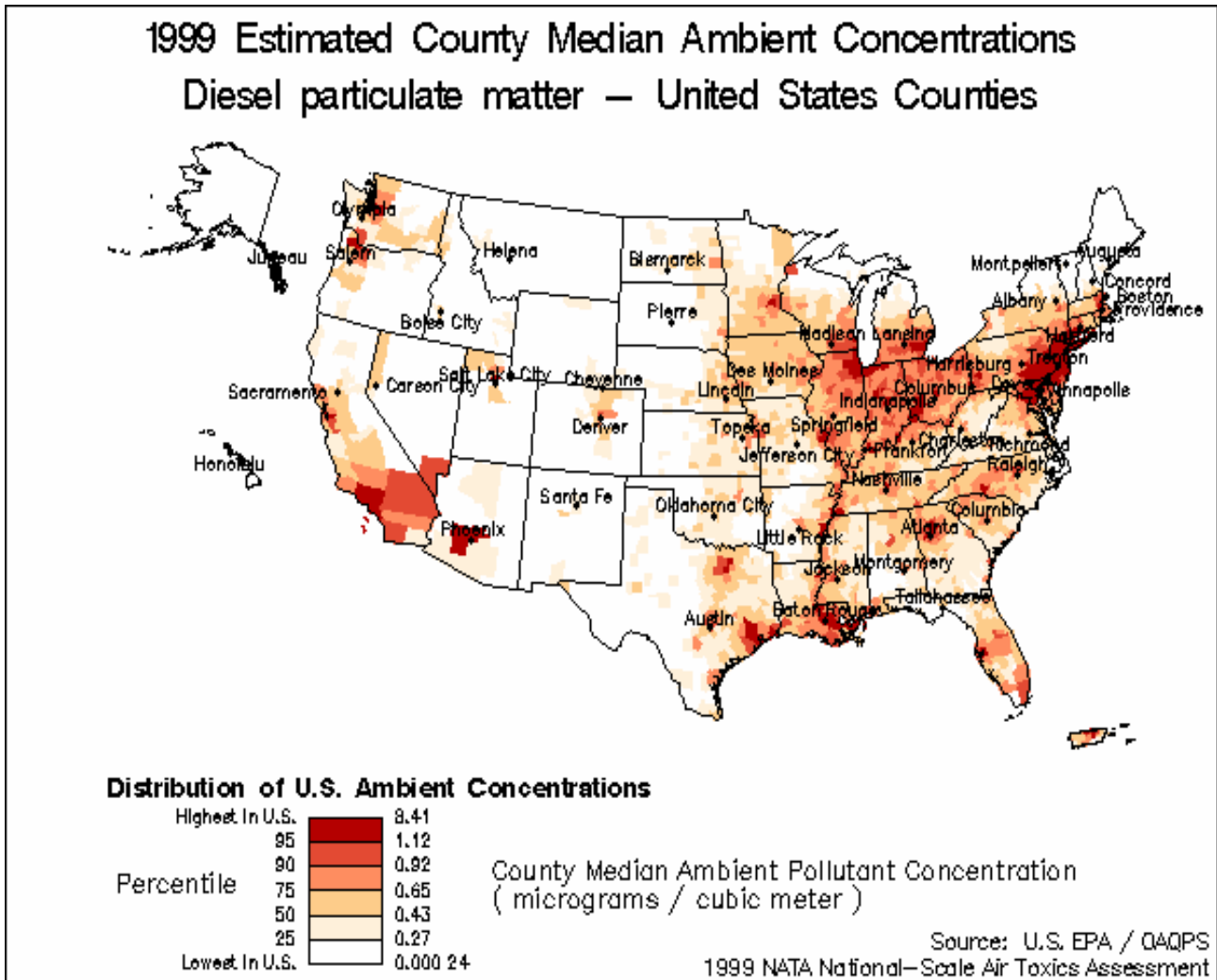
In addition to the general ambient PM modeling conducted for this proposal, diesel PM concentrations for 1999 were recently estimated as part of the second National-Scale Air Toxics Assessment (NATA; EPA, 2006). Ambient impacts of mobile source emissions were predicted using the Assessment System for Population Exposure Nationwide (ASPEN) dispersion model.

From the NATA 1999 modeling, overall medium annual national ambient diesel PM levels of .91 $\mu\text{g}/\text{m}^3$ were calculated with a medium of 1.06 in urban counties and 0.43 in rural counties. Table 2-8 below summarizes the distribution of medium ambient concentrations to diesel PM at the national scale. Over half, 62 percent, of the diesel PM and diesel exhaust organic gases can be attributed to nonroad diesels. A map of county median ambient concentrations is provided in Figure 2-8. While the high median concentrations are clustered in the Northeast, Great Lake States California, and the Gulf Coast States, areas of high median concentrations are distributed throughout the U.S.

Table 2-8 Distribution of Median Ambient Concentrations of Diesel PM at the National Scale in the 1999 NATA Assessment.

	Nationwide ($\mu\text{g}/\text{m}^3$)	Urban ($\mu\text{g}/\text{m}^3$)	Rural ($\mu\text{g}/\text{m}^3$)
5th Percentile	0.21	0.22	0.08
25 th Percentile	0.54	.70	0.28
Medium	0.91	1.06	0.43
75 th Percentile	1.41	1.56	0.62
95th Percentile	2.91	3.21	.96
Onroad Contribution to Mean	0.43	0.49	0.20
Nonroad Contribution to Mean	0.78	0.90	0.28

Figure 2-10 Estimated County Median Ambient Concentration of Diesel Particulate Matter



2.3.1.4 Diesel Exhaust PM Exposures

Exposure of people to diesel exhaust depends on their various activities, the time spent in those activities, the locations where these activities occur, and the levels of diesel exhaust pollutants in those locations. The major difference between ambient levels of diesel particulate and exposure levels for diesel particulate is that exposure accounts for a person moving from location to location, proximity to the emission source, and whether the exposure occurs in an enclosed environment.

2.3.1.4.1 Occupational Exposures

Occupational exposures to diesel exhaust from mobile sources, including locomotive engines and marine diesel engines, can be several orders of magnitude greater than typical exposures in the non-occupationally exposed population.

Over the years, diesel particulate exposures have been measured for a number of occupational groups resulting in a wide range of exposures from 2 to 1,280 $\mu\text{g}/\text{m}^3$ for a variety of occupations. Studies have shown that miners and railroad workers typically have higher diesel exposure levels than other occupational groups studied, including firefighters, truck dock workers, and truck drivers (both short and long haul).²⁰⁰ A 1988 study²⁰¹ estimated that U.S. railroad workers received an estimated occupational exposure/concentration of between 39 -191 $\mu\text{g}/\text{m}^3$ which resulted in an equivalent environmental exposure of 8-40 $\mu\text{g}/\text{m}^3$. As discussed in the Diesel HAD, the National Institute of Occupational Safety and Health (NIOSH) has estimated a total of 1,400,000 workers are occupationally exposed to diesel exhaust from on-road and nonroad vehicles including locomotive and marine diesel engines.

2.3.1.4.1.1 Elevated Concentrations and Ambient Exposures in Mobile Source-Impacted Areas

While occupational studies indicate that those in closest proximity to diesel exhaust experience the greatest health effects, recent studies are showing that human populations living near large diesel emission sources such as major roadways,²⁰² rail yards and marine ports²⁰³ are also likely to experience greater diesel exhaust exposure levels than the overall population putting them at greater health risks.

Regions immediately downwind of rail yards and marine ports may experience elevated ambient concentrations of directly-emitted $\text{PM}_{2.5}$ from diesel engines. Due to the unique nature of rail yards and marine ports, emissions from a large number of diesel engines are concentrated in a small area. Furthermore, emissions occur at or near ground level, allowing emissions of diesel engines to reach nearby receptors without fully mixing with background air.

A recent study conducted by the California Air Resources Board (CARB) examined the air quality impacts of railroad operations at the J.R. Davis Rail Yard, the largest rail facility in the western United States.²⁰⁴ The yard occupies 950 acres along a one-quarter mile wide and four mile long section of land in Roseville, CA. The study developed an emissions inventory for the facility for the year 2000 and modeled ambient concentrations of diesel PM using a well-accepted dispersion model (ISCST3). The study found substantially elevated concentrations in an area 5,000 meters from the facility, with higher concentrations closer to the rail yard. Using local meteorological data, annual average contributions from the rail yard to ambient diesel PM concentrations under prevailing wind conditions were 1.74, 1.18, 0.80, and 0.25 $\mu\text{g}/\text{m}^3$ at receptors located 200, 500, 1000, and 5000 meters from the yard, respectively. Several tens of thousands of people live within the area experiencing substantial increases in annual average ambient $\text{PM}_{2.5}$ as a result of emissions from the yard.

Another study from CARB evaluated air quality impacts of diesel engine emissions within the Ports of Long Beach and Los Angeles in California, one of the largest ports in the U.S. ²⁰⁵ Like the earlier rail yard study, the port study employed the ISCST3 dispersion model. Also using local meteorological data, annual average concentrations were substantially elevated over an area exceeding 200,000 acres. Because they are located near heavily-populated areas, the modeling indicated that over 700,000 people lived in areas with at least $0.3 \mu\text{g}/\text{m}^3$ of port-related diesel PM in ambient air, about 360,000 people lived in areas with at least $0.6 \mu\text{g}/\text{m}^3$ of diesel PM, and about 50,000 people lived in areas with at least $1.5 \mu\text{g}/\text{m}^3$ of ambient diesel PM directly from the port. Figure 2-11 provides an aerial shot of the Port of Long beach and Los Angeles in California.

Figure 2-11 Aerial Shot – Port of LA and Long Beach, California



While these studies focus on two large marine port and one large rail yard facility, these studies do highlight the substantial contribution these facilities make to elevated ambient concentrations in large, densely populated areas.

We have recently initiated a study to better understand the populations that are living near rail yards and marine ports. As part of the study, a computer geographic information system (GIS) is being used to identify the locations and property boundaries of a sampling of these facilities nationally, and to determine the size and demographic characteristics of the population living near these facilities. We anticipate that the results of this study will be complete in early 2007 and we intend to add this report to the public docket in advance of the final rulemaking. Figure 2.-12 to 2.-14 provides a sampling of aerial photos of the rail yards and marine ports that are part of this study.

Figure 2-12 2006 aerial photo Port of Cleveland, Cleveland Ohio

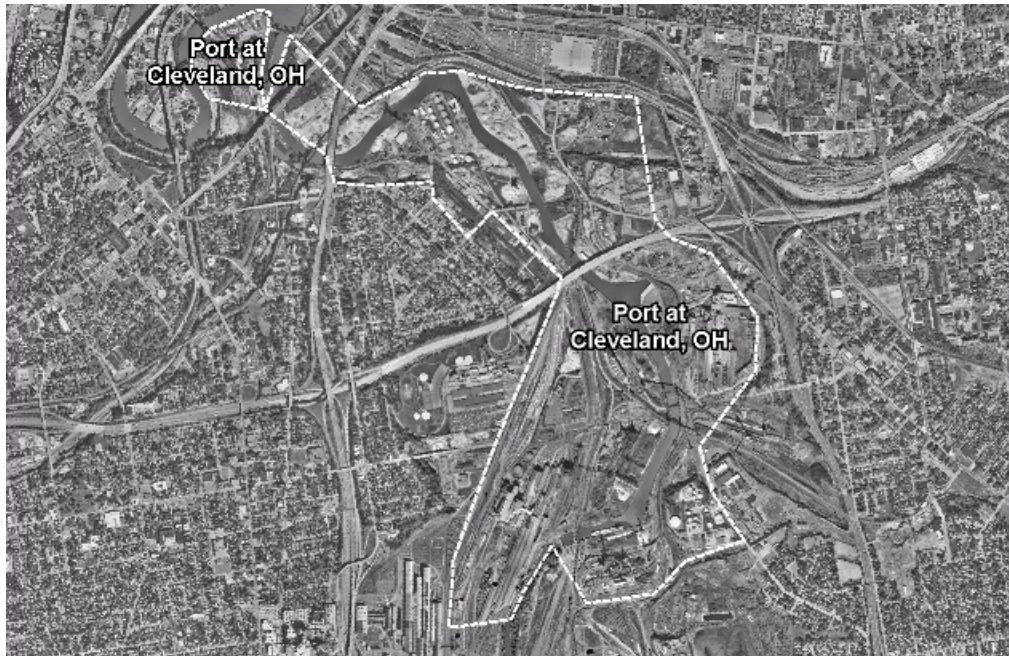
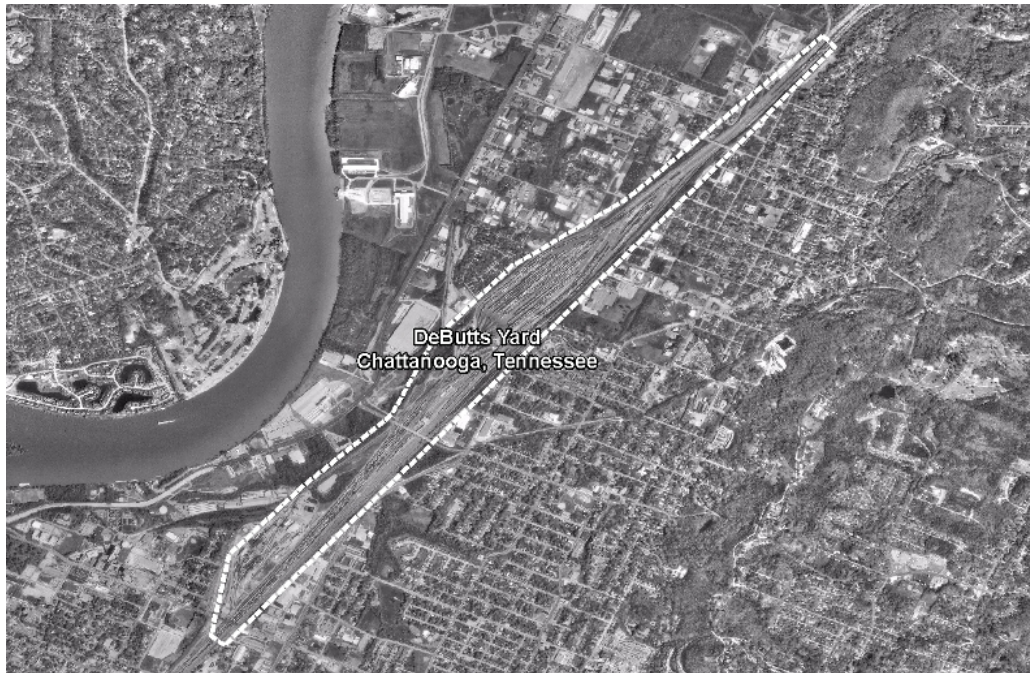


Figure 2-13 2006 aerial photo Argentine Rail Yard, Kansas City, Missouri



Figure 2-14. 2006 aerial photo DeButts Rail Yard, Chattanooga, Tennessee



2.4 Gaseous Air Toxics—benzene, 1,3-butadiene, formaldehyde, acetaldehyde, acrolein, POM, naphthalene

Locomotive and marine diesel engine emissions contribute to ambient levels of other air toxics known or suspected as human or animal carcinogens, or that have non-cancer health effects. These other compounds include benzene, 1,3-butadiene, formaldehyde, acetaldehyde, acrolein, polycyclic organic matter (POM), and naphthalene. All of these compounds, except acetaldehyde, were identified as national or regional risk drivers in the 1999 National-Scale Air Toxics Assessment (NATA) and have significant inventory contributions from mobile sources. Table 2 – 9 provides the mobile source contributions associated with these compounds. The reductions in locomotive and marine diesel engine emissions proposed in this rulemaking would help reduce exposure to these harmful substances.

Table 2-9 Mobile Source Contribution to 1999 NATA Risk Drivers

1999 NATA Risk Drivers	Percent Contribution from ALL Mobile Sources	Percent Contribution for Non-road Mobile Sources
Benzene	68%	19%
1,2-Butadiene	58%	17%
Formaldehyde	47%	20%
Acrolein	25%	11%

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Polycyclic organic matter (POM)*	6%	3%
Naphthalene	27%	6%
Diesel PM and Diesel exhaust organic gases	100%	62%

*This POM inventory includes the 15 POM compounds: benzo[b]fluoranthene, benz[a]anthracene, indeno(1,2,3-c,d)pyrene, benzo[k]fluoranthene, chrysene, benzo[a]pyrene, dibenz(a,h)anthracene, anthracene, pyrene, benzo(g,h,i)perylene, fluoranthene, acenaphthylene, phenanthrene, fluorine, and acenaphthene.

Air toxics can cause a variety of cancer and noncancer health effects. A number of the mobile source air toxic pollutants described in this section are known or likely to pose a cancer hazard in humans. Many of these compounds also cause adverse noncancer health effects resulting from chronic,²⁰⁶ subchronic,²⁰⁷ or acute²⁰⁸ inhalation exposures. These include neurological, cardiovascular, liver, kidney, and respiratory effects as well as effects on the immune and reproductive systems.

Benzene: The EPA's IRIS database lists benzene as a known human carcinogen (causing leukemia) by all routes of exposure, and that exposure is associated with additional health effects, including genetic changes in both humans and animals and increased proliferation of bone marrow cells in mice.^{209, 210, 211} EPA states in its IRIS database that data indicate a causal relationship between benzene exposure and acute lymphocytic leukemia and suggests a relationship between benzene exposure and chronic non-lymphocytic leukemia and chronic lymphocytic leukemia. A number of adverse noncancer health effects including blood disorders, such as preleukemia and aplastic anemia, have also been associated with long-term exposure to benzene.^{212, 213} The most sensitive noncancer effect observed in humans, based on current data, is the depression of the absolute lymphocyte count in blood.^{214, 215} In addition, recent work, including studies sponsored by the Health Effects Institute (HEI), provides evidence that biochemical responses are occurring at lower levels of benzene exposure than previously known.^{216, 217, 218, 219} EPA's IRIS program has not yet evaluated these new data

1,3-Butadiene: EPA has characterized 1,3-butadiene as carcinogenic to humans by inhalation.^{220, 221} The specific mechanisms of 1,3-butadiene-induced carcinogenesis are unknown. However, it is virtually certain that the carcinogenic effects are mediated by genotoxic metabolites of 1,3-butadiene. Animal data suggest that females may be more sensitive than males for cancer effects; while there are insufficient data in humans from which to draw conclusions about sensitive subpopulations. 1,3-Butadiene also causes a variety of reproductive and developmental effects in mice; no human data on these effects are available. The most sensitive effect was ovarian atrophy observed in a lifetime bioassay of female mice.²²²

Formaldehyde: Since 1987, EPA has classified formaldehyde as a probable human carcinogen based on evidence in humans and in rats, mice, hamsters, and monkeys.²²³ EPA's current IRIS summary provides an upper bound cancer unit risk estimate of 1.3×10^{-5} per

$\mu\text{g}/\text{m}^3$. In other words, there is an estimated risk of about thirteen excess leukemia cases in one million people exposed to $1 \mu\text{g}/\text{m}^3$ of formaldehyde over a lifetime. EPA is currently reviewing recently published epidemiological data. For instance, research conducted by the National Cancer Institute (NCI) found an increased risk of nasopharyngeal cancer and lymphohematopoietic malignancies such as leukemia among workers exposed to formaldehyde.^{224, 225} NCI is currently performing an update of these studies. A recent National Institute of Occupational Safety and Health (NIOSH) study of garment workers also found increased risk of death due to leukemia among workers exposed to formaldehyde.²²⁶ Extended follow-up of a cohort of British chemical workers did not find evidence of an increase in nasopharyngeal or lymphohematopoietic cancers, but a continuing statistically significant excess in lung cancers was reported.²²⁷

Based on the developments of the last decade, in 2004, the working group of the International Agency for Research on Cancer (IARC) concluded that formaldehyde is carcinogenic to humans (Group 1), on the basis of sufficient evidence in humans and sufficient evidence in experimental animals—a higher classification than previous IARC evaluations. The Agency is currently conducting a reassessment of the human hazard and dose-response associated with formaldehyde.

In the past 15 years there has been substantial research on the inhalation dosimetry for formaldehyde in rodents and primates by the CIIT Centers for Health Research (formerly the Chemical Industry Institute of Toxicology), with a focus on use of rodent data for refinement of the quantitative cancer dose-response assessment.^{228, 229, 230} CIIT's risk assessment of formaldehyde incorporated mechanistic and dosimetric information on formaldehyde. The risk assessment analyzed carcinogenic risk from inhaled formaldehyde using approaches that are consistent with EPA's draft guidelines for carcinogenic risk assessment. In 2001, Environment Canada relied on this cancer dose-response assessment in their assessment of formaldehyde.²³¹ In 2004, EPA also relied on this cancer unit risk estimate during the development of the plywood and composite wood products national emissions standards for hazardous air pollutants (NESHAPs).²³² In these rules, EPA concluded that the CIIT work represented the best available application of the available mechanistic and dosimetric science on the dose-response for portal of entry cancers due to formaldehyde exposures. EPA is reviewing the recent work cited above from the NCI and NIOSH, as well as the analysis by the CIIT Centers for Health Research and other studies, as part of a reassessment of the human hazard and dose-response associated with formaldehyde.

Formaldehyde exposure also causes a range of noncancer health effects, including irritation of the eyes (tearing of the eyes and increased blinking) and mucous membranes.

Acetaldehyde: Acetaldehyde is classified in EPA's IRIS database as a probable human carcinogen, based on nasal tumors in rats, and is considered toxic by the inhalation, oral, and intravenous routes.²³³ The primary acute effect of exposure to acetaldehyde vapors is irritation of the eyes, skin, and respiratory tract.²³⁴ The agency is currently conducting a reassessment of the health hazards from inhalation exposure to acetaldehyde.

Acrolein: Acrolein is intensely irritating to humans when inhaled, with acute exposure resulting in upper respiratory tract irritation and congestion. EPA determined in

2003 using the 1999 draft cancer guidelines that the human carcinogenic potential of acrolein could not be determined because the available data were inadequate. No information was available on the carcinogenic effects of acrolein in humans and the animal data provided inadequate evidence of carcinogenicity.²³⁵

Polycyclic Organic Matter (POM): POM is generally defined as a large class of organic compounds which have multiple benzene rings and a boiling point greater than 100 degrees Celsius. Many of the compounds included in the class of compounds known as POM are classified by EPA as probable human carcinogens based on animal data. One of these compounds, naphthalene, is discussed separately below.

Recent studies have found that maternal exposures to PAHs, in a population of pregnant women were associated with several adverse birth outcomes, including low birth weight and reduced length at birth as well as impaired cognitive development at age three.^{236, 237} EPA has not yet evaluated these recent studies.

Naphthalene: Naphthalene is found in small quantities in gasoline and diesel fuels but is primarily a product of combustion. Naphthalene emissions have been measured in larger quantities in both gasoline and diesel exhaust and evaporative emissions from mobile sources. EPA recently released an external review draft of a reassessment of the inhalation carcinogenicity of naphthalene based on a number of recent animal carcinogenicity studies.²³⁸ The draft reassessment recently completed external peer review.²³⁹ California EPA has released a new risk assessment for naphthalene, and the IARC has reevaluated naphthalene and re-classified it as Group 2B: possibly carcinogenic to humans.²⁴⁰ Naphthalene also causes a number of chronic non-cancer effects in animals, including abnormal cell changes and growth in respiratory and nasal tissues.²⁴¹

In addition to reducing substantial amounts of NO_x and PM_{2.5} emissions from locomotive and marine diesel engines the standards being proposed today would also reduce air toxics emitted from these engines thereby helping to mitigate some of the adverse health effects associated with operation of these engines.

Chapter 2 Appendices: Air Quality and Resulting Health and Welfare Effects

Appendix 2A PM_{2.5} Nonattainment

Table 2A PM_{2.5} Nonattainment Areas and Populations (Data is current through October 2006 and Population Numbers are from 2000 Census Data)

County	Area Name	County NA Whole/Part	Design Value (µg/m³)	Pop (2000)
ALABAMA				
Jackson Co	Chattanooga, AL-TN-GA	Part	16.1	1,578
Jefferson Co	Birmingham, AL	Whole	17.3	662,047
Shelby Co	Birmingham, AL	Whole	17.3	143,293
Walker Co	Birmingham, AL	Part	17.3	2,272
CALIFORNIA				
Fresno Co	San Joaquin Valley, CA	Whole	21.8	799,407
Kern Co	San Joaquin Valley, CA	Part	21.8	550,220
Kings Co	San Joaquin Valley, CA	Whole	21.8	129,461
Los Angeles Co	Los Angeles-South Coast Air Basin, CA	Part	27.8	9,222,280
Madera Co	San Joaquin Valley, CA	Whole	21.8	123,109
Merced Co	San Joaquin Valley, CA	Whole	21.8	210,554
Orange Co	Los Angeles-South Coast Air Basin, CA	Whole	27.8	2,846,289
Riverside Co	Los Angeles-South Coast Air Basin, CA	Part	27.8	1,194,859
San Bernardino Co	Los Angeles-South Coast Air Basin, CA	Part	27.8	1,330,159
San Joaquin Co	San Joaquin Valley, CA	Whole	21.8	563,598
Stanislaus Co	San Joaquin Valley, CA	Whole	21.8	446,997
Tulare Co	San Joaquin Valley, CA	Whole	21.8	368,021
CONNECTICUT				
Fairfield Co	New York-N. New Jersey-Long Island, NY-NJ-CT	Whole	17.7	882,567
New Haven Co	New York-N. New Jersey-Long Island, NY-NJ-CT	Whole	17.7	824,008
DELAWARE				
New Castle Co	Philadelphia-Wilmington, PA-NJ-DE	Whole	16.2	500,265
DISTRICT OF COLUMBIA				
Entire District	Washington, DC-MD-VA	Whole	15.8	572,059
GEORGIA				
Barrow Co	Atlanta, GA	Whole	18	46,144
Bartow Co	Atlanta, GA	Whole	18	76,019
Bibb Co	Macon, GA	Whole	15.2	153,887
Carroll Co	Atlanta, GA	Whole	18	87,268
Catoosa Co	Chattanooga, AL-TN-GA	Whole	16.1	53,282
Cherokee Co	Atlanta, GA	Whole	18	141,903

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County	Area Name	County NA Whole/Part	Design Value ($\mu\text{g}/\text{m}^3$)	Pop (2000)
Clayton Co	Atlanta, GA	Whole	18	236,517
Cobb Co	Atlanta, GA	Whole	18	607,751
Coweta Co	Atlanta, GA	Whole	18	89,215
De Kalb Co	Atlanta, GA	Whole	18	665,865
Douglas Co	Atlanta, GA	Whole	18	92,174
Fayette Co	Atlanta, GA	Whole	18	91,263
Floyd Co	Rome, GA	Whole	15.6	90,565
Forsyth Co	Atlanta, GA	Whole	18	98,407
Fulton Co	Atlanta, GA	Whole	18	816,006
Gwinnett Co	Atlanta, GA	Whole	18	588,448
Hall Co	Atlanta, GA	Whole	18	139,277
Heard Co	Atlanta, GA	Part	18	170
Henry Co	Atlanta, GA	Whole	18	119,341
Monroe Co	Macon, GA	Part	15.2	950
Newton Co	Atlanta, GA	Whole	18	62,001
Paulding Co	Atlanta, GA	Whole	18	81,678
Putnam Co	Atlanta, GA	Part	18	3,088
Rockdale Co	Atlanta, GA	Whole	18	70,111
Spalding Co	Atlanta, GA	Whole	18	58,417
Walker Co	Chattanooga, AL-TN-GA	Whole	16.1	61,053
Walton Co	Atlanta, GA	Whole	18	60,687
ILLINOIS				
Cook Co	Chicago-Gary-Lake County, IL-IN	Whole	17.7	5,376,741
DuPage Co	Chicago-Gary-Lake County, IL-IN	Whole	17.7	904,161
Grundy Co	Chicago-Gary-Lake County, IL-IN	Part	17.7	6,309
Kane Co	Chicago-Gary-Lake County, IL-IN	Whole	17.7	404,119
Kendall Co	Chicago-Gary-Lake County, IL-IN	Part	17.7	28,417
Lake Co	Chicago-Gary-Lake County, IL-IN	Whole	17.7	644,356
Madison Co	St. Louis, MO-IL	Whole	17.5	258,941
Mc Henry Co	Chicago-Gary-Lake County, IL-IN	Whole	17.7	260,077
Monroe Co	St. Louis, MO-IL	Whole	17.5	27,619
Randolph Co	St. Louis, MO-IL	Part	17.5	3,627
St Clair Co	St. Louis, MO-IL	Whole	17.5	256,082
Will Co	Chicago-Gary-Lake County, IL-IN	Whole	17.7	502,266
INDIANA				

Chapter 2 Appendices: Air Quality and Resulting Health and Welfare Effects

County	Area Name	County NA Whole/Part	Design Value ($\mu\text{g}/\text{m}^3$)	Pop (2000)
Clark Co	Louisville, KY-IN	Whole	16.9	96,472
Dearborn Co	Cincinnati-Hamilton, OH-KY- IN	Part	17.8	10,434
Dubois Co	Evansville, IN	Whole	16.2	39,674
Floyd Co	Louisville, KY-IN	Whole	16.9	70,823
Gibson Co	Evansville, IN	Part	16.2	3,698
Hamilton Co	Indianapolis, IN	Whole	16.7	182,740
Hendricks Co	Indianapolis, IN	Whole	16.7	104,093
Jefferson Co	Louisville, KY-IN	Part	16.9	16,770
Johnson Co	Indianapolis, IN	Whole	16.7	115,209
Lake Co	Chicago-Gary-Lake County, IL-IN	Whole	17.7	484,564
Marion Co	Indianapolis, IN	Whole	16.7	860,454
Morgan Co	Indianapolis, IN	Whole	16.7	66,689
Pike Co	Evansville, IN	Part	16.2	4,633
Porter Co	Chicago-Gary-Lake County, IL-IN	Whole	17.7	146,798
Spencer Co	Evansville, IN	Part	16.2	5,092
Vanderburgh Co	Evansville, IN	Whole	16.2	171,922
Warrick Co	Evansville, IN	Whole	16.2	52,383
KENTUCKY				
Boone Co	Cincinnati-Hamilton, OH-KY- IN	Whole	17.8	85,991
Boyd Co	Huntington-Ashland, WV-KY- OH	Whole	17.2	49,752
Bullitt Co	Louisville, KY-IN	Whole	16.9	61,236
Campbell Co	Cincinnati-Hamilton, OH-KY- IN	Whole	17.8	88,616
Jefferson Co	Louisville, KY-IN	Whole	16.9	693,604
Kenton Co	Cincinnati-Hamilton, OH-KY- IN	Whole	17.8	151,464
Lawrence Co	Huntington-Ashland, WV-KY- OH	Part	17.2	1,050
MARYLAND				
Anne Arundel Co	Baltimore, MD	Whole	16.6	489,656
Baltimore (City)	Baltimore, MD	Whole	16.6	651,154
Baltimore Co	Baltimore, MD	Whole	16.6	754,292
Carroll Co	Baltimore, MD	Whole	16.6	150,897
Charles Co	Washington, DC-MD-VA	Whole	15.8	120,546
Frederick Co	Washington, DC-MD-VA	Whole	15.8	195,277
Harford Co	Baltimore, MD	Whole	16.6	218,590
Howard Co	Baltimore, MD	Whole	16.6	247,842
Montgomery Co	Washington, DC-MD-VA	Whole	15.8	873,341

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County	Area Name	County NA Whole/Part	Design Value ($\mu\text{g}/\text{m}^3$)	Pop (2000)
Prince George's Co	Washington, DC-MD-VA	Whole	15.8	801,515
Washington Co	Martinsburg, WV-Hagerstown, MD	Whole	16.3	131,923
MICHIGAN				
Livingston Co	Detroit-Ann Arbor, MI	Whole	19.5	156,951
Macomb Co	Detroit-Ann Arbor, MI	Whole	19.5	788,149
Monroe Co	Detroit-Ann Arbor, MI	Whole	19.5	145,945
Oakland Co	Detroit-Ann Arbor, MI	Whole	19.5	1,194,156
St Clair Co	Detroit-Ann Arbor, MI	Whole	19.5	164,235
Washtenaw Co	Detroit-Ann Arbor, MI	Whole	19.5	322,895
Wayne Co	Detroit-Ann Arbor, MI	Whole	19.5	2,061,162
MISSOURI				
Franklin Co	St. Louis, MO-IL	Whole	17.5	93,807
Jefferson Co	St. Louis, MO-IL	Whole	17.5	198,099
St Charles Co	St. Louis, MO-IL	Whole	17.5	283,883
St Louis	St. Louis, MO-IL	Whole	17.5	348,189
St Louis Co	St. Louis, MO-IL	Whole	17.5	1,016,315
MONTANA				
Lincoln Co	Libby, MT	Part	16.2	2,626
NEW JERSEY				
Bergen Co	New York-N. New Jersey- Long Island, NY-NJ-CT	Whole	17.7	884,118
Burlington Co	Philadelphia-Wilmington, PA- NJ-DE	Whole	16.2	423,394
Camden Co	Philadelphia-Wilmington, PA- NJ-DE	Whole	16.2	508,932
Essex Co	New York-N. New Jersey- Long Island, NY-NJ-CT	Whole	17.7	793,633
Gloucester Co	Philadelphia-Wilmington, PA- NJ-DE	Whole	16.2	254,673
Hudson Co	New York-N. New Jersey- Long Island, NY-NJ-CT	Whole	17.7	608,975
Mercer Co	New York-N. New Jersey- Long Island, NY-NJ-CT	Whole	17.7	350,761
Middlesex Co	New York-N. New Jersey- Long Island, NY-NJ-CT	Whole	17.7	750,162
Monmouth Co	New York-N. New Jersey- Long Island, NY-NJ-CT	Whole	17.7	615,301
Morris Co	New York-N. New Jersey- Long Island, NY-NJ-CT	Whole	17.7	470,212
Passaic Co	New York-N. New Jersey- Long Island, NY-NJ-CT	Whole	17.7	489,049
Somerset Co	New York-N. New Jersey- Long Island, NY-NJ-CT	Whole	17.7	297,490
Union Co	New York-N. New Jersey-	Whole	17.7	522,541

Chapter 2 Appendices: Air Quality and Resulting Health and Welfare Effects

County	Area Name	County NA Whole/Part	Design Value ($\mu\text{g}/\text{m}^3$)	Pop (2000)
	Long Island, NY-NJ-CT			
New York				
Bronx Co	New York-N. New Jersey-Long Island, NY-NJ-CT	Whole	17.7	1,332,650
Kings Co	New York-N. New Jersey-Long Island, NY-NJ-CT	Whole	17.7	2,465,326
Nassau Co	New York-N. New Jersey-Long Island, NY-NJ-CT	Whole	17.7	1,334,544
New York Co	New York-N. New Jersey-Long Island, NY-NJ-CT	Whole	17.7	1,537,195
Orange Co	New York-N. New Jersey-Long Island, NY-NJ-CT	Whole	17.7	341,367
Queens Co	New York-N. New Jersey-Long Island, NY-NJ-CT	Whole	17.7	2,229,379
Richmond Co	New York-N. New Jersey-Long Island, NY-NJ-CT	Whole	17.7	443,728
Rockland Co	New York-N. New Jersey-Long Island, NY-NJ-CT	Whole	17.7	286,753
Suffolk Co	New York-N. New Jersey-Long Island, NY-NJ-CT	Whole	17.7	1,419,369
Westchester Co	New York-N. New Jersey-Long Island, NY-NJ-CT	Whole	17.7	923,459
NORTH CAROLINA				
Catawba Co	Hickory, NC	Whole	15.5	141,685
Davidson Co	Greensboro-Winston Salem-High Point, NC	Whole	15.8	147,246
Guilford Co	Greensboro-Winston Salem-High Point, NC	Whole	15.8	421,048
OHIO				
Adams Co	Huntington-Ashland, WV-KY-OH	Part	17.2	2,374
Ashtabula Co	Cleveland-Akron-Lorain, OH	Part	18.3	23,239
Belmont Co	Wheeling, WV-OH	Whole	15.7	70,226
Butler Co	Cincinnati-Hamilton, OH-KY-IN	Whole	17.8	332,807
Clark Co	Dayton-Springfield, OH	Whole	15.2	144,742
Clermont Co	Cincinnati-Hamilton, OH-KY-IN	Whole	17.8	177,977
Coshocton Co	Columbus, OH	Part	16.7	1,286
Cuyahoga Co	Cleveland-Akron-Lorain, OH	Whole	18.3	1,393,978
Delaware Co	Columbus, OH	Whole	16.7	109,989
Fairfield Co	Columbus, OH	Whole	16.7	122,759
Franklin Co	Columbus, OH	Whole	16.7	1,068,978
Gallia Co	Huntington-Ashland, WV-KY-OH	Part	17.2	3,625
Greene Co	Dayton-Springfield, OH	Whole	15.2	147,886
Hamilton Co	Cincinnati-Hamilton, OH-KY-	Whole	17.8	845,303

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County	Area Name	County NA Whole/Part	Design Value ($\mu\text{g}/\text{m}^3$)	Pop (2000)
	IN			
Jefferson Co	Steubenville-Weirton, OH-WV	Whole	17.8	73,894
Lake Co	Cleveland-Akron-Lorain, OH	Whole	18.3	227,511
Lawrence Co	Huntington-Ashland, WV-KY- OH	Whole	17.2	62,319
Licking Co	Columbus, OH	Whole	16.7	145,491
Lorain Co	Cleveland-Akron-Lorain, OH	Whole	18.3	284,664
Medina Co	Cleveland-Akron-Lorain, OH	Whole	18.3	151,095
Montgomery Co	Dayton-Springfield, OH	Whole	15.2	559,062
Portage Co	Cleveland-Akron-Lorain, OH	Whole	18.3	152,061
Scioto Co	Huntington-Ashland, WV-KY- OH	Whole	17.2	79,195
Stark Co	Canton-Massillon, OH	Whole	17.3	378,098
Summit Co	Cleveland-Akron-Lorain, OH	Whole	18.3	542,899
Warren Co	Cincinnati-Hamilton, OH-KY- IN	Whole	17.8	158,383
Washington Co	Parkersburg-Marietta, WV-OH	Whole	16	63,251
PENNSYLVANIA				
Allegheny Co	Liberty-Clairton, PA	Part	21.2	21,600
Allegheny Co	Pittsburgh-Beaver Valley, PA	Part	16.9	1,260,066
Armstrong Co	Pittsburgh-Beaver Valley, PA	Part	16.9	3,691
Beaver Co	Pittsburgh-Beaver Valley, PA	Whole	16.9	181,412
Berks Co	Reading, PA	Whole	16.4	373,638
Bucks Co	Philadelphia-Wilmington, PA- NJ-DE	Whole	16.2	597,635
Butler Co	Pittsburgh-Beaver Valley, PA	Whole	16.9	174,083
Cambria Co	Johnstown, PA	Whole	15.8	152,598
Chester Co	Philadelphia-Wilmington, PA- NJ-DE	Whole	16.2	433,501
Cumberland Co	Harrisburg-Lebanon-Carlisle, PA	Whole	15.7	213,674
Dauphin Co	Harrisburg-Lebanon-Carlisle, PA	Whole	15.7	251,798
Delaware Co	Philadelphia-Wilmington, PA- NJ-DE	Whole	16.2	550,864
Greene Co	Pittsburgh-Beaver Valley, PA	Part	16.9	1,714
Indiana Co	Johnstown, PA	Part	15.8	11,833
Lancaster Co	Lancaster, PA	Whole	17	470,658
Lawrence Co	Pittsburgh-Beaver Valley, PA	Part	16.9	1,198
Lebanon Co	Harrisburg-Lebanon-Carlisle, PA	Whole	15.7	120,327
Montgomery Co	Philadelphia-Wilmington, PA- NJ-DE	Whole	16.2	750,097
Philadelphia Co	Philadelphia-Wilmington, PA-	Whole	16.2	1,517,550

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County	Area Name	County NA Whole/Part	Design Value ($\mu\text{g}/\text{m}^3$)	Pop (2000)
	NJ-DE			
Washington Co	Pittsburgh-Beaver Valley, PA	Whole	16.9	202,897
Westmoreland Co	Pittsburgh-Beaver Valley, PA	Whole	16.9	369,993
York Co	York, PA	Whole	17	381,751
TENNESSEE				
Anderson Co	Knoxville, TN	Whole	16.4	71,330
Blount Co	Knoxville, TN	Whole	16.4	105,823
Hamilton Co	Chattanooga, AL-TN-GA	Whole	16.1	307,896
Knox Co	Knoxville, TN	Whole	16.4	382,032
Loudon Co	Knoxville, TN	Whole	16.4	39,086
Roane Co	Knoxville, TN	Part	16.4	737
VIRGINIA				
Alexandria	Washington, DC-MD-VA	Whole	15.8	128,283
Arlington Co	Washington, DC-MD-VA	Whole	15.8	189,453
Fairfax	Washington, DC-MD-VA	Whole	15.8	21,498
Fairfax Co	Washington, DC-MD-VA	Whole	15.8	969,749
Falls Church	Washington, DC-MD-VA	Whole	15.8	10,377
Loudoun Co	Washington, DC-MD-VA	Whole	15.8	169,599
Manassas	Washington, DC-MD-VA	Whole	15.8	35,135
Manassas Park	Washington, DC-MD-VA	Whole	15.8	10,290
Prince William Co	Washington, DC-MD-VA	Whole	15.8	280,813
WEST VIRGINIA				
Berkeley Co	Martinsburg, WV-Hagerstown, MD	Whole	16.3	75,905
Brooke Co	Steubenville-Weirton, OH-WV	Whole	17.8	25,447
Cabell Co	Huntington-Ashland, WV-KY- OH	Whole	17.2	96,784
Hancock Co	Steubenville-Weirton, OH-WV	Whole	17.8	32,667
Kanawha Co	Charleston, WV	Whole	17.1	200,073
Marshall Co	Wheeling, WV-OH	Whole	15.7	35,519
Mason Co	Huntington-Ashland, WV-KY- OH	Part	17.2	2,774
Ohio Co	Wheeling, WV-OH	Whole	15.7	47,427
Pleasants Co	Parkersburg-Marietta, WV-OH	Part	16	1,675
Putnam Co	Charleston, WV	Whole	17.1	51,589
Wayne Co	Huntington-Ashland, WV-KY- OH	Whole	17.2	42,903
Wood Co	Parkersburg-Marietta, WV-OH	Whole	16	87,986
TOTAL	208 Counties			88,394,361

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Appendix 2B: Current 8-Hour Ozone Nonattainment Areas

Table 2B 8-Hour Ozone Nonattainment Areas and Populations (Data is current through October 2006 and Population Numbers are from 2000 Census Data)

8-hour Ozone Nonattainment	State	Classification ^{a,b}	County Name	Whole /Part	2000 Cty Pop
	NY	Subpart 1	Albany Co	W	294,565
Albany-Schenectady-Troy Area	NY	Subpart 1	Greene Co	W	48,195
Albany-Schenectady-Troy Area	NY	Subpart 1	Montgomery Co	W	49,708
Albany-Schenectady-Troy Area	NY	Subpart 1	Rensselaer Co	W	152,538
Albany-Schenectady-Troy Area	NY	Subpart 1	Saratoga Co	W	200,635
Albany-Schenectady-Troy Area	NY	Subpart 1	Schenectady Co	W	146,555
Albany-Schenectady-Troy Area	NY	Subpart 1	Schoharie Co	W	31,582
Allegan County Area	MI	Subpart 1	Allegan Co	W	105,665
Allentown-Bethlehem-Easton Area	PA	Subpart 1	Carbon Co	W	58,802
Allentown-Bethlehem-Easton Area	PA	Subpart 1	Lehigh Co	W	312,090
Allentown-Bethlehem-Easton Area	PA	Subpart 1	Northampton Co	W	267,066
Altoona Area	PA	Subpart 1	Blair Co	W	129,144
Amador and Calaveras Counties (Central Mountain Counties) Area	CA	Subpart 1	Amador Co	W	35,100
Amador and Calaveras Counties (Central Mountain Counties) Area	CA	Subpart 1	Calaveras Co	W	40,554
Atlanta Area	GA		Barrow Co	W	46,144
Atlanta Area	GA	Subpart 2/Marginal	Bartow Co	W	76,019
Atlanta Area	GA	Subpart 2/Marginal	Carroll Co	W	87,268
Atlanta Area	GA	Subpart 2/Marginal	Cherokee Co	W	141,903
Atlanta Area	GA	Subpart 2/Marginal	Clayton Co	W	236,517
Atlanta Area	GA	Subpart 2/Marginal	Cobb Co	W	607,751
Atlanta Area	GA	Subpart 2/Marginal	Coweta Co	W	89,215
Atlanta Area	GA	Subpart 2/Marginal	De Kalb Co	W	665,865
Atlanta Area	GA	Subpart 2/Marginal	Douglas Co	W	92,174
Atlanta Area	GA	Subpart 2/Marginal	Fayette Co	W	91,263
Atlanta Area	GA	Subpart 2/Marginal	Forsyth Co	W	98,407
Atlanta Area	GA	Subpart 2/Marginal	Fulton Co	W	816,006
Atlanta Area	GA	Subpart 2/Marginal	Gwinnett Co	W	588,448
Atlanta Area	GA	Subpart 2/Marginal	Hall Co	W	139,277
Atlanta Area	GA	Subpart 2/Marginal	Henry Co	W	119,341
Atlanta Area	GA	Subpart 2/Marginal	Newton Co	W	62,001
Atlanta Area	GA	Subpart 2/Marginal	Paulding Co	W	81,678

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8-hour Ozone Nonattainment	State	Classification ^{a,b}	County Name	Whole /Part	2000 Cty Pop
Atlanta Area	GA	Subpart 2/Marginal	Rockdale Co	W	70,111
Atlanta Area	GA	Subpart 2/Marginal	Spalding Co	W	58,417
Atlanta Area	GA	Subpart 2/Marginal	Walton Co	W	60,687
Baltimore Area	MD	Subpart 2/Moderate	Anne Arundel Co	W	489,656
Baltimore Area	MD	Subpart 2/Moderate	Baltimore (City)	W	651,154
Baltimore Area	MD	Subpart 2/Moderate	Baltimore Co	W	754,292
Baltimore Area	MD	Subpart 2/Moderate	Carroll Co	W	150,897
Baltimore Area	MD	Subpart 2/Moderate	Harford Co	W	218,590
Baltimore Area	MD	Subpart 2/Moderate	Howard Co	W	247,842
Baton Rouge Area	LA	Subpart 2/Marginal	Ascension Par	W	76,627
Baton Rouge Area	LA	Subpart 2/Marginal	East Baton Rouge Par	W	412,852
Baton Rouge Area	LA	Subpart 2/Marginal	Iberville Par	W	33,320
Baton Rouge Area	LA	Subpart 2/Marginal	Livingston Par	W	91,814
Baton Rouge Area	LA	Subpart 2/Marginal	West Baton Rouge Par	W	21,601
Beaumont-Port Arthur Area	TX	Subpart 2/Marginal	Hardin Co	W	48,073
Beaumont-Port Arthur Area	TX	Subpart 2/Marginal	Jefferson Co	W	252,051
Beaumont-Port Arthur Area	TX	Subpart 2/Marginal	Orange Co	W	84,966
Benton Harbor Area	MI	Subpart 1	Berrien Co	W	162,453
Benzie County Area	MI	Subpart 1	Benzie Co	W	15,998
Berkeley and Jefferson Counties Area	WV	Subpart 1 - EAC	Berkeley Co	W	75,905
Berkeley and Jefferson Counties Area	WV	Subpart 1 - EAC	Jefferson Co	W	42,190
Boston-Lawrence-Worcester (E. Mass) Area	MA	Subpart 2/Moderate	Barnstable Co	W	222,230
Boston-Lawrence-Worcester (E. Mass) Area	MA	Subpart 2/Moderate	Bristol Co	W	534,678
Boston-Lawrence-Worcester (E. Mass) Area	MA	Subpart 2/Moderate	Dukes Co	W	14,987
Boston-Lawrence-Worcester (E. Mass) Area	MA	Subpart 2/Moderate	Essex Co	W	723,419
Boston-Lawrence-Worcester (E. Mass) Area	MA	Subpart 2/Moderate	Middlesex Co	W	1,465,396
Boston-Lawrence-Worcester (E. Mass) Area	MA	Subpart 2/Moderate	Nantucket Co	W	9,520
Boston-Lawrence-Worcester (E. Mass) Area	MA	Subpart 2/Moderate	Norfolk Co	W	650,308
Boston-Lawrence-Worcester (E. Mass) Area	MA	Subpart 2/Moderate	Plymouth Co	W	472,822
Boston-Lawrence-Worcester (E. Mass) Area	MA	Subpart 2/Moderate	Suffolk Co	W	689,807
Boston-Lawrence-Worcester (E. Mass) Area	MA	Subpart 2/Moderate	Worcester Co	W	750,963
Boston-Manchester-Portsmouth (SE) Area	NH	Subpart 2/Moderate	Hillsborough Co	P	336,518
Boston-Manchester-Portsmouth (SE) Area	NH	Subpart 2/Moderate	Merrimack Co	P	11,721
Boston-Manchester-Portsmouth (SE) Area	NH	Subpart 2/Moderate	Rockingham Co	P	266,340

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8-hour Ozone Nonattainment	State	Classification ^{a,b}	County Name	Whole /Part	2000 Cty Pop
Boston-Manchester-Portsmouth (SE) Area	NH	Subpart 2/Moderate	Strafford Co	P	82,134
Buffalo-Niagara Falls Area	NY	Subpart 1	Erie Co	W	950,265
Buffalo-Niagara Falls Area	NY	Subpart 1	Niagara Co	W	219,846
Canton-Massillon Area	OH	Subpart 1	Stark Co	W	378,098
Cass County Area	MI	Subpart 2/Marginal	Cass Co	W	51,104
Charlotte-Gastonia-Rock Hill Area	NC	Subpart 2/Moderate	Cabarrus Co	W	131,063
Charlotte-Gastonia-Rock Hill Area	NC	Subpart 2/Moderate	Gaston Co	W	190,365
Charlotte-Gastonia-Rock Hill Area	NC	Subpart 2/Moderate	Iredell Co	P	39,885
Charlotte-Gastonia-Rock Hill Area	NC	Subpart 2/Moderate	Lincoln Co	W	63,780
Charlotte-Gastonia-Rock Hill Area	NC	Subpart 2/Moderate	Mecklenburg Co	W	695,454
Charlotte-Gastonia-Rock Hill Area	NC	Subpart 2/Moderate	Rowan Co	W	130,340
Charlotte-Gastonia-Rock Hill Area	NC	Subpart 2/Moderate	Union Co	W	123,677
Charlotte-Gastonia-Rock Hill Area	SC	Subpart 2/Moderate	York Co	P	102,000
Chattanooga Area	GA	Subpart 1 - EAC	Catoosa Co	W	53,282
Chattanooga Area	TN	Subpart 1 - EAC	Hamilton Co	W	307,896
Chattanooga Area	TN	Subpart 1 - EAC	Meigs Co	W	11,086
Chicago-Gary-Lake County Area	IL	Subpart 2/Moderate	Cook Co	W	5,376,741
Chicago-Gary-Lake County Area	IL	Subpart 2/Moderate	Du Page Co	W	904,161
Chicago-Gary-Lake County Area	IL	Subpart 2/Moderate	Grundy Co	P	6,309
Chicago-Gary-Lake County Area	IL	Subpart 2/Moderate	Kane Co	W	404,119
Chicago-Gary-Lake County Area	IL	Subpart 2/Moderate	Kendall Co	P	28,417
Chicago-Gary-Lake County Area	IL	Subpart 2/Moderate	Lake Co	W	644,356
Chicago-Gary-Lake County Area	IL	Subpart 2/Moderate	Mc Henry Co	W	260,077
Chicago-Gary-Lake County Area	IL	Subpart 2/Moderate	Will Co	W	502,266
Chicago-Gary-Lake County Area	IN	Subpart 2/Moderate	Lake Co	W	484,564
Chicago-Gary-Lake County Area	IN	Subpart 2/Moderate	Porter Co	W	146,798
Chico Area	CA	Subpart 1	Butte Co	W	203,171
Cincinnati-Hamilton Area	IN	Subpart 1	Dearborn Co	P	10,434
Cincinnati-Hamilton Area	KY	Subpart 1	Boone Co	W	85,991
Cincinnati-Hamilton Area	KY	Subpart 1	Campbell Co	W	88,616
Cincinnati-Hamilton Area	KY	Subpart 1	Kenton Co	W	151,464

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8-hour Ozone Nonattainment	State	Classification ^{a,b}	County Name	Whole /Part	2000 Cty Pop
Cincinnati-Hamilton Area	OH	Subpart 1	Butler Co	W	332,807
Cincinnati-Hamilton Area	OH	Subpart 1	Clermont Co	W	177,977
Cincinnati-Hamilton Area	OH	Subpart 1	Clinton Co	W	40,543
Cincinnati-Hamilton Area	OH	Subpart 1	Hamilton Co	W	845,303
Cincinnati-Hamilton Area	OH	Subpart 1	Warren Co	W	158,383
Clearfield and Indiana Counties Area	PA	Subpart 1	Clearfield Co	W	83,382
Clearfield and Indiana Counties Area	PA	Subpart 1	Indiana Co	W	89,605
Cleveland-Akron-Lorain Area	OH	Subpart 2/Moderate	Ashtabula Co	W	102,728
Cleveland-Akron-Lorain Area	OH	Subpart 2/Moderate	Cuyahoga Co	W	1,393,978
Cleveland-Akron-Lorain Area	OH	Subpart 2/Moderate	Geauga Co	W	90,895
Cleveland-Akron-Lorain Area	OH	Subpart 2/Moderate	Lake Co	W	227,511
Cleveland-Akron-Lorain Area	OH	Subpart 2/Moderate	Lorain Co	W	284,664
Cleveland-Akron-Lorain Area	OH	Subpart 2/Moderate	Medina Co	W	151,095
Cleveland-Akron-Lorain Area	OH	Subpart 2/Moderate	Portage Co	W	152,061
Cleveland-Akron-Lorain Area	OH	Subpart 2/Moderate	Summit Co	W	542,899
Columbia Area	SC	Subpart 1 - EAC	Lexington Co	P	181,265
Columbia Area	SC	Subpart 1 - EAC	Richland Co	P	313,253
Columbus Area	OH	Subpart 1	Delaware Co	W	109,989
Columbus Area	OH	Subpart 1	Fairfield Co	W	122,759
Columbus Area	OH	Subpart 1	Franklin Co	W	1,068,978
Columbus Area	OH	Subpart 1	Knox Co	W	54,500
Columbus Area	OH	Subpart 1	Licking Co	W	145,491
Columbus Area	OH	Subpart 1	Madison Co	W	40,213
Dallas-Fort Worth Area	TX	Subpart 2/Moderate	Collin Co	W	491,675
Dallas-Fort Worth Area	TX	Subpart 2/Moderate	Dallas Co	W	2,218,899
Dallas-Fort Worth Area	TX	Subpart 2/Moderate	Denton Co	W	432,976
Dallas-Fort Worth Area	TX	Subpart 2/Moderate	Ellis Co	W	111,360
Dallas-Fort Worth Area	TX	Subpart 2/Moderate	Johnson Co	W	126,811
Dallas-Fort Worth Area	TX	Subpart 2/Moderate	Kaufman Co	W	71,313
Dallas-Fort Worth Area	TX	Subpart 2/Moderate	Parker Co	W	88,495
Dallas-Fort Worth Area	TX	Subpart 2/Moderate	Rockwall Co	W	43,080
Dallas-Fort Worth Area	TX	Subpart 2/Moderate	Tarrant Co	W	1,446,219
Dayton-Springfield Area	OH	Subpart 1	Clark Co	W	144,742
Dayton-Springfield Area	OH	Subpart 1	Greene Co	W	147,886
Dayton-Springfield Area	OH	Subpart 1	Miami Co	W	98,868
Dayton-Springfield Area	OH	Subpart 1	Montgomery Co	W	559,062
Denver-Boulder-Greeley-Ft. Collins-Love. Area	CO	Subpart 1 - EAC	Adams Co	W	348,618
Denver-Boulder-Greeley-Ft. Collins-Love. Area	CO	Subpart 1 - EAC	Arapahoe Co	W	487,967
Denver-Boulder-Greeley-Ft. Collins-Love. Area	CO	Subpart 1 - EAC	Boulder Co	W	269,814
Denver-Boulder-Greeley-Ft. Collins-Love. Area	CO	Subpart 1 - EAC	Broomfield Co	W	38,272
Denver-Boulder-Greeley-Ft. Collins-Love. Area	CO	Subpart 1 - EAC	Denver Co	W	554,636

Draft Regulatory Impact Analysis

8-hour Ozone Nonattainment	State	Classification ^{a,b}	County Name	Whole /Part	2000 Cty Pop
Collins-Love. Area					
Denver-Boulder-Greeley-Ft. Collins-Love. Area	CO	Subpart 1 - EAC	Douglas Co	W	175,766
Denver-Boulder-Greeley-Ft. Collins-Love. Area	CO	Subpart 1 - EAC	Jefferson Co	W	525,507
Denver-Boulder-Greeley-Ft. Collins-Love. Area	CO	Subpart 1 - EAC	Larimer Co	P	239,000
Denver-Boulder-Greeley-Ft. Collins-Love. Area	CO	Subpart 1 - EAC	Weld Co	P	172,000
Detroit-Ann Arbor Area	MI	Subpart 2/Marginal	Lenawee Co	W	98,890
Detroit-Ann Arbor Area	MI	Subpart 2/Marginal	Livingston Co	W	156,951
Detroit-Ann Arbor Area	MI	Subpart 2/Marginal	Macomb Co	W	788,149
Detroit-Ann Arbor Area	MI	Subpart 2/Marginal	Monroe Co	W	145,945
Detroit-Ann Arbor Area	MI	Subpart 2/Marginal	Oakland Co	W	1,194,156
Detroit-Ann Arbor Area	MI	Subpart 2/Marginal	St Clair Co	W	164,235
Detroit-Ann Arbor Area	MI	Subpart 2/Marginal	Washtenaw Co	W	322,895
Detroit-Ann Arbor Area	MI	Subpart 2/Marginal	Wayne Co	W	2,061,162
Door County Area	WI	Subpart 1	Door Co	W	27,961
Erie Area	PA	Subpart 1	Erie Co	W	280,843
Essex County (Whiteface Mtn.) Area	NY	Subpart 1	Essex Co	P	1,000
Fayetteville Area	NC	Subpart 1 - EAC	Cumberland Co	W	302,963
Flint Area	MI	Subpart 1	Genesee Co	W	436,141
Flint Area	MI	Subpart 1	Lapeer Co	W	87,904
Fort Wayne Area	IN	Subpart 1	Allen Co	W	331,849
Franklin County Area	PA	Subpart 1	Franklin Co	W	129,313
Frederick County Area	VA	Subpart 1 - EAC	Frederick Co	W	59,209
Frederick County Area	VA	Subpart 1 - EAC	Winchester	W	23,585
Grand Rapids Area	MI	Subpart 1	Kent Co	W	574,335
Grand Rapids Area	MI	Subpart 1	Ottawa Co	W	238,314
Greater Connecticut Area	CT	Subpart 2/Moderate	Hartford Co	W	857,183
Greater Connecticut Area	CT	Subpart 2/Moderate	Litchfield Co	W	182,193
Greater Connecticut Area	CT	Subpart 2/Moderate	New London Co	W	259,088
Greater Connecticut Area	CT	Subpart 2/Moderate	Tolland Co	W	136,364
Greater Connecticut Area	CT	Subpart 2/Moderate	Windham Co	W	109,091
Greene County Area	PA	Subpart 1	Greene Co	W	40,672
Greensboro-Winston-Salem-High Point Area	NC	Subpart 2/Marginal - EAC	Alamance Co	W	130,800
Greensboro-Winston-Salem-High Point Area	NC	Subpart 2/Marginal - EAC	Caswell Co	W	23,501
Greensboro-Winston-Salem-High Point Area	NC	Subpart 2/Marginal - EAC	Davidson Co	W	147,246
Greensboro-Winston-Salem-High Point Area	NC	Subpart 2/Marginal - EAC	Davie Co	W	34,835
Greensboro-Winston-Salem-High Point Area	NC	Subpart 2/Marginal - EAC	Forsyth Co	W	306,067
Greensboro-Winston-Salem-High Point Area	NC	Subpart 2/Marginal - EAC	Guilford Co	W	421,048

Chapter 2 Appendices: Air Quality and Resulting Health and Welfare Effects

8-hour Ozone Nonattainment	State	Classification ^{a,b}	County Name	Whole /Part	2000 Cty Pop
Greensboro-Winston-Salem-High Point Area	NC	Subpart 2/Marginal - EAC	Randolph Co	W	130,454
Greensboro-Winston-Salem-High Point Area	NC	Subpart 2/Marginal - EAC	Rockingham Co	W	91,928
Greenville-Spartanburg-Anderson Area	SC	Subpart 1 - EAC	Anderson Co	W	165,740
Greenville-Spartanburg-Anderson Area	SC	Subpart 1 - EAC	Greenville Co	W	379,616
Greenville-Spartanburg-Anderson Area	SC	Subpart 1 - EAC	Spartanburg Co	W	253,791
Hancock, Knox, Lincoln and Waldo Counties (Central Maine Coast) Area	ME	Subpart 1	Hancock Co	P	29,805
Hancock, Knox, Lincoln and Waldo Counties (Central Maine Coast) Area	ME	Subpart 1	Knox Co	P	33,563
Hancock, Knox, Lincoln and Waldo Counties (Central Maine Coast) Area	ME	Subpart 1	Lincoln Co	P	28,504
Hancock, Knox, Lincoln and Waldo Counties (Central Maine Coast) Area	ME	Subpart 1	Waldo Co	P	604
Harrisburg-Lebanon-Carlisle Area	PA	Subpart 1	Cumberland Co	W	213,674
Harrisburg-Lebanon-Carlisle Area	PA	Subpart 1	Dauphin Co	W	251,798
Harrisburg-Lebanon-Carlisle Area	PA	Subpart 1	Lebanon Co	W	120,327
Harrisburg-Lebanon-Carlisle Area	PA	Subpart 1	Perry Co	W	43,602
Haywood and Swain Counties (Great Smoky NP) Area	NC	Subpart 1	Haywood Co	P	28
Haywood and Swain Counties (Great Smoky NP) Area	NC	Subpart 1	Swain Co	P	260
Hickory-Morganton-Lenoir Area	NC	Subpart 1 - EAC	Alexander Co	W	33,603
Hickory-Morganton-Lenoir Area	NC	Subpart 1 - EAC	Burke Co	P	69,970
Hickory-Morganton-Lenoir Area	NC	Subpart 1 - EAC	Caldwell Co	P	64,254
Hickory-Morganton-Lenoir Area	NC	Subpart 1 - EAC	Catawba Co	W	141,685
Houston-Galveston-Brazoria Area	TX	Subpart 2/Moderate	Brazoria Co	W	241,767
Houston-Galveston-Brazoria Area	TX	Subpart 2/Moderate	Chambers Co	W	26,031
Houston-Galveston-Brazoria Area	TX	Subpart 2/Moderate	Fort Bend Co	W	354,452
Houston-Galveston-Brazoria Area	TX	Subpart 2/Moderate	Galveston Co	W	250,158
Houston-Galveston-Brazoria Area	TX	Subpart 2/Moderate	Harris Co	W	3,400,578

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8-hour Ozone Nonattainment	State	Classification ^{a,b}	County Name	Whole /Part	2000 Cty Pop
Houston-Galveston-Brazoria Area	TX	Subpart 2/Moderate	Liberty Co	W	70,154
Houston-Galveston-Brazoria Area	TX	Subpart 2/Moderate	Montgomery Co	W	293,768
Houston-Galveston-Brazoria Area	TX	Subpart 2/Moderate	Waller Co	W	32,663
Huntington-Ashland Area	KY	Subpart 1	Boyd Co	W	49,752
Huron County Area	MI	Subpart 1	Huron Co	W	36,079
Imperial County Area	CA	Subpart 2/Marginal	Imperial Co	W	142,361
Indianapolis Area	IN	Subpart 1	Boone Co	W	46,107
Indianapolis Area	IN	Subpart 1	Hamilton Co	W	182,740
Indianapolis Area	IN	Subpart 1	Hancock Co	W	55,391
Indianapolis Area	IN	Subpart 1	Hendricks Co	W	104,093
Indianapolis Area	IN	Subpart 1	Johnson Co	W	115,209
Indianapolis Area	IN	Subpart 1	Madison Co	W	133,358
Indianapolis Area	IN	Subpart 1	Marion Co	W	860,454
Indianapolis Area	IN	Subpart 1	Morgan Co	W	66,689
Indianapolis Area	IN	Subpart 1	Shelby Co	W	43,445
Jamestown Area	NY	Subpart 1	Chautauqua Co	W	139,750
Jefferson County Area	NY	Subpart 2/Moderate	Jefferson Co	W	111,738
Johnson City-Kingsport-Bristol Area	TN	Subpart 1 - EAC	Hawkins Co	W	53,563
Johnson City-Kingsport-Bristol Area	TN	Subpart 1 - EAC	Sullivan Co	W	153,048
Johnstown Area	PA	Subpart 1	Cambria Co	W	152,598
Kalamazoo-Battle Creek Area	MI	Subpart 1	Calhoun Co	W	137,985
Kalamazoo-Battle Creek Area	MI	Subpart 1	Kalamazoo Co	W	238,603
Kalamazoo-Battle Creek Area	MI	Subpart 1	Van Buren Co	W	76,263
Kent and Queen Anne's Counties Area	MD	Subpart 2/Marginal	Kent Co	W	19,197
Kent and Queen Anne's Counties Area	MD	Subpart 2/Marginal	Queen Annes Co	W	40,563
Kern County (Eastern Kern) Area	CA	Subpart 1	Kern Co	P	99,251
Kewaunee County Area	WI	Subpart 1	Kewaunee Co	W	20,187
Knoxville Area	TN	Subpart 1	Anderson Co	W	71,330
Knoxville Area	TN	Subpart 1	Blount Co	W	105,823
Knoxville Area	TN	Subpart 1	Cocke Co	P	20
Knoxville Area	TN	Subpart 1	Jefferson Co	W	44,294
Knoxville Area	TN	Subpart 1	Knox Co	W	382,032
Knoxville Area	TN	Subpart 1	Loudon Co	W	39,086
Knoxville Area	TN	Subpart 1	Sevier Co	W	71,170
La Porte County Area	IN	Subpart 2/Marginal	La Porte Co	W	110,106
Lancaster Area	PA	Subpart 2/Marginal	Lancaster Co	W	470,658
Lansing-East Lansing Area	MI	Subpart 1	Clinton Co	W	64,753
Lansing-East Lansing Area	MI	Subpart 1	Eaton Co	W	103,655
Lansing-East Lansing Area	MI	Subpart 1	Ingham Co	W	279,320

Chapter 2 Appendices: Air Quality and Resulting Health and Welfare Effects

8-hour Ozone Nonattainment	State	Classification ^{a,b}	County Name	Whole /Part	2000 Cty Pop
Las Vegas Area	NV	Subpart 1	Clark Co	P	1,348,864
Lima Area	OH	Subpart 1	Allen Co	W	108,473
Los Angeles and San Bernardino Counties (W Mojave Desert) Area	CA	Subpart 2/Moderate	Los Angeles Co	P	297,058
Los Angeles and San Bernardino Counties (W Mojave Desert) Area	CA	Subpart 2/Moderate	San Bernardino Co	P	359,350
Los Angeles-South Coast Air Basin Area	CA	Subpart 2/Severe 17	Los Angeles Co	P	9,222,280
Los Angeles-South Coast Air Basin Area	CA	Subpart 2/Severe 17	Orange Co	W	2,846,289
Los Angeles-South Coast Air Basin Area	CA	Subpart 2/Severe 17	Riverside Co	P	1,194,859
Los Angeles-South Coast Air Basin Area	CA	Subpart 2/Severe 17	San Bernardino Co	P	1,330,159
Louisville Area	IN	Subpart 1	Clark Co	W	96,472
Louisville Area	IN	Subpart 1	Floyd Co	W	70,823
Louisville Area	KY	Subpart 1	Bullitt Co	W	61,236
Louisville Area	KY	Subpart 1	Jefferson Co	W	693,604
Louisville Area	KY	Subpart 1	Oldham Co	W	46,178
Macon Area	GA	Subpart 1	Bibb Co	W	153,887
Macon Area	GA	Subpart 1	Monroe Co	P	50
Manitowoc County Area	WI	Subpart 1	Manitowoc Co	W	82,887
Mariposa and Tuolumne Counties (Southern Mountain Counties) Area	CA	Subpart 1	Mariposa Co	W	17,130
Mariposa and Tuolumne Counties (Southern Mountain Counties) Area	CA	Subpart 1	Tuolumne Co	W	54,501
Mason County Area	MI	Subpart 1	Mason Co	W	28,274
Memphis Area	AR	Subpart 2/Marginal	Crittenden Co	W	50,866
Memphis Area	TN	Subpart 2/Marginal	Shelby Co	W	897,472
Milwaukee-Racine Area	WI	Subpart 2/Moderate	Kenosha Co	W	149,577
Milwaukee-Racine Area	WI	Subpart 2/Moderate	Milwaukee Co	W	940,164
Milwaukee-Racine Area	WI	Subpart 2/Moderate	Ozaukee Co	W	82,317
Milwaukee-Racine Area	WI	Subpart 2/Moderate	Racine Co	W	188,831
Milwaukee-Racine Area	WI	Subpart 2/Moderate	Washington Co	W	117,493
Milwaukee-Racine Area	WI	Subpart 2/Moderate	Waukesha Co	W	360,767
Murray County (Chattahoochee Nat Forest) Area	GA	Subpart 1	Murray Co	P	1,000
Muskegon Area	MI	Subpart 2/Marginal	Muskegon Co	W	170,200
Nashville Area	TN	Subpart 1 - EAC	Davidson Co	W	569,891
Nashville Area	TN	Subpart 1 - EAC	Rutherford Co	W	182,023
Nashville Area	TN	Subpart 1 - EAC	Sumner Co	W	130,449
Nashville Area	TN	Subpart 1 - EAC	Williamson Co	W	126,638
Nashville Area	TN	Subpart 1 - EAC	Wilson Co	W	88,809

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8-hour Ozone Nonattainment	State	Classification ^{a,b}	County Name	Whole /Part	2000 Cty Pop
Nevada County (Western part) Area	CA	Subpart 1	Nevada Co	P	77,735
New York-N. New Jersey-Long Island Area	CT	Subpart 2/Moderate	Fairfield Co	W	882,567
New York-N. New Jersey-Long Island Area	CT	Subpart 2/Moderate	Middlesex Co	W	155,071
New York-N. New Jersey-Long Island Area	CT	Subpart 2/Moderate	New Haven Co	W	824,008
New York-N. New Jersey-Long Island Area	NJ	Subpart 2/Moderate	Bergen Co	W	884,118
New York-N. New Jersey-Long Island Area	NJ	Subpart 2/Moderate	Essex Co	W	793,633
New York-N. New Jersey-Long Island Area	NJ	Subpart 2/Moderate	Hudson Co	W	608,975
New York-N. New Jersey-Long Island Area	NJ	Subpart 2/Moderate	Hunterdon Co	W	121,989
New York-N. New Jersey-Long Island Area	NJ	Subpart 2/Moderate	Middlesex Co	W	750,162
New York-N. New Jersey-Long Island Area	NJ	Subpart 2/Moderate	Monmouth Co	W	615,301
New York-N. New Jersey-Long Island Area	NJ	Subpart 2/Moderate	Morris Co	W	470,212
New York-N. New Jersey-Long Island Area	NJ	Subpart 2/Moderate	Passaic Co	W	489,049
New York-N. New Jersey-Long Island Area	NJ	Subpart 2/Moderate	Somerset Co	W	297,490
New York-N. New Jersey-Long Island Area	NJ	Subpart 2/Moderate	Sussex Co	W	144,166
New York-N. New Jersey-Long Island Area	NJ	Subpart 2/Moderate	Union Co	W	522,541
New York-N. New Jersey-Long Island Area	NJ	Subpart 2/Moderate	Warren Co	W	102,437
New York-N. New Jersey-Long Island Area	NY	Subpart 2/Moderate	Bronx Co	W	1,332,650
New York-N. New Jersey-Long Island Area	NY	Subpart 2/Moderate	Kings Co	W	2,465,326
New York-N. New Jersey-Long Island Area	NY	Subpart 2/Moderate	Nassau Co	W	1,334,544
New York-N. New Jersey-Long Island Area	NY	Subpart 2/Moderate	New York Co	W	1,537,195
New York-N. New Jersey-Long Island Area	NY	Subpart 2/Moderate	Queens Co	W	2,229,379
New York-N. New Jersey-Long Island Area	NY	Subpart 2/Moderate	Richmond Co	W	443,728
New York-N. New Jersey-Long Island Area	NY	Subpart 2/Moderate	Rockland Co	W	286,753
New York-N. New Jersey-Long Island Area	NY	Subpart 2/Moderate	Suffolk Co	W	1,419,369
New York-N. New Jersey-Long Island Area	NY	Subpart 2/Moderate	Westchester Co	W	923,459
Norfolk-Virginia Beach-Newport News (Hampton	VA	Subpart 2/Marginal	Chesapeake	W	199,184

Chapter 2 Appendices: Air Quality and Resulting Health and Welfare Effects

8-hour Ozone Nonattainment	State	Classification ^{a,b}	County Name	Whole /Part	2000 Cty Pop
Roads) Area					
Norfolk-Virginia Beach-Newport News (Hampton Roads) Area	VA	Subpart 2/Marginal	Gloucester Co	W	34,780
Norfolk-Virginia Beach-Newport News (Hampton Roads) Area	VA	Subpart 2/Marginal	Hampton	W	146,437
Norfolk-Virginia Beach-Newport News (Hampton Roads) Area	VA	Subpart 2/Marginal	Isle Of Wight Co	W	29,728
Norfolk-Virginia Beach-Newport News (Hampton Roads) Area	VA	Subpart 2/Marginal	James City Co	W	48,102
Norfolk-Virginia Beach-Newport News (Hampton Roads) Area	VA	Subpart 2/Marginal	Newport News	W	180,150
Norfolk-Virginia Beach-Newport News (Hampton Roads) Area	VA	Subpart 2/Marginal	Norfolk	W	234,403
Norfolk-Virginia Beach-Newport News (Hampton Roads) Area	VA	Subpart 2/Marginal	Poquoson	W	11,566
Norfolk-Virginia Beach-Newport News (Hampton Roads) Area	VA	Subpart 2/Marginal	Portsmouth	W	100,565
Norfolk-Virginia Beach-Newport News (Hampton Roads) Area	VA	Subpart 2/Marginal	Suffolk	W	63,677
Norfolk-Virginia Beach-Newport News (Hampton Roads) Area	VA	Subpart 2/Marginal	Virginia Beach	W	425,257
Norfolk-Virginia Beach-Newport News (Hampton Roads) Area	VA	Subpart 2/Marginal	Williamsburg	W	11,998
Norfolk-Virginia Beach-Newport News (Hampton Roads) Area	VA	Subpart 2/Marginal	York Co	W	56,297
Parkersburg-Marietta Area	OH	Subpart 1	Washington Co	W	63,251
Parkersburg-Marietta Area	WV	Subpart 1	Wood Co	W	87,986
Philadelphia-Wilmington-Atlantic City Area	DE	Subpart 2/Moderate	Kent Co	W	126,697
Philadelphia-Wilmington-Atlantic City Area	DE	Subpart 2/Moderate	New Castle Co	W	500,265
Philadelphia-Wilmington-Atlantic City Area	DE	Subpart 2/Moderate	Sussex Co	W	156,638
Philadelphia-Wilmington-Atlantic City Area	MD	Subpart 2/Moderate	Cecil Co	W	85,951
Philadelphia-Wilmington-Atlantic City Area	NJ	Subpart 2/Moderate	Atlantic Co	W	252,552
Philadelphia-Wilmington-Atlantic City Area	NJ	Subpart 2/Moderate	Burlington Co	W	423,394

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8-hour Ozone Nonattainment	State	Classification ^{a,b}	County Name	Whole /Part	2000 Cty Pop
Philadelphia-Wilmington-Atlantic City Area	NJ	Subpart 2/Moderate	Camden Co	W	508,932
Philadelphia-Wilmington-Atlantic City Area	NJ	Subpart 2/Moderate	Cape May Co	W	102,326
Philadelphia-Wilmington-Atlantic City Area	NJ	Subpart 2/Moderate	Cumberland Co	W	146,438
Philadelphia-Wilmington-Atlantic City Area	NJ	Subpart 2/Moderate	Gloucester Co	W	254,673
Philadelphia-Wilmington-Atlantic City Area	NJ	Subpart 2/Moderate	Mercer Co	W	350,761
Philadelphia-Wilmington-Atlantic City Area	NJ	Subpart 2/Moderate	Ocean Co	W	510,916
Philadelphia-Wilmington-Atlantic City Area	NJ	Subpart 2/Moderate	Salem Co	W	64,285
Philadelphia-Wilmington-Atlantic City Area	PA	Subpart 2/Moderate	Bucks Co	W	597,635
Philadelphia-Wilmington-Atlantic City Area	PA	Subpart 2/Moderate	Chester Co	W	433,501
Philadelphia-Wilmington-Atlantic City Area	PA	Subpart 2/Moderate	Delaware Co	W	550,864
Philadelphia-Wilmington-Atlantic City Area	PA	Subpart 2/Moderate	Montgomery Co	W	750,097
Philadelphia-Wilmington-Atlantic City Area	PA	Subpart 2/Moderate	Philadelphia Co	W	1,517,550
Phoenix-Mesa Area	AZ	Subpart 1	Maricopa Co	P	3,054,504
Phoenix-Mesa Area	AZ	Subpart 1	Pinal Co	P	31,541
Pittsburgh-Beaver Valley Area	PA	Subpart 1	Allegheny Co	W	1,281,666
Pittsburgh-Beaver Valley Area	PA	Subpart 1	Armstrong Co	W	72,392
Pittsburgh-Beaver Valley Area	PA	Subpart 1	Beaver Co	W	181,412
Pittsburgh-Beaver Valley Area	PA	Subpart 1	Butler Co	W	174,083
Pittsburgh-Beaver Valley Area	PA	Subpart 1	Fayette Co	W	148,644
Pittsburgh-Beaver Valley Area	PA	Subpart 1	Washington Co	W	202,897
Pittsburgh-Beaver Valley Area	PA	Subpart 1	Westmoreland Co	W	369,993
Portland Area	ME	Subpart 2/Marginal	Androscoggin Co	P	3,390
Portland Area	ME	Subpart 2/Marginal	Cumberland Co	P	252,907
Portland Area	ME	Subpart 2/Marginal	Sagadahoc Co	W	35,214
Portland Area	ME	Subpart 2/Marginal	York Co	P	164,997
Poughkeepsie Area	NY	Subpart 2/Moderate	Dutchess Co	W	280,150
Poughkeepsie Area	NY	Subpart 2/Moderate	Orange Co	W	341,367
Poughkeepsie Area	NY	Subpart 2/Moderate	Putnam Co	W	95,745
Providence (all of RI) Area	RI	Subpart 2/Moderate	Bristol Co	W	50,648
Providence (all of RI) Area	RI	Subpart 2/Moderate	Kent Co	W	167,090
Providence (all of RI) Area	RI	Subpart 2/Moderate	Newport Co	W	85,433

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8-hour Ozone Nonattainment	State	Classification ^{a,b}	County Name	Whole /Part	2000 Cty Pop
Providence (all of RI) Area	RI	Subpart 2/Moderate	Providence Co	W	621,602
Providence (all of RI) Area	RI	Subpart 2/Moderate	Washington Co	W	123,546
Raleigh-Durham-Chapel Hill Area	NC	Subpart 1	Chatham Co	P	21,320
Raleigh-Durham-Chapel Hill Area	NC	Subpart 1	Durham Co	W	223,314
Raleigh-Durham-Chapel Hill Area	NC	Subpart 1	Franklin Co	W	47,260
Raleigh-Durham-Chapel Hill Area	NC	Subpart 1	Granville Co	W	48,498
Raleigh-Durham-Chapel Hill Area	NC	Subpart 1	Johnston Co	W	121,965
Raleigh-Durham-Chapel Hill Area	NC	Subpart 1	Orange Co	W	118,227
Raleigh-Durham-Chapel Hill Area	NC	Subpart 1	Person Co	W	35,623
Raleigh-Durham-Chapel Hill Area	NC	Subpart 1	Wake Co	W	627,846
Reading Area	PA	Subpart 1	Berks Co	W	373,638
Richmond-Petersburg Area	VA	Subpart 2/Marginal	Charles City Co	W	6,926
Richmond-Petersburg Area	VA	Subpart 2/Marginal	Chesterfield Co	W	259,903
Richmond-Petersburg Area	VA	Subpart 2/Marginal	Colonial Heights	W	16,897
Richmond-Petersburg Area	VA	Subpart 2/Marginal	Hanover Co	W	86,320
Richmond-Petersburg Area	VA	Subpart 2/Marginal	Henrico Co	W	262,300
Richmond-Petersburg Area	VA	Subpart 2/Marginal	Hopewell	W	22,354
Richmond-Petersburg Area	VA	Subpart 2/Marginal	Petersburg	W	33,740
Richmond-Petersburg Area	VA	Subpart 2/Marginal	Prince George Co	W	33,047
Richmond-Petersburg Area	VA	Subpart 2/Marginal	Richmond	W	197,790
Riverside County (Coachella Valley) Area	CA	Subpart 2/Serious	Riverside Co	P	324,750
Roanoke Area	VA	Subpart 1 - EAC	Botetourt Co	W	30,496
Roanoke Area	VA	Subpart 1 - EAC	Roanoke	W	94,911
Roanoke Area	VA	Subpart 1 - EAC	Roanoke Co	W	85,778
Roanoke Area	VA	Subpart 1 - EAC	Salem	W	24,747
Rochester Area	NY	Subpart 1	Genesee Co	W	60,370
Rochester Area	NY	Subpart 1	Livingston Co	W	64,328
Rochester Area	NY	Subpart 1	Monroe Co	W	735,343
Rochester Area	NY	Subpart 1	Ontario Co	W	100,224
Rochester Area	NY	Subpart 1	Orleans Co	W	44,171
Rochester Area	NY	Subpart 1	Wayne Co	W	93,765
Rocky Mount Area	NC	Subpart 1	Edgecombe Co	W	55,606
Rocky Mount Area	NC	Subpart 1	Nash Co	W	87,420
Sacramento Metro Area	CA	Subpart 2/Serious	El Dorado Co	P	124,164
Sacramento Metro Area	CA	Subpart 2/Serious	Placer Co	P	239,978
Sacramento Metro Area	CA	Subpart 2/Serious	Sacramento Co	W	1,223,499
Sacramento Metro Area	CA	Subpart 2/Serious	Solano Co	P	197,034
Sacramento Metro Area	CA	Subpart 2/Serious	Sutter Co	P	25,013
Sacramento Metro Area	CA	Subpart 2/Serious	Yolo Co	W	168,660

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8-hour Ozone Nonattainment	State	Classification ^{a,b}	County Name	Whole /Part	2000 Cty Pop
San Antonio Area	TX	Subpart 1 - EAC	Bexar Co	W	1,392,931
San Antonio Area	TX	Subpart 1 - EAC	Comal Co	W	78,021
San Antonio Area	TX	Subpart 1 - EAC	Guadalupe Co	W	89,023
San Diego Area	CA	Subpart 1	San Diego Co	P	2,813,431
San Francisco Bay Area	CA	Subpart 2/Marginal	Alameda Co	W	1,443,741
San Francisco Bay Area	CA	Subpart 2/Marginal	Contra Costa Co	W	948,816
San Francisco Bay Area	CA	Subpart 2/Marginal	Marin Co	W	247,289
San Francisco Bay Area	CA	Subpart 2/Marginal	Napa Co	W	124,279
San Francisco Bay Area	CA	Subpart 2/Marginal	San Francisco Co	W	776,733
San Francisco Bay Area	CA	Subpart 2/Marginal	San Mateo Co	W	707,161
San Francisco Bay Area	CA	Subpart 2/Marginal	Santa Clara Co	W	1,682,585
San Francisco Bay Area	CA	Subpart 2/Marginal	Solano Co	P	197,508
San Francisco Bay Area	CA	Subpart 2/Marginal	Sonoma Co	P	413,716
San Joaquin Valley Area	CA	Subpart 2/Serious	Fresno Co	W	799,407
San Joaquin Valley Area	CA	Subpart 2/Serious	Kern Co	P	550,220
San Joaquin Valley Area	CA	Subpart 2/Serious	Kings Co	W	129,461
San Joaquin Valley Area	CA	Subpart 2/Serious	Madera Co	W	123,109
San Joaquin Valley Area	CA	Subpart 2/Serious	Merced Co	W	210,554
San Joaquin Valley Area	CA	Subpart 2/Serious	San Joaquin Co	W	563,598
San Joaquin Valley Area	CA	Subpart 2/Serious	Stanislaus Co	W	446,997
San Joaquin Valley Area	CA	Subpart 2/Serious	Tulare Co	W	368,021
Scranton-Wilkes-Barre Area	PA	Subpart 1	Lackawanna Co	W	213,295
Scranton-Wilkes-Barre Area	PA	Subpart 1	Luzerne Co	W	319,250
Scranton-Wilkes-Barre Area	PA	Subpart 1	Monroe Co	W	138,687
Scranton-Wilkes-Barre Area	PA	Subpart 1	Wyoming Co	W	28,080
Sheboygan Area	WI	Subpart 2/Moderate	Sheboygan Co	W	112,646
South Bend-Elkhart Area	IN	Subpart 1	Elkhart Co	W	182,791
South Bend-Elkhart Area	IN	Subpart 1	St Joseph Co	W	265,559
Springfield (W. Mass) Area	MA	Subpart 2/Moderate	Berkshire Co	W	134,953
Springfield (W. Mass) Area	MA	Subpart 2/Moderate	Franklin Co	W	71,535
Springfield (W. Mass) Area	MA	Subpart 2/Moderate	Hampden Co	W	456,228
Springfield (W. Mass) Area	MA	Subpart 2/Moderate	Hampshire Co	W	152,251
St. Louis Area	IL	Subpart 2/Moderate	Jersey Co	W	21,668
St. Louis Area	IL	Subpart 2/Moderate	Madison Co	W	258,941
St. Louis Area	IL	Subpart 2/Moderate	Monroe Co	W	27,619
St. Louis Area	IL	Subpart 2/Moderate	St Clair Co	W	256,082
St. Louis Area	MO	Subpart 2/Moderate	Franklin Co	W	93,807
St. Louis Area	MO	Subpart 2/Moderate	Jefferson Co	W	198,099
St. Louis Area	MO	Subpart 2/Moderate	St Charles Co	W	283,883
St. Louis Area	MO	Subpart 2/Moderate	St Louis	W	348,189
St. Louis Area	MO	Subpart 2/Moderate	St Louis Co	W	1,016,315
State College Area	PA	Subpart 1	Centre Co	W	135,758
Steubenville-Weirton Area	OH	Subpart 1	Jefferson Co	W	73,894
Steubenville-Weirton Area	WV	Subpart 1	Brooke Co	W	25,447
Steubenville-Weirton Area	WV	Subpart 1	Hancock Co	W	32,667

Chapter 2 Appendices: Air Quality and Resulting Health and Welfare Effects

8-hour Ozone Nonattainment	State	Classification ^{a,b}	County Name	Whole /Part	2000 Cty Pop
Sutter County (part) (Sutter Buttes) Area	CA	Subpart 1	Sutter Co	P	1
Tioga County Area	PA	Subpart 1	Tioga Co	W	41,373
Toledo Area	OH	Subpart 1	Lucas Co	W	455,054
Toledo Area	OH	Subpart 1	Wood Co	W	121,065
Ventura County (part) Area	CA	Subpart 2/Moderate	Ventura Co	P	753,197
Washington Area	DC	Subpart 2/Moderate	Entire District	W	572,059
Washington Area	MD	Subpart 2/Moderate	Calvert Co	W	74,563
Washington Area	MD	Subpart 2/Moderate	Charles Co	W	120,546
Washington Area	MD	Subpart 2/Moderate	Frederick Co	W	195,277
Washington Area	MD	Subpart 2/Moderate	Montgomery Co	W	873,341
Washington Area	MD	Subpart 2/Moderate	Prince George's Co	W	801,515
Washington Area	VA	Subpart 2/Moderate	Alexandria	W	128,283
Washington Area	VA	Subpart 2/Moderate	Arlington Co	W	189,453
Washington Area	VA	Subpart 2/Moderate	Fairfax	W	21,498
Washington Area	VA	Subpart 2/Moderate	Fairfax Co	W	969,749
Washington Area	VA	Subpart 2/Moderate	Falls Church	W	10,377
Washington Area	VA	Subpart 2/Moderate	Loudoun Co	W	169,599
Washington Area	VA	Subpart 2/Moderate	Manassas	W	35,135
Washington Area	VA	Subpart 2/Moderate	Manassas Park	W	10,290
Washington Area	VA	Subpart 2/Moderate	Prince William Co	W	280,813
Washington County (Hagerstown) Area	MD	Subpart 1 - EAC	Washington Co	W	131,923
Wheeling Area	OH	Subpart 1	Belmont Co	W	70,226
Wheeling Area	WV	Subpart 1	Marshall Co	W	35,519
Wheeling Area	WV	Subpart 1	Ohio Co	W	47,427
York Area	PA	Subpart 1	Adams Co	W	91,292
York Area	PA	Subpart 1	York Co	W	381,751
Youngstown-Warren-Sharon Area	OH	Subpart 1	Columbiana Co	W	112,075
Youngstown-Warren-Sharon Area	OH	Subpart 1	Mahoning Co	W	257,555
Youngstown-Warren-Sharon Area	OH	Subpart 1	Trumbull Co	W	225,116
Youngstown-Warren-Sharon Area	PA	Subpart 1	Mercer Co	W	120,293

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²¹ Currently, two established receptor models are widely used for source apportionment studies: the Chemical Mass Balance (CMB) model and Positive Matrix Factorization (PMF). The CMB receptor model relies on measured source profiles as well as ambient species measurements to produce a source contribution estimate at the receptor location, while the

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