Chapter 2: Air Quality Analysis

Synopsis

This chapter describes the NOx emissions, NO_2 monitoring network, and approach used to calculate 2020 baseline near-roadway NO_2 design values and the amount of emissions reductions needed to attain alternative levels of the 1-hour NO_2 NAAQS. We first describe data on NO_2 emission sources contained in available EPA emission inventories. We then provide an overview of data sources for air quality measurement, and finally the methodology used to project NO_2 levels to 2020. For a more in-depth discussion of NO_2 emissions and air quality data, see the Integrated Science Assessment for the NO_2 NAAQS (EPA, 2007a).

2.1 Sources of NO_2

The primary data source for this discussion is the National Emissions Inventory (NEI) for 2002 (USEPA, 2007b). Ambient levels of NO_2 are the product of both direct NO_2 emissions and emissions of other NOx (e.g., NO), which can then be converted to NO_2 through reaction with ozone. Nationally, anthropogenic sources account for approximately 87% of total NOx emissions. (Apart from these anthropogenic sources, there are also natural sources of NOx including microbial activity in soils, lightning, and wildfires.)

Stationary sources (e.g., electrical utilities and industry) account for about 40% of the national anthropogenic NOx emissions in the 2002 NEI. The main stationary sources of NOx emissions in the 2002 NEI are combustion-related emissions and industrial process-related emissions. Table 2-1 presents emissions estimates for stationary sources grouped into descriptive categories. Presence and relative position of a source category on this list does not necessarily provide an indication of the significance of the emissions from individual sources within the source category. A source category, for example, may be composed of many small (i.e., low-emitting) sources, or of just a few very large (high-emitting) sources.

Mobile sources (both on-road and non-road) account for about 60% of the national anthropogenic NOx emissions in the 2002 NEI. Highway vehicles represent the major mobile source component. In the United States, approximately half the mobile source emissions are contributed by diesel engines and half are emitted by gasoline-fueled vehicles and other sources.

As a result of Clean Air Act requirements, emissions standards promulgated for many source categories have taken effect since 2002, including numerous mobile source standards for gasoline and diesel vehicles/engines, and are projected to result in much lower emissions of both direct NO_2 and other NOx at the current time or in the near future.

Table 2-1. NOx Sources (2002 NEI)

NOx Source Category	Emissions (tons/year)
Electric Utility Fuel Combustion	3,792,292
Industrial Fuel Combustion	1,897,944
Fuel Combustion, other	730,259
Chemical and Allied Product Manufacturing	60,901
Metals Processing	66,173
Petroleum and Related Industries	358,223
Industrial Processes, other	482,007
Solvent Utilization	4,365
Storage and Transport	16,109
Waste Disposal and Recycling	145,678
Highway Vehicles	6,491,821
Off-highway Vehicles	6,027,085
Miscellaneous Source Categories	270,913
Total	**20,343,770**

2.2 Air Quality Monitoring Data

2.2.1 Background on NO_2 monitoring network

From its inception in the late 1970's through the present (2008), the NO_2 network has remained relatively stable with regard to the number of monitoring sites (see memo by Watkins, 2008). As of October 2008, there were 409 NOx monitors within the U.S. actively reporting NO_2 data to the air quality system AQS. The NO_2 network was originally deployed to support implementation of the NO_2 NAAQS established in 1971. The first requirements for NO_2 monitoring were issued in May 1979. At that time, 40 CFR Part 58, Appendix D, section 3.5 stated:

> "Nitrogen Dioxide NAMS [National Ambient Monitoring Stations, now a defunct term] will be required in those areas of the country which have a population greater than 1,000,000. These areas will have two NO_2 NAMS. It is felt that stations in these major metropolitan areas would provide sufficient data for a national analysis of the data, and also because NO_2 problems occur in areas of greater than 1,000,000. Within urban areas requiring [NO_2] NAMS, two permanent monitors are sufficient. The first station (category (a), middle scale or neighborhood scale) would be to measure the photochemical production of NO_2 and would best be located in that part of the urban

> area where the emission density of NOx is the highest. The second station (category (b) urban scale), would be to measure the NO_2 produced from the reaction of NO with O_3 and should be downwind of the area peak NOx emission areas."

In the October, 2006 monitoring rule, these NO_2 monitoring requirements were removed from the CFR due in part to the absence of any NO_2 non-attainment problems under the current standards. In the 2006 rule, EPA rewrote 40 CFR Part 58, Appendix D, section 4.3 to state that:

> "There are no minimum requirements for the number of NO_2 monitoring sites. Continued operation of existing SLAMS [State and Local Ambient Monitoring Station] NO_2 sites using FRM [Federal Reference Method] or FEM [Federal Equivalent Method] is required until discontinuation is approved by the EPA Regional Administrator. Where SLAMS NO_2 monitoring is ongoing, at least one NO_2 site in the area must be located to measure the maximum concentration of NO_2."

As noted earlier, the size of the NO_2 network has been fairly stable through time, even though an actual requirement for state and local air agencies to monitor NO_2, other than for Photochemical Assessment Monitoring Stations (PAMS) or Prevention of Significant Deterioration (PSD), was removed in the 2006 monitoring rule. The maintenance of the NO_2 monitoring network has been driven by several factors, including the need to support ozone modeling and forecasting, the need to track PM precursors, and a general desire on the part of states to continue to understand trends in ambient NO_2.

To characterize the current NO_2 network, staff has reviewed the NO_2 network meta-data. The data reviewed are those available from AQS in October 2008, for monitors reporting data in 2008. The meta-data fields are typically created by state and local agencies when a monitor site is initiated, moved, or re-characterized. While these files are useful for characterizing specific monitors, there is some uncertainty surrounding this meta-data given that there is no routine or enforced process for updating or correcting meta-data fields. With this uncertainty in mind, staff has compiled information on the monitoring objectives and measurement scales for monitors in the NO_2 network.

The monitor objective meta-data field describes the purpose of the monitor. For example the purpose of a particular monitor could be to characterize health effects, photochemical activity, transport, and/or welfare effects. As of October 2008, there were 489 records of NO_2 monitor objective values (some monitors have multiple monitor objectives). Table 2-2 lists the distribution of monitoring objectives across the network. There are 12 categories of monitor objectives for NO_2 monitors within AQS. The “other” category is for sites likely addressing a state or local need outside of the routine objectives, and the “unknown” category represents missing meta-data. The remaining categories stem directly from categorizations of site types within CFR. In 40 CFR Part 58 Appendix D, there are six examples of NO_2 site types:

1. Sites located to determine the highest concentration expected to occur in the area covered by the network (Highest Concentration).
2. Sites located to measure typical concentrations in areas of high population (Population Exposure).
3. Sites located to determine the impact of significant sources or source categories on air quality (Source Oriented).
4. Sites located to determine general background concentration levels (General Background).
5. Sites located to determine the extent of regional pollutant transport among populated areas; and in support of secondary standards (Regional Transport).
6. Sites located to measure air pollution impacts on visibility, vegetation damage, or other welfare-based impacts (Welfare Related Impacts).

The remaining four categories represent available site types for Photochemical Assessment Monitoring Stations (PAMS) network. These PAMS site types are described in 40 CFR Part 58 Appendix D:

1. Type 1 sites are established to characterize upwind background and transported ozone and its precursor concentrations entering the area and will identify those areas which are subjected to transport (Upwind Background).
2. Type 2 sites are established to monitor the magnitude and type of precursor emissions in the area where maximum precursor emissions are expected to impact and are suited for the monitoring of urban air toxic pollutants (Maximum Precursor Impact).
3. Type 3 sites are intended to monitor maximum ozone concentrations occurring downwind from the area of maximum precursor emissions (Maximum Ozone Concentration).
4. Type 4 sites are established to characterize the downwind transported ozone and its precursor concentrations exiting the area and will identify those areas which are potentially contributing to overwhelming transport in other areas (Extreme Downwind).

Table 2-2: NO_2 Network Distribution of Monitor Objectives.

NO_2 Monitor Objective	Number of Monitor Objective Records	Percent Distribution
Population Exposure	177	36.20
Highest Concentration	58	11.86
General Background	51	10.43
Max. Precursor Impact (PAMS Type 2 Site)	21	4.29
Source Oriented	19	3.89
Upwind Background (PAMS Type 1 Site)	18	3.68
Regional Transport	12	2.45
Other	9	1.84
Max. Ozone Concentration (PAMS Type 3 Site)	8	1.64
Extreme Downwind (PAMS Type 4 Site)	3	0.61
Welfare Related Impacts	1	0.20
Unknown	112	22.90
Totals:	**489**	**100%**

The meta-data for the NO_2 network also indicate the measurement scale represented by each particular monitor. The definitions of measurement scales can be found in 40 CFR Part 58, Appendix D, Section 1 "Monitoring Objectives and Spatial Scales." This part of the regulation spells out what data from a monitor can represent in terms of air volumes associated with area dimensions:

Microscale -	0 to 100 meters
Middle Scale -	100 to 500 meters
Neighborhood Scale -	500 meters to 4 kilometers
Urban Scale -	4 to 50 kilometers
Regional Scale -	50 kilometers up to 1000km

There are 386 NO_2 monitor records in AQS with available measurement scale data. Table 2-3 shows the measurement scale distribution across all NO_2 sites form the available data in AQS of monitors reporting data in 2008.

Table 2-3: NO_2 Network Distribution across Measurement Scales.

Measurement Scale	Number of Measurement Scale Records	Percent Distribution
Microscale	3	0.78
Middle Scale	23	5.96
Neighborhood	212	54.92
Urban Scale	119	30.83
Regional Scale	29	7.51
Totals:	**386**	**100%**

Many of the monitors used in the analyses presented here, especially for the near-road adjustment calculations, are defined as area-wide monitors. These are monitors that would meet the following criteria:

- Neighborhood, urban, or regional scale (based on measurement scale)
- Not a site identified as being operated by industry
- If the monitor is a neighborhood scale monitor, its monitor objective is not highest concentration and its dominant source is not a point source.

The criteria above will be used to identify monitors to adjust for near-road conditions in Section 2.3.2.2. More details about monitor classification can be found in Appendix 2.

In summary, the NO_2 network is primarily targeting public health and photochemical process monitoring objectives. Nearly half of the monitor objective records are directly targeting public health through the population exposure (36.2%) and highest concentration (11.8%) categories alone. The other categories serve to inform public health concerns, but also address photochemistry issues where NOx serves as a precursor to ozone. Further, it appears that approximately 10% of NO_2 monitors are in place to serve the PAMS network. In reality, a large majority of sites likely could serve both public health and photochemistry related objectives due to their proximity to urban areas. The exceptions would likely be categories such as upwind background, extreme downwind, regional transport, and possibly maximum O_3 concentration. These four categories only represent approximately 7% of the NO_2 network, and have a higher likelihood of being rural and regional in scale.

2.2.2 Trends in and characterizations of ambient concentrations of NO_2

As noted above, NO_2 is monitored largely in urban areas and, therefore, data from the NO_2 monitoring network is generally more representative of urban areas than rural areas. According to monitoring data, nationwide levels of ambient NO_2 (annual average) decreased 41% between 1980 and 2006 (ISA, Figure 2.4-15). Between 2003 and 2005, national mean concentrations of NO_2 were about 15 ppb for averaging periods ranging from a day to a year. The average daily maximum hourly NO_2 concentrations were approximately 30 ppb. These values are about twice as high as the 24-h averages. The highest maximum hourly concentrations (~200 ppb) between 2003 and 2005 are more than a factor of ten higher than the mean hourly or 24-h concentrations (ISA, Figure 2.4-13). The monthly highest levels of NO_2 in the United States can be found in and around Los Angeles, in the Midwest, and in the Northeast. Local maxim around Denver, CO, Salt Lake City, UT, and El Paso, TX can also be found (ISA, Figure 2.4-14) Policy-relevant background concentrations, which are those concentrations that would occur in the United States in the absence of anthropogenic emissions in continental North America (defined here as the United States, Canada, and Mexico), are estimated to range from only 0.1 ppb to 0.3 ppb on an annual basis (ISA, section 2.4.6).

Ambient levels of NO_2 exhibit both seasonal and diurnal variation. In southern cities, such as Atlanta, higher concentrations are found during winter, consistent with the lowest mixing layer heights being found during that time of the year. Lower concentrations are found during summer, consistent with higher mixing layer heights and increased rates of photochemical oxidation of NO_2. For cities in the Midwest and Northeast, such as Chicago and New York City, higher levels tend to be found from late winter to early spring with lower levels occurring from summer though the fall. In Salt Lake City, higher concentrations tend to be found in winter in association with winter temperature inversions. In Los Angeles the highest levels tend to occur from autumn though early winter and the lowest levels from spring though early summer. Mean and peak concentrations in winter can be up to a factor of two larger than in the summer at sites in Los Angeles. In terms of daily variability, NO_2 levels typically peak during the morning rush hours. Monitor siting plays a key role in evaluating diurnal variability as monitors located further away from traffic will show cycles that are less pronounced over the course of a day than monitors located closer to traffic.

2.2.3 Uncertainty Associated with the Ambient NO_2 Monitoring Method

As has been acknowledged by the Agency and the scientific community for some time, the most prevalently used measurement method for estimating ambient NO_2 levels (i.e., subtraction of NO from a measure of total NOx) is subject to interference by NOx oxidation products. Limited evidence from some studies suggests that these interferences could result in an overestimate of NO_2 levels by roughly 20 to 25% at typical ambient levels. However, smaller relative errors are estimated to occur in measurements taken near strong NOx sources since most of the mass emitted as NO or NO_2 would not yet have been further oxidized. Relatively larger errors appear in locations more distant from strong local NOx sources. Two additional sources of uncertainty in NO_2 measurements can result from monitor siting. First, many NO_2 monitors are located above ground level in the cores of large cities. Because most sources of NO_2 are near ground level (i.e., combustion emissions from traffic), higher levels NO_2 concentrations exist near ground level and lower levels being detected at the elevated monitors. One comparison has found an average of a 2.5-fold increase in NO_2 concentration measured at 4 meters above the ground compared to 15 meters above the ground. The ISA notes that levels are likely even higher at elevations below 4 meters (ISA, section 2.5.3.3). Second, NO_2 monitors are currently sited to determine annual regional levels rather than to capture small-scale variability in NO_2 concentrations near sources such as roadway traffic. Significant gradients in NO_2 concentrations near roadways have been observed in several studies, and NO_2 concentrations have been found to be negatively correlated with distance from roadway and traffic volume (ISA, section 2.5.3.2).

2.3 Air Quality Analysis

The principle objective of this air quality analysis is to estimate 2020 design values[1] that reflect maximum concentrations, compare these estimates to alternative levels of the NO_2 NAAQS, and determine the emission reductions required to reduce NO_2 air quality concentrations to below these various levels. Two challenges exist: estimating future levels given reductions from promulgated control programs and determining these future levels in locations where we expect maximum short term concentrations to occur. The first challenge is typical of RIA analyses and the second is unique to NO_2 because the monitoring network is not currently optimized to represent maximum short term levels. Such levels are expected to occur near roads but the monitoring network, while urban in its orientation, is oriented to area-wide measurements. In order to overcome the absence of a current near road monitoring network, we have used scientific literature on the gradients between near road levels and those locations at various distances from roads to estimate near road levels. In other words, we are adjusting NO_2 levels from area wide locations to attempt to approximate near road conditions.

The alternative levels of the NO_2 NAAQS being analyzed are 80, 100, and 125 ppb based on design values calculated using the 3-year average of the 98th percentile 1-hour daily maximum concentrations based on the monitoring network described in section 2.2 with adjustments for a near-road network. The projected 2020 baseline NO_2 design values are used to identify 2020 nonattainment counties and to calculate, for each such county, the amount of reduction in NO_2 concentration necessary to attain the alternative levels of the NAAQS. This section also describes the approach for calculating "ppb NO_2 concentration per ton NOx

[1] A design value is a statistic that describes the air quality status of a given area relative to the level of the National Ambient Air Quality Standards (NAAQS). http://www.epa.gov/airtrends/values.html

emissions" ratios that are used to estimate the amount of NOx emissions reductions that may be needed to provide for attainment of the alternative NO_2 standards. As described below, the air quality analysis relies on NO_2 predictions from simulations of the Community Multiscale Air Quality (CMAQ) model coupled with ambient 2005-2007 design values and emissions data to project 2020 NO_2 design value concentrations and the "ppb per ton" ratios. A description of CMAQ is provided in the Ozone NAAQS RIA Air Quality Modeling Platform Document (U.S. EPA, 2008a).

2.3.1 2005-2007 Design Values

The form of the final NO_2 standard is the 3-year average of the 98th concentration of the daily 1-hour maximum concentration for each year using measurements from the monitoring network described in Section 2.2. The first step in calculating the 3-year 2005-2007 design values is to identify the maximum 1-hour concentration for every day during the three years 2005 through 2007. Next, the 98th percentile concentration of these daily 1-hour maximum concentrations is calculated for each year. The 98th percentile concentrations for each year are averaged to determine a 3-year average concentration. Only monitors that had valid measurements for at least 75% of the day, 75% of the days in a quarter, and all 4 quarters for all three years were considered to have sufficient data completeness to be representative and were thereby included in the analysis[2]. In 2007, there were 435 monitors (259 counties) for NO_2 nationwide. Of those 435 monitors, 256 monitors (160 counties) met the criteria described above. Appendix 2a contains the complete list of 2005-2007 design values used in calculation of the 2020 design values. Note that Hawaiian monitors were excluded from the air quality analysis because there was no CMAQ data over Hawaii. This decreased the number of monitors and counties used in the analysis to 255 monitors and 159 counties

In Figure 2-1, the Core Based Statistical Areas' (CBSA) with populations greater 350,000 people are shown along with the number of monitors in each CBSA (CBSAs outside the continental U.S. are not included). Those with zero monitors have no monitors because: 1) no monitor was in the CBSA or 2) the monitors in the CBSA did not meet the completeness criteria described above. The number of monitors in Figure 2-1 represents 210 of the 255 monitors. The remaining 45 monitors were either in CBSAs with populations less than 350,000 people or not located in a CBSA. Figure 2-2 shows the population of the CBSAs shown in Figure 2-1. Figure 2-3 shows the population of the CBSAs within several population categories for CBSAs with population greater than 350,000 people. Shown are populations for CBSAs with monitors in the 2005-2007 period (green bars), those that have monitors but were excluded due to data completeness (yellow bars) and those CBSAs currently not monitored (orange bars). Also shown

[2] Email from Rhonda Thompson to James Thurman, January 22, 2009.

in each bar are the number of CBSAs in each population category. As can be seen by Figure 2-3, approximately 160 million people are in CBSAs that have at least one monitor in 2005-2007. Also, CBSAs with populations greater than 1 million people are represented in the analyses presented here. The large urban centers such as New York, Los Angeles, and Chicago are represented. Notable CBSAs not included in the analyses are: Detroit, Baltimore, Las Vegas and Seattle. While Detroit, Baltimore, and Las Vegas do have monitors, they were excluded due to incomplete data in 2005-2007, Seattle is currently the largest CBSA without monitors. As part of the new monitoring requirements, Seattle will have at least two monitors as the population of the CBSA is over 2.5 million.

Table 2a-1 in Appendix 2a lists the CBSAs with and the number of monitors from each area used in the analysis and Table 2a-2 lists the CBSAs with populations greater than 350,000 people not included in the analyses. In Table 2a-2 in Appendix A, the CBSA area for each of the 255 monitors is also listed.

Figure 2-1: Number of monitors per CBSA for CBSAs with 2007 population greater than 350,000 people

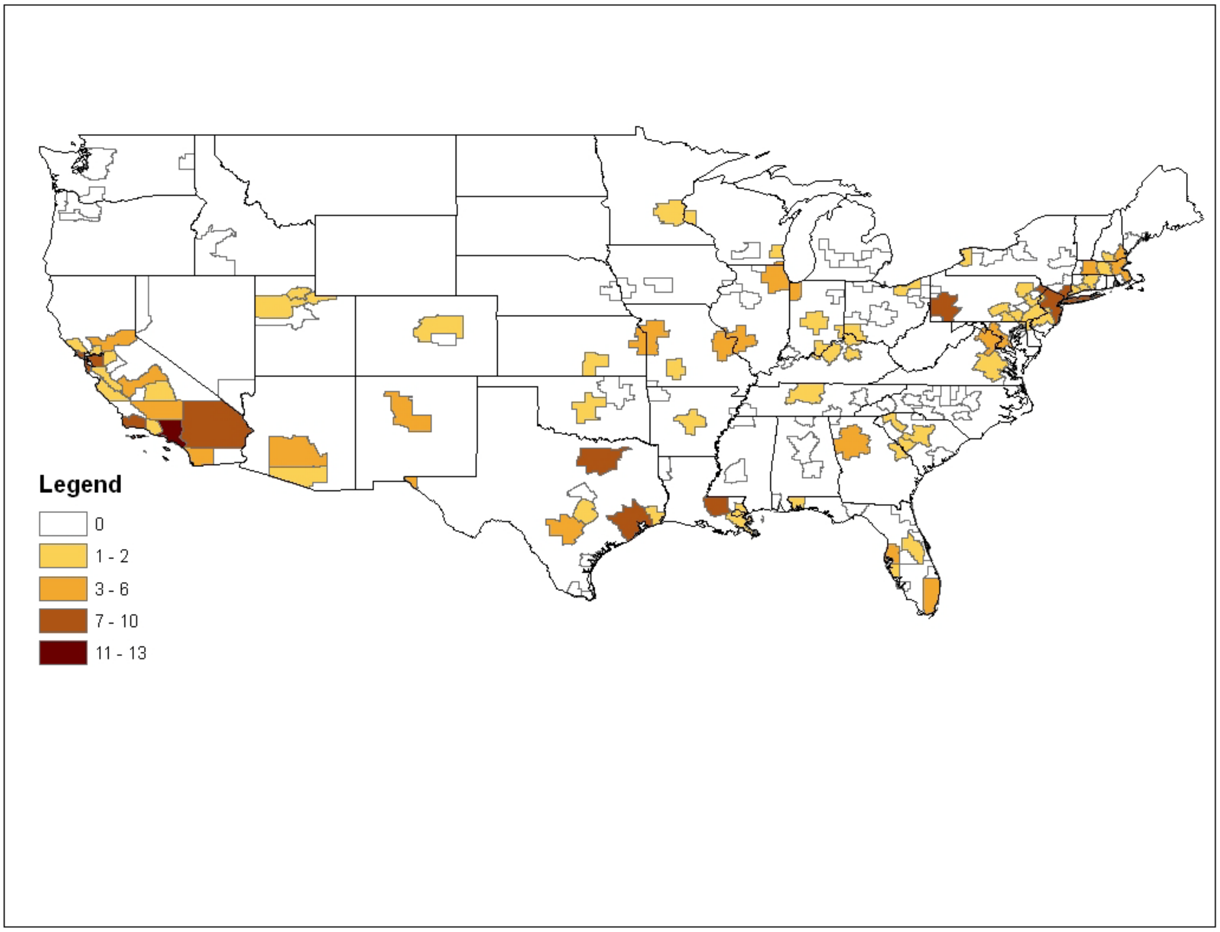


Figure 2-2: Populations of CBSAs with 2007 populations greater than 350,000 people

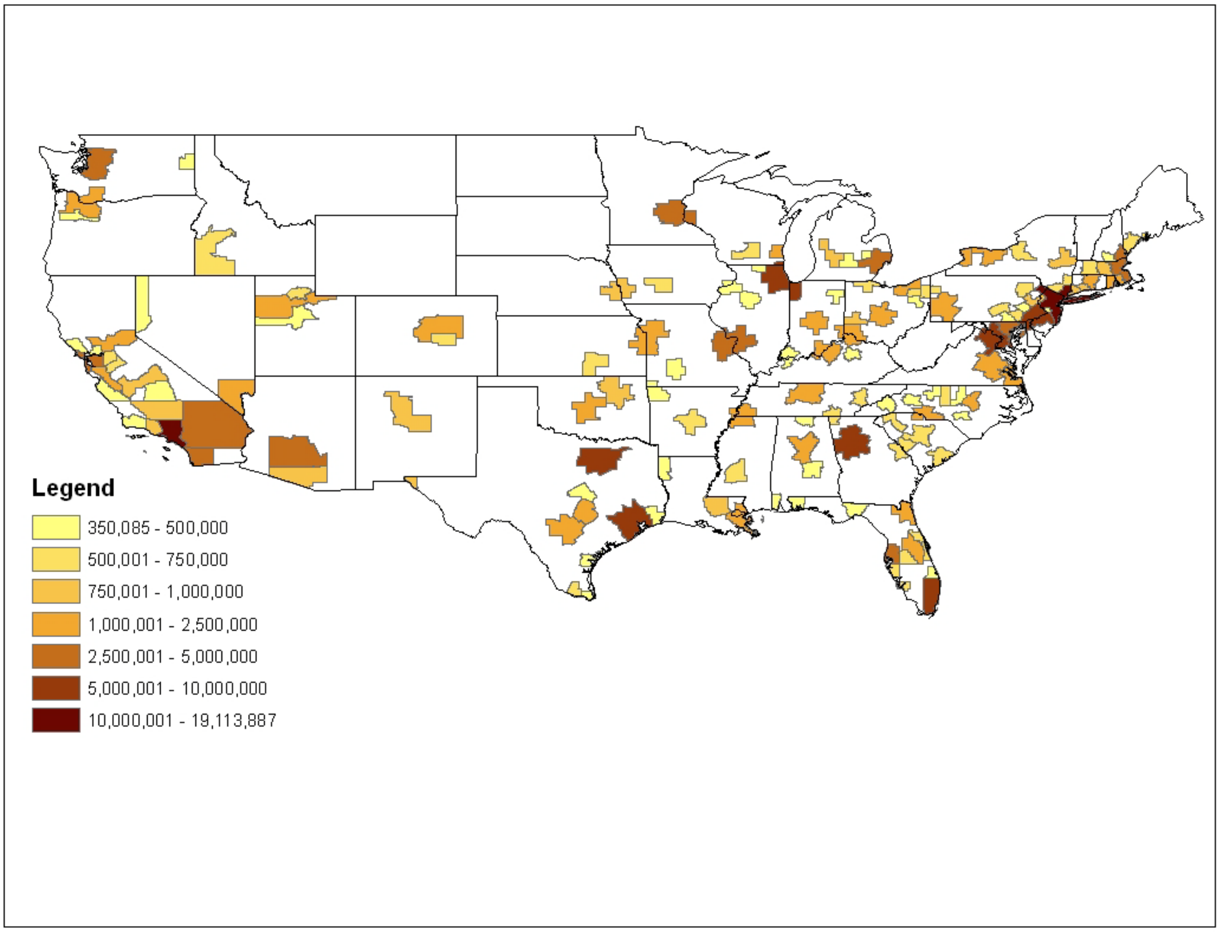
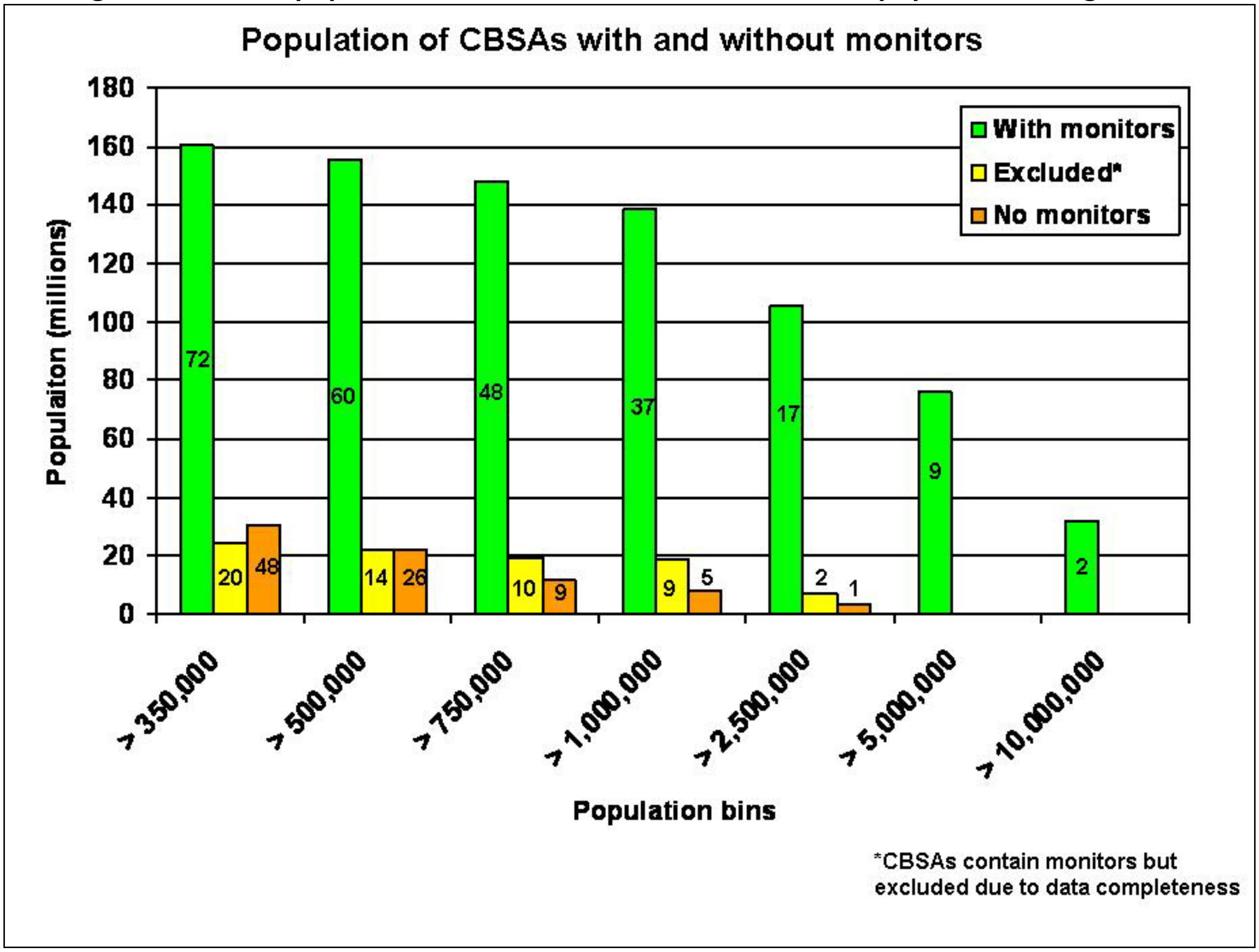


Figure 2-3: Total population and number of CBSAs for several population categories



2.3.2 Calculation of 2020 Projected Design Values

The 2020 baseline design values were calculated in a two step process. First, the 2005-2007 design values, which represented area-wide design values, were projected to 2020 using CMAQ concentrations and county-level emissions. This yielded a 2020 area-wide design value. Second, the projected 2020 area-wide design values were adjusted to simulate near-road concentrations. This adjustment involves two steps: (1) using concentrations gradients at distances from roadways from the scientific literature (i.e., 30, 65, and 100%); and an adjustment to account for the greater efficacy of onroad controls to near-road monitors in the future. This section describes the processing in the projection of 2005-2007 design values to 2020 near-road design values.

2.3.2.1 Calculation of 2020 area-wide design values

The 2020 baseline area-wide design values were determined using CMAQ concentrations for 2002 and 2020 and county emissions for 2002, 2006, and 2020. The CMAQ daily 1-hour maximum concentrations from 2002 and 2020 were used to calculate a relative response factor (RRF). The daily 1-hour maximum NO_2 concentrations in 2002 and 2020 were obtained from CMAQ runs performed for the ozone RIA (U.S. EPA, 2008b). The modeled NOx emissions in the CMAQ runs reflect reductions from federal programs including the Clean Air Interstate Rule (EPA, 2005a), the Clean Air Mercury Rule (EPA, 2005b), the Clean Air Visibility Rule (EPA, 2005c), the Clean Air Nonroad Diesel Rule (EPA, 2004), the Light-Duty Vehicle Tier 2 Rule (EPA, 1999), the Heavy Duty Diesel Rule (EPA, 2000); proposed rules for Locomotive and Marine Vessels (EPA, 2007c) and for Small Spark-Ignition Engines (EPA, 2007d); and national, state and local level mobile and stationary source controls identified for additional reductions in emissions for the purpose of attaining the current PM 2.5 and Ozone standards[3].

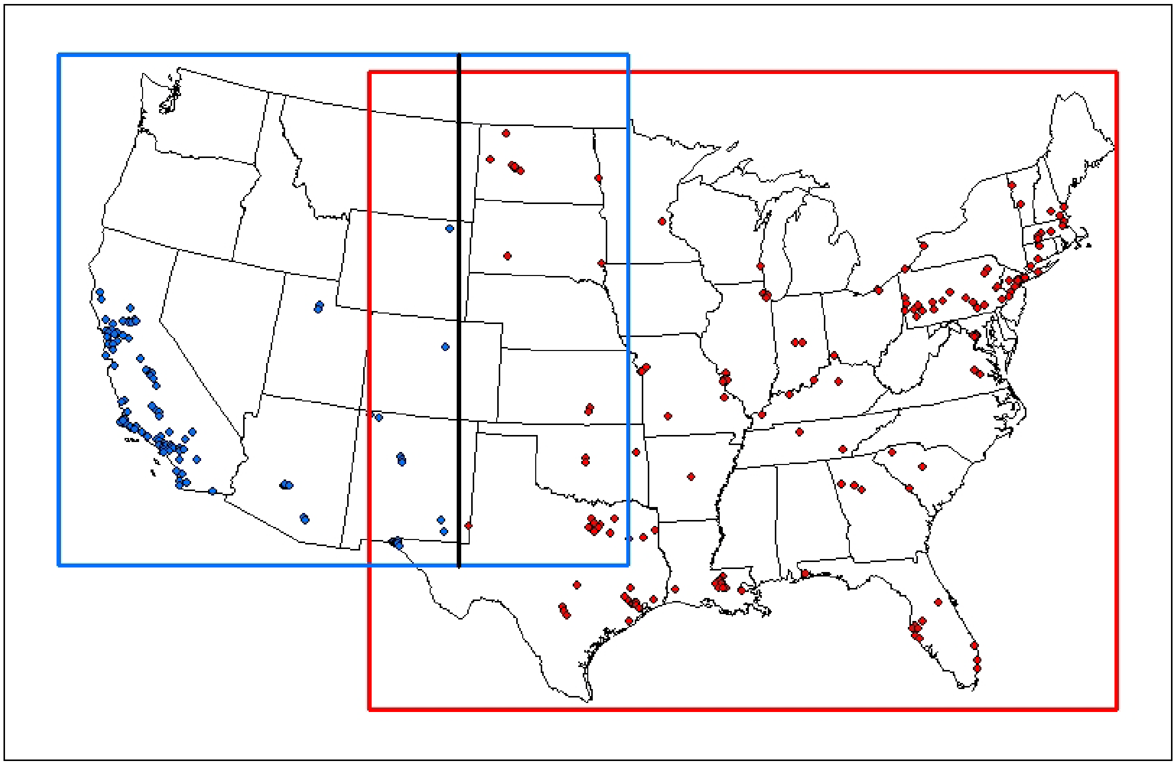
In brief, these CMAQ runs were performed at 12 km horizontal resolution for two modeling domains which, collectively, cover the lower 48 States and adjacent portions of Canada and Mexico. The boundaries of these two domains are shown in Figure 2-4. For 2020 we used CMAQ-predicted NO_2 concentrations from the Ozone NAAQS RIA control case. The CMAQ output represents concentrations based on emissions needed to attain an ozone standard of 0.070 ppm. We will refer to these concentrations and associated emissions as

[3] It should be noted that the emission reductions modeled for the PM2.5 and Ozone standards represent one possible control scenario, while the actual control strategies and resulting levels of emission reductions will be determined as part of the process of developing and implementing state implementation plans over the coming years. We should also note that since the finalization of these recent NAAQS standards, several of the proposed mobile source rules mentioned above have been finalized with updated analyses showing slightly greater levels of expected NOx reductions.

2020_070. As is standard analytic practice used in other RIAs previously, in order to align the base year modeled NO_2 data with the mid-point of the 2005-2007 design value period, we used the relationship between 2002 and 2006 NOx emissions used to estimate the 2002 NO_2 model-predicted concentrations to 2006. In addition to NOx emissions for the modeled 2002_070 (base emissions used in the projected 2020 0.070 ppm standard case) scenario, we calculated emissions for the 2020 baseline scenario, based on an emissions forecast described in Chapter 4 of the ozone RIA (EPA, 2008b). We refer to this inventory as 2020_075. This inventory contains emission reductions for 21 counties that did not meet the 0.070 ppm standard or less stringent 0.075 ppm standard (EPA, 2008b). In these 21 counties, across the board reductions of 30%, 60%, and 90% were made in the areas encompassing parts of California, Texas (centered on Houston), the Midwest (Chicago and Detroit areas), and the Northeast (portions of eastern Pennsylvania, New York, New Jersey, Maryland, Delaware, and Connecticut. These reductions were made to in an attempt to attain the 0.070 ppm standard. These are referred to as Phase I areas in the ozone RIA and can be seen in Figure 4.1 of the ozone RIA (EPA, 2008b. The RRF values and emissions were used to forecast 2020 design values and the amount of residual nonattainment at each monitored location.

The following are the steps used in calculating 2020 baseline NO_2 design values from the 2005-2007 monitor design values and CMAQ NO_2 concentrations for the 2002 and 2020_070 scenarios. Ambient monitored data were assigned to CMAQ grid cells using ArcGIS. Since there were areas of the country where the eastern and western domains overlapped, monitors in these overlapping areas were assigned to the eastern or western grid cells by using a "combined grid." This combined grid was a mesh of the eastern and western domains, with overlapping areas assigned eastern grid cells or western grid cells based on the location relative to the dividing line shown in Figure 2-4. Figure 2-4 also shows the assignment of monitors to the two domains. An example of monitors in both domains was the El Paso County monitors. These monitors were assigned to the western domain.

Figure 2-4: CMAQ 12 km domains and monitors used in air quality analyses. The western domain is outlined in blue and the eastern domain outlined in red. The black vertical line denotes the dividing line to assign monitoring sites to either the eastern or western domains. Monitors in red were assigned to the eastern domain and monitors in blue were assigned to the western domain.



The steps in projecting the 2020 area-wide design values were:

1. Beginning with 12-km CMAQ output, we calculated daily 1-hour maximum concentrations for each grid cell for 2002 and 2020_070 model output. For each grid cell, the top 10 daily 1-hour maximum concentrations for 2002 were averaged (C_{2002}). For 2020_070, the daily 1-hour maximum concentrations for the same calendar days corresponding to the top ten days in 2002 were also averaged (C_{2020_070}).

2. Relative response factors (RRF_C) were calculated by dividing the average of the 2020_070 concentrations by the average of the 2002 concentrations from step 1 (Equation 2.1).

$$RRF_C = \frac{C_{2020\_070}}{C_{2002}} \tag{2.1}$$

3. Monitors were assigned 2002, 2006, 2020_070, and 2020_075 county-wide emissions for the counties in which they were located. The 2020 baseline area-wide design values (i.e., using 2020_075 scenario emissions) were calculated by:

    a. An emissions relative response factor ($RRF_{E:2020\_070}$) was calculated to represent the emission changes from 2002 (E_{2002}) to 2020_070 (E_{2020_070}) as

$$RRF_{E:2020\_070} = \frac{E_{2020\_070}}{E_{2002}} \tag{2.2}$$

    Where E_{2020_070} are the 2020_070 county emissions, E_{2002} are the 2002 county emissions used in the modeling to yield the concentrations used in Steps 1 and 2. The emissions relative response factor is essentially the magnitude of 2020 emissions relative to 2002. If $RRF_{E:2020\_070}$ equals 0.9, that means the 2020_070 emissions are 90% of the 2002 emissions.

    b. We then calculated an emissions relative response factor ($RRF_{E:2020\_075}$) for emissions changes from 2006 (E_{2006}) to 2020_075 (E_{2020_075}) as

$$RRF_{E:2020\_075} = \frac{E_{2020\_075}}{E_{2006}} \tag{2.3}$$

    c. By assuming that the ratio of reduction in concentrations and reduction in emissions from 2002 to 2020_070 would be equal for a change from 2006 to 2020_075,

$$\frac{1 - RRF_{C}}{1 - RRF_{E:2020\_070}} = \frac{1 - RRF_{C:2020\_075}}{1 - RRF_{E:2020\_075}} \tag{2.4a}$$

    we calculated a concentration RRF for 2020_075 ($RRF_{C:2020\_075}$) as

$$RRF_{C:2020\_075} = 1 - \left[\left(\left(1 - RRF_{C}\right) \Big/ \left(1 - RRF_{E:2020\_070}\right) \right) \times \left(1 - RRF_{E:2020\_075}\right) \right] \tag{2.4b}$$

    A concentration RRF for 2020_075 must be calculated from this relationship because we do not have modeled 2006 concentrations or 2020 concentrations under the 0.075 ppm scenario.

d. Using the results from above, a 2020 area-wide 98th percentile design value (DV_{2020_075}) was calculated by multiplying the 2020_075 concentration RRF by the monitor's 2005-2007 98th percentile design values ($DV_{2005\text{-}07}$) by the concentration RRF ($RRF_{C:2020\_075}$) calculated in Equation 2.4b

$$DV_{2020\_075} = RRF_{C:2020\_075} \times DV_{2005-07} \qquad (2.5)$$

4. Once 2020_075 98th percentile design values were calculated, changes in concentrations relative to emissions (ppb/ton) between 2020_075 and 2006 were calculated as:

$$ppb/ton = \left(DV_{2020\_075} - DV_{2005-2007}\right) \Big/ \left(E_{2020\_075} - E_{2006}\right) \qquad (2.6)$$

2.3.2.2 Near-road adjustment of area-wide design values

Once 2020 area-wide design values were calculated, they were adjusted to simulate near-road concentrations.

2.3.2.2.1 Identification of monitors for adjustment

To identify monitors that, accounting for the gradient in concentrations away from the roadway, could inform near-road conditions, OAQPS used (1) monitor characteristics (i.e., metadata) in the AQS database, (2) visual inspection by using Google Earth geospatial software, and (3) the condition that only Core Based Statistical Areas (CBSAs) with populations of 350,000 or greater would be required to have at least one maximum concentration site near roadways consistent with the final NO_2 NAAQS rulemaking.

We first select "area-wide" monitors to adjust to approximate near-roadway conditions. The monitors included in this analysis are those considered to be representative of "area-wide" conditions; i.e. those monitors to which it would be appropriate to apply the gradient to scale from area-wide to near-roadway conditions. Specifically, we did not select monitors that are microscale or middle scale, source oriented, non-EPA (one federal monitor in Yosemite National Park), or those affected by a dominant source, including roadways, in this analysis[4].

[4] This process excluded no monitoring sites; it merely identified those monitors relevant to adjust for a near-roadway approximation. Monitors not selected for adjustment were still included in the overall analysis.

Next, to address the limitations of the monitors' metadata, we conducted a visual inspection and geospatial analysis using Google Earth of the remaining monitors. The analysis reviewed where the site was physically located in an urban area, checked its proximity to major roads (such as interstates, freeways, and major arterial roads), and its proximity to identifiable sources such as industrial complexes and facilities, commercial facilities (such as trucking depots), or proximity to other area sources (such as airports or shipping ports).

Finally, we did not scale up any sites that were not in CBSAs with a population of 350,000 or greater to be consistent with the population based thresholds that trigger minimum required near-road monitors in the NO_2 NAAQS and monitoring package.

Using the list of area-wide monitors appropriate for near-roadway adjustment, we included only those monitors with sufficient data completeness to estimate a 2020 design value (see Section 2.3.1 for details). One hundred seventy-three monitors were considered appropriate for near-road adjustment and eighty-two were considered inappropriate for scaling up. For more details about the monitor selection methodology see Appendix 2a, and for the full list of monitors with criteria see Table 2-3a of Appendix 2a.

2.3.2.2.2 Adjustment methodology

Reflecting scientific literature on the roadway gradient discussed in the final NAAQs rule's preamble (i.e., near road monitors can be from 30% to 100% greater than the area wide monitors), we adjust our estimated design values at area-wide locations for the future year of 2020 by 130%, 165%, and 200% (30%, 65%, and 100% gradients respectively).

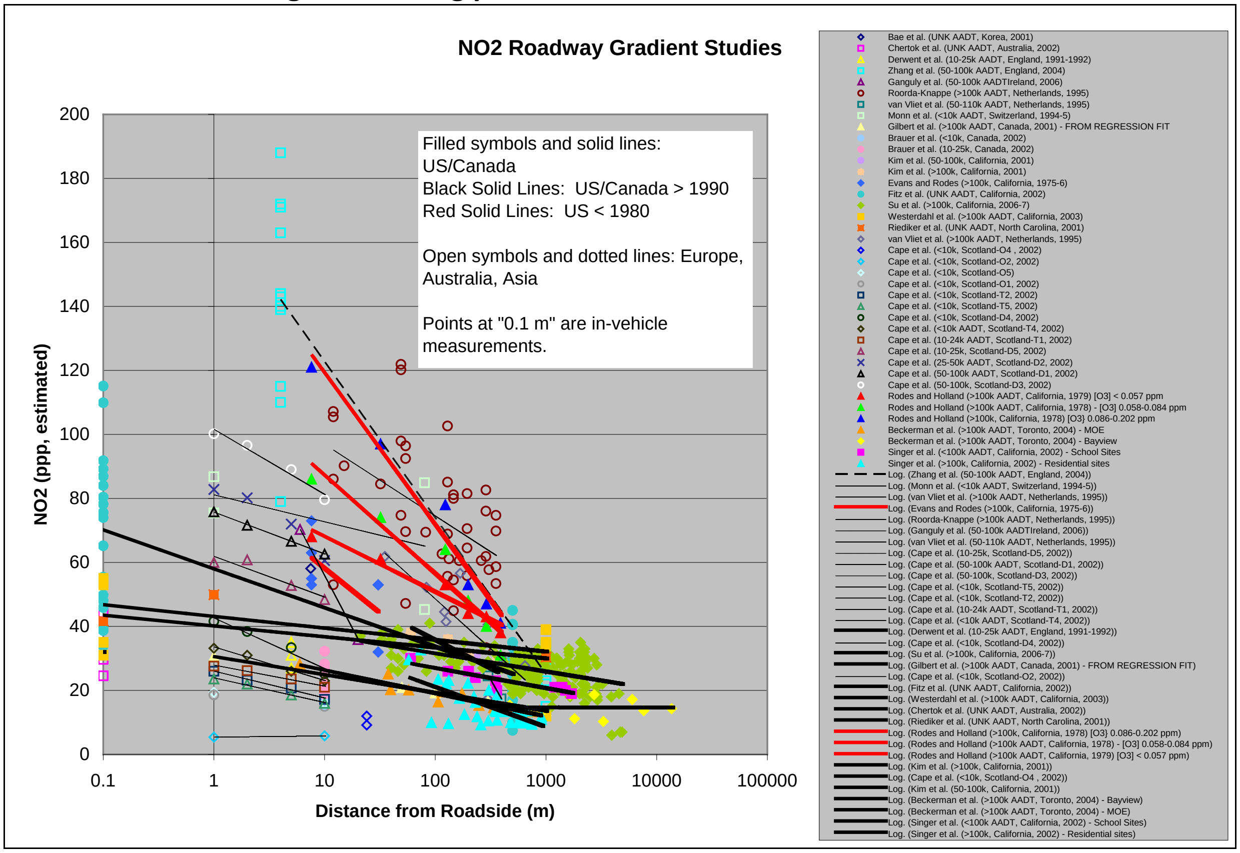
One significant limitation of attempting to approximate near road conditions by simply multiplying by the gradient alone is that the range may not account for the expected future design values near roads (i.e., we believe this approach may over-estimate future design values near roads and may suggest that the future nonattainment problem is worse than it might be, and that the costs and benefits of addressing the residual nonattainment problem in the future are greater than they will actually be). This potential overestimation results from the fact that CMAQ averages the reductions from all sources over the 12km grid which effectively smoothes the concentration changes of source-specific emissions reductions that would have a greater effect at any specific location within the grid, e.g., mobile source emissions reductions near roads. We presume that future near-roadway peaks are reduced more than future area-wide peaks because (1) the near-road proxy monitors are by definition located near the roadway; and (2) on-road mobile source emission reductions between 2006 and 2020 are expected to be significant due to a number of previously-cited Federal mobile source regulations. This suggests that we should consider an appropriate adjustment of the 2020 design values at 'near roadway' proxy monitors to account for the dilution of mobile emission reductions across entire grid squares by CMAQ.

To adjust for the fact that air quality peak design values near roadways will be affected more significantly by mobile source emission reductions than will air quality peak design values in area-wide locations, we start with the design values adjusted to account for the near road gradients described previously and , based on available data, we calculated a relative effectiveness metric for each county reflecting the greater efficacy of mobile source emissions reductions (i.e., ppb/ton) at those locations than predicted by CMAQ for area wide monitor locations. We then apply the resulting national average metric (1.20) across all monitors calculated above to adjust the 2020 design values at the 'near-roadway' proxy monitors consistently.

Reviews of roadway studies indicate that a second adjustment is also reasonable. An analysis of U.S. studies before 1980 and U.S. and Canadian studies after 1990 indicate that the slope of the concentration gradient from the roadway becomes less steep with time (Figure 2-5). The red lines are the concentration gradients for U.S. studies before 1980, while the black lines are concentration gradients for U.S. and Canadian studies after 1990. The black lines are flatter than the red lines, indicating that with time, concentration gradients may decrease. Average NO2 concentrations for US and Canadian studies from 1970-1979 and 2000-2009 for AADT > 100K, show concentration gradient changes from approximately 65% (1970-79) to approximately 30% (2000-2009) for concentrations > 200 m from road when compared to concentrations < 50 m from road. In other words, in the 1970-1979 period, concentrations near the road (50 m) would be 165% higher than concentrations farther from the road (200 m). In 2000-2009, concentrations near the road (50 m) would be 130% higher than concentrations farther from the road (200 m). The difference between the gradients in this context is then approximately 20%. Therefore, the change in gradients with time provides justification for our use of a factor of 1.2 to adjust the required reductions in roadside emissions downward.[5]

[5] That the two adjustment factors have the same value is coincidental.

Figure 2-5: Log plots of NO_2 vs. distance from roadside



While we believe this approach is conceptually sound, it is a new methodology developed out of necessity to complete this assessment for near-roadway monitor locations in the absence of such a monitoring network and based on limited data and modeling results, i.e., information not designed to address near-road situations. Furthermore, the use of a national average adjustment as opposed to a county-specific adjustment makes the adjustment more straight forward but does result in some specific under- and over-adjustments at particular locations.

Following is the methodology to develop the national adjustment factor, 1.20 for the adjustment of the 2020 area-wide design values to near-road design values. The national adjustment factor is based on the use of the 98th percentile design values for 2005-2007 and 2020. The following calculations were performed for monitors that were appropriate for scaling:

1. First we calculated the 2005-2007 ($DV_{on:2005\text{-}2007}$) and 2020 ($DV_{on:2020}$) onroad components of the 2005-2007 and 2020 98th percentile area-wide design values by

multiplying the area-wide design values by the ratio of county onroad ($E_{onroad:2006}$ and $E_{onroad:2020}$) to county total emissions ($E_{total:2006}$ and $E_{total::2020}$) for each scalable monitor:

$$DV_{on:2005-2007} = DV_{2005-2007} \times \frac{E_{onroad:2006}}{E_{total:2006}} \tag{2.9}$$

$$DV_{on:2020} = DV_{2020} \times \frac{E_{onroad:2020}}{E_{total:2020}} \tag{2.10}$$

The county emissions for both 2006 and 2020 are the county emissions used to calculate the 2020 area-wide design values as described in Section 2.3.2.1. The 2020 emissions are the 2020 emissions used to meet the 0.075 ppm ozone standard [See Chapter 4 of the ozone RIA (EPA, 2008)].

2. After calculating the onroad components of the area-wide design values for 2005-2007 and 2020, the onroad ppb/ton estimate, ppb/ton_{onroad}, was calculated as:

$$ppb/ton_{onroad} = \frac{DV_{on:2020} - DV_{on:2005-2007}}{E_{on:2020} - E_{on:2006}} \tag{2.11}$$

3. Next, the ratio of onroad to total ppb/ton metric, $Ratio_{ppb/ton}$ was calculated as:

$$Ratio_{ppb/ton} = \frac{ppb/ton_{onroad}}{ppb/ton_{total}} \tag{2.12}$$

   Where ppb/ton_{onroad} is as defined above and ppb/ton_{total} is defined as in Equation 2.6 of Section 2.3.2.1.

4. Finally, we calculate the national average of $Ratio_{ppb/ton}$ across all monitors appropriate for scale-up as

$$\frac{\sum_{i=1}^{N} Ratio_{ppb/toni}}{N} = 1.2 \tag{2.13}$$

   Where N is the number of monitors appropriate for scale-up

To simplify the analysis, we used the average $Ratio_{ppb/ton}$ in step 4 above across all scalable monitors in the final adjustment for the near-road proxy monitors. The national average ratio was calculated as 1.2, meaning that onroad emissions reductions were approximately 20% more effective at reducing near-roadway concentrations than total emission reductions in the county.

After calculating the national average ratio in step 3, the final near-roadway adjusted 2020 design values were calculated as:

$$DV_{NR:GRAD} = \frac{DV_{2020:} \times GRAD}{1.2} \tag{2.14}$$

Where $DV_{NR2:GRAD}$ is the 2020 near-roadway adjusted concentration for each gradient with GRAD equal to 1.3, 1.65, or 2 (i.e., reflecting 30%, 65%, or 100% increase respectively), and DV_{2020} is the 2020 area-wide design value for the 98th percentile. The 1.2 factor in the denominator is the national average ratio calculated in Equation 2.13. For the eighty-two monitors that were not deemed appropriate for adjustment, the near-road design value was set equal to the 2020 area-wide design value.

Once the near-roadway design values were calculated for 2020 for each of the three gradient increases (30%, 65%, and 100%), residual concentration improvements needed to result in levels below the NAAQS were calculated for three alternative levels of the standard (in ppb): 80, 100, and 125. Nonattainment was calculated as:

$$NA_{GRAD:AS} = DV_{NR:GRAD} - AS \tag{2.15}$$

Where $NA_{GRAD:AS}$ is the residual nonattainment (ppb) for GRAD equal to 30, 65, or 100% increase for alternative standard AS of 80, 100, or 125 ppb and $DV_{NR:GRAD}$ is the 2020 near-roadway adjusted design value for the 30%, 65%, or 100% increase for the 98th percentile. For locations exceeding a particular alternative standard AS, the mobile tons needed to reach attainment are calculated as:

$$Tons_{GRAD:AS} = \frac{NA_{GRAD:AS}}{(ppb / ton \times 1.2)} \tag{2.16}$$

Where $Tons_{GRAD:AS}$ are the tons needed for attainment of alternative standard for the near-roadway increase of 30%, 65%, or 100%, $NA_{GRAD:AS}$ is defined in Equation 2.15 above, and ppb/ton is the total (all county emissions) ppb/ton for the 98th percentile design value as calculated in Equation 2.8. The total ppb/ton is multiplied by 1.20 in Equation 2.16 to approximate the onroad ppb/ton based on the national average ratio of onroad ppb/ton to total ppb/ton calculated in Equation 2.13. While, each monitor had its own value of onroad ppb/ton estimates as calculated in Equation 2.11, in order to maintain consistency with the 1.2 adjustment factor (the ratio of onroad ppb/ton to total ppb/ton), the ppb/ton estimate for each monitor was multiplied by 1.2 to approximate the onroad ppb/ton. For locations below a particular alternative standard, AS, tons for control were not calculated and additional emission controls were not needed.

A complete list of 2020 projected design values by monitor can be found in Table 2-1a of Appendix 2a.

2.4 Results

2.4.1 Nonattainment of alternative standards

Figure 2-6 shows the projected design values for 2020 for the 98th percentile NO_2 design value concentrations for the most extreme case, 100% gradient. Shown are the highest projected design values for each county. Counties in white were below the lowest alternative standard, 80 ppb. It should be noted all of the non-adjusted monitors were below 80 ppb. Table 2-4 shows the number of monitors and counties that exceeded the alternative standards for the three gradient increases.

Figure 2-6: 2020 98th percentile design values for the 100% gradient increase

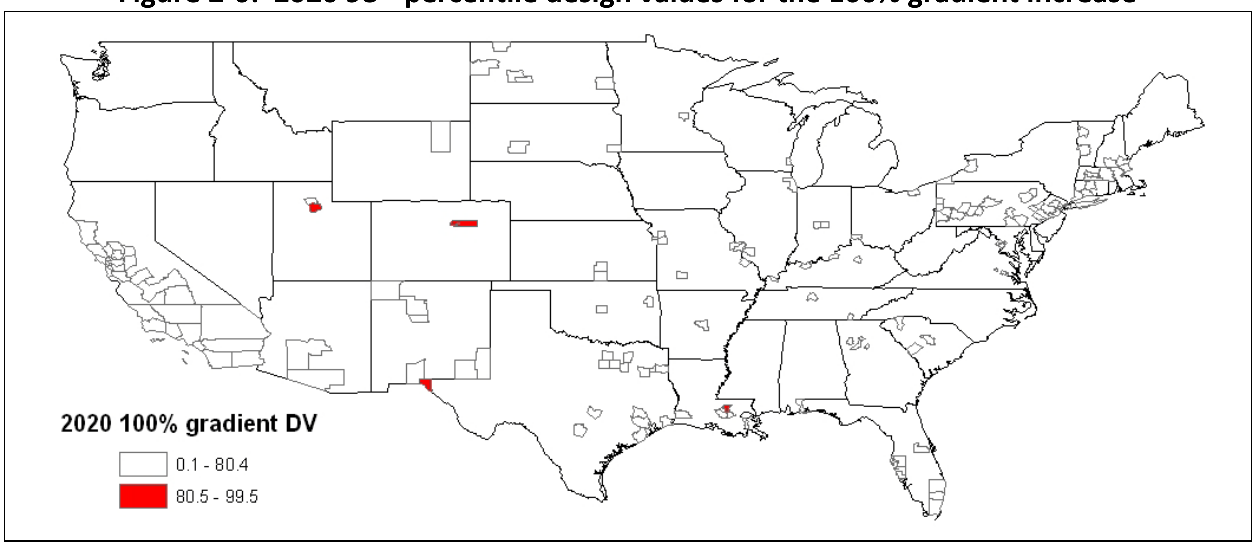


Table 2-4: Summary of 2020 near-road design values exceeding alternative standards for gradient increases

Gradient (%)	Alternative standard	Number of Monitors	Number of Counties
30	80	0	0
	100	0	0
	125	0	0
65	80	1	1
	100	0	0
	125	0	0
100	80	5	4
	100	0	0
	125	0	0

The one county that exceeded 80 ppb for the 65% increase was Adams County, CO with a design value of 82.0 ppb and we estimated a reduction in onroad emissions of 676 tons were needed to attain 80 ppb. The counties that after adjustment for the 100% gradient had NO_2 ambient concentrations projected to be above 80 ppb are shown in Table 2-5.

Table 2-5: Nonattainment counties for 80 ppb for 100% gradient. Onroad mobile tons needed for attainment are also listed

			Tons for control
CO	Adams	99.5	9,861
TX	El Paso	95.8	8,643
UT	Salt Lake	89.0	4,088
LA	East Baton Rouge	80.8	456

While the counties in Table 2-5 were predicted to be in nonattainment in 2020 after adjusting to near-road monitors, there were other sources or events influencing the concentrations before near-road adjustments. In Adams County, CO, the monitor was near a large EGU source and several non-EGU point sources. In El Paso County, TX the monitors were near the international border between the U.S. and Mexico. El Paso is explained in more detail in Section 2.4.2.2. Salt Lake County, UT appeared to be influenced by seasonal inversions, which can lead to higher surface concentrations. In East Baton Rouge, LA, the violating monitor was in the downtown area and located near several non-EGU point sources.

It should be noted that different values of the gradient may be more appropriate for some monitors than other values of the gradient. Many of the monitors may be more influenced by stationary sources than onroad sources or the distance from the roadway may justify the use of a lower gradient. For example, the Charles City County, VA monitor is not located near major roads (within 1 mile), so the 30% gradient may be more appropriate to apply than 65% or 100%. Also, one monitor in Los Angeles County is near the Long Beach Port and Long Beach Municipal Airport. The monitor is located within 500 m of the nearest roadway and most likely already has an influence from the road, so the 30% or 65% gradient may be more appropriate than 100%. However, it should be noted that neither of these monitors exceeded 80 ppb in 2020 when the 100% gradient was applied.

2.4.2 Special cases

After projection of 2005-2007 design values to 2020, some notable results were seen. This section describes the reasons for those values.

2.4.2.1 Non-calculated projected design values

For sixteen monitors (eleven counties), the projected 2020 design values were not calculated for the 98th percentile concentrations (see 2020 concentrations denoted by "*" Table 2a-3 in Appendix 2a). Ten of the counties were in California and one in Pennsylvania. These counties were in regions that were not forecast to meet the 0.070 or 0.075 ozone standard as described in Chapter 4 of the ozone RIA (U.S. EPA, 2008b). These counties received across the board reductions in NOx in addition to the reductions included in the 0.070 ozone analysis. . In the California counties, the 2020_075 emissions were 20% of the 2020_070 emissions, while in Pennsylvania, the 2020_075 emissions were 13% of the 2020_070 emissions. For more details about the emissions reduction see Chapter 4 of the ozone RIA (U.S. EPA, 2008b). Concentrations could not be calculated because 2020_075 emissions were so low that the methodology described in Section 2.3.2.1 did not produce reasonable results. All of the monitors in question were already below the lowest alternative standard of 80 ppb in 2005-2007, so these monitors should not have issues with nonattainment.

2.4.2.2 El Paso County

El Paso County represents a case where future design values for NO_2 above the levels being considered are influenced by international emissions. The 2005-2007 98th percentile design values are shown in Figure 2-7. The three monitors in the black circle were the highest monitors. The 2020 98th percentile design values are shown in Figure 2-8. Area-wide and near-road adjusted projected design values are shown. One monitor was not considered appropriate for adjustment and has no near-road design value listed.

Figure 2-7: 2005-2007 98th percentile design values (ppb) for El Paso County

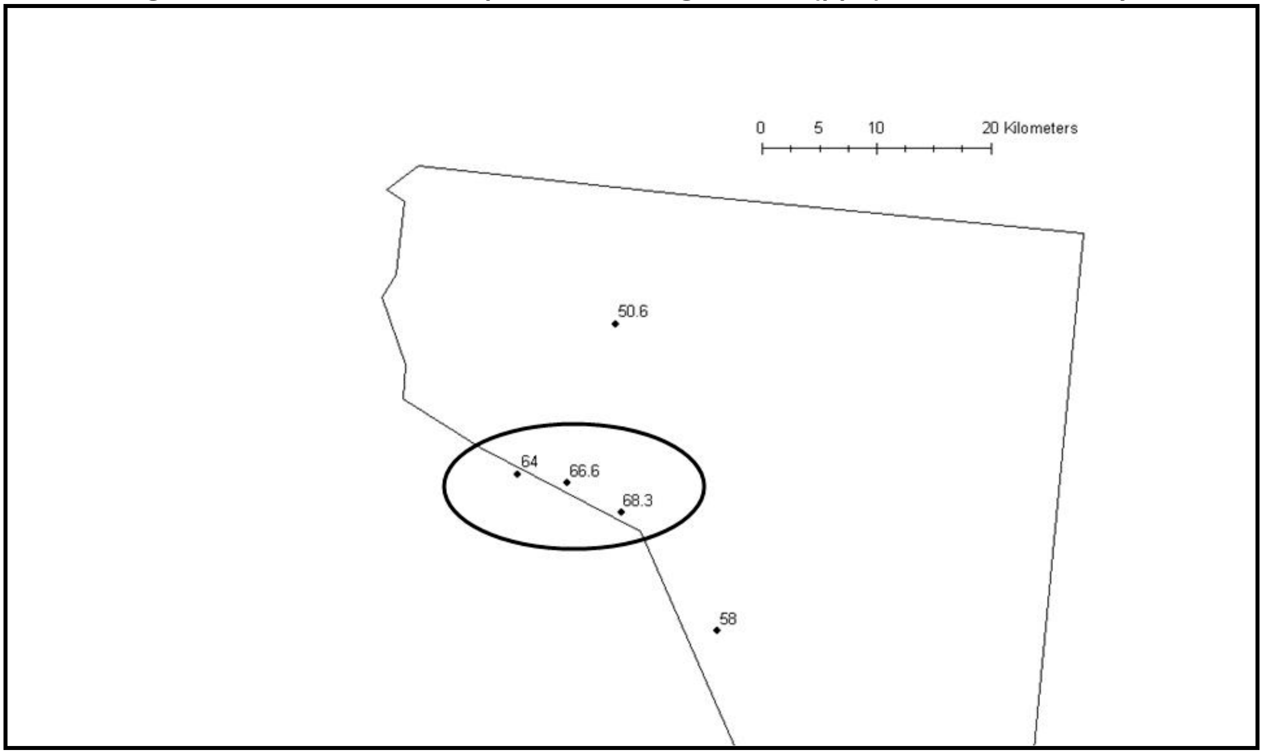
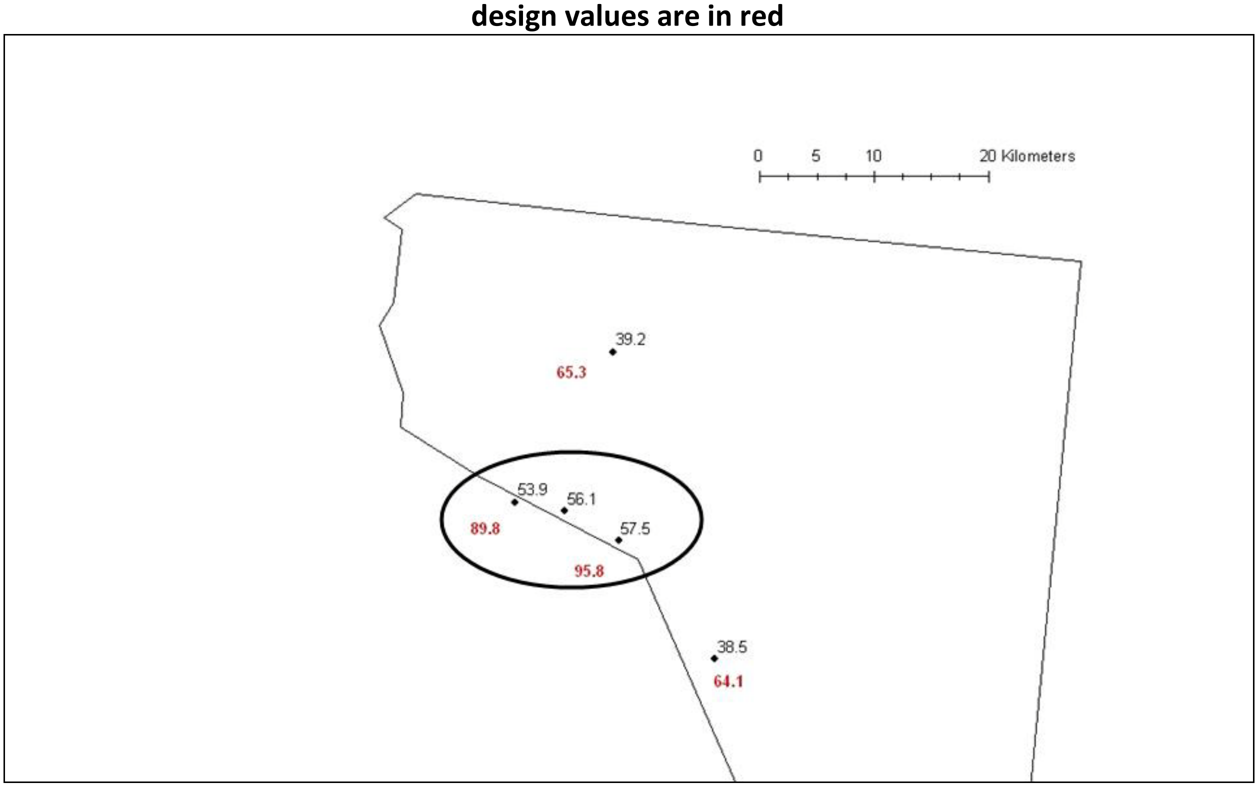
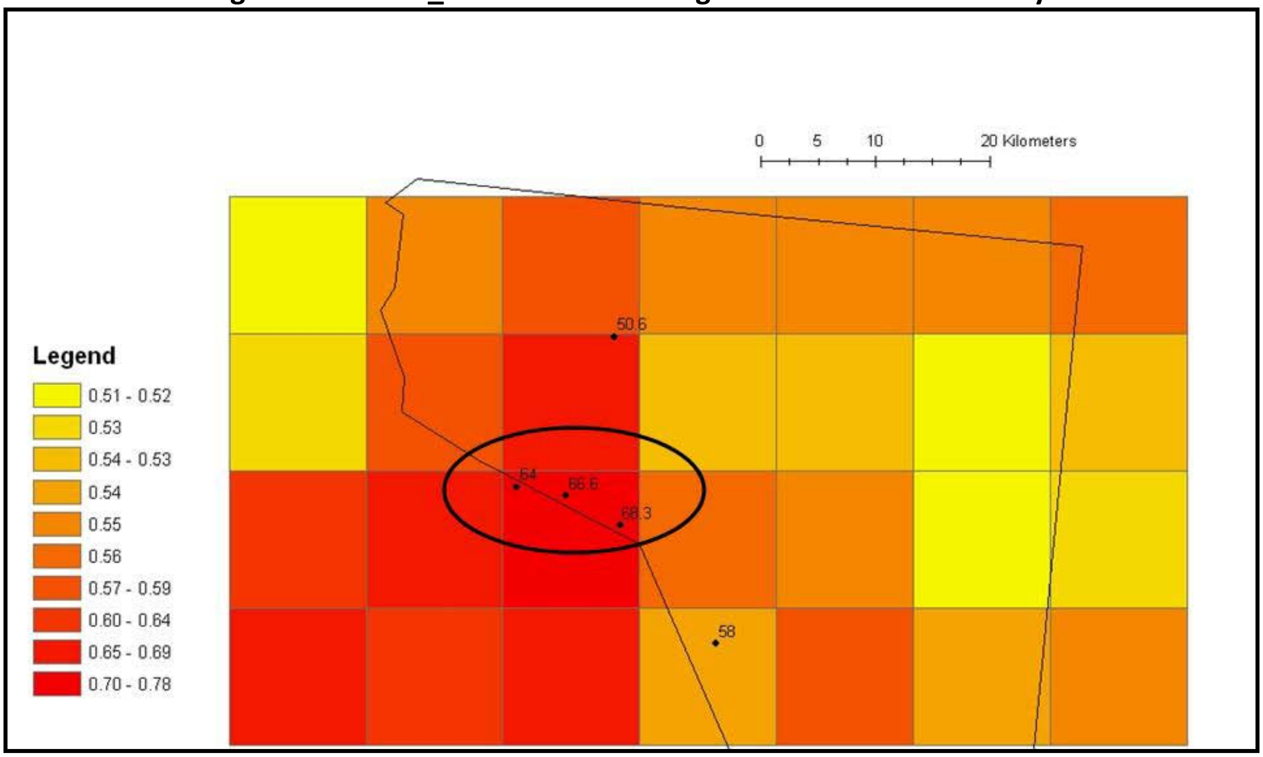


Figure 2-8: 2020 98th percentile design values (ppb) for El Paso County. Area-wide design values are in black and for monitors that were scalable, 100% gradient adjusted near-road design values are in red



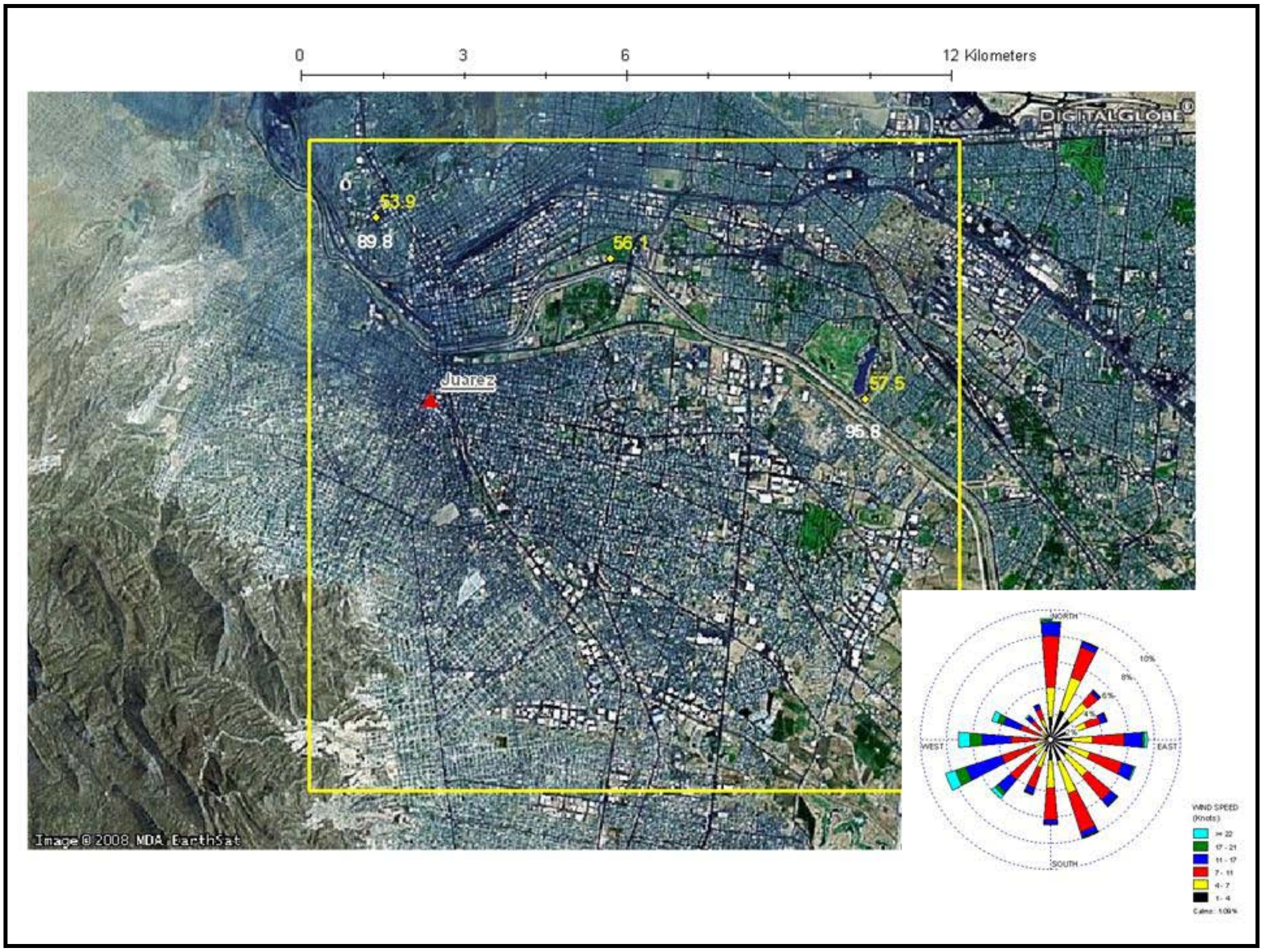
In 2020, two of the near-road design values of near-road adjusted monitors exceeded 80 ppb for the 100% gradient adjustment, 89.8 ppb and 95.8 ppb (Figure 2-8). Examining the average of the top ten daily 1-hour maximum concentrations for 2002 and the average of the daily 1-hour maximum concentrations for the same ten calendar days in 2020 showed that the grid cell containing the top two nonattainment monitors was the highest value among the grid cells in the county containing monitors, 65.6 ppb for 2002 and 51.3 for 2020 (not shown). The resulting RRF was also the highest, 0.78 (Figure 2-9) and the mean daily 1-hour maximum concentration in 2020 was also highest for the county, 31.3 ppb.

Figure 2-9: 2020_070 RRF values for grid cells in El Paso County



Note that these monitors were not only located along the border highway, but they were also very close to the international border with city of Juarez just to the southwest (Figure 2-10). A wind rose from El Paso Airport for 2005-2007 exhibited a relatively high frequency of winds from the east-southeast through west-southwest that would transport pollutants from Juarez toward the three NO_2 monitoring sites across the river in El Paso. The grid cell that contained the two highest monitors is mostly in Mexico. Emissions from across the international border could impact the modeled concentrations of the grid cells containing the monitors. However, for our emission inventories, we do not forecast controls on international emissions over which we have no jurisdiction.

Figure 2-10: Aerial photograph of CMAQ grid cell containing nonattainment monitors for El Paso County. Yellow box is 12 x 12 km grid cell and El Paso 2005-2007 wind rose is shown in lower right corner. Area-wide design values are in yellow and near-road adjusted design values are in white



In summary:

- Two monitors in El Paso County were the highest monitors in the 2005-2007 and 2020 98th percentile design values in the county.
- The grid cell containing the monitors had the highest average of the top 10 daily 1-hour maximum concentrations for 2002 for grid cells containing monitors in El Paso County.
- Also, the monitors' grid cell had the highest average of the 2020_070 daily 1-hour maximum concentrations for the same days as the ten days in the average of the 2002 daily 1-hour maximum concentrations.
- The monitors' grid cell had the highest RRF value for all monitor grid cells in the county.
- Since all of the monitors in the county used the same 2002, 2006, 2020_070, and 2020_075 emissions for emissions RRF calculations (Equations 3.2 and 3.3), the driving factor was the high RRF for the grid cell.

- The grid cell contained international emissions and were not controlled in the 2020_070 inventory, resulting in higher daily 1-hour maximum concentrations when compared to other monitor grid cells.

2.5 Summary

In summary, 2020 NO_2 design value concentrations were projected from 2005-2007 observed design values using CMAQ output from the 2002 and the 2020_070 scenario simulations performed for the ozone NAAQS RIA (U.S. EPA, 2008b). County emissions for 2002, 2006, and 2020 were used in conjunction with the CMAQ output to project the 2005-2007 design values for the 2020 area-wide design values. The 2020 area-wide design values from appropriate monitors were then adjusted to (1) reflect a near-roadway network of monitors using gradient increases of 30%, 65%, and 100%; and (2) to reflect the efficacy of controls on onroad mobile emissions in the future. The 2020 near-roadway concentrations were then compared against three alternative standards of 80, 100, and 125 ppb for each of the three gradient increases. No counties exceeded 80 ppb for the 30% gradient, one county exceeded 80 ppb for the 65% gradient, and four counties exceeded 80ppb, for the 100% gradient. No counties exceeded 100 ppb for any of the three gradients.

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