

Regulatory Impact Analysis: Renewable Fuel Standard Program

Chapter 5 Air Quality Impacts

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Chapter 5: Air Quality Impacts

5.1 Ozone

We performed ozone air quality modeling simulations for the eastern United States using the ozone Response Surface Model (RSM) to estimate the effects of the projected changes in emissions from gasoline vehicles and equipment. The ozone RSM is a screening-level air quality modeling tool that allows users to quickly assess the estimated air quality changes over the modeling domain. The ozone RSM is a model of a full-scale air quality model and is based on statistical relationships between model inputs and outputs obtained from the full-scale air quality model. In other words, the ozone RSM uses statistical techniques to relate a response variable to a set of factors that are of interest, e.g., emissions of precursor pollutants from particular sources and locations. The following section describes the modeling methodology, including the development of the multi-dimensional experimental design for control strategies and implementation and verification of the RSM technique. Additional detail is available in the Air Quality Modeling Technical Support Document (AQMTSD) that was drafted for the Mobile Source Air Toxics Rule Proposal (published March 29, 2006).^{AAAA}

The foundation for the ozone response surface metamodeling analyses was the CAMx modeling done in support of the final Clean Air Interstate Rule (CAIR). The CAIR modeling is fully described in the CAIR Air Quality Modeling Technical Support Document, but a brief description is provided below.^{BBBB} The modeling procedures used in the CAIR analysis (e.g., domain, episodes, meteorology) have been used for several EPA rulemaking analyses over the past five years and are well-established at this point.

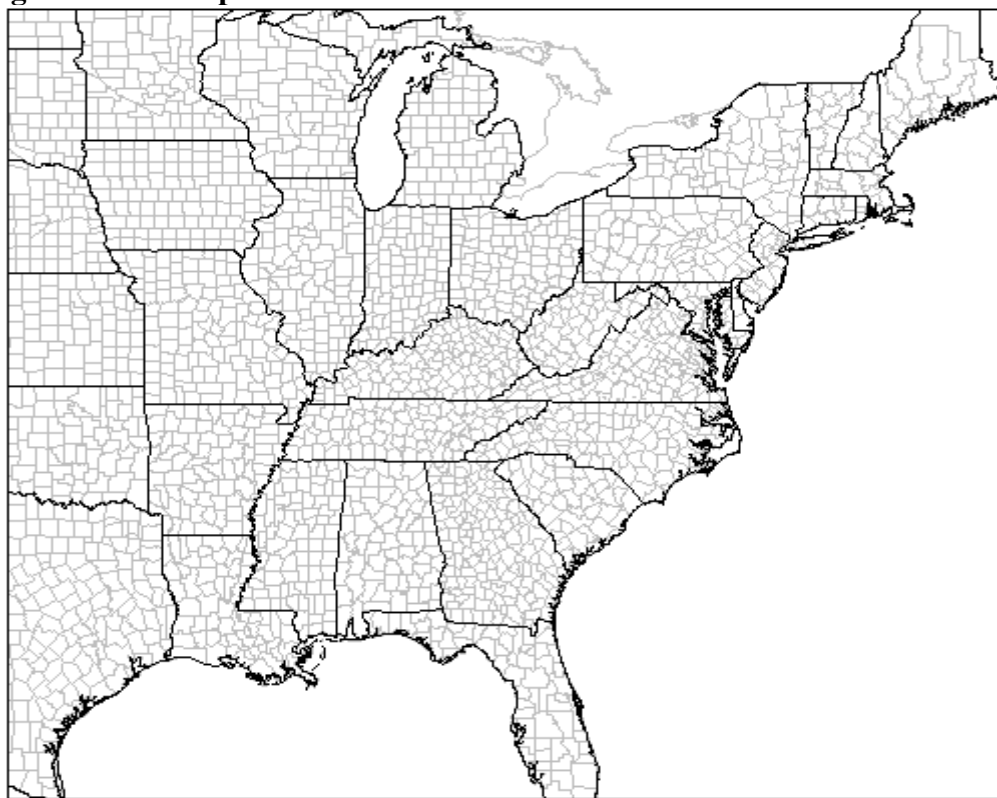
The ozone RSM uses the 2015 controlled CAIR emissions inventory as its baseline, assuming future fuel quality remains unchanged from pre-Act levels, which serves as the baseline for the analysis of the final RFS standards.^{CCCC} We then compare these baseline emissions to the emissions which would have occurred in the future if fuel quality had remained unchanged from pre-Act levels to those which will occur with fuel quality reflecting the increased renewable fuel use projected in the future. This approach differs from that traditionally taken in EPA regulatory impact analyses. Traditionally, we would have compared future emissions with and without the requirement of the Act. However, as described in Chapter 1, we expect that total renewable fuel use in the U.S. in 2012 to exceed 7.5 billion gallons even in the absence of the Renewable Fuel Standard (RFS). Thus, a traditional regulatory impact analysis would have shown no impact on emissions or air quality.

The modeling simulations that comprised the metamodeling were conducted using CAMx version 3.10. It should be noted that because the ozone RSM is built from CAMx air quality model runs, it therefore has the same strengths and limitations of the underlying model and its inputs. CAMx is a non-proprietary computer model that simulates the formation and fate of photochemical oxidants including ozone for given input sets of meteorological conditions and emissions. The gridded meteorological data for three historical episodes were developed using the Regional Atmospheric Modeling System (RAMS), version 3b.^{DDDD} In all, 30 episode days were modeled using frequently-occurring, ozone-conducive, meteorological conditions from the

summer of 1995. Emissions estimates were developed for the evaluation year (1995) as well as a future year (2015).

The CAMx model applications were performed for a domain covering all, or portions of, 37 States (and the District of Columbia) in the Eastern U.S., as shown in Figure 5.1-1. The domain has nested horizontal grids of 36 km and 12 km. However, the output data from the metamodeling is provided at a 12 km resolution (i.e., cells from the outer 36 km cells populate the nine finer scale cells, as appropriate). Although the domain of the ozone RSM is the 37 Eastern states, the expanded use of ethanol in fuel is expected to occur nationwide. Chapter 4 describes the nationwide inventory impacts associated with the proposed standards.

Figure 5.1-1. Map of the CAMx Domain Used for RFS Ozone Metamodeling



The ozone RSM used for assessing the air quality impacts of expanded ethanol use in fuel was developed broadly to look at various control strategies with respect to attaining the 8-hour ozone NAAQS. The experimental design for the ozone RSM covered three key areas: type of precursor emission (NO_x or VOC), emission source type (i.e., onroad vehicles, nonroad vehicles, area sources, electrical generating utility (EGU) sources, and non-utility point sources), and location in or out of a 2015 model-projected residual ozone nonattainment area. This resulted in a set of 14 emissions factors.

The 14 emission factors were randomly varied and used as inputs to CAMx. The experimental design for these 14 factors was developed using a Maximin Latin Hypercube method. Based on a rule of thumb of 10 runs per factor, we developed an overall design with

140 runs (a base case plus 139 control runs). The range of emissions reductions considered within the metamodel ranged from 0 to 120 percent of the 2015 CAIR emissions. This experimental design resulted in a set of CAMx simulations that serve as the inputs to the ozone response surface metamodel.

To develop a response surface approximation to CAMx, we used a multidimensional kriging approach, implemented through the MIXED procedure in SAS. We modeled the predicted changes in ozone in each CAMx grid cell as a function of the weighted average of the modeled responses in the experimental design. A response-surface was then fit for several ozone metrics, namely the ozone design value, the 1-hour maximum value, the 24-hour average value and the average ozone level between 9 am and 5 pm. The effect of changes in VOC and NOx emissions on ozone was estimated in each grid cell covered by the model for each ozone metric except the ozone design value. The ozone design value is the mathematically determined pollutant concentration at a particular monitoring site that must be reduced to, or maintained at or below the National Ambient Air Quality Standard to assume attainment. The 8-hour ozone design value is the 3-year average of the fourth-highest daily maximum 8-hour average ozone concentrations measured at each monitor within an area over each year, which must not exceed 0.08 ppm (85 ppb, considering round-off). Thus, ozone design values only exist for grid cells which contain ozone monitoring stations and where ozone attainment has been an issue. Ozone design values have been developed for 525 of the 2696 counties in the 37 state region. of the 31 these The specific ozone design values used in this analysis are those for 2001, which represent the average of the ozone design values determined for three, three-year periods (1999-2001, 2000-2002, and 2001-2003). Validation was performed and is summarized in the Mobile Source Air Toxics rule Air Quality Modeling Technical Support Document. The validation exercises indicated that the ozone RSM replicates CAMx response to emissions changes very well for most emissions combinations and in most locations.

The ozone RSM limits the number of geographically distinct changes in VOC and NOx emissions which can be simulated. Emissions from motor vehicles and nonroad equipment can be varied separately. Distinct percentage changes in either the motor vehicle or nonroad inventories can also be applied in ozone nonattainment and attainment areas. However, distinct emission impacts cannot be simulated in various ozone nonattainment areas (e.g., Chicago and Houston or New York and Kansas City). This limits our ability to simulate the impact of increased ethanol use in a couple of ways. First, ethanol use is not geographically uniform across the U.S., either currently or in the future. Thus, the emission impacts resulting from changes in ethanol use also varies geographically. Second, the emission impacts of ethanol use are not uniform. Ethanol use in RFG and other areas which do not grant ethanol blends a 1.0 psi RVP waiver will not experience as much of an increase in VOC emissions with increased ethanol use as areas which grant ethanol blends an RVP waiver. Third, the impacts of new ethanol plants will be even more geographically focused. The Ozone RSM cannot generally be applied to model the emission impacts from such local sources for a couple of reasons. One, the location of new ethanol plants is difficult to predict in many cases. Two, the impact of these plants on local emissions can be very large in percentage terms given the absence of a lot of other industrial activity. The Ozone RSM was designed to represent the ozone impact of the same change in VOC or NOx emissions across a broad region (e.g., all attainment areas). Therefore, it cannot be used to model the impact of a large change in one county's emissions

without also assuming the same change in the upwind county's emissions. As not every county will contain a new ethanol plant, the assumptions inherent in the Ozone RSM do not match the situation of a new individual point source, such as an ethanol plant.

We developed a methodology which would best approximate the impact of changes in local emissions on the ozone level in each local area, while maintaining as much of the impact of ozone transport from other areas as possible given the above mentioned limitations. We do this by running the ozone RSM twice for each scenario and drawing the resultant ozone impact from the run which best matched the emission impact expected in a particular local area, considering both the change in emissions modeled for that particular local area, as well as that occurring in upwind areas.

First, as mentioned above, ethanol use is expected to change dramatically in some areas, but not at all in others. Averaging the emission impacts across these two types of areas and estimating the associated ozone impact would be very misleading. No area would be likely to experience the ozone impact predicted. Some areas would experience a much greater impact, while others would experience no impact. Therefore, the first step in using the Ozone RSM to predict the ozone impacts related to the RFS is to estimate the change in VOC and NO_x emissions in those areas ethanol blend market share changed significantly. As was done in the analysis of local emission impacts presented in Section 4.1.3.3 above, we defined a significant change in ethanol blend market share as a change of 50% or more. This focuses the change in emissions in those areas where the change is likely to occur.

As discussed in Chapters 3 and 4, the effect of ethanol use on emissions differs depending on the baseline fuel quality and the applicable RVP standards. In particular, ethanol use has significantly different impacts on emissions in RFG, low RVP and 9 RVP areas. Therefore, in order to better predict the ozone impact likely to occur in specific areas, we estimate the change in VOC and NO_x emissions separately for RFG, low RVP and 9 RVP areas (per above, only for those areas in each case where ethanol blend market share changed by 50% or more).

The Ozone RSM only covers the 37 easternmost states in the U.S. Therefore, we limited the calculation of VOC and NO_x emission impacts to only those states. The Ozone RSM was developed with the year 2015 as the default year. Since we develop most of our impacts of the RFS for the year 2012 and 2015, we chose to run the Ozone RSM for, 2015. The Ozone RSM is designed to accept emission changes in terms of total onroad and total nonroad sources, respectively, and both emission categories include diesels. Therefore, we included estimates of VOC and NO_x emissions from diesel vehicles and equipment in 2015 in our calculation of the emission impacts. These diesel emissions do not change between the various RFS scenarios. However, they do reduce the effective percentage change in VOC and NO_x emissions which is projected to occur. Overall, these analyses are very similar to those described in Section 4.1.3.3 above, with the exceptions of the limitation to 37 states and the inclusion of diesel emissions. The results of these calculations are shown in Table 5.1-1.

**Table 5.1-1.
Emission Inputs to Ozone Modeling: Change in Total Mobile Sources Emissions in 37
Eastern States where Ethanol Use Changes Significantly, July 2015 (percent change)**

		VOC				NOx			
		On Road		Non Road		On Road		Non Road	
		Attain. (9 RVP)	Non- Attain.	Attain. (9 RVP)	Non- Attain.	Attain. (9 RVP)	Non- Attain.	Attain. (9 RVP)	Non- Attain.
Primary Analysis									
RFS	RFG	7.5%	-1.2%	1.5%	1.8%	0.2%	0.1%	3.0%	-1.9%
	LRVP	7.5%	8.9%	1.5%	1.7%	0.2%	0.2%	3.0%	3.3%
EIA	RFG	8.2%	1.7%	2.3%	2.4%	0.2%	0.1%	3.2%	0.7%
	LRVP	8.2%	9.3%	2.3%	2.0%	0.2%	0.2%	3.2%	3.3%
Sensitivity Analysis									
RFS	RFG	6.0%	-5.3%	1.5%	1.8%	2.9%	1.7%	3.0%	-1.9%
	LRVP	6.0%	7.0%	1.5%	1.7%	2.9%	3.5%	3.0%	3.3%
EIA	RFG	6.6%	-1.2%	2.3%	2.4%	3.0%	2.2%	3.2%	0.7%
	LRVP	6.6%	7.4%	2.3%	2.0%	3.0%	3.6%	3.2%	3.3%

Our category of 9 RVP areas is very similar to the set of attainment areas in the Ozone RSM. Therefore, the application of the emission impacts expected in 9 RVP areas in the Ozone RSM was straightforward. However, both RFG and low RVP areas together generally comprise the set of nonattainment areas in the Ozone RSM. As seen in Table 5.1-1, the expected emission impacts of the various RFS scenarios differ significantly depending on whether the area has RFG or low RVP fuel. Both sets of emission impacts could not be run in the Ozone RSM at the same time. Therefore, we ran the Ozone RSM twice. The first run applied the emission impacts estimated for RFG areas to the ozone nonattainment areas in the Ozone RSM and applied the emission impacts for 9.0 RVP areas to the ozone attainment areas in the Ozone RSM. This run should produce satisfactory projections of ozone impacts for all areas except those areas with low RVP, as well as those areas where ethanol use is not expected to change.

The second run applied the emission impacts estimated for Low RVP areas to the ozone nonattainment areas in the Ozone RSM and applied the emission impacts for 9.0 RVP areas to the ozone attainment areas in the Ozone RSM. This run should produce satisfactory projections of ozone impacts for all areas except those areas with RFG, as well as those areas where ethanol use is not expected to change.

For both runs of the Ozone RSM, we set the predicted change in ozone to zero in those counties not expected to experience a significant change in ethanol use. This ignores any impact from ozone transport from other areas where ethanol use did change. However, we believe that the ozone impacts due to transport are much smaller than those associated with changes in local emissions. This is particularly true in this case, where the percentage change in emissions would be the same in both the local and upwind areas.

We merged the results of the two runs by attributing ozone impacts from each county according to their nonattainment designation and fuel type. For non-attainment areas, this choice was obvious. Non-attainment counties with RFG programs were assigned the ozone impacts from the first run (i.e., the run where the changes in VOC and NO_x emissions were the average of those observed for RFG areas). Non-attainment counties with Low RVP programs were assigned the ozone impacts from the second run (i.e., the run where the changes in VOC and NO_x emissions were the average of those observed for Low RVP areas). For attainment areas (i.e., 9 RVP areas), the results of either run could be used, as both runs of the Ozone RSM applied the same emission changes to attainment areas. Thus, the local emission impacts would be identical in the two Ozone RSM runs. Ozone transport is also likely identical for the vast majority of these counties, given that they are likely downwind from other attainment area counties. The only difference occurs if an attainment area is downwind of a RFG or Low RVP area. For a nationwide analysis such as this one, we were not able to determine for each attainment area whether a potential upwind area was more likely to be an RFG or Low RVP area. Therefore, we chose to use the ozone impacts results from the first Ozone RSM run of the model (i.e., where the emission impacts for RFG areas were applied to ozone nonattainment areas) for all attainment areas. We chose this run because RFG areas tend to have the higher ozone levels than Low RVP areas and thus, would be more likely to affect areas downwind. We present the ozone impacts of increased ethanol use resulting from this methodology in the following section.

5.1.1 Ozone Response Surface Metamodel Results

This section summarizes the results of our modeling of ozone air quality impacts in the future with and without the expanded use of ethanol in fuel. The impact of increased ethanol use on the 8-hour ozone design values in 2015 are presented in Table 5.1-2. The changes presented in Table 5.1-2 are for those counties with 2001 modeled design values.⁵⁷ The Chapter 5 Appendix presents the impacts of increased ethanol use on a number of alternative measures of ambient ozone concentration.

⁵⁷ 2001 design values were calculated as an average of the 1999-2001, 2000-2002 and 2001-2003 design values at each monitoring site. Please see the Air Quality Modeling Technical Support Document for the final Clean Air Interstate Rule for additional information.

**Table 5.1-2.
Impact of Increased Ethanol Use on 8-hour Ozone Design Values in 2015 (ppb)**

	RFS Case	EIA Case
Primary Analysis		
Minimum Change	-0.015	0.000
Maximum Change	0.329	0.337
Average Change Across 37 States	0.057	0.079
Population-Weighted Change Across 37 States	0.052	0.056
Average Change Where Ethanol Use Changed Significantly States	0.153	0.181
Population-Weighted Change Where Ethanol Use Changed Significantly States	0.154	0.183
Sensitivity Analysis		
Minimum Change	-0.115	0.000
Maximum Change	0.624	0.549
Average Change Across 37 States	0.111	0.142
Population-Weighted Change Across 37 States	0.092	0.096
Average Change Where Ethanol Use Changed Significantly States	0.300	0.325
Population-Weighted Change Where Ethanol Use Changed Significantly States	0.272	0.315

As can be seen, ozone levels generally increase with increased ethanol use. This is likely due to the projected increases in both VOC and NO_x emissions. Some areas do see a small decrease in ozone levels. In our primary analysis, where exhaust emissions from Tier 1 and later onroad vehicles are assumed to be unaffected by ethanol use, the population-weighted increase in ambient ozone levels is 0.052-0.056 ppb. Since the 8-hour ambient ozone standard is 0.08 ppm (85 ppb), this increase represents about 0.06 percent of the standard, a very small percentage⁵⁸. While small, this figure includes essentially zero changes in ozone in areas where ethanol use did not change. When we focus just on those areas where the market share of ethanol blends changed by 50 percent or more, the population-weighted increase in ambient ozone levels rises to 0.154-0.183 ppb. This increase represents about 0.2 percent of the standard.

In our sensitivity analysis, where exhaust emissions from Tier 1 and later onroad vehicles are assumed to respond to ethanol like Tier 0 vehicles, the population-weighted increase in ambient ozone levels across the entire 37 state area is slightly less than twice as high, or 0.092-0.096 ppb. This increase represents about 0.11 percent of the standard. When we focus just on those areas where the market share of ethanol blends changed by 50 percent or more, the population-weighted increase in ambient ozone levels rises to 0.272-0.315 ppb. This increase represents about 0.35 percent of the standard.

For the primary analysis, we also present the counties with the largest increases in the ozone design value. Table 5.1-3 presents the county level ozone design value impacts of the RFS case, while Table 5.1-4 presents the same information for the EIA case. It is important to note that the results of this ozone response surface metamodeling exercise is meant for screening-level purposes only and does not represent the results that would be obtained from

⁵⁸ Appendix I of 40 CFR Part 50.

full-scale photochemical ozone modeling. It is also important to note that the ozone RSM results indicate that the counties which are projected to experience the greatest increase in ozone design values are generally counties that are projected to have ambient concentrations well below the 0.08 ppm ozone standard in the 2015 baseline.

**Table 5.1-3. RFS Case, Primary Analysis:
2015 Ozone Response Surface Metamodeling Results^a for Counties with
Largest Increases in Ozone 8hr Design Value (ppb) Due to Increased Use of Ethanol**

State Name	County Name	2015 Baseline (Post-CAIR) ^b	2015 RFS Case	Effect of Expanded Ethanol Use (ppb)	2015 Population
Arkansas	Crittenden Co	78	78.3289	0.3	53,852
Ohio	Geauga Co	82.5	82.7972	0.3	108,600
Ohio	Mahoning Co	74.7	74.9943	0.3	248,545
Indiana	Lake Co	80.7	80.9929	0.3	490,796
Ohio	Medina Co	72	72.2909	0.3	187,686
Ohio	Summit Co	77.4	77.6901	0.3	557,892
Maine	York Co	77.6	77.8825	0.3	210,006
Ohio	Stark Co	71.7	71.9707	0.3	384,672
Ohio	Clinton Co	75.7	75.9705	0.3	50,635
Louisiana	West Baton Rouge Parish	77.4	77.6685	0.3	23,202
Louisiana	Livingston Parish	76.6	76.8656	0.3	141,807
Illinois	Cook Co	81.1	81.3605	0.3	5,362,932
Indiana	Shelby Co	76.2	76.4587	0.3	47,904
Ohio	Knox Co	71.4	71.6541	0.3	62,138
Alabama	Mobile Co	68	68.2514	0.3	430,341
Louisiana	Jefferson Parish	77.1	77.351	0.3	512,963

^a The Ozone RSM is meant for screening-level purposes only and does not represent results that would be obtained from full-scale photochemical ozone modeling. In particular, the model does not account for changes in CO emissions or VOC reactivity, both of which should decrease with increased ethanol use and directionally reduce ozone. in areas where ozone formation is VOC-limited.

^b The Clean Air Interstate Rule (CAIR) modeling is fully described in the CAIR Air Quality Modeling Technical Support Document (Docket EPA-HQ-OAR-2005-0036).

**Table 5.1-4. EIA Case, Primary Analysis:
2015 Ozone Response Surface Metamodeling Results for Counties with
Largest Increases in Ozone 8hr Design Value (ppb) Due to Increased Use of Ethanol**

State Name	County Name	2015 Baseline (Post-CAIR) ^c	2015 EIA Case	Effect of Expanded Ethanol Use (ppb)	2015 Population
Ohio	Geauga Co	82.5	82.8369	0.3	108,600
Ohio	Clinton Co	75.7	76.0218	0.3	50,635
Ohio	Mahoning Co	74.7	75.0213	0.3	248,545
Arkansas	Crittenden Co	78	78.3204	0.3	53,852
Ohio	Summit Co	77.4	77.7175	0.3	557,892
Mississippi	Adams Co	67.2	67.5164	0.3	33,495
Ohio	Stark Co	71.7	72.0153	0.3	384,672
Indiana	Shelby Co	76.2	76.5115	0.3	47,904
Maine	York Co	77.6	77.902	0.3	210,006
New York	Wayne Co	71.6	71.8926	0.3	103,846
Texas	Travis Co	69.4	69.6912	0.3	1,022,772
Ohio	Medina Co	72	72.2909	0.3	187,686
Maine	Hancock Co	76.8	77.0904	0.3	55,606
Maine	Kennebec Co	64.9	65.1903	0.3	122,363
Louisiana	Livingston Parish	76.6	76.8883	0.3	141,807
Louisiana	West Baton Rouge Parish	77.4	77.6869	0.3	23,202
Louisiana	Lafourche Parish	72.7	72.984	0.3	95,881
Mississippi	Warren Co	56.2	56.4827	0.3	52,075
Michigan	Huron Co	71.9	72.1766	0.3	37,530
Ohio	Franklin Co	77	77.2716	0.3	1,181,578
Ohio	Trumbull Co	80	80.2713	0.3	227,546
Louisiana	Jefferson Parish	77.1	77.3707	0.3	512,963
Florida	Pinellas Co	62.3	62.5639	0.3	998,864
Ohio	Delaware Co	72.1	72.3606	0.3	149,341
Ohio	Knox Co	71.4	71.6579	0.3	62,138
Florida	Duval Co	50.6	50.8568	0.3	895,678
Indiana	Marion Co	74.6	74.8565	0.3	889,645
Indiana	Madison Co	72.9	73.1564	0.3	140,575
Massachusetts	Middlesex Co	75.8	76.0564	0.3	1,498,849
Michigan	Oakland Co	79.2	79.4542	0.3	1,355,671
Pennsylvania	Beaver Co	70.5	70.7528	0.3	184,649
New York	Monroe Co	74.3	74.5521	0.3	754,405
Mississippi	Harrison Co	69.3	69.5517	0.3	216,926

There are a number of important caveats concerning our estimated ozone impacts using the Ozone RSM. The Ozone RSM does not account for changes in CO emissions. As shown in Chapter 4, ethanol use should reduce CO emissions significantly, directionally reducing ambient ozone levels in areas where ozone formation is VOC-limited. Accounting for the reduction in CO emissions in NO_x-limited areas, however, may have little impact on the ozone impact of ethanol use.

The Ozone RSM also does not account for changes in VOC reactivity. With additional ethanol use, the ethanol content of VOC should increase. Ethanol is less reactive than the average VOC. Therefore, this change should also reduce ambient ozone levels in a way not addressed by the Ozone RSM. Again, like the impact of reduced CO emissions, this effect applies to those areas where ozone formation is VOC-limited. Another limitation is the RSM's inability to simulate the spatial distribution of emission impacts associated with the proposed standard. Instead, we are forced to make simplifying assumptions about the geographic uniformity of RFS emissions impacts, explained above. The caveats and limitations associated with the RSM highlight the fact that it should only be used as a screening-level tool to characterize broad trends associated with changes in different source categories of ozone precursors.

Finally, our application of the Ozone RSM here does not include the impact of emissions from new ethanol plants. Directionally, this will increase ozone levels in the vicinity of the new plant. As discussed in Chapter 4, the overall VOC and NO_x emission impacts of new ethanol plants are only slightly lower than the emission impacts resulting from increased use of ethanol in vehicles and equipment. Given the concentrated nature of these impacts, the ozone impacts of these new plants should be a focus of further study in the future.

Keeping these limitations in mind, the expanded use of ethanol will impact the national emissions inventory of precursors to ozone, such as VOCs and NO_x, as described in Chapter 4. Exposure to ozone has been linked to a variety of respiratory effects including premature mortality, hospital admissions and illnesses resulting in school absences. Ozone can also adversely affect the agricultural and forestry sectors by decreasing yields of crops and forests.

The Wisconsin Department of Natural Resources (DNR) recently performed a similar study of the impact of increased ethanol use on ozone.^{EEEE} They estimated that the conversion of gasoline outside of RFG areas in Wisconsin to E10 blends would increase ozone in these areas on the order of 1 ppb to as much as 2 ppb. (RFG areas in Wisconsin already contain 10 vol% ethanol.) This ozone increase was due to the predicted increase in NO_x emissions associated with ethanol use, since the non-RFG areas in Wisconsin are generally NO_x limited for ozone formation.

The Wisconsin DNR estimated the ozone impact for calendar year 2003 and assumed that all vehicles experience the increase in NO_x emissions. Thus, their results are more comparable to our sensitivity analysis, than our primary analysis. For the two increased ethanol use scenarios, our sensitivity analysis projects increased ozone levels for several Wisconsin counties of 0.35-40 ppb. Because the Wisconsin DNR analyzed calendar year 2003 emissions and air quality, their base emission levels are much higher than those estimated here for the year 2015. Emission standards applicable to new vehicles and equipment are continually reducing emissions over time. Per the emission models used here and by the State of Wisconsin (NONROAD and MOBILE6), the effect of fuel quality is generally estimated in terms of a percentage change in the base emission level. As emissions from vehicles and equipment decrease over time, the absolute impact of fuel quality changes decreases at the same rate. Thus, the absolute emission changes predicted here for 2015 could easily be a factor of two lower than those predicted by

Wisconsin for 2003. This is likely the primary cause of the difference in the two sets of projected ozone impacts.

We received a comment from the Missouri Department of Natural Resources on the ozone impact analysis performed for the NPRM which expressed a concern that our approach of zeroing out the ozone impact in areas which did not experience a significant change in ethanol use had the effect of ignoring the impact of ozone transport due to increased ethanol use in upwind areas. This comment is correct. In a national analysis such as this one, it is not practical to go through over 3100 counties to determine which counties might have not experienced a change in ethanol use in a particular ethanol use case, but is downwind of an area which did. Still, the issue is a potentially relevant one and of reasonable interest particularly to those tasked with air quality management.

In an attempt to approximate the impact of ozone transport from areas which did experience a change in ethanol use on ozone in areas which did not, we performed one additional run of the Ozone RSM. This additional run applied the changes in VOC and NO_x emissions estimated above for attainment areas from our sensitivity analysis for the EIA case to emissions in attainment areas, and applied no change in emissions in non-attainment areas. We then compared the resulting ozone levels to those from the base case, focusing on the difference in ozone levels in non-attainment areas. Emissions in non-attainment areas were the same in both cases (no change from the base case). Thus, the difference in ozone levels in non-attainment areas should only be due to changes in emissions and ozone levels in upwind attainment areas.

The results of this comparison indicated that, in terms of the 8-hour ozone design value, ozone levels in non-attainment areas (i.e., RFG or Low RVP areas) decreased by 0.03 ppb. Thus, the average impact due to ozone transport is a reduction in ozone in downwind areas. However, the standard deviation in the ozone impact was 0.05 ppb, indicating that a significant number of areas experienced an increase, though most experienced a decrease. This is not surprising given that ozone in some attainment areas is VOC limited and may be decreasing in this fuel case, while others are NO_x limited and may be increasing. The maximum ozone reduction was 0.17 ppb, while the maximum increase was 0.12. More precise local atmospheric dispersion modeling will be needed in order to estimate this type of impact for specific non-attainment areas.

In summary, we estimate that the measurable changes in VOC and NO_x which are a result of increased ethanol use will, on average, result in small increases in ambient ozone formation. As we discussed above, the ozone modeling results in a net increase in the average population weighted ozone design value metric measured within the modeled domain (37 Eastern states and the District of Columbia). In Appendix A, we also present the impacts of increased ethanol use on a number of alternative measures of ambient ozone concentration. We acknowledge, however, that to the extent it occurs, increased future levels of ambient concentrations of ozone related to the increased use of ethanol may result in detrimental health and welfare effects due to ozone.

5.2 Particulate Matter

5.2.1 Impact of Changes in Direct PM Emissions

The amount of data evaluating the impact of ethanol and MTBE blending on direct emissions of PM from gasoline-fueled vehicles is extremely limited, as discussed in Chapter 3. Most studies do not test PM emissions from vehicles fueled with unleaded gasoline, because the level of PM emissions from properly operating vehicles is usually very low, less than 0.1 g/mi.

Two studies indicate that the addition of ethanol might reduce direct PM emissions from gasoline vehicles^{FFFF,GGGG}. However, both studies were performed under wintertime conditions and one at high altitude. One of the studies only consisted of three vehicles. The available data indicate that ethanol blending might reduce exhaust PM emissions under very cold weather conditions (i.e., 0 F or less), particularly at high altitude. There is no indication of PM emission reductions at higher temperatures or under warmed up conditions. Thus, the data are certainly too limited to support a quantitative estimate of the effect of ethanol on PM emissions.

5.2.2 Potential Impact of Changes in Secondary PM Formation

In addition to being emitted directly from a combustion source, fine particles can be formed through a series of chemical reactions in the atmosphere when SO₂, NO_x, and VOC oxidize or otherwise react to form a wide variety of secondary PM. For example, SO₂ oxidizes to SO₃ and sulfuric acid and NO_x oxidizes to NO₃ and nitric acid which, in turn, react with ammonia in the atmosphere to form ammonium sulfate and ammonium nitrate. Particles generated through this gas to particle conversion are referred to as secondary aerosols (SA) and represent a significant portion of ambient fine particulate matter. Studies have shown that as much as 70% of the total organic carbon in urban particulate matter can be attributed to secondary organic aerosol (SOA) formation although the amount can also be less.^{HHHH} Secondary PM tends to form more in the summer with higher temperatures and more intense sunlight.

Source-receptor modeling studies conducted in the Los Angeles area in 1993 by Schauer et al^{IIII} indicate that as much as 67% of the fine particulate matter collected could not be attributed to primary sources. The authors concluded that much of this unidentifiable organic matter is secondary organic aerosol formed in the atmosphere. This is consistent with previous studies conducted by Turpin and Huntzicker in 1991 who concluded that 70% of the total organic carbon in urban PM measurements made in southern California can be attributed to SOA.

Gas phase VOCs are oxidized by OH, NO₂, peroxyacetyl nitrate (PAN), and ozone in the atmosphere, but their propensity to condense in the particle phase is a function of two factors: volatility and reactivity. To accumulate as an aerosol, a reaction product must first be formed in the gas phase at a concentration equal to its saturation concentration. This requirement will not be met if the relevant gas-phase reactions of the VOC are too slow or if the vapor pressure of the reaction product is higher than the initial concentration of its VOC precursor.^{JJJJ} Limited data for reaction rate constants determined both experimentally and estimated by structural

relationships are available in the published literature. However, the atmospheric chemistry behind SOA reaction rates and the estimated aerosol yield is highly complex and carries with it a great deal of uncertainty. Research in this area is ongoing and thus the capacity to quantitatively model SOA formation is not yet a straightforward process.

In general, all reactive VOC are oxidized by OH or other compounds. Additionally, alkenes, cycloalkenes, and other olefinic compounds can react with ozone and NO₂ to form secondary aerosols. In fact, ozone is responsible for nearly all the SOA formation from olefins, while OH plays little or no role at all (Grosjean and Seinfeld, 1989; Izumi and Fukuyama, 1990). Many VOC, however, will never form secondary organic aerosol under atmospheric conditions regardless of their reactivity. This is because the products of reactions of these compounds have vapor pressures that are too high to form aerosols at atmospheric temperatures and pressures. These include all alkanes and alkenes with up to 6 carbon atoms, benzene and many low-molecular weight carbonyls, chlorinated compounds and oxygenated solvents (Grosjean, 1992).

The VOC that have the greatest propensity to form SOA include aromatic hydrocarbons (such as toluene but even including benzene), higher molecular weight olefins and cyclic olefins, and higher molecular weight paraffins. Kleindienst et al suggest that a high fraction of SOA is due to aromatic hydrocarbon precursors. Furthermore, “aromatic products having a single alkyl group on the aromatic ring were found to represent a ‘high-yield’ family (e.g., toluene, ethylbenzene); compounds having multiple methyl groups (e.g., m-xylene, 1,2,4-trimethylbenzene) were found to represent a ‘low-yield’ family” (Kleindienst, 269). All of the above mentioned VOC precursors are important either because there are large amounts of these particular VOC emitted per day, or because a large fraction of the VOC reacts, or a combination of the two. Based on VOC emissions inventory data collected in the Los Angeles area, the most important aerosol precursors (in the LA area using 1982 VOC emissions inventories) are listed in Table 5.2-1 below:

**Table 5.2-1.
Predicted In Situ SOA Formation
During a Smog Episode in Los Angeles**

VOC Functional Group	kg emitted daily*	Secondary PM Produced (kg)*	% yield
<u>Aromatics</u>	223985	3061	1.37
<u>Olefins</u>			
Alkenes	31163	608	1.95
Cyclic Olefins	3220	144	4.47
Terpenes	6000	626	10.43
<u>Paraffins</u>			
Alkanes	140493	368	0.26
Cycloalkanes	37996	96	0.25

*Source: Grosjean et al, 1992

These predictions are a function of input data collected in the Los Angeles area, and assume ambient levels of [ozone] = 100 ppb, [OH]=1.0x10⁶ molecules/cm³, and [NO₃]=0 with 6 hours of reaction time. Aromatics are the largest functional group in terms of the absolute

quantities of VOC emitted daily, and thus they eventually form the most SOA. Likewise, many high molecular weight paraffins (alkanes) form SOA on a significant scale simply because their emissions are high. However, the relative fraction of paraffins that react is less than that of aromatics in smog chamber experiments simulating SOA formation in the atmosphere. For olefins, the alkenes exhibit a combination of both relatively high emissions, and a high fraction of VOC reacted to form SOA. Cyclic olefins, in contrast, are emitted in relatively low levels, but a high fraction of these VOC react and the end result is a proportionally higher SOA yield than with the alkenes. Lastly, there are several “miscellaneous” compounds and terpenes that are emitted on a relatively small scale (in southern California), but that produce a substantial amount of secondary organic aerosol.

Researchers at EPA recently completed a field study in the Raleigh/Durham area of North Carolina that investigated the contribution of various sources to ambient PM 2.5 concentrations.^{KKKK} In the study they identified toluene as an SOA precursor. They estimate that mobile sources contribute nearly 90% of the total toluene emissions in that region based on a chemical mass balance approach. At the same time, however, SOA attributable to non-fuel-related VOC (i.e., biogenic emissions) was found to be an even larger contributor to SOA (i.e., toluene was not likely the dominant source of SOA in this area). This study is currently undergoing peer review and will be published shortly. Qualitatively, however, this information is still quite useful since the study identifies a contributing source of SOA that is attributable almost entirely due to mobile sources.

VOC reaction rates increase with increasing ambient temperature and sunlight intensity, so the level of SOA formed is much higher in summer than in winter. Even in the more temperate coastal climates of southern CA, studies have found the summertime concentration of SOA calculated through Chemical Mass Balance models show SOA formation to be anywhere from 2 – 5 times higher in summer than winter. In a study conducted at both urban and rural locations in the southeastern United States, the concentration of SOA in the summer and early fall was roughly 2-3 times that of colder months^{LLLL}.

As mentioned in Chapters 2 and 3, the addition of ethanol should reduce aromatics in gasoline, which will in turn reduce the aromatics emitted in the exhaust. However, quantifying the emission reduction is not possible at this time due to a lack of speciated exhaust data for newer vehicles running on ethanol blends. In addition, increased NOx emissions resulting from the increased use of ethanol could increase the formation of nitrate PM.

Based on the following, we believe that it is likely that the decrease in secondary PM from organic aromatic hydrocarbons is likely to exceed the increase in secondary nitrate PM. In 1999, NOx emissions from gasoline-fueled vehicles and equipment comprised about 20% of national NOx emissions from all sources^{MMMM}. In areas where ethanol use increases dramatically, NOx emissions from gasoline-fueled vehicles and equipment increases roughly 5-10%. This is roughly equivalent to a 1-2% increase in NOx emissions nationwide.

In contrast, gasoline-fueled vehicles and equipment comprised over 60% of all national gaseous aromatic VOC emissions⁵⁹. In areas where ethanol use increases dramatically, fuel

aromatic content decreases by about 4 vol% in the summertime, averaged across conventional gasoline and RFG. This represents about a 15% reduction from a base level of around 27 vol%. Assuming a proportional relationship between fuel aromatics and aromatic emissions, this represents about a 24% reduction in aromatic emissions nationwide.

In most urban areas, ambient levels of excess summer carbonaceous PM (a reasonable estimate of secondary organic PM) tend to exceed those of secondary nitrate PM. Thus, directionally, it appears likely that a net reduction in ambient PM levels will result from increased ethanol use. However, this should be considered a rough comparison at this time. A more precise comparison will have to await the incorporation of secondary organic aerosol formation into models, such as CMAx.

The research to facilitate this incorporation is currently underway. EPA ORD scientists are currently carrying out a wide variety of laboratory studies to refine the SOA chemistry mechanisms for use in the next version of the CMAQ model, which is expected to be completed in 2007 and submitted for peer review. This information should be available in time for the comprehensive study of the Act's fuel requirements which is due in 2009.⁶⁰

⁵⁹ Based on internal analyses of emissions inventories.

⁶⁰ Subject to funding.

Chapter 5: Appendix

Table 5A-1. 2015 Ozone Response Surface Metamodeling Summary Statistics for the RFS Rule^a; Primary Scenario

<i>8hr Design Value (ppb)</i>		
Statistic	RFS Scenario	EIA Scenario
Minimum Change	-0.015	0.000
Maximum Change	0.329	0.337
Average Change	0.057	0.079
Standard Deviation	0.086	0.100
Population-Weighted Change	0.052	0.056
<i>24hr Average (ppb)</i>		
Statistic	RFS Scenario	EIA Scenario
Minimum Change	-0.168	-0.162
Maximum Change	0.197	0.074
Average Change	0.008	0.013
Standard Deviation	0.018	0.021
Population-Weighted Change	0.014	0.013
<i>1hr Maximum (ppb)</i>		
Statistic	RFS Scenario	EIA Scenario
Minimum Change	-0.104	-0.094
Maximum Change	0.213	0.180
Average Change	0.015	0.024
Standard Deviation	0.033	0.040
Population-Weighted Change	0.029	0.033
<i>Average 9-to-5 (ppb)</i>		
Statistic	RFS Scenario	EIA Scenario
Minimum Change	-0.107	-0.097
Maximum Change	0.203	0.141
Average Change	0.012	0.019
Standard Deviation	0.027	0.031
Population-Weighted Change	0.022	0.024
<i>Average 10-to-3 (ppb)</i>		
Statistic	RFS Scenario	EIA Scenario
Minimum Change	-0.108	-0.107
Maximum Change	0.207	0.149
Average Change	0.012	0.019
Standard Deviation	0.027	0.032
Population-Weighted Change	0.023	0.024

^a Note that the statistics presented here represent ethanol use changes across the entire 37-state ozone RSM domain.

Table 5A-2. 2015 Ozone Response Surface Metamodeling Summary Statistics for the RFS Rule^a; Sensitivity Scenario

<i>8hr Design Value (ppb)</i>		
Statistic	RFS Scenario	EIA Scenario
Minimum Change	-0.115	0.000
Maximum Change	0.624	0.549
Average Change	0.111	0.142
Standard Deviation	0.158	0.170
Population-Weighted Change	0.092	0.096
<i>24hr Average (ppb)</i>		
Statistic	RFS Scenario	EIA Scenario
Minimum Change	-0.181	-0.173
Maximum Change	0.184	0.142
Average Change	0.015	0.024
Standard Deviation	0.034	0.041
Population-Weighted Change	0.025	0.028
<i>1hr Maximum (ppb)</i>		
Statistic	RFS Scenario	EIA Scenario
Minimum Change	-0.150	-0.133
Maximum Change	0.498	0.346
Average Change	0.027	0.043
Standard Deviation	0.060	0.072
Population-Weighted Change	0.052	0.062
<i>Average 9-to-5 (ppb)</i>		
Statistic	RFS Scenario	EIA Scenario
Minimum Change	-0.177	-0.163
Maximum Change	0.400	0.260
Average Change	0.022	0.034
Standard Deviation	0.049	0.057
Population-Weighted Change	0.040	0.046
<i>Average 10-to-3 (ppb)</i>		
Statistic	RFS Scenario	EIA Scenario
Minimum Change	-0.182	-0.167
Maximum Change	0.431	0.273
Average Change	0.022	0.035
Standard Deviation	0.050	0.058
Population-Weighted Change	0.041	0.047

^a Note that the statistics presented here reflect the impact of ethanol use changes across the entire eastern U.S. 37-state ozone RSM domain.