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# Clean Air Act Section 211(v)(1) Anti-backsliding Study

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

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## 1 Glossary of Acronyms

ASTM	American Society of Testing and Materials
CAA	Clean Air Act
CAP	Criteria Air Pollutants
CARB	California Air Resources Board
CG	Conventional Gasoline
CMAQ	Community Multiscale Air Quality Model
CMAS	Community Modeling and Analysis System
CNG	Compressed Natural Gas
CONUS	Continental United States
EIA	Energy Information Administration
EISA	Energy Independence and Security Act
EPAct	Energy Policy Act of 2005
eVNA	enhanced Veronoi Neighbor Average
HAP	Hazardous Air Pollutants
HOTELING	Hours of extended idle
LPG	Liquid Petroleum Gas
MCIP	Meteorology-Chemistry Interface Processor
MOVES	MOtor Vehicle Emission Simulator
MSAT	Mobile Source Air Toxics
NATA	National Air Toxics Assessment
NOAA	National Oceanic and Atmospheric Administration
OC	Organic Carbon
OM	Organic Matter
PAN	Peroxyacetyl Nitrate
PM	Particulate Matter
PMC	Coarse Particulate Matter
POA	Primary Organic Aerosols
RFG	Reformulated Gasoline
RFS	Renewable Fuel Standard
RPD	Rate-per-distance
RPH	Rate-per-hour
RPP	Rate-per-profile
RPV	Rate-per-vehicle
RVP	Reid Vapor Pressure
SCC	Source Classification Code
SMAT-CE	Software for the Modeled 8-30 Attainment Test – Community Edition
SMOKE	Sparse Matrix Operator Kernel Emissions
SMPS	Scanning Mobility Particle Sizer
SOA	Secondary Organic Aerosols
TSD	Technical Support Document
VMT	Vehicle Miles Traveled
VOC	Volatile Organic Compound

VPOP  
WRF

Vehicle Population  
Weather Research Forecasting Model

## 2 Executive Summary

As required by Clean Air Act section 211(v)(1), this study examined the impacts on air quality as a result of changes in vehicle and engine emissions resulting from required renewable fuel volumes under the Renewable Fuel Standard (RFS). Specifically, this study compared two scenarios for calendar year 2016, one with actual ethanol and biodiesel volumes (“with-RFS” scenario) and another with ethanol and biodiesel use approximating 2005 levels (the “pre-RFS” scenario).

The “with-RFS” scenario assumed 10 percent ethanol (E10) was used nationwide in all onroad and nonroad gasoline-fueled vehicles and engines, and biodiesel was used at a five percent blend (B5) in all onroad diesel vehicles nationwide. This was compared to the “pre-RFS” scenario, which assumed E10 usage only in the 2016 reformulated gasoline (RFG) areas and no biodiesel usage (except in California). In California, we assumed that the “pre-RFS” scenario was the same as the “with-RFS” scenario and therefore did not model any emissions changes there. Everything was held constant between the two scenarios except the fuel supplies for onroad and nonroad engines; “upstream” emissions from producing, storing, and transporting fuels and feedstocks were also held constant in both scenarios at 2016 levels.

As described in the following paragraphs, this study has a number of limitations. It is narrowly focused on the impacts of statutorily-required renewable fuel volumes on concentrations of criteria and toxic pollutants due to changes in vehicle and engine emissions; this study is not an examination of the lifecycle impacts of renewable fuels on air quality, greenhouse gases, or other environmental impacts. Clean Air Act section 211(v)(1)(A) specifically refers to air quality impacts “as a result of changes in vehicle and engine emissions.”

This anti-backsliding study examines the impacts of required renewable volumes as compared to a hypothetical case where renewable fuel usage in 2016 was approximately the same as it had been in 2005, before EPA’s Act was enacted. Because of the very limited data on 2005 fuel properties and their spatial distribution, the “pre-RFS” scenario is only a general approximation of 2005 fuels. In addition, in the absence of consistent and reliable data on biodiesel use across the country, this study assumes in the “with-RFS” scenario that biodiesel was at a B5 blend level in all onroad diesel fuel nationwide; it does not capture the impacts of higher or lower biodiesel blends that may have been used in specific areas. As a result, the air quality modeling results are illustrative at a broad geographic scale (rather than being locally specific). For this reason, this study does not attempt to estimate changes in the status of individual areas’ attainment of the National Ambient Air Quality Standards. With respect to estimating the effects of renewable fuels on emissions, data for nonroad gasoline engines and diesel vehicles and engines is much more limited than the data available for onroad gasoline vehicles.

This study examines impacts for a single retrospective year (2016). By analyzing calendar year 2016, EPA was able to use an existing modeling platform that includes known renewable fuel volumes for 2016 and fuel properties based on actual data (aggregation of refinery batch reports and fuel surveys). The study does not project future renewable fuel volumes and their impacts, and thus it does not account for the impacts of the Tier 3 Motor Vehicle Emissions and Fuel Standards, which took effect in 2017. These Tier 3 standards have lowered the sulfur content of

gasoline and tightened the emissions standards for onroad motor vehicles, resulting in lower emissions of criteria and toxic pollutants and precursors in 2017 and into the future as more Tier 3-compliant vehicles enter the fleet. The Tier 3 standards are projected to reduce concentrations of ozone, PM<sub>2.5</sub>, NO<sub>2</sub>, toxics (such as acetaldehyde, formaldehyde, acrolein, benzene, 1,3-butadiene, and naphthalene), and other pollutants into the future. In addition, the analysis year of 2016 used in this study does not reflect the full turnover of the diesel fleet to the most recent highway standards. Because emissions data does not indicate that biodiesel affects emissions from engines subject to the most recent standards, this study reflects emissions changes from older engines that would be projected to decline in the future as the fleet turns over.

Despite these limitations, this study satisfies the requirements of Clean Air Act section 211(v)(1) to determine the air quality impacts of changes in vehicle and engine emissions associated with required renewable fuel volumes.

Compared to the “pre-RFS” scenario, the 2016 “with-RFS” scenario increased ozone concentrations (eight-hour maximum average) across the eastern U.S. and in some areas in the western U.S., with some decreases in localized areas. In the 2016 “with-RFS” scenario, concentrations of fine particulate matter (PM<sub>2.5</sub>) were relatively unchanged in most areas, with increases in some areas and decreases in some localized areas. The 2016 “with-RFS” scenario increased concentrations of nitrogen dioxide (NO<sub>2</sub>) across the eastern U.S. and in some areas in the western U.S., with larger increases in some urban areas. The 2016 “with-RFS” scenario decreased concentrations of carbon monoxide (CO) across the eastern U.S. and in some areas in the western U.S., with larger decreases in some areas.

Compared to the “pre-RFS” scenario, the 2016 “with-RFS” scenario increased concentrations of acetaldehyde across much of the eastern U.S. and some areas in the western U.S., and resulted in widespread increases in formaldehyde concentrations. The 2016 “with-RFS” scenario decreased benzene concentrations across most of the U.S., as compared to the “pre-RFS” scenario. The 2016 “with-RFS” scenario also resulted in decreased concentrations of 1,3-butadiene in many urban areas. The 2016 “with-RFS” scenario resulted in geographically limited increases and decreases in concentrations of acrolein and naphthalene.



### 3 Introduction

#### 3.1 Statutory Requirement for Anti-backsliding Study

In 2007, the Energy Independence and Security Act (EISA) amended the Clean Air Act (CAA) to include a requirement (CAA section 211(v)(1)) that EPA complete a study to determine whether the renewable fuel volumes required by CAA section 211(o) would adversely impact air quality as a result of changes in vehicle and engine emissions. This required study is commonly known as the “anti-backsliding study.” Section 211(v)(1)(B) requires the study to include consideration of different blend levels, types of renewable fuels, and available vehicle technologies, as well as appropriate national, regional, and local air quality control measures.

After considering the results of the study, EPA must either (1) promulgate fuel regulations to implement appropriate measures to mitigate, to the greatest extent achievable, any adverse impacts on air quality; or (2) determine that no such measures are necessary (CAA section 211(v)(2)).

#### 3.2 Background on Renewable Fuel Standard Program

The Renewable Fuel Standard (RFS) program (CAA section 211(o)) was created by the Energy Policy Act of 2005 (EPAct) and expanded by EISA in 2007. CAA section 211(o) establishes targets for renewable fuel volumes and requires EPA to set volume requirements annually. In 2010 EPA finalized a rulemaking putting in place the regulations to implement 211(o) for EISA.<sup>1</sup> In doing so EPA projected the potential renewable fuel types and volumes that at the time we thought might be used to meet the statutory requirement. However, since that time it has become clear that those volumes were not attainable, and EPA has been using other authorities provided in 211(o) to reduce the volumes to achievable levels in subsequent rulemakings on an annual basis.

#### 3.3 Overview of Vehicle and Engine Emissions

Vehicles and engines contribute to air pollution through both exhaust and evaporative emissions. Exhaust emissions include hydrocarbons (HC), nitrogen oxides (NO<sub>x</sub>), particulate matter (PM), carbon monoxide (CO), sulfur dioxide (SO<sub>2</sub>), and various toxic pollutants, such as benzene, acetaldehyde, and formaldehyde. Evaporative emissions are hydrocarbon compounds, some of which are toxic pollutants (e.g., benzene).

Vehicle and engine emissions depend on the design and functionality of the engine and the associated exhaust and evaporative emission controls, in concert with the properties of the fuel on which it is operating. Relevant gasoline properties include fuel-content parameters as well as bulk properties. Fuel-content parameters include levels of oxygenate, ethanol, olefins, aromatics, benzene, and sulfur. Bulk properties include vapor pressure and distillation properties, expressed

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<sup>1</sup> Regulation of Fuels and Fuel Additives: Changes to Renewable Fuel Standard Program, 75 FR 14799-14808, March 26, 2010.

as temperatures (T50, T90) or as volumes evaporated at specific distillation temperatures (E200, E300). For diesel fuel, properties relevant to emissions include sulfur content and biodiesel content. The impact of renewable fuels on vehicle and engine emissions results from differences in these fuel properties.

EPA's emissions model, MOtor Vehicle Emission Simulator (MOVES), estimates the emissions from onroad vehicles and nonroad engines, considering fuel properties and many other factors. MOVES models the impact of ethanol and a wide range of other gasoline properties on exhaust and evaporative emissions in onroad gasoline vehicles, and models the impact of biodiesel blends on the emissions of diesel onroad vehicles.<sup>2,3,4</sup> Note that MOVES does not model any emissions effects of biodiesel in onroad heavy-duty engines that are model year 2007 and later because no significant and consistent effects have been observed.<sup>2</sup> For nonroad equipment, MOVES models the HC, CO, and NO<sub>x</sub> emissions impacts of oxygenate (such as ethanol) for nonroad gasoline engines, as well as the impacts on toxics emissions.<sup>5, 6</sup> MOVES does not model a fuel effect on direct PM emissions from nonroad equipment due to insufficient data. Also, MOVES does not model emissions impacts of biodiesel on nonroad diesel equipment. Little data on emissions in nonroad engines using biodiesel exists, and an EPA analysis of this limited data could not determine emission effects or conclude with confidence that nonroad engines respond to biodiesel similarly to highway engines.<sup>7</sup>

## 4 Overview of Study Approach

This study assessed the air quality impact of vehicle and engine emissions in 2016 under two fuel scenarios:

- “Pre-RFS”: Renewable fuel use at approximately 2005 (pre-EPA) levels
- “With-RFS”: Renewable fuel use at 2016 levels

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<sup>2</sup> USEPA (2016). *Fuel Effects on Exhaust Emissions from On-road Vehicles in MOVES2014*. EPA-420-R-16-001. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI. February 2016. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100O5W2.pdf>

<sup>3</sup> USEPA (2016). *Air Toxic Emissions from On-road Vehicles in MOVES2014*. EPA-420-R-16-016. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI. November 2016. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100PUNO.pdf>

<sup>4</sup> USEPA (2014). *Evaporative Emissions from On-road Vehicles in MOVES2014*. EPA-420-R-14-014. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI. September 2014. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100KB5V.pdf>

<sup>5</sup> USEPA (2005). *Exhaust Emission Effects of Fuel Sulfur and Oxygen on Gasoline Nonroad Engines*. EPA-420-R-05-016. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI. December 2005. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1004L80.pdf>

<sup>6</sup> USEPA (2018). *Speciation Profiles and Toxic Emission Factors for Nonroad Engines in MOVES2014b*. EPA-420-R-18-011. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI. July 2018. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100UXK7.pdf>

<sup>7</sup> USEPA (2002). *A Comprehensive Analysis of Biodiesel Impacts on Exhaust Emissions (Draft Report)*. EPA-420-P-02-001. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI. October 2002. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1001ZA0.pdf>

Everything in these two scenarios was held constant except the fuels; for example, the vehicle and engine population, activity, base emission rates, and meteorological data were the same in both scenarios (reflecting calendar year 2016 conditions).

As described in more detail in Section 5.2, the “pre-RFS” fuel scenario assumed for 2016 that vehicles used E10 in Reformulated Gasoline (RFG) areas and E0 everywhere else, which approximates the ethanol usage patterns before EPA’s Act was enacted in 2005. Except in California, the “pre-RFS” scenario did not assume any biodiesel use. The 2016 “with-RFS” scenario (described in Section 5.1) used EPA’s 2016 air quality modeling platform (2016v7.2 beta)<sup>8</sup>, including its fuel supply and emission inventory.

By analyzing calendar year 2016, EPA was able to use this existing modeling platform that includes known renewable fuel volumes for 2016 and fuel properties based on actual data (aggregation of refinery batch reports and fuel surveys). However, this retrospective analysis for 2016 does not account for the effects of the Tier 3 Motor Vehicle Emissions and Fuel Standards,<sup>9</sup> which took effect in 2017.

Except for California, the 2016 mobile source inventory was generated using MOVES2014b, the latest public version available when the study was initiated in 2019.<sup>10</sup> EPA’s Motor Vehicle Emission Simulator (MOVES) is a state-of-the-science emission modeling system that estimates emissions for mobile sources at the national, county, and project level for criteria air pollutants, greenhouse gases, and air toxics. MOVES computes exhaust emissions from onroad vehicles and nonroad equipment<sup>11,12</sup> and the evaporative emissions from onroad vehicles and nonroad equipment.<sup>13,14,15</sup> MOVES is the standard tool used to generate U.S. onroad and nonroad emission inventories. As detailed in Sections 5 and 6, California fuels were the same in both scenarios and the mobile source inventories for California were developed separately.

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<sup>8</sup> USEPA (2019). *Technical Support Document: Preparation of Emissions Inventories for the Version 7.2 2016 North American Emissions Modeling Platform*. Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency, Research Triangle Park, NC. September 2019. [https://www.epa.gov/sites/production/files/2019-09/documents/2016v7.2\\_regionalhaze\\_emismod\\_tsd\\_508.pdf](https://www.epa.gov/sites/production/files/2019-09/documents/2016v7.2_regionalhaze_emismod_tsd_508.pdf)

<sup>9</sup> 79 FR 23414, April 28, 2014.

<sup>10</sup> <https://www.epa.gov/moves/latest-version-motor-vehicle-emission-simulator-moves>

<sup>11</sup> USEPA (2016). *Fuel Effects on Exhaust Emissions from On-road Vehicles in MOVES2014*. EPA-420-R-16-001. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI. February 2016. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100O5W2.pdf>

<sup>12</sup> USEPA (2005). *Exhaust Emission Effects of Fuel Sulfur and Oxygen on Gasoline Nonroad Engines*. EPA-420-R-05-016. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI. December 2005. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1004L80.pdf>

<sup>13</sup> USEPA (2014). *Evaporative Emissions from On-road Vehicles in MOVES2014*. EPA-420-R-14-014. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI. September 2014. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100KB5V.pdf>

<sup>14</sup> USEPA (2010). *Nonroad Evaporative Emission Rates*. EPA-420-R-10-021. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI. July 2010. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100820I.pdf>

<sup>15</sup> USEPA (2004). *Refueling Emissions for Nonroad Engine Modeling*. EPA-420-P-04-013. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI. April 2004. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100021U.pdf>

Consistent with the requirements of CAA section 211(v)(1)(A), which refers specifically to adverse air quality impacts of “changes in vehicle and engine emissions,” this study did not consider any changes to “upstream” emissions, i.e., emissions associated with fuel production and distribution. As a result, this study does not represent a comprehensive consideration of the air quality impacts of required renewable fuel volumes.

This study assessed the changes in emissions from motor vehicles and nonroad engines and equipment (as estimated by MOVES) and used the Community Multiscale Air Quality model (CMAQ) to estimate the resulting impacts on concentrations of ozone, PM, NO<sub>2</sub>, CO, and some air toxics (including acetaldehyde, acrolein, benzene, 1,3-butadiene, formaldehyde, and naphthalene). CMAQ, a photochemical model, allows us to understand how changes in emissions impact air quality, especially for those pollutants which are formed secondarily and/or are transported long distances.

This study satisfies the CAA section 211(v)(1)(A) requirement to determine the impacts of required renewable fuel volumes, by comparing two scenarios with known renewable fuel volumes (i.e., before and after the renewable fuel requirements). Other potential study approaches would have involved highly uncertain estimates of fuel volumes and would have been less informative. As required by CAA section 211(v)(1)(B), this study considered different types and blends of renewable fuels (e.g., ethanol and biodiesel) and available vehicle technologies (i.e., all vehicle and nonroad engine technologies in the 2016 fleet). The 2016 air quality modeling platform includes national, regional, and local air quality control measures that were relevant for that year.

## 5 Renewable Fuel Scenarios

### 5.1 2016 “With-RFS” Scenario

The “with-RFS” scenario used the beta version of the 2016 air quality modeling platform (2016v7.2 beta), including the MOVES2014b fuel supply. The model has 11 fuel regions based on the general structure of the gasoline distribution system and overlays additional detail for state and local fuel controls; this results in a total of 22 sets of fuel properties used to compute fuel-based emission adjustments. Table 5.1 gives summary information on these fuel regions. The properties in these regions were determined from detailed fuel property data that EPA collects from refiners as part of gasoline production compliance batch reporting, and they were validated against survey data in local gasoline markets where available. Development of the MOVES fuel supply is described in detail in the MOVES Fuel Supply Defaults technical report.<sup>16</sup>

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<sup>16</sup> USEPA (2018). *Fuel Supply Defaults: Regional Fuels and the Fuel Wizard in MOVES2014b*. EPA-420-R-18-008. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI, July 2018. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockkey=P100UXDZ.pdf>

**Table 5.1 Summary Description of MOVES2014 Fuel Regions**

Region ID	Base Region ID#	Base Region Name	Maximum Summer RVP (psi) or 00 for ASTM	E10 RVP Waiver Status (01=No 1-psi Waiver)	Minimum Ethanol Volume, %
100000000	1	East Coast	0.0	0	0
100010000			0.0	1	0
170000000			7.0	0	0
178000000			7.8	0	0
178010000			7.8	1	0
200000000	2	Midwest	0.0	0	0
270000000			7.0	0	0
278000000			7.8	0	0
278010000			7.8	1	0
300000000	3	South	0.0	0	0
370000000			7.0	0	0
370010000			7.0	1	0
400000000	4	North	0.0	0	0
500000000	5	Rocky Mtns	0.0	0	0
578000000			7.8	0	0
600000000	6	CA/NV/AR/All Others	0.0	0	0
678000000			7.8	0	0
1170011000	11	East Coast RFG	7.0	1	10
1270011000	12	MD/VA RFG	7.0	1	10
1370011000	13	Texas RFG	7.0	1	10
1470011000	14	Midwest RFG	7.0	1	10
1570011000	15	California	7.0	1	10

Renewable fuel volumes for the 2016 “with-RFS” scenario were based on the simplifying assumption that all non-E85 gasoline was E10 (both conventional gasoline (CG) and reformulated gasoline (RFG)) nationwide. This was reasonable because volumes of other blends (E0 and E15) were small and we do not have consistent and reliable data on their volume or location.

Similarly, biodiesel was assumed to be at a B5 blend level (5 percent biodiesel) in all diesel nationwide. We lack consistent and reliable data on biodiesel across the country, and this is a reasonable simplifying assumption that is generally consistent with aggregate usage figures. In addition, under ASTM D975, blends up to 5% can be labeled as “diesel fuel,” confounding the ability to determine where biodiesel is used.

A copy of the fuel formulation table used with MOVES for the 2016v7.2 beta platform is available as part of the online supplemental materials.<sup>17</sup>

<sup>17</sup> <https://www.epa.gov/renewable-fuel-standard-program/anti-backsliding-determination-and-study>

## 5.2 2016 “Pre-RFS” Scenario

Because EPCRA, which created the RFS program, was signed into law in late 2005, ethanol and biodiesel consumption from that year was chosen for the “pre-RFS” scenario. EIA’s Monthly Energy Review presents U.S. ethanol consumption volumes on a monthly and annual basis.<sup>18</sup> Computing the annualized fourth quarter rate for 2005 gives a volume of 4.5 billion gallons, which closely matches the amount of ethanol required to blend all 2016 RFG, including California gasoline, to E10 (about 4.7 billion gallons).<sup>19</sup> Preferential use of ethanol in RFG is reasonable because these areas invested heavily in ethanol blending infrastructure and logistics leading up to and immediately following EPCRA, quickly converting over entirely from MTBE use to E10 use.<sup>20,21</sup> While there likely was some continued use of E10 in some non-RFG areas in this timeframe, the market was changing quickly and we do not have good data on precisely which areas did and did not have E10. Allocating the E10 to RFG areas represents a reasonable approximation of where ethanol was primarily used in this timeframe. Figure 5.1 shows the E10 RFG areas for this scenario.

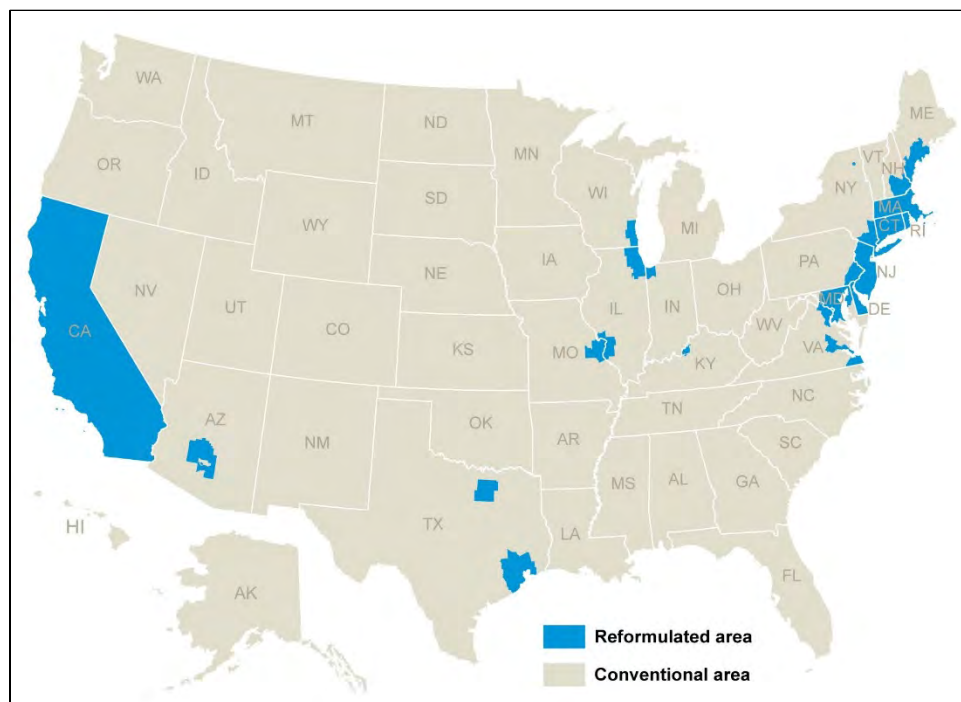


Figure 5.1 RFG areas where E10 was allocated in the "pre-RFS" scenario

<sup>18</sup> US Energy Information Administration. Monthly Energy Review, Table 10.3. Retrieved September 2018.

<sup>19</sup> US Energy Information Administration. U.S. Prime Supplier Sales Volumes of Petroleum Products, Table 2. Retrieved April 2018.

<sup>20</sup> US EPA Office of Transportation and Air Quality. Fuels Trends Report: Gasoline 1995-2005, Figure 6. EPA-420-R-08-002. January 2008.

<sup>21</sup> US EPA Office of Transportation and Air Quality. Fuels Trends Report: Gasoline 2006-2016, Figure 8. EPA-420-R-17-005. October 2017.



Therefore, in order to create the “pre-RFS” scenario for 2016, the 2016 “with-RFS” fuel supply was modified to make all CG E0 while all RFG remained E10 (including all California counties). The market share of E85 blends was left unmodified (i.e., the same as 2016 “with-RFS” levels) as a simplifying assumption, because they represent a small volume in the context of the overall fuel supply. Other blend levels, such as E15, were assumed to be zero in the 2016 “pre-RFS” scenario because E15 was not in use in 2005.

According to the EIA Monthly Energy Review, biodiesel use in 2005 was 91 million gallons.<sup>22</sup> Biodiesel use is modeled in MOVES as a single blend level nationwide, so dividing this volume by the total 2016 onroad diesel use of approximately 45 billion gallons results in a blend level of 0.2%.<sup>23</sup> For the purpose of the “pre-RFS” scenario in this analysis, this blend level was treated as B0 (zero) in all states except California. We maintained California at the B5 blend level in the “pre-RFS” scenario under the assumption that the state Low Carbon Fuel Standard<sup>24</sup> would have maintained biodiesel blending there even in the absence of EPA Act. The fuel volume scenarios are summarized in Table 5.2.

Removing ethanol from all 2016 CG required adjustment of several other fuel properties to account for the fact that ethanol blending increases RVP, decreases distillation mid-point temperature (T50), and dilutes aromatics and other constituents. These adjustments were done using the E10 factors shown in Table 4 of the MOVES Fuel Supply Defaults technical report, reproduced here as Table 5.3. These are applied as additive adjustments, and the sign has been reversed here relative to the technical report because the scenario involved removing E10 from CG. For RVP, an adjustment of 1.0 psi downward was applied in both summer and winter when going from E10 to E0, except for summertime areas with “no waiver” flags in the MOVES fuel supply (see Table 5.1). In these cases, we assumed that, in the absence of ethanol, additional butanes/pentanes would be used in these fuels up to the allowed RVP limit. No changes were made to RFG fuels (including California). A copy of the fuel formulation table modified for use in the “pre-RFS” scenario is available as part of the online supplemental materials.<sup>25</sup>

**Table 5.2 Summary of Fuel Cases (2016 Calendar Year)**

Scenario	E0	E10	Biodiesel
2016 “With-RFS”	None	All gasoline	B5 in all onroad diesel
2016 “Pre-RFS”	All conventional gasoline	All reformulated gasoline	B5 in California; B0 elsewhere

<sup>22</sup> US Energy Information Administration. Monthly Energy Review, Table 10.4. Retrieved September 2018.

<sup>23</sup> US Energy Information Administration. Monthly Energy Review, Tables 3.5 and 3.7c. Retrieved February 2020.

<sup>24</sup> Low Carbon Fuel Standard Regulation, title 17, California Code of Regulations, sections 95480-95503.

<sup>25</sup> <https://www.epa.gov/renewable-fuel-standard-program/anti-backsliding-determination-and-study>

**Table 5.3 MOVES2014 Fuel Property Adjustment Factors Moving from E10 to E0<sup>a</sup>**

<b>Fuel</b>	<b>RVP (psi)</b>	<b>Sulfur (ppm)</b>	<b>Aromatics (vol%)</b>	<b>Olefins (vol%)</b>	<b>Benzene (vol%)</b>	<b>E200 (vol%)</b>	<b>E300 (vol%)</b>	<b>T50 (deg.F)</b>	<b>T90 (deg.F)</b>
E10 summer	-1.00	-	2.02	0.46	-	-3.11	-0.39	6.34	1.77
E10 winter	-1.00	-	3.65	2.07	-	-4.88	-0.54	9.96	2.45

<sup>a</sup> Sulfur and benzene content are controlled by downstream regulations, and therefore aren't expected to change with addition of ethanol.

## 6 Emissions Inventories and Air Quality Modeling

### 6.1 Overview of Analysis Methods

This analysis utilizes EPA's 2016v7.2 beta emissions modeling platform,<sup>26</sup> which includes a suite of 2016 inventories, ancillary emissions data, and scripts and software for preparing emissions for air quality modeling. The onroad and nonroad inventories in the 2016v7.2 beta platform represents the "with-RFS" scenario. These inventories were then recalculated using modified fuels information, described in Section 5.2, to represent the "pre-RFS" scenario. Section 6.2 describes the methodology for developing onroad mobile emissions inventories for both scenarios; Section 6.3 describes this methodology for the nonroad mobile sector. In addition to emissions from onroad and nonroad mobile sources, air quality modeling requires emissions from all inventory sectors (e.g., biogenic, point, nonpoint sources); these sectors are discussed in Section 6.5. Finally, Sections 6.6 and 6.7 provide an overview of the air quality modeling methodology, including how emissions inventories are processed for air quality modeling.

### 6.2 Onroad Mobile Emissions Inventory

This section focuses on the approach and data sources used to develop gridded, hourly emissions for the onroad mobile sector that are suitable for input to an air quality model in terms of the format, grid resolution, and chemical species. While the fuel supplies used to develop emissions for the "with-RFS" and "pre-RFS" scenarios differed, the approach and all other (non-fuel supply) data sources used to calculate emissions for both scenarios were identical.

Onroad mobile source emission factors for all states except California were generated with MOVES2014b, the latest public version available when the study was initiated in 2019. For this analysis, MOVES2014b estimated onroad exhaust and evaporative emissions at the county level. The MOVES2014b onroad emission estimates are based on a detailed analysis of in-use

<sup>26</sup>USEPA (2019). *Technical Support Document: Preparation of Emissions Inventories for the Version 7.2 2016 North American Emissions Modeling Platform*. Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency, Research Triangle Park, NC. September 2019. [https://www.epa.gov/sites/production/files/2019-09/documents/2016v7.2\\_regionalhaze\\_emismod\\_tsd\\_508.pdf](https://www.epa.gov/sites/production/files/2019-09/documents/2016v7.2_regionalhaze_emismod_tsd_508.pdf)



emissions from tens of thousands of light duty vehicles<sup>27</sup> and hundreds of heavy-duty trucks.<sup>28</sup> MOVES2014b also incorporates data from a wide range of test programs and other sources, including data on the emissions effects of fuel properties such as gasoline sulfur and ethanol,<sup>29,30</sup> and data on evaporative emissions from fuel leaks and from vehicles parked for multiple days.<sup>31</sup> The impact of biodiesel blends on HC, CO, NO<sub>x</sub> and direct PM emissions from pre-2007 onroad diesel engines is based on a detailed analysis of hundreds of emissions tests.<sup>32</sup> Note that no biodiesel effects are modelled for 2007 and later vehicles because no significant and consistent effects have been observed.<sup>33</sup>

The MOVES-generated onroad emission factors were then combined with activity data to produce emissions within the Sparse Matrix Operator Kernel Emissions (SMOKE) modeling system. The collection of programs that compute the onroad mobile source emissions are known as SMOKE-MOVES. SMOKE-MOVES uses a combination of vehicle activity data, emission factors from MOVES, meteorology data, and temporal allocation information needed to estimate hourly onroad emissions. Additional types of ancillary data are used for the emissions processing, such as spatial surrogates which ensure emissions are developed on the grid used in air quality modeling.

California is the only state for which EPA does not generate onroad mobile emissions based on MOVES output. Instead, the California Air Resources Board uses the EPA-approved California-specific model, EMFAC, to generate onroad mobile emissions in California. Because California fuels were identical in the “pre-RFS” and “with-RFS” scenarios, a separate “pre-RFS” scenario was not implemented for California. California, therefore, is not included in the inventory comparisons.

National onroad emission summaries for key pollutants are provided in Section 6.4.

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<sup>27</sup> USEPA (2015). *Exhaust Emission Rates for Light-Duty On-Road Vehicles in MOVES2014*. EPA-420-R-15-005. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI, October 2015. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100NNVN.pdf>

<sup>28</sup> USEPA (2015). *Exhaust Emission Rates for Heavy-Duty On-Road Vehicles in MOVES2014*. EPA-420-R-15-015a. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI, November 2015. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100NO46.pdf>

<sup>29</sup> USEPA (2016). *Fuel Effects on Exhaust Emissions for On-Road Vehicles in MOVES2014*. EPA-420-R-16-001. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI, February 2016. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100O5W2.pdf>

<sup>30</sup> USEPA (2016b). *Air Toxic Emissions from On-road Vehicles in MOVES2014*. EPA-420-R-16-016. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI, November 2016. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100PUNO.pdf>

<sup>31</sup> USEPA (2014). *Evaporative Emissions from On-Road Vehicles in MOVES2014*. EPA-420-R-14-014. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI, September 2014. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100KB5V.pdf>

<sup>32</sup> USEPA (2010). *Regulatory Impact Analysis: Renewable Fuel Standard Program (RFS2)*. EPA-420-R-10-006. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI, February 2010. (Appendix A).

<sup>33</sup> USEPA (2016). *Fuel Effects on Exhaust Emissions for On-Road Vehicles in MOVES2014*. EPA-420-R-16-001. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI, February 2016. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100O5W2.pdf>

## 6.2.1 Onroad Mobile Emissions Inventory Methodology

Onroad mobile source emissions result from motorized vehicles that normally operate on public roadways, including passenger cars, motorcycles, minivans, sport-utility vehicles, light-duty trucks, heavy-duty trucks, and buses. These sources are further divided between diesel, gasoline, E-85, and compressed natural gas (CNG) vehicles. The onroad mobile emissions sector characterizes emissions from parked vehicle processes (e.g., starts, hot soak, and hoteling) as well as from “on-network” processes (i.e., from vehicles as they move along the roads). Except for California, all onroad emissions are generated using the SMOKE-MOVES emissions modeling framework that leverages MOVES-generated emission factors, county and Source Classification Code (SCC)-specific activity data, and hourly meteorological data.

SMOKE-MOVES uses emission rate (i.e., “lookup”) tables generated by MOVES as input. These tables differentiate emissions by process (i.e., running, start, vapor venting, etc.), vehicle type, road type, temperature, speed, hour of day, etc. To generate the MOVES emission rates that could be applied across the U.S., an automated process is used to run MOVES to produce emission factors for a series of temperatures and speeds for a set of “representative counties,” to which every other county in the country is mapped. Representative counties are used because it is impractical to generate a full suite of emission factors for every county in the U.S. The representative counties for which emission factors are generated are selected according to their state, elevation, fuels, age distribution, ramp fraction, and inspection and maintenance programs. Each county is then mapped to a representative county based on its similarity to the representative county with respect to those attributes. For age distributions and vehicle fuel types, rather than choosing values specific to each representative county, a weighted average is computed for all counties represented by each representative county, and the mean of those averages was used. For the 2016v7.2 beta modeling platform used in this analysis, there are 303 representative counties, which is same as in the 2014v7.1 emissions modeling platform. A detailed discussion of the selection of representative counties used in this analysis is available in the 2014NEIv2 Technical Support Document (TSD), Section 6.8.2.<sup>34</sup>

Once representative counties are identified, emission factors are generated by running MOVES for each representative county for two “fuel months” – January to represent winter months and July to represent summer months – because different types of fuels are used in each season. MOVES is run for the range of temperatures that occur in each representative county for each season. SMOKE selects the appropriate MOVES emissions rates for each county, hourly temperature, SCC, and speed bin and multiplies the emission rate by appropriate activity data: VMT (vehicle miles traveled), VPOP (vehicle population), or HOTELING (hours of extended idle) to produce emissions. These calculations are done for every county and grid cell in the continental U.S. for every hour of the year. SMOKE-MOVES accounts for the temperature sensitivity of the onroad emissions in each county by using the gridded hourly temperature information available from the meteorological model outputs used for air quality modeling.

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<sup>34</sup> USEPA (2018). *Technical Support Document: 2014 National Emissions Inventory, version 2*. Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency, Research Triangle Park, NC. July 2018. [https://www.epa.gov/sites/production/files/2018-07/documents/nei2014v2\\_tsd\\_05jul2018.pdf](https://www.epa.gov/sites/production/files/2018-07/documents/nei2014v2_tsd_05jul2018.pdf)

In summary, the SMOKE-MOVES process for creating the model-ready onroad emissions for all states except California consists of the following steps:

- 1) Determine which counties will be used to represent other counties in the MOVES runs.
- 2) Determine which months will be used to represent other months' fuel characteristics.
- 3) Create inputs required by MOVES: county-specific information on vehicle populations, age distributions, speed distribution, temporal profiles, and inspection-maintenance programs for each of the representative counties.
- 4) Create inputs needed both by MOVES and SMOKE, including temperatures and activity data.
- 5) Run MOVES to create emission factor tables for the temperatures and speeds that exist in each county during the modeled period.
- 6) Run SMOKE to apply the emission factors to activity data (VMT, VPOP, and HOTELING) to calculate emissions based on the gridded hourly temperatures in the meteorological data.
- 7) Aggregate the results to the county-SCC level.

The onroad emissions are processed as four components that are merged together into the final onroad sector emissions:

- rate-per-distance (RPD) uses VMT as the activity data plus speed and speed profile information to compute on-network emissions from exhaust, evaporative, permeation, refueling, and brake and tire wear processes;
- rate-per-vehicle (RPV) uses VPOP activity data to compute off-network emissions from exhaust, evaporative, permeation, and refueling processes;
- rate-per-profile (RPP) uses VPOP activity data to compute off-network emissions from evaporative fuel vapor venting, including hot soak (immediately after a trip) and diurnal (vehicle parked for a long period) emissions; and
- rate-per-hour (RPH) uses HOTELING hours activity data to compute off-network emissions for idling of long-haul trucks from extended idling and auxiliary power unit process.

### 6.3 Nonroad Mobile Emissions Inventory

This section details the approach and data sources used to develop the 2016 emissions inventory for the mobile nonroad equipment sector. This sector includes all mobile source emissions that do not operate on roads, excluding commercial marine vehicles, railways, and aircraft. Types of nonroad equipment include recreational vehicles, pleasure craft, and construction, mining, and lawn and garden equipment.

Nonroad mobile emissions were generated using MOVES2014b without any state-provided inputs, except for in California, where inventories are provided by the California Air Resources Board (CARB), using an EPA-approved model. MOVES2014b incorporates EPA's previous

NONROAD2008 model,<sup>35</sup> including the fuel effects,<sup>36</sup> but improves nonroad engine population growth rates,<sup>37</sup> nonroad Tier 4 engine emission rates,<sup>38</sup> and sulfur levels of nonroad diesel fuels.<sup>39</sup> MOVES2014b models the HC, CO and NOx emissions impacts of oxygenate (such as ethanol) for nonroad gasoline engines, as well as the impacts on toxics emissions.<sup>40, 41</sup> MOVES does not model a fuel effect on direct PM emissions from nonroad equipment due to insufficient data. MOVES2014b does not model emissions impacts of biodiesel on nonroad diesel equipment. Little data on emissions in nonroad engines using biodiesel exists, and an EPA analysis of this data could not determine emission effects or conclude with confidence that nonroad engines respond to biodiesel similarly to highway engines.<sup>42</sup>

For areas other than California, monthly MOVES2014b inventory outputs were used after a limited amount of post-processing. Nonroad inventories were processed with the Sparse Matrix Operating Kernel Emissions (SMOKE) modeling system version 4.6. SMOKE creates emissions in a format that can be input into air quality models.

National nonroad emission summaries for key pollutants are provided in Section 6.4.

### 6.3.1 Nonroad Mobile Emissions Inventory Methodology

For all states except California, MOVES2014b was run to create a monthly emissions inventory for criteria air pollutants (CAPs), a full set of hazardous air pollutants (HAPs), and additional pollutants such as ethanol and total organic compounds less the sum of the HAPs (NONHAPTOG), which are used for speciation. MOVES2014b provides estimates of NONHAPTOG along with the speciation profile code for the NONHAPTOG emission source.

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<sup>35</sup> USEPA (2009). *Frequently Asked Questions about NONROAD2008*. EPA-420-F-09-021. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI, April 2009. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1003E8H.pdf>

<sup>36</sup> USEPA (2005). *Exhaust Emission Effects of Fuel Sulfur and Oxygen on Gasoline Nonroad Engines*. EPA-420-R-05-016. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI, December 2005. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1004L80.pdf>

<sup>37</sup> USEPA (2018). *Nonroad Engine Population Growth Estimates in MOVES2014b*. EPA-420-R-18-010. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI, December July 2018. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100UXJK.pdf>

<sup>38</sup> USEPA (2018). *Exhaust and Crankcase Emission Factors for Nonroad Compression-Ignition Engines in MOVES2014b*. EPA-420-R-18-009. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI, December July 2018. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100UXEN.pdf>

<sup>39</sup> USEPA (2018). *Fuel Supply Defaults: Regional Fuels and the Fuel Wizard in MOVES2014b*. EPA-420-R-18-008. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI, July 2018. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100UXDZ.pdf>

<sup>40</sup> USEPA (2005). *Exhaust Emission Effects of Fuel Sulfur and Oxygen on Gasoline Nonroad Engines*. EPA-420-R-05-016. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI, December 2005. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1004L80.pdf>

<sup>41</sup> USEPA (2018). *Speciation Profiles and Toxic Emission Factors for Nonroad Engines in MOVES2014b*. EPA-420-R-18-011. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI, July 2018. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100UXK7.pdf>

<sup>42</sup> USEPA (2002). *A Comprehensive Analysis of Biodiesel Impacts on Exhaust Emissions (Draft Report)*. EPA-420-P-02-001. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI, October 2002. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1001ZA0.pdf>

MOVES2014b also provides estimates of PM<sub>2.5</sub> by speciation profile code for the PM<sub>2.5</sub> emission source. To facilitate calculation of coarse particulate matter (PMC) within SMOKE, and to help create emissions summaries, an additional pollutant representing total PM<sub>2.5</sub> (called PM25TOTAL) was added to the inventory.

MOVES2014b outputs emissions data in county-specific databases, and a post-processing script converts the data into Flat File 10 (FF10) format. Additional post-processing steps were performed as follows:

- 1) County-specific FF10s were combined into a single FF10 file.
- 2) To reduce the file size of the inventory, HAPs that are not needed for air quality modeling, such as dioxins and furans, were removed from the inventory.
- 3) To reduce the file size of the inventory further, all emissions for sources (identified by county/SCC) for which total CAP emissions are less than  $1 \times 10^{-10}$  were removed from the inventory. For technical reasons, MOVES2014b attributes a very tiny amount of emissions to sources that generate zero emissions, for example, snowmobile emissions in Florida. Removing these sources from the inventory reduces the total file size of the inventory by 7%.
- 4) Gas and particulate components of HAPs that MOVES outputs separately, such as naphthalene, were combined.
- 5) VOC was renamed VOC\_INV so that SMOKE does not speciate both VOC (volatile organic compounds) and NONHAPTOG, which would result in double counting.
- 6) PM25TOTAL, referenced above, was also created at this stage of the process.
- 7) Emissions for airport ground support vehicles (SCCs ending in -8005), and oil field equipment (SCCs ending in -10010), were removed from the inventory at this stage, to prevent a double count with the ptnonipm and np\_oilgas sectors, respectively.

California is the only state for which EPA does not generate nonroad mobile emissions based on MOVES output. Instead, the California Air Resources Board uses the EPA-approved California-specific suite of offroad modeling tools to generate nonroad mobile emissions in California. Because California fuels were identical in the “pre-RFS” and “with-RFS” scenarios, a separate “pre-RFS” scenario was not implemented for California. California, therefore, is not included in the inventory comparisons.

## 6.4 Onroad and Nonroad Emissions Inventory Summaries

National onroad and nonroad emissions totals for 2016 for the continental United States (CONUS; excluding California) by pollutant and by sector for the “with-RFS” and “pre-RFS” cases are provided in Table 6.1 and Table 6.2.

Comparison of state (CONUS; excluding California) total annual (2016) emissions (tons per year) by pollutant for the "with-RFS" and "pre-RFS" scenarios are provided in Table 6.3 through

Table 6.8. Detailed state-level emission summaries by pollutant and by sector are included with the online supplemental materials.<sup>43</sup>

**Table 6.1 Comparison of national (CONUS; excluding California) total annual (2016) emissions (tons per year) for the "with-RFS" and "pre-RFS" scenarios for criteria pollutants**

<b>Pollutant</b>	<b>Sector</b>	<b>"with-RFS"</b>	<b>"pre-RFS"</b>	<b>diff</b>	<b>% diff</b>
NOx	Total (Gasoline + Diesel)	4,684,218	4,542,487	141,731	3.1 %
	Gasoline - onroad	2,013,967	1,900,020	113,947	6 %
	Gasoline - nonroad	90,858	69,821	21,037	30.1 %
	Diesel - onroad	1,819,928	1,813,180	6,748	0.4 %
	Diesel - nonroad	759,466	759,466	0	0 %
VOC	Total (Gasoline + Diesel)	2,600,189	2,495,537	104,652	4.2 %
	Gasoline - onroad	1,708,443	1,601,992	106,451	6.6 %
	Gasoline - nonroad	649,764	646,758	3,006	0.5 %
	Diesel - onroad	171,044	175,850	-4,806	-2.7 %
	Diesel - nonroad	70,938	70,938	0	0 %
PM <sub>10</sub>	Total (Gasoline + Diesel)	345,532	347,558	-2,026	-0.6 %
	Gasoline - onroad	144,320	143,605	715	0.5 %
	Gasoline - nonroad	33,181	33,181	0	0 %
	Diesel - onroad	106,511	109,253	-2,742	-2.5 %
	Diesel - nonroad	61,519	61,519	0	0 %
PM <sub>2.5</sub>	Total (Gasoline + Diesel)	210,458	212,347	-1,889	-0.9 %
	Gasoline - onroad	48,693	48,059	634	1.3 %
	Gasoline - nonroad	30,526	30,526	0	0 %
	Diesel - onroad	71,565	74,088	-2,523	-3.4 %
	Diesel - nonroad	59,674	59,674	0	0 %
SO <sub>2</sub>	Total (Gasoline + Diesel)	27,577	26,914	663	2.5 %
	Gasoline - onroad	22,148	21,485	663	3.1 %
	Gasoline - nonroad	728	728	0	0 %
	Diesel - onroad	3,680	3,680	0	0 %
	Diesel - nonroad	1,021	1,021	0	0 %
CO	Total (Gasoline + Diesel)	28,193,369	31,084,459	-2,891,090	-9.3 %
	Gasoline - onroad	18,653,076	19,761,434	-1,108,358	-5.6 %
	Gasoline - nonroad	8,310,708	10,069,942	-1,759,234	-17.5 %
	Diesel - onroad	858,309	881,807	-23,498	-2.7 %
	Diesel - nonroad	371,276	371,276	0	0 %

<sup>43</sup> <https://www.epa.gov/renewable-fuel-standard-program/anti-backsliding-determination-and-study>

**Table 6.2 Comparison of national (CONUS; excluding California) total annual (2016) emissions (tons per year) for the "with-RFS" and "pre-RFS" scenarios for air toxics**

<b>Pollutant</b>	<b>Sector</b>	<b>"with-RFS"</b>	<b>"pre-RFS"</b>	<b>diff</b>	<b>% diff</b>
Acetaldehyde	Total (Gasoline + Diesel)	30,882	21,875	9,007	41.2 %
	Gasoline - onroad	15,357	7,312	8,045	110 %
	Gasoline - nonroad	2,571	1,438	1,133	78.8 %
	Diesel - onroad	6,659	6,830	-171	-2.5 %
	Diesel - nonroad	6,295	6,295	0	0 %
Acrolein	Total (Gasoline + Diesel)	3,779	3,700	79	2.1 %
	Gasoline - onroad	890	820	70	8.5 %
	Gasoline - nonroad	216	176	40	22.7 %
	Diesel - onroad	1,157	1,189	-32	-2.7 %
	Diesel - nonroad	1,516	1,516	0	0 %
Benzene	Total (Gasoline + Diesel)	64,021	74,184	-10,163	-13.7 %
	Gasoline - onroad	43,354	49,463	-6,109	-12.4 %
	Gasoline - nonroad	16,680	20,696	-4,016	-19.4 %
	Diesel - onroad	1,444	1,482	-38	-2.6 %
	Diesel - nonroad	2,543	2,543	0	0 %
1,3-Butadiene	Total (Gasoline + Diesel)	10,052	11,064	-1,012	-9.1 %
	Gasoline - onroad	6,579	7,492	-913	-12.2 %
	Gasoline - nonroad	2,935	3,019	-84	-2.8 %
	Diesel - onroad	407	421	-14	-3.3 %
	Diesel - nonroad	132	132	0	0 %
Formaldehyde	Total (Gasoline + Diesel)	51,743	51,427	316	0.6 %
	Gasoline - onroad	13,037	12,137	900	7.4 %
	Gasoline - nonroad	4,526	4,733	-207	-4.4 %
	Diesel - onroad	16,506	16,882	-376	-2.2 %
	Diesel - nonroad	17,675	17,675	0	0 %
Naphthalene	Total (Gasoline + Diesel)	5,192	5,258	-66	-1.3 %
	Gasoline - onroad	2,258	2,263	-5	-0.2 %
	Gasoline - nonroad	1,012	1,029	-17	-1.7 %
	Diesel - onroad	1,657	1,700	-43	-2.5 %
	Diesel - nonroad	266	266	0	0 %

**Table 6.3 Comparison of state (CONUS; excluding California) total annual (2016) NOx and VOC emissions (tons per year) for the "with-RFS" and "pre-RFS" scenarios<sup>†</sup>**

State	NOx				VOC			
	"with-RFS"	"pre-RFS"	diff	% diff	"with-RFS"	"pre-RFS"	diff	% diff
Alabama	129,385	123,457	5,928	4.8 %	81,209	74,778	6,431	8.6 %
Arizona	117,980	115,305	2,675	2.3 %	71,372	67,945	3,427	5 %
Arkansas	87,840	84,912	2,928	3.4 %	42,234	39,619	2,615	6.6 %
Colorado	87,698	83,460	4,238	5.1 %	60,001	58,313	1,688	2.9 %
Connecticut	32,042	32,017	25	0.1 %	26,834	26,855	-21	-0.1 %
Delaware	13,801	13,787	14	0.1 %	13,239	13,247	-8	-0.1 %
D.C.	4,264	4,259	5	0.1 %	3,208	3,213	-5	-0.2 %
Florida	296,038	281,020	15,018	5.3 %	229,949	211,582	18,367	8.7 %
Georgia	200,730	192,480	8,250	4.3 %	113,134	105,857	7,277	6.9 %
Idaho	50,297	48,158	2,139	4.4 %	30,875	29,671	1,204	4.1 %
Illinois	181,457	178,222	3,235	1.8 %	108,380	106,140	2,240	2.1 %
Indiana	161,569	156,409	5,160	3.3 %	79,929	76,474	3,455	4.5 %
Iowa	92,042	89,141	2,901	3.3 %	46,821	44,819	2,002	4.5 %
Kansas	87,336	84,685	2,651	3.1 %	38,691	36,592	2,099	5.7 %
Kentucky	97,725	94,565	3,160	3.3 %	53,741	51,139	2,602	5.1 %
Louisiana	90,054	86,343	3,711	4.3 %	54,174	50,152	4,022	8 %
Maine	26,653	25,265	1,388	5.5 %	26,837	26,122	715	2.7 %
Maryland	71,762	70,843	919	1.3 %	45,943	45,368	575	1.3 %
Massachusetts	53,671	53,621	50	0.1 %	43,186	43,224	-38	-0.1 %
Michigan	132,215	124,963	7,252	5.8 %	120,141	115,718	4,423	3.8 %
Minnesota	118,743	113,021	5,722	5.1 %	97,407	94,554	2,853	3 %
Mississippi	75,429	72,404	3,025	4.2 %	41,360	38,283	3,077	8 %
Missouri	168,040	163,448	4,592	2.8 %	80,935	77,339	3,596	4.6 %
Montana	44,541	42,964	1,577	3.7 %	22,706	21,892	814	3.7 %
Nebraska	66,524	64,558	1,966	3 %	29,318	27,887	1,431	5.1 %
Nevada	53,123	51,227	1,896	3.7 %	28,569	26,880	1,689	6.3 %
New Hampshire	17,797	17,350	447	2.6 %	16,434	16,143	291	1.8 %
New Jersey	78,396	78,295	101	0.1 %	50,705	50,779	-74	-0.1 %
New Mexico	71,235	69,178	2,057	3 %	28,019	26,366	1,653	6.3 %
New York	149,458	145,826	3,632	2.5 %	112,298	110,316	1,982	1.8 %
North Carolina	161,062	152,638	8,424	5.5 %	106,244	99,305	6,939	7 %
North Dakota	57,558	56,601	957	1.7 %	15,786	15,518	268	1.7 %
Ohio	166,708	159,309	7,399	4.6 %	109,372	105,021	4,351	4.1 %
Oklahoma	90,011	86,413	3,598	4.2 %	52,111	48,747	3,364	6.9 %
Oregon	76,487	72,675	3,812	5.2 %	51,201	49,315	1,886	3.8 %
Pennsylvania	171,017	165,922	5,095	3.1 %	100,329	97,230	3,099	3.2 %
Rhode Island	12,631	12,611	20	0.2 %	7,329	7,342	-13	-0.2 %
South Carolina	107,013	101,930	5,083	5 %	66,149	61,335	4,814	7.8 %
South Dakota	39,152	38,155	997	2.6 %	16,014	15,463	551	3.6 %
Tennessee	146,310	140,179	6,131	4.4 %	83,413	78,090	5,323	6.8 %
Texas	423,981	414,507	9,474	2.3 %	204,951	196,808	8,143	4.1 %
Utah	74,015	71,518	2,497	3.5 %	37,523	36,113	1,410	3.9 %
Vermont	11,237	10,812	425	3.9 %	8,523	8,321	202	2.4 %
Virginia	129,334	125,669	3,665	2.9 %	81,244	78,622	2,622	3.3 %
Washington	134,715	128,199	6,516	5.1 %	85,766	82,662	3,104	3.8 %
West Virginia	36,462	35,084	1,378	3.9 %	19,917	18,947	970	5.1 %
Wisconsin	112,680	108,443	4,237	3.9 %	82,775	80,588	2,187	2.7 %
Wyoming	28,563	27,696	867	3.1 %	12,467	12,087	380	3.1 %

<sup>†</sup> State annual totals only include emissions from gasoline and diesel vehicles and engines and exclude emissions from alternative fuels (i.e., CNG, LPG).



**Table 6.4 Comparison of state (CONUS; excluding California) total annual (2016) PM<sub>10</sub> and PM<sub>2.5</sub> emissions (tons per year) for the "with-RFS" and "pre-RFS" scenarios<sup>†</sup>**

State	PM <sub>10</sub>				PM <sub>2.5</sub>			
	"with-RFS"	"pre-RFS"	diff	% diff	"with-RFS"	"pre-RFS"	diff	% diff
Alabama	8,131	8,182	-51	-0.6 %	4,742	4,790	-48	-1 %
Arizona	7,610	7,663	-53	-0.7 %	4,950	4,998	-48	-1 %
Arkansas	5,232	5,275	-43	-0.8 %	3,576	3,616	-40	-1.1 %
Colorado	6,401	6,422	-21	-0.3 %	3,970	3,990	-20	-0.5 %
Connecticut	2,818	2,828	-10	-0.4 %	1,595	1,604	-9	-0.6 %
Delaware	947	952	-5	-0.5 %	589	594	-5	-0.8 %
D.C.	615	617	-2	-0.3 %	249	251	-2	-0.8 %
Florida	26,276	26,372	-96	-0.4 %	14,247	14,338	-91	-0.6 %
Georgia	13,306	13,380	-74	-0.6 %	7,706	7,775	-69	-0.9 %
Idaho	2,911	2,937	-26	-0.9 %	2,026	2,049	-23	-1.1 %
Illinois	15,637	15,721	-84	-0.5 %	9,417	9,495	-78	-0.8 %
Indiana	11,724	11,790	-66	-0.6 %	7,292	7,353	-61	-0.8 %
Iowa	6,070	6,100	-30	-0.5 %	4,556	4,583	-27	-0.6 %
Kansas	5,342	5,375	-33	-0.6 %	3,923	3,954	-31	-0.8 %
Kentucky	6,025	6,078	-53	-0.9 %	3,794	3,843	-49	-1.3 %
Louisiana	6,260	6,315	-55	-0.9 %	3,607	3,657	-50	-1.4 %
Maine	2,066	2,080	-14	-0.7 %	1,371	1,384	-13	-0.9 %
Maryland	6,334	6,376	-42	-0.7 %	3,439	3,478	-39	-1.1 %
Massachusetts	6,105	6,127	-22	-0.4 %	3,146	3,166	-20	-0.6 %
Michigan	10,403	10,421	-18	-0.2 %	6,306	6,323	-17	-0.3 %
Minnesota	9,118	9,136	-18	-0.2 %	6,797	6,814	-17	-0.2 %
Mississippi	4,356	4,386	-30	-0.7 %	2,668	2,696	-28	-1 %
Missouri	10,345	10,435	-90	-0.9 %	7,070	7,153	-83	-1.2 %
Montana	2,550	2,568	-18	-0.7 %	1,945	1,962	-17	-0.9 %
Nebraska	4,033	4,054	-21	-0.5 %	3,095	3,114	-19	-0.6 %
Nevada	4,192	4,207	-15	-0.4 %	2,747	2,760	-13	-0.5 %
New Hampshire	1,516	1,523	-7	-0.5 %	971	978	-7	-0.7 %
New Jersey	7,291	7,333	-42	-0.6 %	4,276	4,315	-39	-0.9 %
New Mexico	3,736	3,784	-48	-1.3 %	2,441	2,485	-44	-1.8 %
New York	16,839	16,926	-87	-0.5 %	8,971	9,051	-80	-0.9 %
North Carolina	11,212	11,243	-31	-0.3 %	6,448	6,478	-30	-0.5 %
North Dakota	3,967	3,992	-25	-0.6 %	3,316	3,339	-23	-0.7 %
Ohio	13,505	13,547	-42	-0.3 %	8,028	8,067	-39	-0.5 %
Oklahoma	5,787	5,830	-43	-0.7 %	3,581	3,621	-40	-1.1 %
Oregon	5,006	5,032	-26	-0.5 %	3,224	3,248	-24	-0.7 %
Pennsylvania	13,403	13,483	-80	-0.6 %	8,104	8,178	-74	-0.9 %
Rhode Island	1,084	1,093	-9	-0.8 %	618	627	-9	-1.4 %
South Carolina	6,678	6,729	-51	-0.8 %	4,120	4,167	-47	-1.1 %
South Dakota	2,489	2,504	-15	-0.6 %	2,014	2,028	-14	-0.7 %
Tennessee	9,039	9,095	-56	-0.6 %	5,360	5,413	-53	-1 %
Texas	30,456	30,701	-245	-0.8 %	18,093	18,320	-227	-1.2 %
Utah	5,098	5,151	-53	-1 %	3,085	3,135	-50	-1.6 %
Vermont	1,145	1,146	-1	-0.1 %	752	753	-1	-0.1 %
Virginia	9,089	9,136	-47	-0.5 %	5,425	5,469	-44	-0.8 %
Washington	8,764	8,809	-45	-0.5 %	5,498	5,540	-42	-0.8 %
West Virginia	2,216	2,237	-21	-0.9 %	1,436	1,456	-20	-1.4 %
Wisconsin	7,761	7,809	-48	-0.6 %	5,286	5,330	-44	-0.8 %
Wyoming	1,416	1,434	-18	-1.3 %	983	999	-16	-1.6 %

<sup>†</sup> State annual totals only include emissions from gasoline and diesel vehicles and engines and exclude emissions from alternative fuels (i.e., CNG, LPG).

**Table 6.5 Comparison of state (CONUS; excluding California) total annual (2016) CO and SO<sub>2</sub> emissions (tons per year) for the "with-RFS" and "pre-RFS" scenarios<sup>†</sup>**

State	CO				SO <sub>2</sub>			
	"with-RFS"	"pre-RFS"	diff	% diff	"with-RFS"	"pre-RFS"	diff	% diff
Alabama	771,116	861,426	-90,310	-10.5 %	805	779	26	3.3 %
Arizona	715,015	758,134	-43,119	-5.7 %	402	397	5	1.3 %
Arkansas	398,809	450,934	-52,125	-11.6 %	365	355	10	2.8 %
Colorado	632,377	733,034	-100,657	-13.7 %	419	406	13	3.2 %
Connecticut	285,881	285,989	-108	0 %	312	312	0	0 %
Delaware	117,531	117,579	-48	0 %	110	110	0	0 %
D.C.	34,996	35,016	-20	-0.1 %	56	56	0	0 %
Florida	2,421,201	2,772,409	-351,208	-12.7 %	2,592	2,504	88	3.5 %
Georgia	1,206,612	1,365,641	-159,029	-11.6 %	1,366	1,321	45	3.4 %
Idaho	243,103	279,835	-36,732	-13.1 %	155	151	4	2.6 %
Illinois	1,183,863	1,243,093	-59,230	-4.8 %	794	784	10	1.3 %
Indiana	851,946	947,199	-95,253	-10.1 %	687	669	18	2.7 %
Iowa	414,609	472,698	-58,089	-12.3 %	354	344	10	2.9 %
Kansas	383,024	432,490	-49,466	-11.4 %	327	318	9	2.8 %
Kentucky	529,833	575,891	-46,058	-8 %	378	370	8	2.2 %
Louisiana	500,605	566,472	-65,867	-11.6 %	591	572	19	3.3 %
Maine	183,160	213,297	-30,137	-14.1 %	176	174	2	1.1 %
Maryland	535,309	550,106	-14,797	-2.7 %	639	636	3	0.5 %
Massachusetts	502,940	503,133	-193	0 %	661	661	0	0 %
Michigan	1,056,388	1,207,516	-151,128	-12.5 %	752	727	25	3.4 %
Minnesota	791,305	913,050	-121,745	-13.3 %	476	462	14	3 %
Mississippi	404,359	447,946	-43,587	-9.7 %	465	450	15	3.3 %
Missouri	791,294	863,227	-71,933	-8.3 %	646	632	14	2.2 %
Montana	185,935	210,590	-24,655	-11.7 %	116	113	3	2.7 %
Nebraska	268,706	304,913	-36,207	-11.9 %	220	214	6	2.8 %
Nevada	300,472	345,712	-45,240	-13.1 %	193	188	5	2.7 %
New Hampshire	148,806	158,297	-9,491	-6 %	152	151	1	0.7 %
New Jersey	646,004	646,311	-307	0 %	809	809	0	0 %
New Mexico	256,351	287,121	-30,770	-10.7 %	253	246	7	2.8 %
New York	1,217,731	1,305,719	-87,988	-6.7 %	1,504	1,481	23	1.6 %
North Carolina	1,137,222	1,289,318	-152,096	-11.8 %	1,299	1,254	45	3.6 %
North Dakota	146,692	169,134	-22,442	-13.3 %	125	123	2	1.6 %
Ohio	1,171,741	1,337,454	-165,713	-12.4 %	924	895	29	3.2 %
Oklahoma	518,057	590,345	-72,288	-12.2 %	464	450	14	3.1 %
Oregon	489,879	568,738	-78,859	-13.9 %	297	288	9	3.1 %
Pennsylvania	1,055,311	1,170,165	-114,854	-9.8 %	1,102	1,077	25	2.3 %
Rhode Island	79,013	79,070	-57	-0.1 %	91	91	0	0 %
South Carolina	651,879	738,968	-87,089	-11.8 %	633	613	20	3.3 %
South Dakota	138,140	156,214	-18,074	-11.6 %	97	95	2	2.1 %
Tennessee	833,412	935,039	-101,627	-10.9 %	754	730	24	3.3 %
Texas	2,415,363	2,603,866	-188,503	-7.2 %	2,779	2,725	54	2 %
Utah	332,276	380,154	-47,878	-12.6 %	275	268	7	2.6 %
Vermont	71,758	83,978	-12,220	-14.6 %	92	89	3	3.4 %
Virginia	902,061	959,852	-57,791	-6 %	950	935	15	1.6 %
Washington	807,113	924,660	-117,547	-12.7 %	513	497	16	3.2 %
West Virginia	188,275	212,394	-24,119	-11.4 %	171	166	5	3 %
Wisconsin	696,371	778,282	-81,911	-10.5 %	464	454	10	2.2 %
Wyoming	99,493	112,775	-13,282	-11.8 %	83	81	2	2.5 %

<sup>†</sup> State annual totals only include emissions from gasoline and diesel vehicles and engines and exclude emissions from alternative fuels (i.e., CNG, LPG).

**Table 6.6 Comparison of state (CONUS; excluding California) total annual (2016) acetaldehyde and acrolein emissions (tons per year) for the "with-RFS" and "pre-RFS" scenarios<sup>†</sup>**

State	Acetaldehyde				Acrolein			
	"with-RFS"	"pre-RFS"	diff	% diff	"with-RFS"	"pre-RFS"	diff	% diff
Alabama	774	462	312	67.5 %	83	81	2	2.5 %
Arizona	758	616	142	23.1 %	97	96	1	1 %
Arkansas	514	355	159	44.8 %	68	66	2	3 %
Colorado	648	375	273	72.8 %	73	70	3	4.3 %
Connecticut	270	271	-1	-0.4 %	29	29	0	0 %
Delaware	109	110	-1	-0.9 %	11	11	0	0 %
D.C.	32	33	-1	-3 %	4	4	0	0 %
Florida	1,988	1,235	753	61 %	244	234	10	4.3 %
Georgia	1,212	771	441	57.2 %	143	138	5	3.6 %
Idaho	330	212	118	55.7 %	40	39	1	2.6 %
Illinois	1,379	1,179	200	17 %	177	175	2	1.1 %
Indiana	1,060	726	334	46 %	135	132	3	2.3 %
Iowa	656	447	209	46.8 %	96	94	2	2.1 %
Kansas	562	396	166	41.9 %	83	81	2	2.5 %
Kentucky	614	433	181	41.8 %	72	70	2	2.9 %
Louisiana	540	354	186	52.5 %	67	64	3	4.7 %
Maine	217	153	64	41.8 %	25	23	2	8.7 %
Maryland	472	438	34	7.8 %	53	53	0	0 %
Massachusetts	464	465	-1	-0.2 %	52	52	0	0 %
Michigan	1,125	582	543	93.3 %	121	112	9	8 %
Minnesota	994	608	386	63.5 %	130	123	7	5.7 %
Mississippi	435	271	164	60.5 %	50	49	1	2 %
Missouri	1,048	796	252	31.7 %	137	135	2	1.5 %
Montana	313	219	94	42.9 %	44	43	1	2.3 %
Nebraska	456	329	127	38.6 %	69	68	1	1.5 %
Nevada	359	254	105	41.3 %	48	47	1	2.1 %
New Hampshire	152	125	27	21.6 %	17	16	1	6.3 %
New Jersey	570	572	-2	-0.3 %	70	71	-1	-1.4 %
New Mexico	363	253	110	43.5 %	44	44	0	0 %
New York	1,172	942	230	24.4 %	145	141	4	2.8 %
North Carolina	992	568	424	74.6 %	109	105	4	3.8 %
North Dakota	368	315	53	16.8 %	67	67	0	0 %
Ohio	1,249	728	521	71.6 %	146	139	7	5 %
Oklahoma	558	356	202	56.7 %	69	67	2	3 %
Oregon	544	336	208	61.9 %	62	60	2	3.3 %
Pennsylvania	1,154	813	341	41.9 %	136	132	4	3 %
Rhode Island	88	89	-1	-1.1 %	10	10	0	0 %
South Carolina	630	387	243	62.8 %	72	69	3	4.3 %
South Dakota	265	203	62	30.5 %	44	43	1	2.3 %
Tennessee	895	551	344	62.4 %	103	99	4	4 %
Texas	2,361	1,900	461	24.3 %	308	304	4	1.3 %
Utah	436	293	143	48.8 %	54	53	1	1.9 %
Vermont	98	63	35	55.6 %	13	12	1	8.3 %
Virginia	837	659	178	27 %	94	92	2	2.2 %
Washington	948	581	367	63.2 %	108	105	3	2.9 %
West Virginia	223	135	88	65.2 %	25	24	1	4.2 %
Wisconsin	820	553	267	48.3 %	99	94	5	5.3 %
Wyoming	160	114	46	40.4 %	21	20	1	5 %

<sup>†</sup> State annual totals only include emissions from gasoline and diesel vehicles and engines and exclude emissions from alternative fuels (i.e., CNG, LPG).

**Table 6.7 Comparison of state (CONUS; excluding California) total annual (2016) benzene and 1,3-butadiene emissions (tons per year) for the "with-RFS" and "pre-RFS" scenarios<sup>†</sup>**

State	Benzene				1,3-butadiene			
	"with-RFS"	"pre-RFS"	diff	% diff	"with-RFS"	"pre-RFS"	diff	% diff
Alabama	1,887	2,189	-302	-13.8 %	272	301	-29	-9.6 %
Arizona	1,678	1,810	-132	-7.3 %	249	261	-12	-4.6 %
Arkansas	1,003	1,176	-173	-14.7 %	156	173	-17	-9.8 %
Colorado	1,625	1,997	-372	-18.6 %	254	291	-37	-12.7 %
Connecticut	644	644	0	0 %	122	122	0	0 %
Delaware	311	311	0	0 %	57	57	0	0 %
D.C.	68	68	0	0 %	12	12	0	0 %
Florida	5,388	6,310	-922	-14.6 %	764	836	-72	-8.6 %
Georgia	2,763	3,247	-484	-14.9 %	407	453	-46	-10.2 %
Idaho	731	876	-145	-16.6 %	115	126	-11	-8.7 %
Illinois	2,637	2,846	-209	-7.3 %	422	439	-17	-3.9 %
Indiana	1,945	2,294	-349	-15.2 %	295	330	-35	-10.6 %
Iowa	1,168	1,390	-222	-16 %	193	216	-23	-10.6 %
Kansas	965	1,140	-175	-15.4 %	153	173	-20	-11.6 %
Kentucky	1,246	1,420	-174	-12.3 %	192	208	-16	-7.7 %
Louisiana	1,257	1,456	-199	-13.7 %	184	200	-16	-8 %
Maine	576	662	-86	-13 %	106	104	2	1.9 %
Maryland	1,112	1,158	-46	-4 %	200	203	-3	-1.5 %
Massachusetts	1,089	1,089	0	0 %	208	209	-1	-0.5 %
Michigan	2,825	3,386	-561	-16.6 %	463	503	-40	-8 %
Minnesota	2,338	2,817	-479	-17 %	344	367	-23	-6.3 %
Mississippi	962	1,115	-153	-13.7 %	139	154	-15	-9.7 %
Missouri	1,930	2,185	-255	-11.7 %	308	335	-27	-8.1 %
Montana	574	684	-110	-16.1 %	87	98	-11	-11.2 %
Nebraska	740	876	-136	-15.5 %	119	134	-15	-11.2 %
Nevada	710	840	-130	-15.5 %	107	118	-11	-9.3 %
New Hampshire	382	412	-30	-7.3 %	70	70	0	0 %
New Jersey	1,277	1,278	-1	-0.1 %	237	237	0	0 %
New Mexico	636	747	-111	-14.9 %	103	116	-13	-11.2 %
New York	2,740	3,026	-286	-9.5 %	479	499	-20	-4 %
North Carolina	2,584	3,069	-485	-15.8 %	385	428	-43	-10 %
North Dakota	422	505	-83	-16.4 %	56	62	-6	-9.7 %
Ohio	2,804	3,406	-602	-17.7 %	444	502	-58	-11.6 %
Oklahoma	1,230	1,456	-226	-15.5 %	196	219	-23	-10.5 %
Oregon	1,313	1,600	-287	-17.9 %	204	230	-26	-11.3 %
Pennsylvania	2,505	2,914	-409	-14 %	416	454	-38	-8.4 %
Rhode Island	182	182	0	0 %	35	35	0	0 %
South Carolina	1,572	1,838	-266	-14.5 %	231	253	-22	-8.7 %
South Dakota	409	488	-79	-16.2 %	54	61	-7	-11.5 %
Tennessee	1,988	2,337	-349	-14.9 %	297	330	-33	-10 %
Texas	4,784	5,305	-521	-9.8 %	762	812	-50	-6.2 %
Utah	902	1,085	-183	-16.9 %	140	157	-17	-10.8 %
Vermont	206	246	-40	-16.3 %	34	36	-2	-5.6 %
Virginia	1,975	2,172	-197	-9.1 %	329	347	-18	-5.2 %
Washington	2,215	2,681	-466	-17.4 %	340	384	-44	-11.5 %
West Virginia	482	577	-95	-16.5 %	75	84	-9	-10.7 %
Wisconsin	1,923	2,247	-324	-14.4 %	295	309	-14	-4.5 %
Wyoming	285	340	-55	-16.2 %	46	50	-4	-8 %

<sup>†</sup> State annual totals only include emissions from gasoline and diesel vehicles and engines and exclude emissions from alternative fuels (i.e., CNG, LPG).

**Table 6.8 Comparison of state (CONUS; excluding California) total annual (2016) formaldehyde and naphthalene emissions (tons per year) for the "with-RFS" and "pre-RFS" scenarios<sup>†</sup>**

State	Formaldehyde				Naphthalene			
	"with-RFS"	"pre-RFS"	diff	% diff	"with-RFS"	"pre-RFS"	diff	% diff
Alabama	1,234	1,215	19	1.6 %	139	139	0	0 %
Arizona	1,376	1,371	5	0.4 %	128	129	-1	-0.8 %
Arkansas	969	964	5	0.5 %	85	86	-1	-1.2 %
Colorado	1,007	1,002	5	0.5 %	117	120	-3	-2.5 %
Connecticut	413	415	-2	-0.5 %	46	46	0	0 %
Delaware	156	157	-1	-0.6 %	22	22	0	0 %
D.C.	49	50	-1	-2 %	5	5	0	0 %
Florida	3,415	3,389	26	0.8 %	368	372	-4	-1.1 %
Georgia	2,077	2,061	16	0.8 %	222	224	-2	-0.9 %
Idaho	566	561	5	0.9 %	67	68	-1	-1.5 %
Illinois	2,412	2,404	8	0.3 %	219	220	-1	-0.5 %
Indiana	1,892	1,879	13	0.7 %	168	170	-2	-1.2 %
Iowa	1,227	1,215	12	1 %	98	99	-1	-1 %
Kansas	1,071	1,066	5	0.5 %	80	81	-1	-1.2 %
Kentucky	1,006	998	8	0.8 %	109	110	-1	-0.9 %
Louisiana	946	939	7	0.7 %	107	108	-1	-0.9 %
Maine	347	336	11	3.3 %	55	54	1	1.9 %
Maryland	749	751	-2	-0.3 %	87	88	-1	-1.1 %
Massachusetts	736	739	-3	-0.4 %	78	78	0	0 %
Michigan	1,632	1,572	60	3.8 %	224	226	-2	-0.9 %
Minnesota	1,724	1,681	43	2.6 %	187	189	-2	-1.1 %
Mississippi	719	709	10	1.4 %	77	77	0	0 %
Missouri	1,935	1,927	8	0.4 %	174	176	-2	-1.1 %
Montana	582	580	2	0.3 %	52	53	-1	-1.9 %
Nebraska	899	894	5	0.6 %	63	63	0	0 %
Nevada	699	697	2	0.3 %	56	57	-1	-1.8 %
New Hampshire	237	233	4	1.7 %	32	32	0	0 %
New Jersey	994	999	-5	-0.5 %	101	102	-1	-1 %
New Mexico	655	653	2	0.3 %	69	70	-1	-1.4 %
New York	2,015	2,001	14	0.7 %	220	222	-2	-0.9 %
North Carolina	1,567	1,546	21	1.4 %	177	178	-1	-0.6 %
North Dakota	869	871	-2	-0.2 %	43	44	-1	-2.3 %
Ohio	1,991	1,965	26	1.3 %	204	207	-3	-1.4 %
Oklahoma	950	944	6	0.6 %	102	103	-1	-1 %
Oregon	912	911	1	0.1 %	100	102	-2	-2 %
Pennsylvania	1,940	1,928	12	0.6 %	205	208	-3	-1.4 %
Rhode Island	141	142	-1	-0.7 %	15	15	0	0 %
South Carolina	1,051	1,040	11	1.1 %	120	121	-1	-0.8 %
South Dakota	552	550	2	0.4 %	37	38	-1	-2.6 %
Tennessee	1,485	1,467	18	1.2 %	161	162	-1	-0.6 %
Texas	4,357	4,358	-1	0 %	413	418	-5	-1.2 %
Utah	764	763	1	0.1 %	89	91	-2	-2.2 %
Vermont	181	176	5	2.8 %	17	17	0	0 %
Virginia	1,322	1,315	7	0.5 %	143	143	0	0 %
Washington	1,562	1,553	9	0.6 %	169	172	-3	-1.7 %
West Virginia	356	352	4	1.1 %	41	41	0	0 %
Wisconsin	1,357	1,329	28	2.1 %	172	173	-1	-0.6 %
Wyoming	300	298	2	0.7 %	35	35	0	0 %

<sup>†</sup> State annual totals only include emissions from gasoline and diesel vehicles and engines and exclude emissions from alternative fuels (i.e., CNG, LPG).

## 6.5 Other Inventory Sectors

Emissions for all other inventory sectors (Table 6.9) were developed for the 2016v7.2 beta modeling platform. Along with the above-described onroad and nonroad mobile emissions, emissions from these sectors are used as inputs for the air quality modeling discussed in Section 6.7 and Section 7. Emissions from these sectors were unchanged between the “with-RFS” and “pre-RFS” scenarios. Documentation detailing the development of these emissions inventories is available from the 2016 National Emissions Collaborative wiki page.<sup>44</sup>

**Table 6.9 Inventory sectors included in the 2016v7.2 beta emissions modeling platform**

<b>Inventory Sector</b>	<b>Sector Description</b>
Biogenic	VOC emissions from trees, shrubs, grasses, and soils
Mobile - Nonroad	See Section 6.3
Mobile - Onroad	See Section 6.2
Mobile – Commercial Marine Vessels	Commercial marine vessels with Category 1, 2, and 3 engines
Mobile - Rail	Class I, II, and III railroad emissions (including yards and switchers), commuter rail, Amtrak
Nonpoint - Agriculture	NH <sub>3</sub> and VOC emissions from livestock and fertilizer sources
Nonpoint – Area Fugitive Dust	PM emissions from paved roads, unpaved roads and airstrips, construction, agriculture production, and mining and quarrying
Nonroad – Residential Wood Combustion	Residential wood burning devices such as fireplaces, woodstoves, pellet stoves, indoor furnaces, outdoor burning in fire pits and chimneys
Nonpoint - Other	All nonpoint sources not included in other sectors, including solvents, industrial processes, waste disposal, storage and transport of chemicals and petroleum, waste disposal, commercial cooking, miscellaneous area sources
Oil & Gas – Nonpoint and Point	Oil and gas exploration and production, both onshore and offshore
Point – Electrical Generating Units	Fossil fuel fired electrical generating units
Point – Fires – Agricultural	Agricultural burning
Point – Fires – Wild and Prescribed	Wildfires and prescribed burns
Canada – Mobile - Onroad	Onroad emissions in Canada
Mexico – Mobile - Onroad	Onroad emissions in Mexico
Canada/Mexico - Point	Canadian and Mexican point source emissions
Canada/Mexico – Point - Fires	Canadian and Mexican fire emissions
Canada/Mexico - Nonpoint	Canadian and Mexican nonpoint emissions
Canada – Nonpoint – Area Fugitive Dust	Area fugitive dust emissions in Canada
Canada – Point - Dust	Dust emissions in Canada

## 6.6 Emissions Modeling

The CMAQ air quality model requires hourly emissions of specific gas and particle species for the horizontal and vertical grid cells contained within the modeled region (i.e., modeling domain). To provide emissions in the form and format required by the model, it is necessary to “pre-process” the “raw” emissions (i.e., emissions input to SMOKE) for the sectors described above in Sections 6.2 – 6.5. The process of emissions modeling transforms the emissions

<sup>44</sup> <http://views.cira.colostate.edu/wiki/wiki/10197>

inventories from their original temporal resolution, pollutant resolution, and spatial resolution into the hourly, speciated, gridded resolution required by the air quality model. Emissions modeling includes chemical speciation, temporal allocation, and spatial allocation of emissions.

SMOKE version 4.6 was used to process the raw emissions inventories into emissions inputs for each modeling sector in a format compatible with CMAQ. When preparing emissions for CMAQ, emissions for each sector are processed separately through SMOKE, and then merged to combine the model-ready, sector-specific 2-D gridded emissions across sectors.

The emissions modeling step for chemical speciation creates the “model species” needed by CMAQ for a specific chemical mechanism. These model species are either individual chemical compounds (i.e., “explicit species”) or groups of species (i.e., “lumped species”). The chemical mechanism used for the 2016v7.2 beta platform is the CB6 mechanism<sup>45</sup>. This platform generates the PM<sub>2.5</sub> model species associated with the CMAQ Aerosol Module version 6 (AE6). See Section 3.2 of the *Preparation of Emissions Inventories for the Version 7.2 2016 North American Emissions Modeling Platform Technical Support Document*<sup>46</sup> for more information about chemical speciation in the 2016v7.2 beta platform.

Temporal allocation is the process of distributing aggregated emissions to a finer temporal resolution, thereby converting annual emissions to hourly emissions as is required by CMAQ. While the total emissions are important, the timing of the occurrence of emissions is also essential for accurately simulating ozone, PM, and other pollutant concentrations in the atmosphere. Many emissions inventories are annual or monthly in nature. Temporal allocation takes these aggregated emissions and distributes the emissions to the hours of each day. This process is typically done by applying temporal profiles to the inventories in this order: monthly, day of the week, and diurnal, with monthly and day-of-week profiles applied only if the inventory is not already at that level of detail. See Section 3.3 of the *Preparation of Emissions Inventories for the Version 7.2 2016 North American Emissions Modeling Platform Technical Support Document*<sup>46</sup> for more information about the profiles used to temporally allocate emissions to the 2016v7.2 beta platform.

Spatial allocation is the process of distributing aggregated emissions to a finer spatial resolution, as is required by CMAQ. There are more than 100 spatial surrogates available for spatially allocating U.S. county-level emissions to the 12-km grid cells used by the air quality model. See Section 3.4 of the *Preparation of Emissions Inventories for the Version 7.2 2016 North American Emissions Modeling Platform Technical Support Document*<sup>46</sup> for a description of the spatial surrogates used for allocating county-level emissions in the 2016v7.2 beta platform.

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<sup>45</sup> Yarwood, G., et al. (2010) Updates to the Carbon Bond Chemical Mechanism for Version 6 (CB6). Presented at the 9th Annual CMAS Conference, Chapel Hill, NC. Available at [https://www.cmascenter.org/conference/2010/abstracts/emery\\_updates\\_carbon\\_2010.pdf](https://www.cmascenter.org/conference/2010/abstracts/emery_updates_carbon_2010.pdf)

<sup>46</sup> USEPA (2019). *Technical Support Document: Preparation of Emissions Inventories for the Version 7.2 2016 North American Emissions Modeling Platform*. Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency, Research Triangle Park, NC. September 2019. [https://www.epa.gov/sites/production/files/2019-09/documents/2016v7.2\\_regionalhaze\\_emismod\\_tsd\\_508.pdf](https://www.epa.gov/sites/production/files/2019-09/documents/2016v7.2_regionalhaze_emismod_tsd_508.pdf)

## 6.7 Air Quality Modeling Methodology

CMAQ was run to generate hourly concentration predictions for ozone, PM<sub>2.5</sub> component species, nitrogen and sulfate deposition, nitrogen dioxide, carbon monoxide, and a subset of air toxics (formaldehyde, acetaldehyde, acrolein, benzene, 1,3-butadiene, and naphthalene) for each grid cell in the modeling domain.

The 12-kilometer (km) CMAQ modeling domain was modeled for the entire year of 2016. The 12 km domain simulations included a “ramp-up” period, comprised of 10 days before the beginning of the annual simulation, to mitigate the effects of initial concentrations. The ramp-up period is not included in the output analyses.

For the 8-hour ozone results, we are only using modeling results from the period between May 1 and September 30, 2016. This 153-day period generally conforms to the ozone season across most parts of the U.S. and contains the majority of days with observed high ozone concentrations in 2016. Data from the entire year were utilized when estimating nitrogen and sulfate deposition, visibility, and PM<sub>2.5</sub>, nitrogen dioxide, carbon monoxide, and toxics impacts.

### 6.7.1 Air Quality Model

CMAQ is a non-proprietary computer model that simulates the formation and fate of photochemical oxidants, primary and secondary PM concentrations, acid deposition, and air toxics, over regional and urban spatial scales for given inputs of meteorological conditions and emissions. CMAQ includes numerous science modules that simulate the emission, production, decay, deposition and transport of organic and inorganic gas-phase and particle pollutants in the atmosphere. The CMAQ model is a well-known and well-respected tool and has been used in numerous national and international applications.<sup>47,48,49,50,51</sup>

This 2016v7.2 beta platform used the most recent multi-pollutant CMAQ code available at the time of air quality modeling (CMAQ version 5.2.1).<sup>52</sup> The 2016 CMAQ runs utilized the CB6r3 chemical mechanism (Carbon Bond with linearized halogen chemistry) for gas-phase chemistry, and AERO6 (aerosol model with non-volatile primary organic aerosol) for aerosols. CMAQ

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<sup>47</sup> Hogrefe, C., et al. (2004) Simulating regional-scale ozone climatology over the eastern United States: model evaluation results. *Atmos. Environ.*, 38(17), 2627-2638.

<sup>48</sup> USEPA (2016). Air Quality Modeling Technical Support Document: Heavy-Duty Vehicle Greenhouse Gas Phase 2 Final Rule. EPA-420-R-16-007.

<sup>49</sup> Lin, M., et al. (2008) Long range transport of acidifying substances in East Asia Part I: Model evaluation and sensitivity studies. *Atmos. Environ.*, 42(24), 5939-5955.

<sup>50</sup> USEPA (2009). *Technical Support Document for the Proposal to Designate an Emissions Control Area for Nitrogen Oxides, Sulfur Oxides, and Particulate Matter*: EPA-420-R-007, 329 pp. <https://19january2017snapshot.epa.gov/sites/production/files/2016-09/documents/420r09007.pdf>

<sup>51</sup> Simon, H., Baker, K., Phillips, S., 2012: Compilation and interpretation of photochemical model performance statistics published between 2006 and 2012. *Atmos. Environ.* 61, 124-139.

<sup>52</sup> CMAQ version 5.2.1: doi:10.5281/ <https://zenodo.org/record/1212601> Model code for CMAQ v5.2.1 is also available from the Community Modeling and Analysis System (CMAS) at: <http://www.cmascenter.org>



versions 5.0.2 and 5.1 were most recently peer-reviewed in September of 2015 for the U.S. EPA.<sup>53</sup>

## 6.7.2 Model Domain and Configuration

The CMAQ modeling analyses used a domain covering the continental United States, as shown in Figure 6.1. This single domain covers the entire continental U.S. (CONUS) and large portions of Canada and Mexico using 12 km × 12 km horizontal grid spacing. The 2016 simulation used a Lambert Conformal map projection centered at (-97, 40) with true latitudes at 33 and 45 degrees north. The model extends vertically from the surface to 50 millibars (approximately 17,600 meters) using a sigma-pressure coordinate system with 35 vertical layers. Table 6.10 provides some basic geographic information regarding the CMAQ domains and Table 6.11 provides the vertical layer structure for the CMAQ domain.

**Table 6.10 Geographic elements of domains used in air quality modeling**

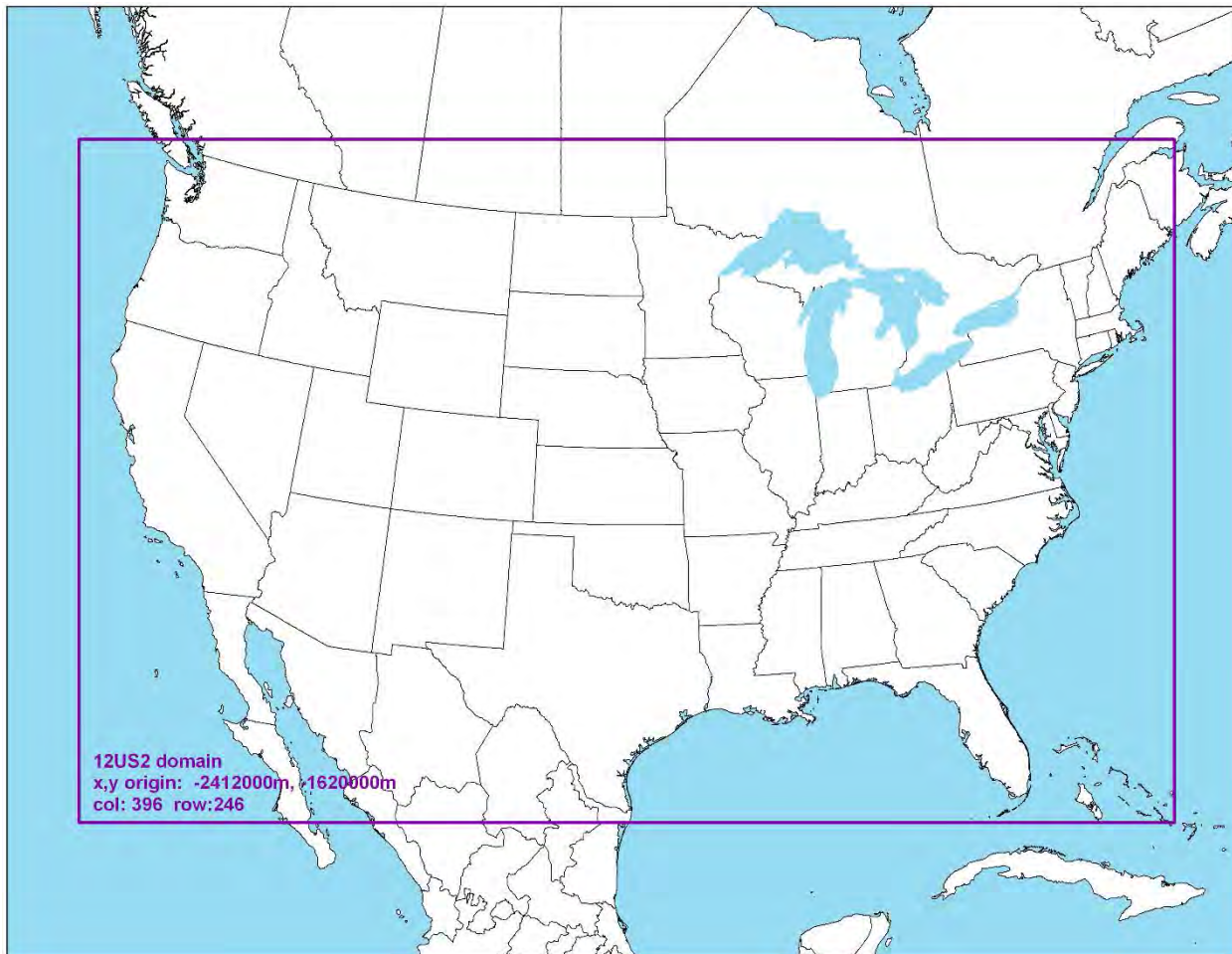
	<b>CMAQ Modeling Configuration</b>
<b>Grid Resolution</b>	12 km National Grid
<b>Map Projection</b>	Lambert Conformal Projection
<b>Coordinate Center</b>	97 deg W, 40 deg N
<b>True Latitudes</b>	33 deg N and 45 deg N
<b>Dimensions</b>	396 × 246 × 35
<b>Vertical extent</b>	35 Layers: Surface to 50 millibar level (see Table 6.11)

**Table 6.11 Vertical layer structure for 2016 CMAQ anti-backsliding simulations**

Vertical Layers	Sigma P	Pressure (mb)	Approximate Height (m)
35	0.0000	50.00	17,556
34	0.0500	97.50	14,780
33	0.1000	145.00	12,822
32	0.1500	192.50	11,282
31	0.2000	240.00	10,002
30	0.2500	287.50	8,901
29	0.3000	335.00	7,932
28	0.3500	382.50	7,064
27	0.4000	430.00	6,275
26	0.4500	477.50	5,553
25	0.5000	525.00	4,885
24	0.5500	572.50	4,264

<sup>53</sup> Moran, M.D., et al. (2015) Final Report: Fifth Peer Review of the CMAQ Model, [https://www.epa.gov/sites/production/files/2016-11/documents/cmaq\\_fifth\\_review\\_final\\_report\\_2015.pdf](https://www.epa.gov/sites/production/files/2016-11/documents/cmaq_fifth_review_final_report_2015.pdf). This peer review was focused on CMAQ v5.0.2, which was released in May 2014, as well as CMAQ v5.1, which was released in October 2015. It is available from the Community Modeling and Analysis System (CMAS) as well as previous peer-review reports at: <http://www.cmascenter.org>

23	0.6000	620.00	3,683
22	0.6500	667.50	3,136
21	0.7000	715.00	2,619
20	0.7400	753.00	2,226
19	0.7700	781.50	1,941
18	0.8000	810.00	1,665
17	0.8200	829.00	1,485
16	0.8400	848.00	1,308
15	0.8600	867.00	1,134
14	0.8800	886.00	964
13	0.9000	905.00	797
12	0.9100	914.50	714
11	0.9200	924.00	632
10	0.9300	933.50	551
9	0.9400	943.00	470
8	0.9500	952.50	390
7	0.9600	962.00	311
6	0.9700	971.50	232
5	0.9800	981.00	154
4	0.9850	985.75	115
3	0.9900	990.50	77
2	0.9950	995.25	38
1	0.9975	997.63	19
0	1.0000	1000.00	0



**Figure 6.1 Map of the CMAQ 12 km modeling domain (noted by the purple box)**

### 6.7.3 Model Inputs

The key inputs to the CMAQ model include emissions from anthropogenic and biogenic sources, meteorological data, and initial and boundary conditions.

The onroad and nonroad emissions inputs used for the 2016 “with-RFS” and “pre-RFS” scenarios are summarized in Section 6.2 and Section 6.3 of this document, respectively, and emissions inputs for other sectors are described in Section 6.5 and in the documentation for the 2016v7.2 beta modeling platform.<sup>54</sup>

<sup>54</sup> USEPA (2019). *Technical Support Document: Preparation of Emissions Inventories for the Version 7.2 2016 North American Emissions Modeling Platform*. Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency, Research Triangle Park, NC. September 2019. [https://www.epa.gov/sites/production/files/2019-09/documents/2016v7.2\\_regionalhaze\\_emismod\\_tsd\\_508.pdf](https://www.epa.gov/sites/production/files/2019-09/documents/2016v7.2_regionalhaze_emismod_tsd_508.pdf)

The CMAQ meteorological input files were derived from simulations of the Weather Research and Forecasting Model (WRF) version 3.8 for the entire 2016 year.<sup>55,56</sup> The WRF Model is a state-of-the-science mesoscale numerical weather prediction system developed for both operational forecasting and atmospheric research applications.<sup>57</sup> The meteorological outputs from WRF were processed to create 12 km model-ready inputs for CMAQ using the Meteorology-Chemistry Interface Processor (MCIP) version 4.3. These inputs included hourly varying horizontal wind components (i.e., speed and direction), temperature, moisture, vertical diffusion rates, and rainfall rates for each grid cell in each vertical layer.<sup>58</sup>

The boundary and initial species concentrations were provided by a northern hemispheric CMAQ modeling platform for the year 2016.<sup>59,60</sup> The hemispheric-scale platform uses a polar stereographic projection at 108 km resolution to completely and continuously cover the northern hemisphere for 2016. Meteorology is provided by WRF v3.8. Details on the emissions used for hemispheric CMAQ can be found in the 2016 hemispheric emissions modeling platform TSD.<sup>61</sup> The atmospheric processing (transformation and fate) was simulated by CMAQ (v5.2.1) using the CB6r3 and the aerosol model with non-volatile primary organic carbon (AE6nvPOA). The CMAQ model also included the on-line windblown dust emission sources (excluding agricultural land), which are not always included in the regional platform but are important for large-scale transport of dust.

#### 6.7.4 Ozone and PM<sub>2.5</sub> Fused Fields

Fused fields are spatial fields, or surfaces, of gridded modeled concentrations (within the 12 km modeling grid used in this analysis) where the model output has been adjusted using monitored data. The fused fields use ambient concentration data from monitors to adjust the modeled concentration in each grid cell to match observed data at locations of monitoring sites. In grid cells where monitor data does not exist, data is interpolated between monitors using the enhanced Veronoi Neighbor Average (eVNA) method to create an uninterrupted surface of monitored concentrations which can be used to adjust the modeled data.

This results in a gridded future-year projection which accounts for measured values and is a way to attempt to minimize model bias problems posed by imperfect model performance on

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<sup>55</sup> Skamarock, W.C., et al. (2008) A Description of the Advanced Research WRF Version 3.

<https://opensky.ucar.edu/islandora/object/technotes:500>

<sup>56</sup> USEPA (2019). Meteorological Model Performance for Annual 2016 Simulation WRF v3.8

[https://www3.epa.gov/ttn/scram/reports/Met\\_Model\\_Performance-2016\\_WRF.pdf](https://www3.epa.gov/ttn/scram/reports/Met_Model_Performance-2016_WRF.pdf). EPA-454/R-19-010.

<sup>57</sup> <https://www.mmm.ucar.edu/weather-research-and-forecasting-model>

<sup>58</sup> Byun, D.W., Ching, J. K.S. (1999). Science algorithms of EPA Models-3 Community Multiscale Air Quality (CMAQ) modeling system, EPA/600/R-99/030, Office of Research and Development. Please also see:

<https://www.cmascenter.org/>

<sup>59</sup> Henderson, B., et al. (2018) Hemispheric-CMAQ Application and Evaluation for 2016, Presented at 2019 CMAS Conference, available [https://cmascenter.org/conference//2018/slides/0850\\_henderson\\_hemispheric-cmaq\\_application\\_2018.pptx](https://cmascenter.org/conference//2018/slides/0850_henderson_hemispheric-cmaq_application_2018.pptx)

<sup>60</sup> Mathur, R., et al. (2017) Extending the Community Multiscale Air Quality (CMAQ) modeling system to hemispheric scales: overview of process considerations and initial applications, Atmos. Chem. Phys., 17, 12449-12474, <https://doi.org/10.5194/acp-17-12449-2017>.

<sup>61</sup> USEPA (2019). Technical Support Document: Preparation of Emissions Inventories for the Version 7.1 2016 Hemispheric Emissions Modeling Platform. Office of Air Quality Planning and Standards.

individual days. The monitoring network does not allow fused fields to be generated for all pollutants, but ozone and PM<sub>2.5</sub> had enough monitored ambient data coverage to make it feasible to create fused field surfaces. Additional information on creating the fused fields can be found in the Appendix, see Section 9.2.

### 6.7.5 CMAQ Evaluation

The CMAQ predictions for ozone, fine particulate matter, sulfate, nitrate, ammonium, organic carbon, elemental carbon, nitrogen and sulfur deposition, and specific air toxics (formaldehyde, acetaldehyde, acrolein, benzene, 1,3-butadiene, and naphthalene) from the 2016 “with-RFS” scenario were compared to measured concentrations in order to evaluate the ability of the modeling platform to replicate observed concentrations. This evaluation was comprised of statistical and graphical comparisons of paired modeled and observed data. Details on the model performance evaluation including a description of the methodology, the model performance statistics, and results are provided in the Appendix, Section 9.3.

## 7 Air Quality Modeling Results

As described in Section 3, this study assessed the air quality impact of vehicle and engine emissions in 2016 for two scenarios, a “pre-RFS” scenario where renewable fuel use was at approximately 2005 levels and a “with-RFS” scenario where renewable fuel use was at 2016 levels. This section of the report presents modeled changes in ambient concentrations of air pollutants when comparing the “pre-RFS” and “with-RFS” scenarios.

- Decreases in concentration mean that the “with-RFS” scenario decreases the pollutant concentration compared to the “pre-RFS” scenario.
- Increases in concentration mean that the “with-RFS” scenario increases the pollutant concentration compared to the “pre-RFS” scenario.

Everything in the two modeled scenarios was held constant except the onroad and nonroad inventories, which reflected the differing fuel supplies used to develop the emissions. This includes the vehicle and engine population, activity, base emission rates, and meteorological data (reflecting calendar year 2016 conditions) and the emissions for all other sources, including boundary conditions and initial conditions used in the air quality modeling methodology. This study assumed that California fuels were the same in both the “pre-RFS” and “with-RFS” scenarios, and as a result, we did not model California.

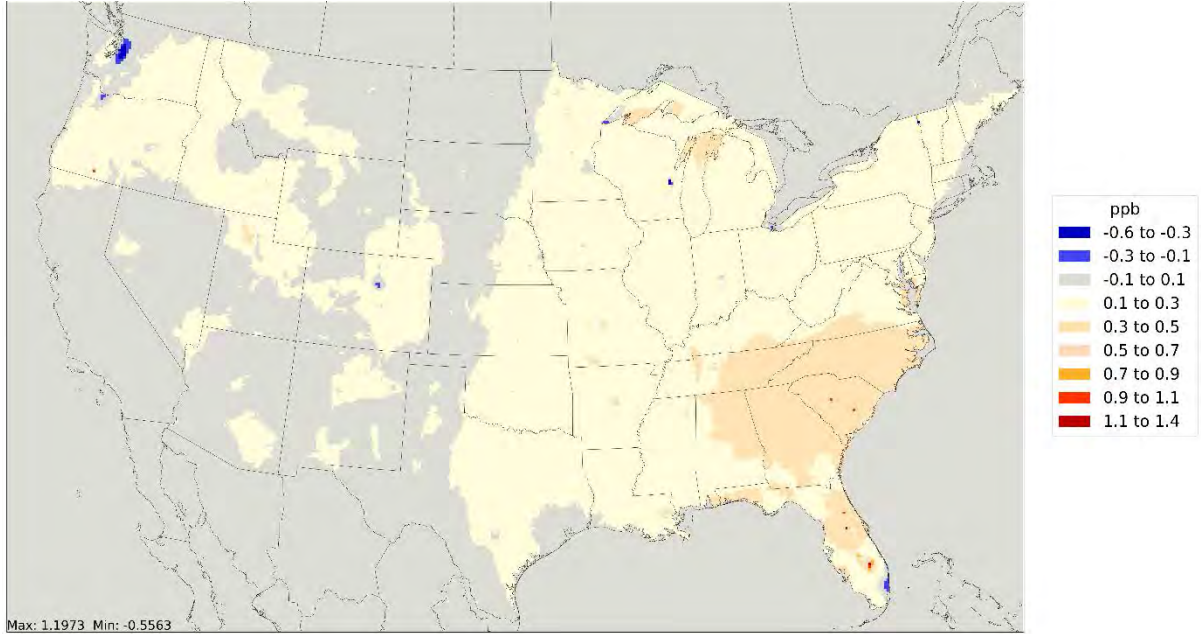
### 7.1 Ozone

Figure 7.1 and Figure 7.2 show the absolute change and percent change in the maximum average 8-hour ozone concentrations when comparing the “with-RFS” scenario to the “pre-RFS” scenario. The results shown are for the time period of May 1<sup>st</sup> through September 30<sup>th</sup>, 2016.<sup>62</sup> Compared to the “pre-RFS” scenario, the “with-RFS” scenario increases ozone concentrations in

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<sup>62</sup> The months of May to September are commonly known as the ozone season and represent the timeframe when most high ozone concentrations occur.

the eastern United States, particularly the southeast, while ozone concentrations are unchanged in some areas of the western United States. Some localized decreases also occur. Tabular results for all grid cells are included with the online supplemental materials.<sup>63</sup>



**Figure 7.1 Change in absolute concentrations of 8-hour maximum average ozone between “pre-RFS” and “with-RFS” scenarios**

<sup>63</sup> <https://www.epa.gov/renewable-fuel-standard-program/anti-backsliding-determination-and-study>





**Figure 7.2 Percent change in concentrations of 8-hour maximum average ozone between “pre-RFS” and “with-RFS” scenarios**

As presented in Sections 6.2 and 6.3, emissions of NO<sub>x</sub> and VOC increase from onroad and nonroad vehicles in the “with-RFS” scenario, as compared to the “pre-RFS” scenario. Relatively small amounts of NO<sub>x</sub> enable ozone to form rapidly when VOC levels are relatively high; such conditions are called “NO<sub>x</sub>-limited.” Rural areas are usually NO<sub>x</sub>-limited, due to the relatively large amounts of biogenic VOC emissions in such areas. The southeastern United States has high levels of biogenic VOC emissions and is known as a NO<sub>x</sub>-limited region, so it is likely that the modeled increases in NO<sub>x</sub> emissions lead to the ozone increases associated with the “with-RFS” scenario.<sup>64</sup>

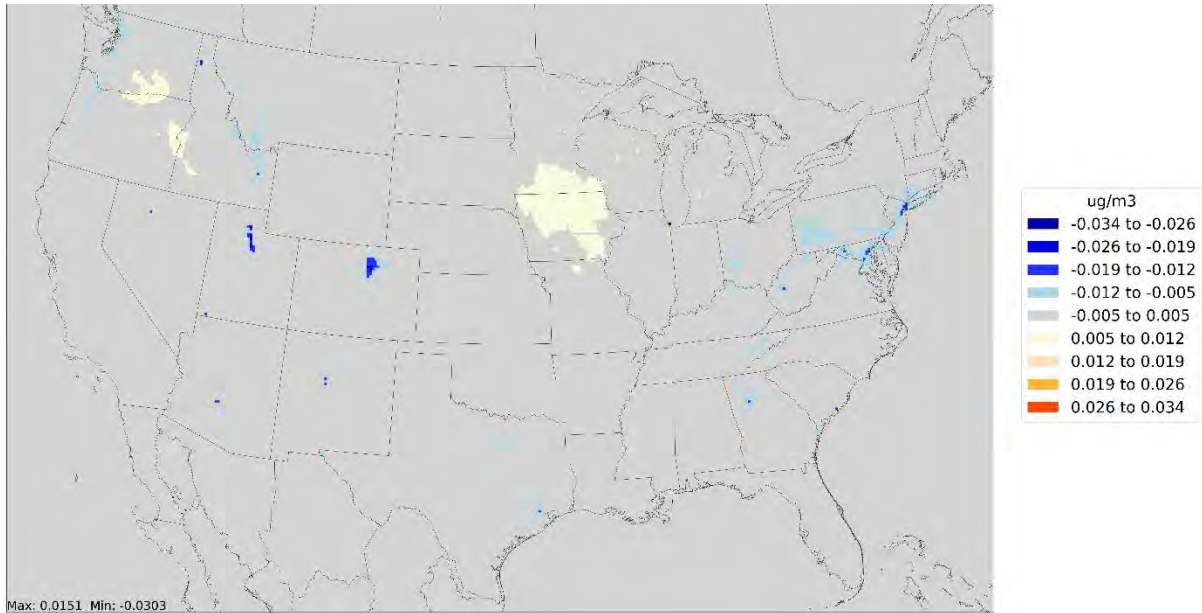
When NO<sub>x</sub> levels are relatively high and VOC levels relatively low, NO<sub>x</sub> forms inorganic nitrates (i.e., particles) but relatively little ozone. Such conditions are called “VOC-limited.” Under these conditions, increases in NO<sub>x</sub> can decrease local ozone. The localized decreases shown in Figure 7.1 and Figure 7.2 are likely occurring in VOC-limited areas due to this tendency for NO<sub>x</sub> to form particles instead of ozone under certain circumstances.

## 7.2 Particulate Matter

Figure 7.3 and Figure 7.4 show the absolute change and percent change that would occur in the modeled annual average 2016 PM<sub>2.5</sub> concentrations when comparing the “with-RFS” scenario to the “pre-RFS” scenario. Figure 7.5 presents the absolute change in the modeled average annual, average January, and average July 2016 PM<sub>2.5</sub> concentrations when the “with-RFS” scenario is compared to the “pre-RFS” scenario. PM<sub>2.5</sub> concentrations remain relatively unchanged in most of the United States, but the “with-RFS” scenario results in PM<sub>2.5</sub> increases in some areas and

<sup>64</sup> Simon *et al.* (2014). Ozone Trends Across the United States Over a Period of Decreasing NO<sub>x</sub> and VOC Emissions. [dx.doi.org/10.1021/es504514z](https://doi.org/10.1021/es504514z) | Environ. Sci. Technol. 2015, 49, 186–195

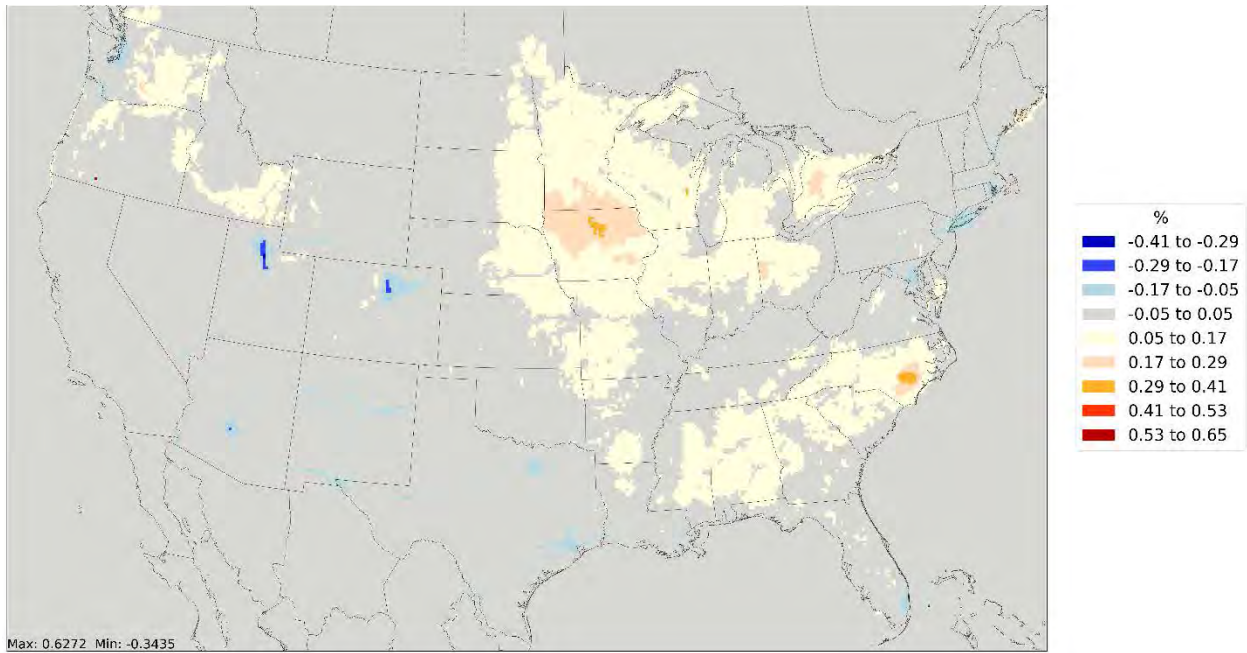
some localized decreases in other areas, as compared to the “pre-RFS” scenario. The monthly maps have larger maximum increases and decreases because they are averaged over a shorter time period. The maximum increases and decreases in January are larger than they are in July, potentially due to winter inversions. Tabular results for all grid cells are included with the online supplemental materials.<sup>65</sup>



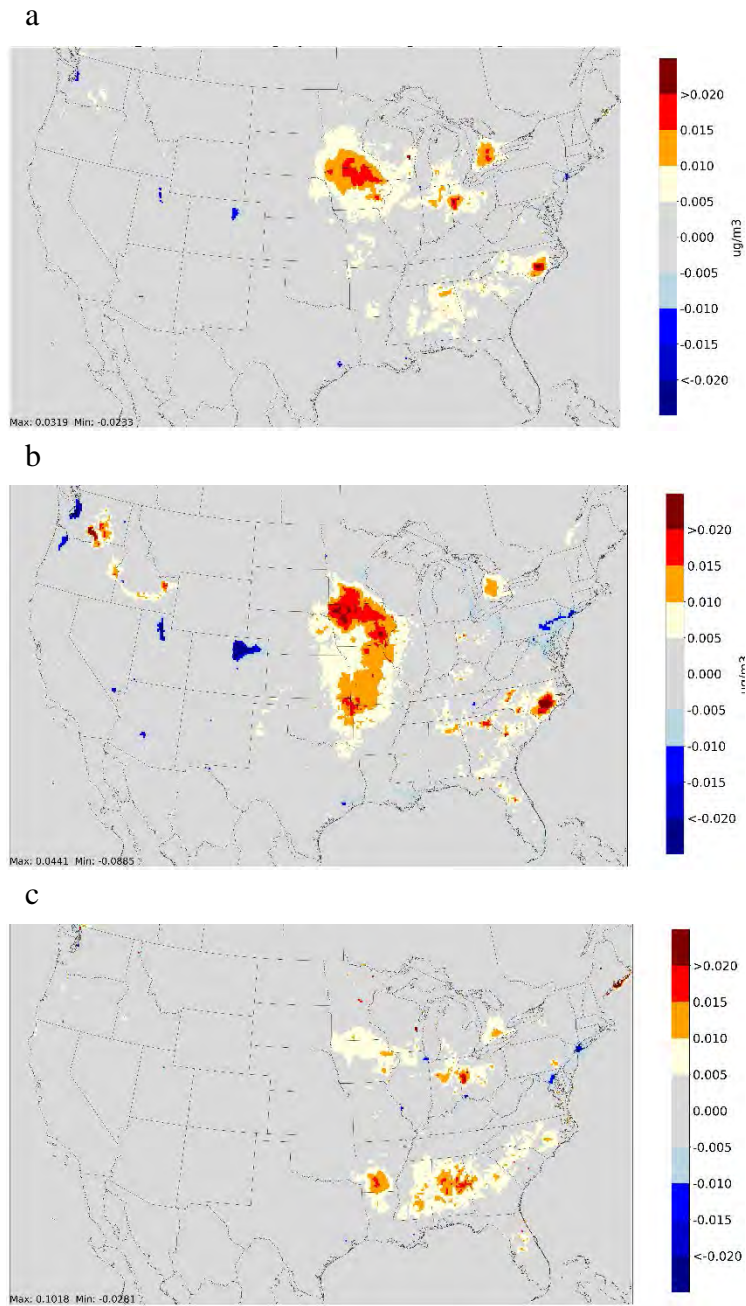
**Figure 7.3 Absolute change in average annual 2016 PM<sub>2.5</sub> concentrations between “pre-RFS” and “with-RFS” scenarios**

<sup>65</sup> <https://www.epa.gov/renewable-fuel-standard-program/anti-backsliding-determination-and-study>





**Figure 7.4 Percent change in average annual 2016 PM<sub>2.5</sub> concentrations between “pre-RFS” and “with-RFS” scenarios**



**Figure 7.5 Absolute difference in (a) annual average, (b) January average, and (c) July average PM<sub>2.5</sub> concentrations between “pre-RFS” and “with-RFS” scenarios for 2016**

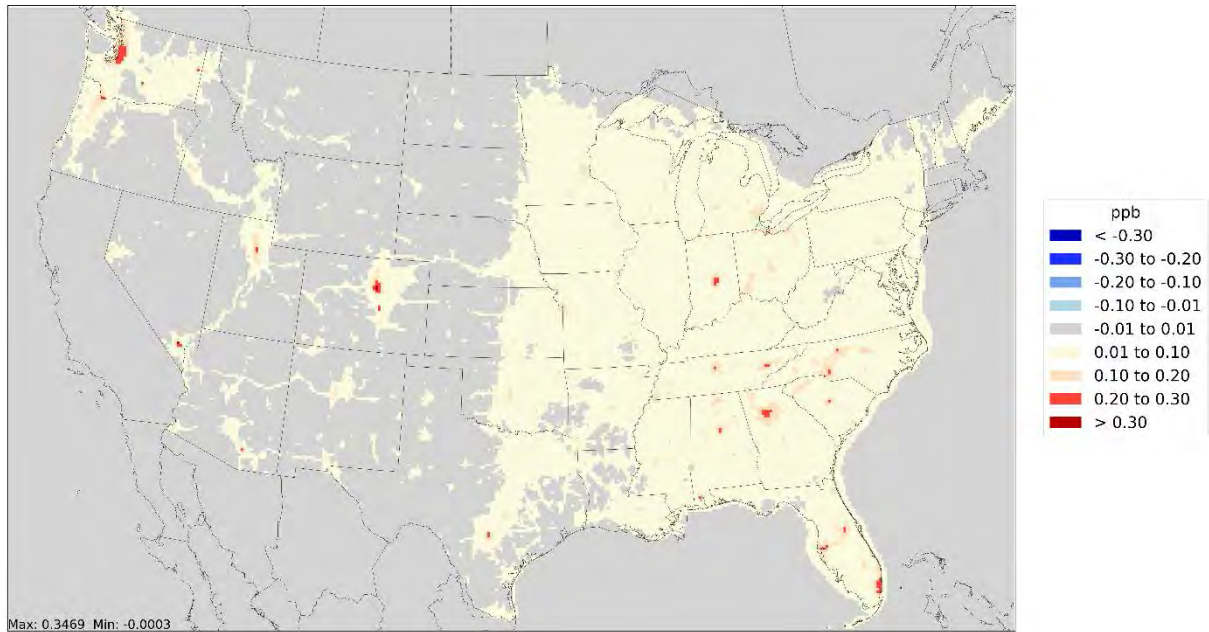
PM<sub>2.5</sub> is emitted directly from vehicles and is also formed through atmospheric chemical reactions of gaseous emissions (e.g., sulfur oxides (SO<sub>x</sub>), nitrogen oxides (NO<sub>x</sub>) and volatile organic compounds (VOCs)); the former is often referred to as “primary” PM<sub>2.5</sub>, and the latter as “secondary” PM<sub>2.5</sub>. Particles can remain in the atmosphere for days to weeks and travel through the atmosphere hundreds to thousands of kilometers, and particle concentration and composition is affected by several weather-related factors, such as temperature, clouds, humidity, and wind. Thus, PM<sub>2.5</sub> concentrations are made up of a complex mixture of different components including sulfates, nitrates, and organic compounds, and these components vary over time and space. On average, the “with-RFS” scenario has higher emissions of NO<sub>x</sub>, SO<sub>2</sub>, and VOC, and lower emissions of primary PM<sub>2.5</sub>, compared to the “pre-RFS” scenario. (See emission summaries in Section 6.) It is likely that the increased PM<sub>2.5</sub> concentrations associated with the “with-RFS” scenario are due to increases in secondary PM<sub>2.5</sub> which outweigh any decreases in primary PM<sub>2.5</sub> concentrations in those areas, and areas with decreased PM<sub>2.5</sub> concentrations are due to decreases in primary PM<sub>2.5</sub> concentrations which outweigh any increases in secondary PM<sub>2.5</sub> concentrations in those areas.

### 7.3 Nitrogen Dioxide

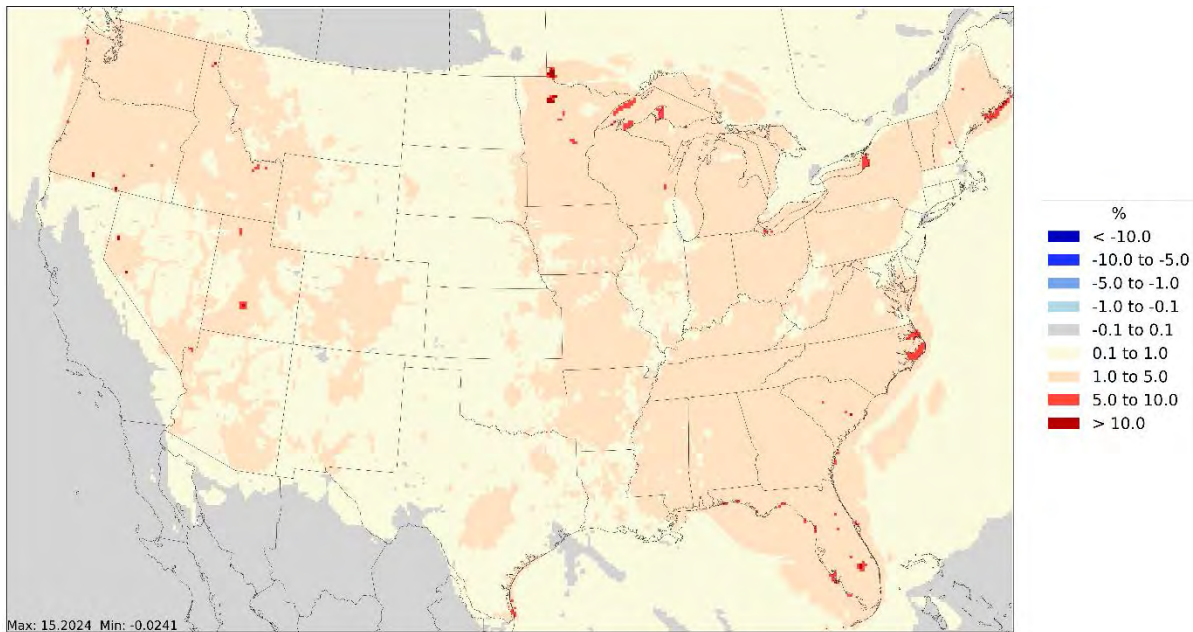
Figure 7.6 and Figure 7.7 show the absolute change and percent change in the annual average NO<sub>2</sub> concentrations when comparing the “with-RFS” scenario to the “pre-RFS” scenario. Tabular results for all grid cells are included with the online supplemental materials.<sup>66</sup> The “with-RFS” scenario results in increases across the eastern U.S. and in some areas in the western U.S.; with larger increases in some urban areas compared to the “pre-RFS” scenario. These absolute increases in annual average NO<sub>2</sub> concentrations are also evident when comparing the “pre-RFS” scenario to the “with-RFS” scenario as a percent difference. Additional monthly (January and July 2016) average NO<sub>2</sub> difference maps, both absolute difference and percent difference, are available in the Appendix, Section 9.4.

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<sup>66</sup> <https://www.epa.gov/renewable-fuel-standard-program/anti-backsliding-determination-and-study>



**Figure 7.6 Absolute change in average annual 2016 NO<sub>2</sub> concentrations between “pre-RFS” and “with-RFS” scenarios**



**Figure 7.7 Percent change in average annual 2016 NO<sub>2</sub> concentrations between “pre-RFS” and “with-RFS” scenarios**

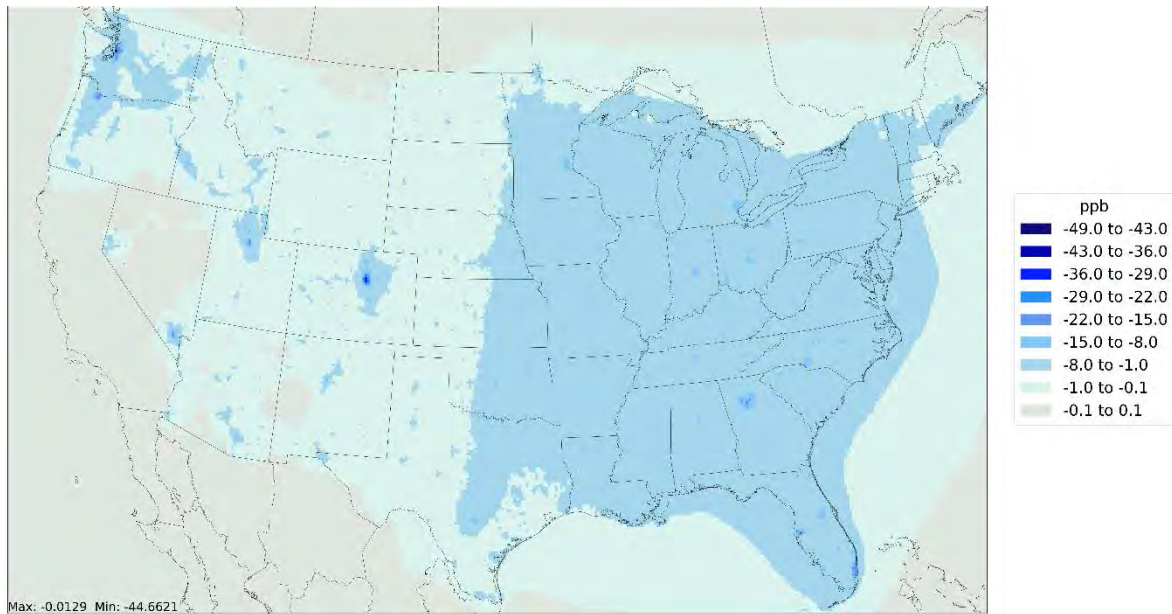
## 7.4 Carbon Monoxide

Figure 7.8 and Figure 7.9 show the absolute change and percent change in the annual average CO concentrations when comparing the “with-RFS” scenario to the “pre-RFS” scenario. Tabular results for all grid cells are included with the online supplemental materials.<sup>67</sup> Compared to the

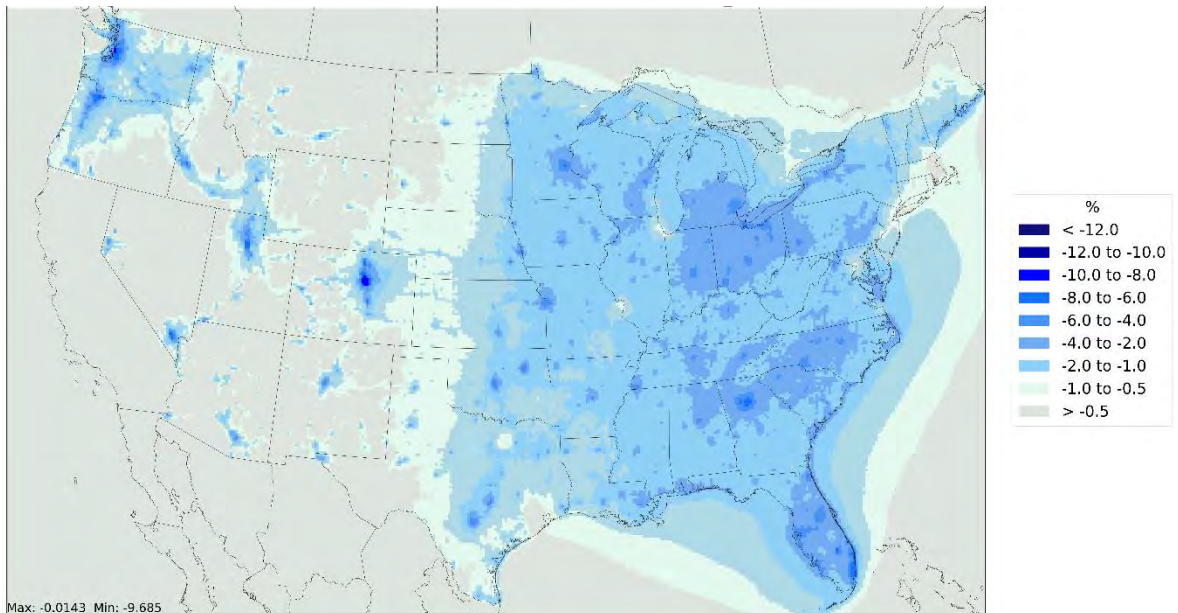
<sup>67</sup> <https://www.epa.gov/renewable-fuel-standard-program/anti-backsliding-determination-and-study>



“pre-RFS” scenario, the “with-RFS” scenario results in decreases across the eastern U.S. and some areas in the western U.S. with some larger decreases in some areas. These absolute decreases in annual average CO concentrations are also reflected in the percent difference map for annual average CO concentrations when comparing the “with-RFS” and “pre-RFS” scenarios. Additional monthly (January and July 2016) average CO difference maps, both absolute difference and percent difference, are available in the Appendix, Section 9.4.



**Figure 7.8 Absolute change in average annual 2016 CO concentrations between “pre-RFS” and “with-RFS” scenarios**



**Figure 7.9 Percent change in average annual 2016 CO concentrations between “pre-RFS” and “with-RFS” scenarios**

## 7.5 Air Toxics (acetaldehyde, acrolein, benzene, 1,3-butadiene, formaldehyde, naphthalene)

In this section we describe results of our modeling of air toxics concentrations for the “pre-RFS” and “with-RFS” scenarios. Although there are many compounds which are considered air toxics, we focused on the following six pollutants: acetaldehyde, acrolein, benzene, 1,3-butadiene, formaldehyde and naphthalene. These pollutants have been identified as national or regional-scale cancer and noncancer risk drivers in the 2014 or past National Air Toxics Assessments (NATAs).<sup>68,69</sup> Ambient levels of air toxics pollutants dominated by primary emissions (or a decay product of a directly emitted pollutant), such as benzene and 1,3-butadiene, have the largest impacts. Air toxics that primarily result from photochemical transformation, such as formaldehyde and acetaldehyde, are not impacted as much as those dominated by direct emissions. Additional monthly (January and July 2016) absolute and percent difference maps are available in the Appendix, 9.4 and tabular results for all grid cells are included with the online supplemental materials.<sup>70</sup>

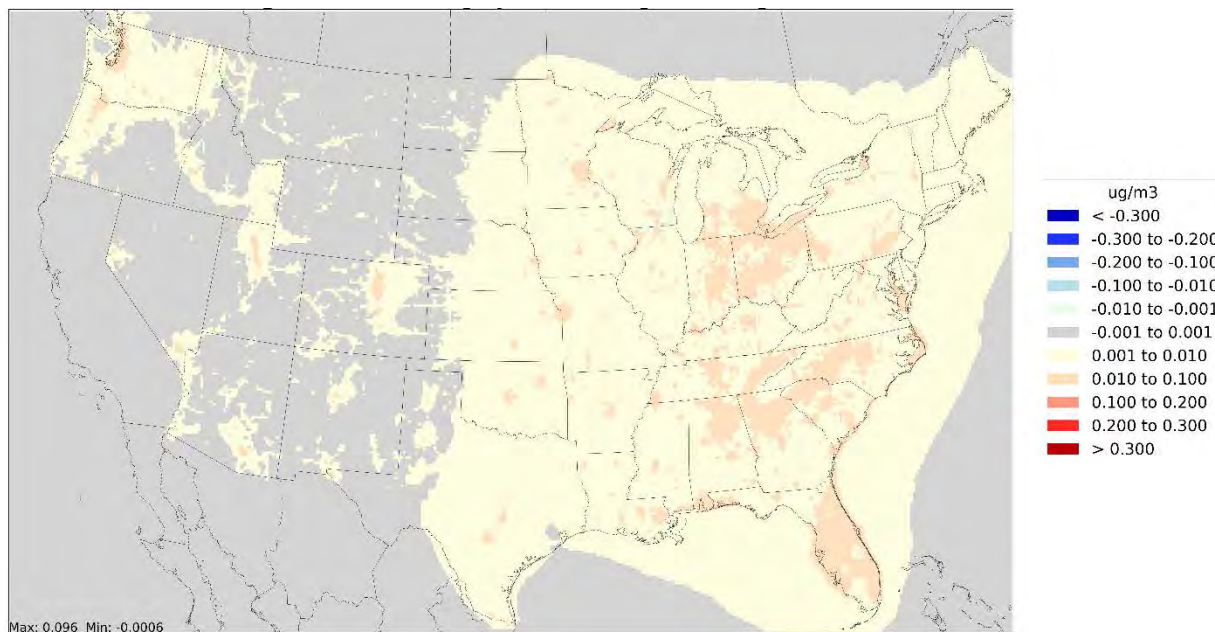
<sup>68</sup> USEPA (2015). 2011 NATA: Assessment Results. <https://www.epa.gov/national-air-toxics-assessment/2011-nata-assessment-results>

<sup>69</sup> USEPA (2018). *Technical Support Document EPA’s 2014 National Air Toxics Assessment*. Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency, Research Triangle Park, NC. August 2018. <https://www.epa.gov/national-air-toxics-assessment/2014-nata-assessment-results>

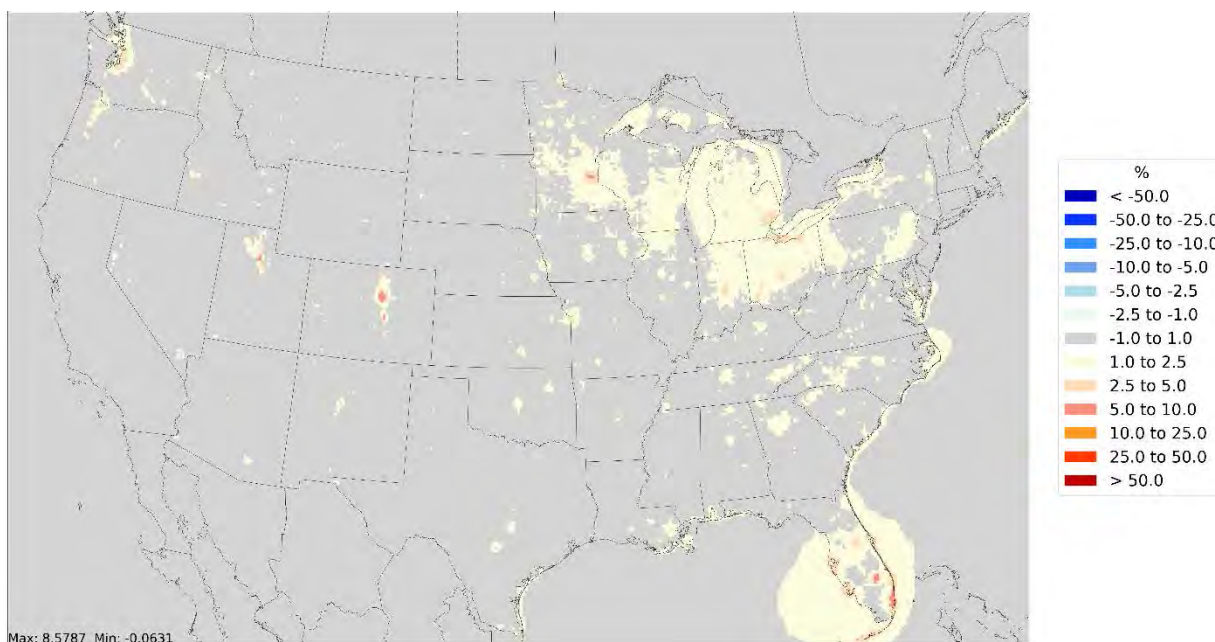
<sup>70</sup> <https://www.epa.gov/renewable-fuel-standard-program/anti-backsliding-determination-and-study>

### 7.5.1 Acetaldehyde

Figure 7.10 and Figure 7.11 show the absolute change and percent change in the annual average acetaldehyde concentrations when comparing the “pre-RFS” scenario to the “with-RFS” scenario. Compared to the “pre-RFS” scenario, the “with-RFS” scenario results in increases across much of the eastern U.S. and some areas in the western U.S., with larger increases in some areas. The percent difference map for annual average acetaldehyde comparing the “with-RFS” and “pre-RFS” scenarios also shows increases with the “with-RFS” scenario, mainly in the upper midwest, Florida, and some urban areas in the western U.S.



**Figure 7.10 Absolute change in average annual 2016 acetaldehyde concentrations between “pre-RFS” and “with-RFS” scenarios**



**Figure 7.11 Percent change in average annual 2016 acetaldehyde concentrations between “pre-RFS” and “with-RFS” scenarios**

## 7.5.2 Acrolein

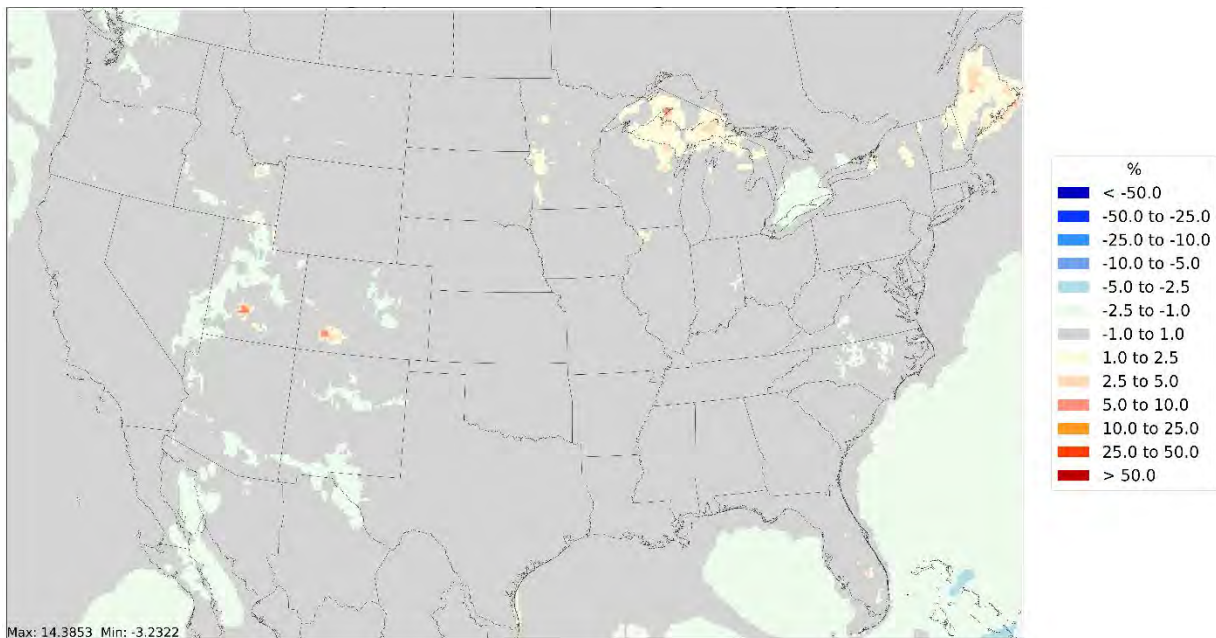
Figure 7.12 and Figure 7.13 show the absolute change and percent change in the annual average acrolein concentrations when comparing the “pre-RFS” scenario to the “with-RFS” scenario. Compared to the “pre-RFS” scenario, the “with-RFS” scenario results in some geographically



limited increases and decreases in the southwestern U.S., upper midwest, and northeastern U.S. on the percent difference map. These percent differences correspond to decreases in absolute concentration of less than 0.001 ug/m<sup>3</sup>, which correspond to the color gray on the map.



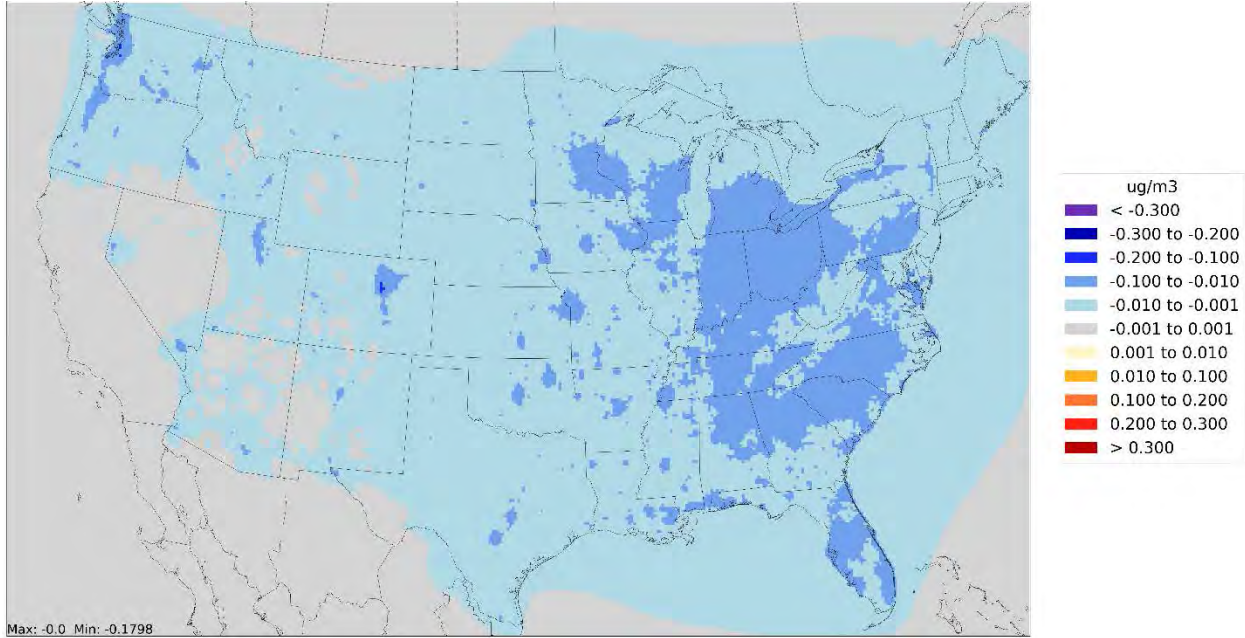
**Figure 7.12 Absolute change in average annual 2016 acrolein concentrations between “pre-RFS” and “with-RFS” scenarios**



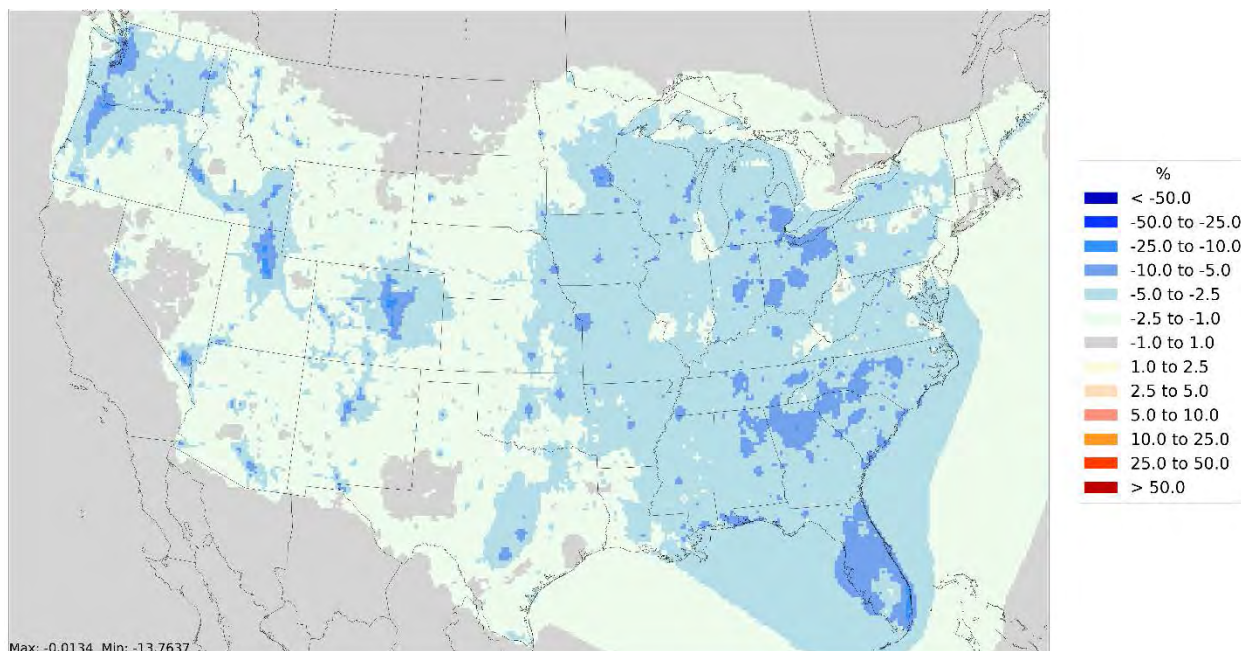
**Figure 7.13 Percent change in average annual 2016 acrolein concentrations between “pre-RFS” and “with-RFS” scenarios**

### 7.5.3 Benzene

Figure 7.14 and Figure 7.15 show the absolute change and percent change in the annual average benzene concentrations when comparing the “with-RFS” scenario to the “pre-RFS” scenario. Compared to the “pre-RFS” scenario, the “with-RFS” scenario results in decreases across much of the country, with larger decreases in some areas. These absolute decreases in annual average benzene concentrations are also reflected in the percent difference map for annual average benzene comparing the “with-RFS” and “pre-RFS” scenarios.



**Figure 7.14 Absolute change in average annual 2016 benzene concentrations between “pre-RFS” and “with-RFS” scenarios**

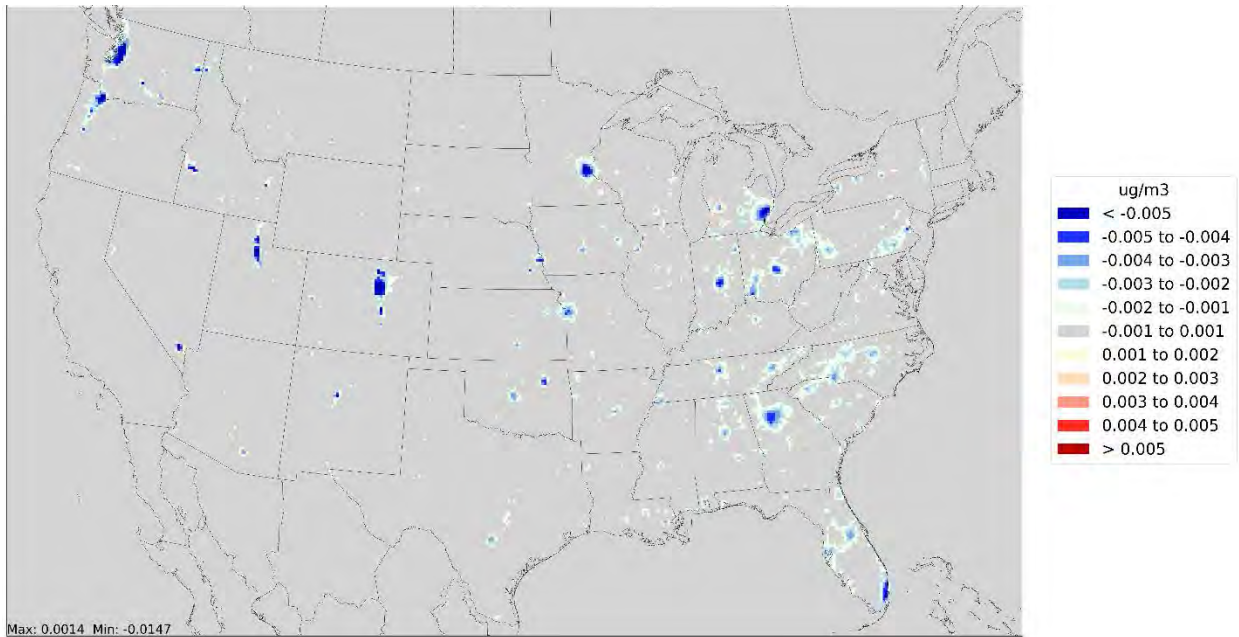


**Figure 7.15 Percent change in average annual 2016 benzene concentrations between “pre-RFS” and “with-RFS” scenarios**

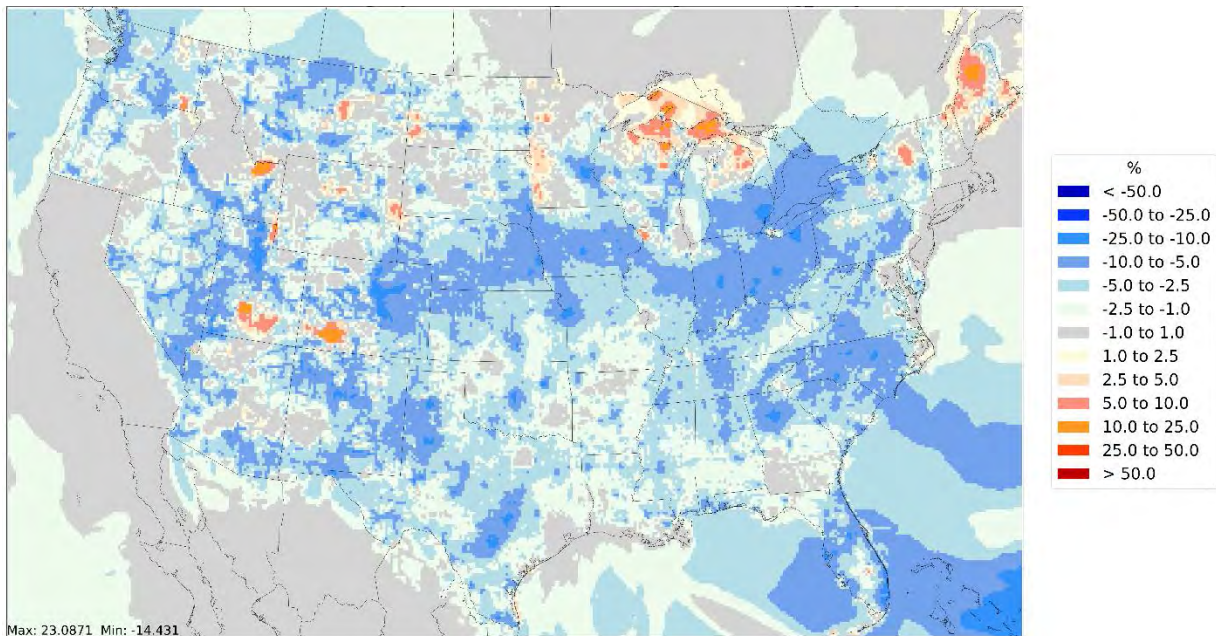
#### 7.5.4 1,3-Butadiene

Figure 7.16 and Figure 7.17 show the absolute change and percent change in the annual average 1,3-butadiene concentrations when comparing the “pre-RFS” scenario to the “with-RFS” scenario. Compared to the “pre-RFS” scenario, the “with-RFS” scenario results in decreases in many urban areas. The 1,3-butadiene annual average percent difference map shows decreases in some areas and increases in other areas when comparing “pre-RFS” and “with-RFS” scenarios.





**Figure 7.16 Absolute change in average annual 2016 1,3-butadiene concentrations between “pre-RFS” and “with-RFS” scenarios**



**Figure 7.17 Percent change in average annual 2016 1,3-butadiene concentrations between “pre-RFS” and “with-RFS” scenarios**

### 7.5.5 Formaldehyde

Figure 7.18 and Figure 7.19 show the absolute change and percent change in the annual average formaldehyde concentrations when comparing the “pre-RFS” scenario to the “with-RFS”

scenario. Compared to the “pre-RFS” scenario, the “with-RFS” scenario results in widespread increases in absolute concentration of formaldehyde across much of the U.S. These absolute differences correspond to percent increases of less than 1% across much of the U.S.



**Figure 7.18 Absolute change in average annual 2016 formaldehyde concentrations between “pre-RFS” and “with-RFS” scenarios**



**Figure 7.19 Percent change in average annual 2016 formaldehyde concentrations between “pre-RFS” and “with-RFS” scenarios**

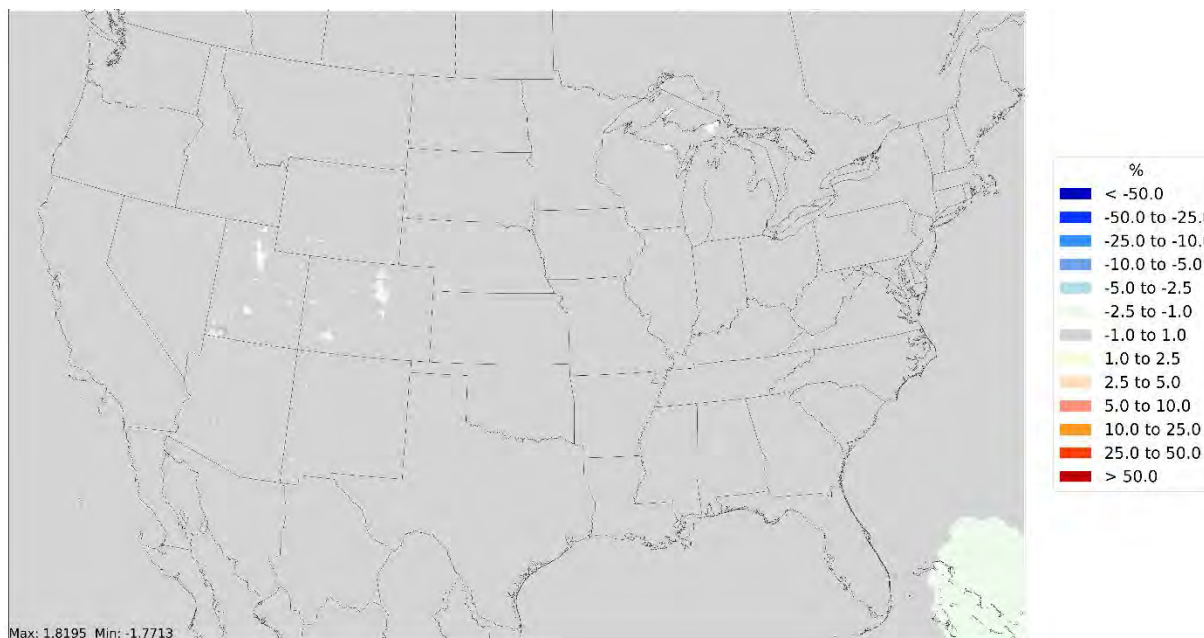
## 7.5.6 Naphthalene

Figure 7.20 and Figure 7.21 show the absolute change and percent change in the annual average naphthalene concentrations when comparing the “pre-RFS” scenario to the “with-RFS” scenario. Compared to the “pre-RFS” scenario, the “with-RFS” scenario results in some geographically limited increases and decreases in the western U.S. and upper midwest on the percent difference map. These percent differences correspond to decreases in absolute concentration of less than  $0.001 \text{ ug/m}^3$ , which correspond to the color gray on the map.



**Figure 7.20 Absolute change in average annual 2016 naphthalene concentrations between “pre-RFS” and “with-RFS” scenarios**





**Figure 7.21 Percent change in average annual 2016 naphthalene concentrations between “pre-RFS” and “with-RFS” scenarios**

## 8 Study Limitations and Uncertainties

In order to assess the air quality impacts of changes in vehicle and engine emissions resulting from required renewable fuel volumes, this study used the best models and methods that were feasible and publicly available for that purpose at the time the study was initiated. However, as with any study, there are inherent limitations and uncertainties. This section identifies some key limitations and uncertainties associated with the study.

### 8.1 Study Scope and Design

This study is narrowly focused on the impacts of required renewable fuel volumes on concentrations of criteria and toxic pollutants due to changes in vehicle and engine emissions; this study is not an examination of the lifecycle impacts of renewable fuels on air quality, greenhouse gases, or other environmental impacts. This study examines only the impacts of renewable fuel volumes on vehicles and engines, and it does not include “upstream” emissions impacts associated with production and distribution of renewable fuels and feedstocks. Specifically, the study holds emissions from all sources constant at 2016 levels in both scenarios, except for gasoline-fueled vehicles and engines (onroad and nonroad) and onroad diesel-fueled vehicles outside the state of California. For these non-California onroad and nonroad sources, only the fuel supply is changed.

This anti-backsliding study examines the impacts of required renewable volumes as compared to a hypothetical case where renewable fuel usage in 2016 was approximately the same as it had been in 2005, before EPCRA was enacted. This study examines impacts for a single retrospective year (2016). By analyzing calendar year 2016, EPA was able to use an existing modeling

platform that includes known renewable fuel volumes for 2016 and fuel properties based on actual data (aggregation of refinery batch reports and fuel surveys). The study does not project future renewable fuel volumes and their impacts, and it does not account for the impacts of the Tier 3 motor vehicle emissions and fuel standards, which took effect in 2017.<sup>71</sup> These standards lowered the sulfur content of gasoline and tightened the emissions standards for onroad motor vehicles, resulting in lower emissions of criteria and toxic pollutants and precursors in 2017 and into the future as more Tier 3-compliant vehicles enter the fleet. These standards are projected to reduce concentrations of ozone, PM<sub>2.5</sub>, NO<sub>2</sub>, toxics (such as acetaldehyde, formaldehyde, acrolein, benzene, 1,3-butadiene, and naphthalene), and other pollutants into the future. By examining 2016, this study also does not reflect the full turnover of the diesel fleet to the most recent highway standards. Thus, the modeling reflects changes in diesel emissions that would be projected to decline into the future.

## 8.2 Data and Model Limitations and Uncertainties

In the absence of consistent and reliable data on biodiesel use across the country, this study assumed in the “with-RFS” scenario that biodiesel usage was at a B5 blend level (5 percent biodiesel) in all onroad diesel fuel nationwide, generally consistent with aggregate usage figures. Under ASTM D975, blends up to 5% can be labeled as “diesel fuel.” This study did not capture the impacts of higher or lower biodiesel blends that may have been occurring in specific areas. This study assumed no biodiesel was being used in the “pre-RFS” scenario, except in California. Furthermore, this study did not assume any E15 in either the “pre-RFS” or “with-RFS” scenario, because of low E15 sales, and lack of data about its use in 2016.

Because California had state fuels regulations that affected fuel properties and usage independent of EPA Act and EISA, this study assumed that California fuels were the same in both the “pre-RFS” and “with-RFS” scenarios, and as a result, we did not model California. More broadly, because of the very limited data on 2005 fuel properties and their spatial distribution, the “pre-RFS” scenario is only a general approximation of 2005 fuels. As a result, the air quality modeling results are illustrative at a broad geographic scale (rather than being locally specific).

With respect to estimating the effects of renewable fuels on emissions, there is much more data available for onroad gasoline vehicles than for nonroad gasoline engines or for diesel vehicles and engines.<sup>72</sup> The impact of ethanol on HC, CO and NO<sub>x</sub> from nonroad gasoline engines is based on three studies from 1991-1997.<sup>73</sup> There is insufficient data to model fuel effects on PM emissions from nonroad gasoline engines. Data on fuel effects for nonroad diesel are also

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<sup>71</sup> USEPA (2014). *Control of Air Pollution From Motor Vehicles: Tier 3 Motor Vehicle Emission and Fuel Standards*, 79 FR 23414 (April 28, 2014).

<sup>72</sup> USEPA (2016). *Fuel Effects on Exhaust Emissions from On-road Vehicles in MOVES2014*. EPA-420-R-16-001. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI. February 2016. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100O5W2.pdf>

<sup>73</sup> USEPA (2005). *Exhaust Emission Effects of Fuel Sulfur and Oxygen on Gasoline Nonroad Engines*. EPA-420-R-05-016. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI. December 2005. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1004L80.pdf>



limited; EPA's last comprehensive evaluation of biodiesel impacts on nonroad exhaust emissions was completed in 2002.<sup>74</sup>

More generally, the MOVES2014b model incorporates the data and analysis available when the model was developed. Updates and improvements have been suggested<sup>75,76,77</sup> and are underway<sup>78</sup> that would likely affect the baseline emissions for both light-duty and heavy-duty emissions, but are unlikely to have a large impact on the individual fuel effects in calendar year 2016.

CMAQ simulates the impact of a very large number of physical and chemical processes on ambient air quality. Physical processes, such as horizontal and vertical mixing, transport, and deposition are all based on data that are continually evolving. In addition, the chemical mechanism specifies a series of chemical reaction pathways and reaction rates. Explicitly tracking the many thousands of chemical compounds and reactions that occur in the atmosphere would be too burdensome computationally for air quality models to be practical. Therefore, a chemical mechanism represents this complexity with only tens or hundreds of "lumped" or representative species and reactions. Such a simplification introduces additional uncertainty.

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<sup>74</sup> USEPA (2002). *A Comprehensive Analysis of Biodiesel Impacts on Exhaust Emissions (Draft Report)*. EPA-420-P-02-001. Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, Ann Arbor, MI. October 2002. <https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1001ZA0.pdf>

<sup>75</sup> Heiken, J. G., M. Hixson and J. Lyons (2016). Review of EPA's MOVES2014 Model. E-101. August 11, 2016. <http://crcsite.wpengine.com/wp-content/uploads/2019/05/FINAL-E101-Report-SR-20160810-w-CRC-Cover-and-Appendices.pdf>

<sup>76</sup> Barth (2018) "MOVES Review Work Group Update", presentation to Mobile Source Technical Subcommittee, May 22, 2018. <https://www.epa.gov/sites/production/files/2018-05/documents/052218-moves-wg-barth.pdf>

<sup>77</sup> Barth (2017) "MOVES Review Work Group Update", presentation to Mobile Source Technical Subcommittee, May 31, 2017. <https://www.epa.gov/sites/production/files/2017-06/documents/053120217-barth.pdf>

<sup>78</sup> Beardsley, et al. (2019) "Updates to EPA's Motor Vehicle Emission Simulator (MOVES)", presentation to CRC Real World Emissions Workshop, March 10, 2019. <https://www.epa.gov/sites/production/files/2019-10/documents/09-17-2019-beardsley.pdf>

## 9 Appendix A: Emissions Modeling and Air Quality Modeling

### 9.1 Chemical Mechanisms in Air Quality Modeling

This analysis examines air quality impacts of criteria pollutants including NO<sub>x</sub>, VOC, CO, PM<sub>2.5</sub>, SO<sub>2</sub>, and air toxics, specifically formaldehyde, acetaldehyde, acrolein, benzene, 1,3-butadiene, and naphthalene. The air toxics were added as explicit model species to the CB6r3 mechanism used in CMAQv5.2.1.<sup>79,80</sup> Emissions of all the pollutants included in the onroad inventories were generated using MOVES VOC emissions and toxic-to-VOC ratios.<sup>81</sup> In addition to direct emissions, photochemical mechanisms are responsible for formation of some of these compounds in the atmosphere from precursor emissions. For some pollutants such as PM, formaldehyde, and acetaldehyde, many photochemical processes are involved. CMAQ therefore also requires inventories for a large number of other air toxics and precursor pollutants. Methods used to develop the air quality inventories can be found on the air emissions modeling platform website.<sup>82</sup>

In the CB6r3 mechanism, the chemistry of thousands of different VOCs in the atmosphere are represented by a much smaller number of model species which characterize the general behavior of a subset of chemical bond types; this condensation is necessary to allow the use of complex photochemistry in a fully three-dimensional air quality model.<sup>83</sup>

Complete combustion of ethanol in fuel produces carbon dioxide (CO<sub>2</sub>) and water (H<sub>2</sub>O). Incomplete combustion produces other air pollutants, such as acetaldehyde and other aldehydes, and the release of unburned ethanol. Ethanol is also present in evaporative emissions. In the atmosphere, ethanol from unburned fuel and evaporative emissions can undergo photodegradation to form aldehydes (acetaldehyde and formaldehyde) and peroxyacetyl nitrate (PAN), and it also plays a role in ground-level ozone formation. Mechanisms for these reactions are included in CMAQ. Additionally, alkenes and other hydrocarbons are considered because any increase in acetyl peroxy radicals due to ethanol increases might be counterbalanced by a decrease in radicals resulting from decreases in other hydrocarbons, particularly alkenes.

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<sup>79</sup> Yarwood, G., Whitten, G. Z., Jung, J., Heo, G., Allen, D. T. (2010). Development, evaluation and testing of version 6 of the Carbon Bond chemical mechanism (CB6), Final report to the Texas Commission on Environmental Quality, Work Order No. 582-7-84005-FY10-26.

<sup>80</sup> Luecken, D.J., Yarwood, G., Hutzell, W.T. (2019). Multipollutant modeling of ozone, reactive nitrogen and HAPs across the continental US with CMAQ-CB6, *Atmos. Environ.*, 201, 15 March 2019, 62-72.

<sup>81</sup> USEPA (2014) Memorandum to Docket EPA-HQ-OAR-2011-0135 by David Choi: Updates to MOVES for the Tier 3 FRM Analysis. Document number EPA-HQ-OAR-2011-0135-5063.

<sup>82</sup> USEPA (2019). *Technical Support Document: Preparation of Emissions Inventories for the Version 7.2 2016 North American Emissions Modeling Platform*. Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency, Research Triangle Park, NC. September 2019. [https://www.epa.gov/sites/production/files/2019-09/documents/2016v7.2\\_regionalhaze\\_emismod\\_tsd\\_508.pdf](https://www.epa.gov/sites/production/files/2019-09/documents/2016v7.2_regionalhaze_emismod_tsd_508.pdf)

<sup>83</sup> Dodge, M.C. (2000). Chemical oxidant mechanisms for air quality modeling: critical review. *Atmospheric Environment* 34, 2103-2130.

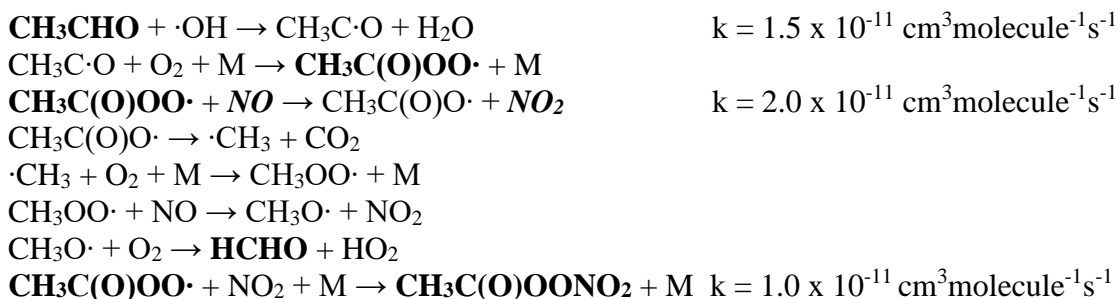
CMAQ includes 63 inorganic reactions to account for the cycling of all relevant oxidized nitrogen species and cycling of radicals, including the termination of NO<sub>2</sub> and formation of nitric acid (HNO<sub>3</sub>) without PAN formation.<sup>84</sup>



The CB6r3 mechanism also includes more than 90 organic reactions and contains numerous updates from CB05 chemical mechanism that include new operators to better represent alkoxy and peroxy radicals, updated reaction rates throughout, additional explicit species to represent long-lived and abundant chemicals and species that form SOA, and additional representation of organic nitrates.<sup>85,86</sup>

### 9.1.1 Acetaldehyde

Acetaldehyde is the main photodegradation product of ethanol, as well as other precursor hydrocarbons. Acetaldehyde is also a product of fuel combustion. In the atmosphere, acetaldehyde can react with the OH radical and O<sub>2</sub> to form the acetyl peroxy radical [CH<sub>3</sub>C(O)OO·].<sup>87</sup> When NO<sub>x</sub> is present in the atmosphere this radical species can then further react with nitric oxide (NO), to produce formaldehyde (HCHO), or with nitrogen dioxide (NO<sub>2</sub>), to produce PAN [CH<sub>3</sub>C(O)OONO<sub>2</sub>]. An overview of these reactions and the corresponding reaction rates are provided below, as published by Atkinson, et al.<sup>88</sup>



Acetaldehyde can also photolyze (hv), which predominantly produces ·CH<sub>3</sub> (which reacts as shown above to form CH<sub>3</sub>OO·) and HCO (which rapidly forms HO<sub>2</sub> and CO):

<sup>84</sup> All rate coefficients presented in this section are listed at 298K and, if applicable, 1 bar of air.

<sup>85</sup> Emery, C., Jung, J., Koo, B., Yarwood, G. (2015). Improvements to CAMx Snow Cover Treatments and Carbon Bond Chemical Mechanism for Winter Ozone. Final report for Utah DAQ, project UDAQ PO 480 52000000001.

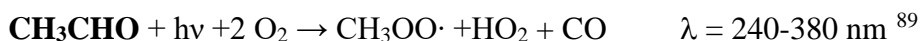
<sup>86</sup> Available at

[https://github.com/USEPA/CMAQ/blob/5.2.1/CCTM/docs/Release\\_Notes/CB6\\_release\\_notes.md#brief-description](https://github.com/USEPA/CMAQ/blob/5.2.1/CCTM/docs/Release_Notes/CB6_release_notes.md#brief-description)

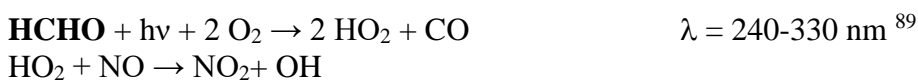
<sup>87</sup> Acetaldehyde is not the only source of acetyl peroxy radicals in the atmosphere. For example, dicarbonyl compounds (methylglyoxal, biacetyl, and others) also form acetyl radicals, which can further react to form peroxyacetyl nitrate (PAN).

<sup>88</sup> Atkinson, R., et al. (2005). Evaluated Kinetic and Photochemical Data for Atmospheric Chemistry - IUPAC Subcommittee on Gas Kinetic Data Evaluation for Atmospheric Chemistry. July 2005 web version.

<http://iupac.pole-ether.fr/>.



As mentioned above,  $\text{CH}_3\text{OO}\cdot$  can react in the atmosphere to produce formaldehyde (HCHO). Formaldehyde is also a product of hydrocarbon combustion. In the atmosphere, the most important reactions of formaldehyde are photolysis and reaction with the OH, with atmospheric lifetimes of approximately 3 hours and 13 hours, respectively.<sup>90,91</sup> Formaldehyde can also react with  $\text{NO}_3$  radical, ground state oxygen atom ( $\text{O}^3\text{P}$ ) and chlorine, although these reactions are much slower. Formaldehyde is removed mainly by photolysis whereas the higher aldehydes, those with two or more carbons such as acetaldehyde, react predominantly with OH radicals. The photolysis of formaldehyde is an important source of new hydroperoxy radicals ( $\text{HO}_2$ ), which can lead to ozone formation and regenerate OH radicals.



Photolysis of HCHO can also proceed by a competing pathway which makes only stable products:  $\text{H}_2$  and CO.

CB6r3 mechanism for acetaldehyde formation warrant a detailed discussion given the increase in vehicle and engine exhaust emissions for this pollutant and ethanol, which can form acetaldehyde in the air. Acetaldehyde is represented explicitly in the CB6r3 chemical mechanism<sup>79,92,93</sup> by the ALD2 model species, which can be both formed from other VOCs and can decay via reactions with oxidants and radicals. The reaction rates for acetaldehyde, as well as for the inorganic reactions that produce and cycle radicals, and the representative reactions of other VOCs have all been updated to be consistent with recommendations in the literature.<sup>89</sup> The decay reactions of acetaldehyde are fewer in number and can be characterized well because they are explicit representations. In CB6r3 acetaldehyde can photolyze or react with molecular oxygen ( $\text{O}^3\text{P}$ ), hydroxyl radical (OH), or nitrate radicals. The reaction rates are based on expert recommendations, and the photolysis rate is from IUPAC recommendations.<sup>89</sup>

In CMAQ v5.2.1, the acetaldehyde that is formed from photochemical reactions is tracked separately from that which is due to direct emission and transport of direct emissions. In CB6r3 there are 25 different reactions that form acetaldehyde in molar yields ranging from 0.02 (ozone reacting with lumped products from isoprene oxidation) to 2.0 (cross reaction of acylperoxy radicals,  $\text{CXO}_3$ ). The specific parent VOCs that contribute the most to acetaldehyde concentrations vary spatially and temporally depending on characteristics of the ambient air, but

<sup>89</sup> Sander, S.P., et al. (2003). Chemical Kinetics and Photochemical Data for use in Atmospheric Studies, Evaluation Number 14. NASA Jet Propulsion Laboratory <https://jpldataeval.jpl.nasa.gov/index.html>.

<sup>90</sup> The 3 h lifetime for HCHO photolysis is determined for Jul 1 at noon and 30° N latitude. In most daylight cases, it is substantially longer than 3 h making the OH reaction far more competitive.

<sup>91</sup> Calvert, J. G., et al. (2011) The mechanisms of atmospheric oxidation of the oxygenates. Oxford University Press, New York/Oxford.

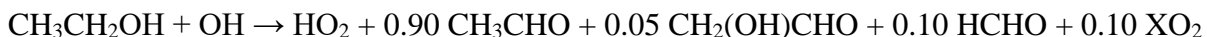
<sup>92</sup> Yarwood, G., Rao, S., Yocke, M., Whitten, G.Z., (2005). Updates to the Carbon Bond Mechanism: CB05. Final Report to the U.S. EPA, RT-0400675. Yocke and Company, Novato, CA.

<sup>93</sup> Luecken, D.J., et al. (2008). Effects of using the CB05 vs. SAPRC99 vs. CB4 chemical mechanism on model predictions: Ozone and gas-phase photochemical precursor concentrations. *Atmos. Environ.* 42, 5805-5820.

alkenes in particular are found to play a large role.<sup>94</sup> The IOLE model species, which represents internal carbon-carbon double bonds, has high emissions and relatively high yields of acetaldehyde. The OLE model species, representing terminal carbon double bonds, also plays a role because it has high emissions although lower acetaldehyde yields. Production from peroxypropional nitrate and other peroxyacylnitrates (PANX) and aldehydes with 3 or more carbon atoms can in some instances increase acetaldehyde, but because they also are a sink of radicals, their effect is smaller. Thus, the amount of acetaldehyde (and formaldehyde as well) formed in the ambient air, as well as emitted in the exhaust (the latter being accounted for in emission inventories), is affected by changes in these precursor compounds due to the addition of ethanol to fuels (e.g., decreases in alkenes would cause some decrease of acetaldehyde, and to a larger extent, formaldehyde).

The reaction of ethanol (CH<sub>3</sub>CH<sub>2</sub>OH) with OH is slower than some other important reactions but can be an important source of acetaldehyde if the emissions are large. Based on kinetic data for molecular reactions, the only important chemical loss process for ethanol (and other alcohols) is reaction with the hydroxyl radical ( $\cdot$ OH).<sup>95</sup> This reaction produces acetaldehyde (CH<sub>3</sub>CHO) with a 90 percent yield.<sup>88</sup> The lifetime of ethanol in the atmosphere can be calculated from the rate coefficient, k, and due to reaction with the OH radical, occurs on the order of a day in polluted urban areas or several days in unpolluted areas. For example, an atmospheric lifetime for acetaldehyde under nominal oxidant conditions, OH of  $1.0 \times 10^{-6} \text{ cm}^3 \text{ molecule}^{-1} \text{ s}^{-1}$ , would be 3.5 days.

In CB6r3 reaction of one molecule of ethanol yields 0.90 molecules of acetaldehyde. It assumes the majority of the reaction occurs through H-atom abstraction of the more weakly-bonded methylene group, which reacts with oxygen to form acetaldehyde and hydroperoxy radical (HO<sub>2</sub>), and the remainder of the reaction occurs at the -CH<sub>3</sub> and -OH groups, creating formaldehyde (HCHO), oxidizing NO to NO<sub>2</sub> (represented by model species XO<sub>2</sub>) and creating glycoaldehyde:



### 9.1.2 Organic Aerosols

Organic aerosol (OA) can be classified as either primary or secondary depending on whether it is emitted into the atmosphere as a particle (primary organic aerosol, POA) or formed in the atmosphere (secondary organic aerosol, SOA). SOA precursors include volatile organic compounds (VOCs) as well as low-volatility compounds that can react to form even lower volatility compounds. Current research suggests SOA contributes significantly to ambient OA concentrations, and in Southeast and Midwest States may make up more than 50 percent (although the contribution varies from area to area) of the organic fraction of PM<sub>2.5</sub> during the

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<sup>94</sup> Luecken, D.J., et al. (2012). Regional sources of atmospheric formaldehyde and acetaldehyde, and implications for atmospheric modeling. *Atmos. Environ.*, 47, 477-490.

<sup>95</sup> Atkinson, R., Arey, J. (2003). Atmospheric Degradation of Volatile Organic Compounds. *Chem. Rev.* 103, 4605-4638.

summer (but less in the winter).<sup>96,97</sup> A wide range of laboratory studies conducted over the past twenty years show that anthropogenic aromatic hydrocarbons and long-chain alkanes, along with biogenic isoprene, monoterpenes, and sesquiterpenes, contribute to SOA formation.<sup>98,99,100,101,102</sup> Modeling studies, as well as carbon isotope measurements, indicate that a significant fraction of SOA results from the oxidation of biogenic hydrocarbons.<sup>103,104</sup> Based on parameters derived from laboratory chamber experiments as well as predicted via structure-activity relationships or computational chemistry, SOA chemical mechanisms have been developed and integrated into air quality models such as the CMAQ model and have been used to predict OA concentrations.<sup>105,106,107</sup>

Secondary organic aerosol (SOA) chemistry in CMAQ v5.2 is documented in the work of Pye et al. (2017) and considers both volatility-driven condensation as well as heterogeneous and aqueous chemistry.<sup>108</sup>

In the analysis presented here, the primary organic aerosol (POA) is nonvolatile and undergoes heterogeneous oxidation.<sup>109</sup> Specifically, primary organic aerosol is tracked separately in terms of its carbon and non-carbon organic matter. Non-carbon organic matter (such as oxygen and hydrogen) is added to the reduced carbon as a result of heterogeneous reaction with OH. Diesel POA is emitted with an organic matter to organic carbon (OM/OC) ratio of 1.25. The ratio

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<sup>96</sup> Lewandowski M., et al. (2008). Primary and secondary contributions to ambient PM in the midwestern United States, *Environ. Sci. Technol.* 42(9), 3303-3309.

<sup>97</sup> Kleindienst T. E., et al. (2007). Estimates of the contributions of biogenic and anthropogenic hydrocarbons to secondary organic aerosol at a southeastern U.S. location, *Atmos. Environ.* 41(37), 8288-8300.

<sup>98</sup> Offenberg J. H., et al. (2007). Contributions of Toluene and  $\alpha$ -pinene to SOA Formed in an Irradiated Toluene/ $\alpha$ -pinene,NO<sub>x</sub>/Air Mixture: Comparison of Results Using <sup>14</sup>C Content and SOA Organic Tracer Methods, *Environ. Sci. Technol.* 41, 3972-3976.

<sup>99</sup> Pandis, S.N., Harley, R.A., Cass, G.R., Seinfeld, J.H. (1992). Secondary organic aerosol formation and transport. *Atmos. Environ.* 26, 2269-2282.

<sup>100</sup> Takekawa, H. Minoura, H. Yamazaki, S. (2003). Temperature dependence of secondary organic aerosol formation by photo-oxidation of hydrocarbons. *Atmos. Environ.* 37, 3413-3424.

<sup>101</sup> Kleeman, M.J., et al. (2007) Source apportionment of secondary organic aerosol during a severe photochemical smog episode. *Atmos. Environ.* 41, 576-591.

<sup>102</sup> Robinson, A. L., et al. (2007). Rethinking organic aerosol: Semivolatile emissions and photochemical aging. *Science* 315, 1259-1262.

<sup>103</sup> Griffin, R. J.; Cocker, D. R.; Seinfeld, J. H.; Dabdub, D. (1999). Estimate of global atmospheric organic aerosol from oxidation of biogenic hydrocarbons. *Geophys. Res. Lett.* 26 (17), 2721- 2724.

<sup>104</sup> Lewis, C. W.; Klouda, G. A.; Ellenson, W. D. (2004). Radiocarbon measurement of the biogenic contribution to summertime PM-2.5 ambient aerosol in Nashville, TN. *Atmos Environ* 38 (35), 6053- 6061.

<sup>105</sup> Byun DW, Schere, KL (2006). Review of the Governing Equations, Computational Algorithms, and Other Components of the Models-3 Community Multiscale Air Quality (CMAQ) Modeling System, *J Applied Mechanics Reviews* 59, 51-76.

<sup>106</sup> Pye, H. O. T., et al. (2013). Epoxide Pathways to Improve Model Predictions of Isoprene Markers and Reveal Key Role of Acidity in Aerosol Formation. *Environ. Sci. Tech.* 47 (19), 11056-11064.

<sup>107</sup> Piletic, I. R., Edney, E. O., Bartolotti, L. J. (2013). A computational study of acid catalyzed aerosol reactions of atmospherically relevant epoxides. *Phys. Chem. Chem. Phys.* 15, 18065-18076

<sup>108</sup> Pye, H. O. T., et al. (2017). On the Implications of aerosol liquid water and phase separation for organic aerosol mass. *Atmos. Chem. Phy.* 17, 343-369.

<sup>109</sup> Simon, H and Bhave, P. (2012). Simulating the Degree of Oxidation in Atmospheric Organic Particles. *Environ. Sci. Technol.*, 46 (1), 331-339.

increases due to exposure with OH. In the absence of removal, this oxidation process results in increasing organic aerosol concentrations. These OM/OC ratios assist with post-processing of model output for comparison with measured OC from routine networks.

Over the past 10 years, ambient OA concentrations have been routinely measured in the U.S. and some of these data have been used to determine, by employing source/receptor methods, the contributions of the major OA sources, including biomass burning and vehicular gasoline and diesel exhaust. Since mobile sources are a significant source of VOC emissions, mobile sources are also an important source of SOA, particularly in populated areas.<sup>110,111</sup>

Toluene is an important contributor to anthropogenic SOA.<sup>112,113</sup> Mobile sources are the most significant contributor to ambient toluene concentrations as shown by analyses done for the 2014 National Air Toxics Assessment (NATA)<sup>114</sup> and the Mobile Source Air Toxics (MSAT) Rule.<sup>115</sup> The 2014 NATA indicates that onroad and nonroad mobile sources accounted for around 70 percent ( $0.51 \mu\text{g}/\text{m}^3$ ) of the total average nationwide ambient concentration of toluene ( $0.71 \mu\text{g}/\text{m}^3$ ).

The amount of toluene in gasoline influences the amount of toluene emitted in vehicle exhaust and evaporative emissions, although, like benzene, some toluene is formed in the combustion process. In turn, levels of toluene and other aromatics in gasoline are potentially influenced by the amount of ethanol blended into the fuel. Due to the high octane of ethanol, it greatly reduces the need for other high-octane components including aromatics such as toluene (which is the major aromatic compound in gasoline). Since toluene contributes to SOA and the toluene level of gasoline is decreasing, it is important to assess the effect of these reductions on ambient PM.

In addition to toluene, other mobile-source hydrocarbons such as benzene, ethylbenzene, xylene, and alkanes form SOA and are treated accordingly in CMAQ v5.2.1.<sup>108,116,117</sup> Similar to toluene, the SOA produced by benzene and xylene from low-NO<sub>x</sub> pathways is expected to be less volatile and be produced in higher yields than SOA from high-NO<sub>x</sub> conditions.<sup>113</sup> Oxidation of alkanes

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<sup>110</sup> Jathar, S. H., et al. (2017). Chemical transport model simulations of organic aerosol in southern California: model evaluation and gasoline and diesel source contributions, *Atmos. Chem. Phys.* 17, 4305–4318.

<sup>111</sup> Jathar, S. H., et al. (2014). Unspeciated organic emissions from combustion sources and their influence on the secondary organic aerosol budget in the United States. *Proc. Natl. Acad. Sci.*, 111 (29), 10473-10478.

<sup>112</sup> Hildebrandt et al. (2009). High formation of secondary organic aerosol from the photo-oxidation of toluene. *Atmos. Chem. Phys.* 9, 2973-2986.

<sup>113</sup> Ng, N. L. et al. (2007). Secondary organic aerosol formation from m-xylene, toluene, and benzene. *Atmos. Chem. Phys.* 7, 3909-3922.

<sup>114</sup> USEPA (2018). Technical Support Document EPA's 2014 National Air Toxics Assessment. Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency, Research Triangle Park, NC. August 2018.. [https://www.epa.gov/sites/production/files/2018-09/documents/2014\\_nata\\_technical\\_support\\_document.pdf](https://www.epa.gov/sites/production/files/2018-09/documents/2014_nata_technical_support_document.pdf)

<sup>115</sup> USEPA (2007). Regulatory Impact Analysis for the Control of Hazardous Air Pollutants from Mobile Sources Rule, Chapter 3, Air Quality and Resulting Health and Welfare Effects of Air Pollution from Mobile Sources. 72 FR 8428, February 26, 2007. <https://nepis.epa.gov/Exe/ZyPdf.cgi?Dockey=P1004LNN.PDF>

<sup>116</sup> Pye, H. O. T.; Pouliot, G. A. (2012). Modeling the role of alkanes, polycyclic aromatic hydrocarbons, and their oligomers in secondary organic aerosol formation. *Environ. Sci. Technol.* 46 (11), 6041-6047.

<sup>117</sup> Carlton, A. G., et al. (2010). Model representation of secondary organic aerosol in CMAQv4.7. *Environ. Sci. Technol.*, 44(22): 8553–8560.



with longer chains as well as cyclic alkanes form SOA with relatively higher yields than small straight-chain alkanes.<sup>118</sup>

It is unlikely that ethanol would form SOA directly or affect SOA formation indirectly through changes in the radical populations due to increasing ethanol exhaust. Nevertheless, scientists at the U.S. EPA's Office of Research and Development directed experiments to investigate ethanol's SOA-forming potential.<sup>119</sup> The experiments were conducted under conditions where peroxy radical reactions would dominate over reaction with NO (i.e., irradiations performed in the absence of NO<sub>x</sub> and OH produced from the photolysis of hydrogen peroxide). This was the most likely scenario under which SOA formation could occur, since a highly oxygenated C<sub>4</sub> organic could form. As expected, no SOA was produced. From these experiments, the upper limit for the aerosol yield is less than 0.01 percent based on scanning mobility particle sizer (SMPS) data. Given the lack of aerosol formation found in these initial smog chamber experiments, these data were not published.

In general, measurements of OA represent the sum of POA and SOA and the fraction of aerosol that is secondary in nature can only be estimated. CMAQ has been evaluated against POA and SOA surrogates estimated from aerosol mass spectrometer (AMS) measurements along with positive matrix factorization (PMF).<sup>108,110,120,121,122,123</sup> AMS methods rely on detection of heavily fragmented structures which limits the ability to determine specific precursors to SOA as well as the role of biogenic vs anthropogenic VOC sources.

Upon release into the atmosphere, numerous VOC compounds can react with free radicals in the atmosphere to form SOA. While this has been investigated in the laboratory, there is relatively little information available on the specific chemical composition of SOA compounds themselves from specific VOC precursors. This absence of complete compositional data from the precursors has made the identification of aromatically-derived SOA in ambient samples challenging, which in turn has prevented observation-based measurements of individual SOA source contributions to ambient PM levels.

As a first step in estimating ambient SOA concentrations, EPA has developed a tracer-based method.<sup>97,98</sup> The method is based on using mass fractions of SOA tracer compounds, measured in smog chamber-generated SOA samples, to convert ambient concentrations of SOA tracer compounds to ambient SOA concentrations. This method consists of irradiating the SOA

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<sup>118</sup> Lim, Y.B., Ziemann, P.J. (2009). Effects of Molecular Structure on Aerosol Yields from OH Radical-Initiated Reactions of Linear, Branched, and Cyclic Alkanes in the Presence of NO<sub>x</sub>. *Environ. Sci. Technol.* 43 (7), 2328-2334.

<sup>119</sup> Kleindienst, T.E. (2008). Hypothetical SOA Production from Ethanol Photooxidation. Memo to the Docket EPA-HQ-OAR-2005-0161.

<sup>120</sup> Pye, H. O. T., et al. (2015). Modeling the current and future roles of particulate organic nitrates in the southeastern United States. *Environ. Sci. Technol.* 49 (24), 14195-14203.

<sup>121</sup> Murphy, B. N., et al. (2017). Semivolatile POA and parameterized total combustion SOA in CMAQv5.2: impacts on source strength and partitioning. *Atmos. Chem. Phys.* 17 (18), 11107-11133.

<sup>122</sup> Woody, M. C., et al. (2016). Understanding sources of organic aerosol during CalNex-2010 using the CMAQ-VBS. *Atmos. Chem. Phys.* 16 (6), 4081-4100.

<sup>123</sup> Baker, K. R. et al. (2015). Gas and aerosol carbon in California: comparison of measurements and model predictions in Pasadena and Bakersfield. *Atmos. Chem. Phys.*, 15, 5243-5258.



precursor of interest in a smog chamber in the presence of NO<sub>x</sub>, collecting the SOA produced on filters, and then analyzing the samples for highly polar compounds using advanced analytical chemistry methods. Employing this method, candidate tracers have been identified for several VOC compounds which are emitted in significant quantities and known to produce SOA in the atmosphere. Some of these SOA-forming compounds include toluene, a variety of monoterpenes, isoprene, and β-caryophyllene, the latter three of which are emitted by vegetation and are more significant sources of SOA than toluene. Smog chamber work can also be used to investigate SOA chemical formation mechanisms.<sup>124,125,126,127</sup>

Although these concentrations are only estimates, due to the assumption that the mass fractions of the smog chamber SOA samples using these tracers are equal to those in the ambient atmosphere, there are presently limited other means available for estimating the SOA concentrations originating from individual SOA precursors. Among the tracer compounds observed in ambient PM<sub>2.5</sub> samples are two tracer compounds that have been identified in smog chamber aromatic SOA samples.<sup>127</sup> To date, these aromatic tracer compounds have been identified in the laboratory for toluene and *m*-xylene SOA. Additional work is underway by the EPA to determine whether these tracers are also formed by benzene and other alkylbenzenes (including *o*-xylene, *p*-xylene, 1,2,4-trimethylbenzene, and ethylbenzene).

One caveat regarding this work is that a large number of VOCs emitted into the atmosphere, which have the potential to form SOA, have not yet been studied in environmental smog chambers. These unstudied compounds could produce SOA species that are being used as tracers for other VOCs thus overestimating the amount of SOA formed in the atmosphere by the VOCs studied to date. This approach may also estimate entire hydrocarbon classes (e.g., all methylsubstituted-monoaromatics or all monoterpenes) and not individual precursor hydrocarbons. Thus, the tracers could be broadly representative and not indicative of individual precursors. This is still unknown. Also, anthropogenic precursors play a role in formation of atmospheric radicals and aerosol acidity, and these factors influence SOA formation from biogenic hydrocarbons.<sup>128,129</sup> This anthropogenic and biogenic interaction, important to EPA and others, needs further study. The issue of SOA formation from aromatic precursors is an important one to which EPA and others are paying significant attention.

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<sup>124</sup> Claeys, M. R., et al (2007). Hydroxydicarboxylic acids: Markers for secondary organic aerosol from the photooxidation of α-pinene. *Environ. Sci. Technol.* 41(5), 1628-1634.

<sup>125</sup> Edney E. O., et al. (2005). Formation of 2-methyl tetrols and 2-methylglyceric acid in secondary organic aerosol from laboratory irradiated isoprene/NO<sub>x</sub>/SO<sub>2</sub>/air mixtures and their detection in ambient PM<sub>2.5</sub> samples collected in the Eastern United States. *Atmos. Environ.* 39, 5281-5289.

<sup>126</sup> Jaoui, M., et al. (2005). Identification and quantification of aerosol polar oxygenated compounds bearing carboxylic or hydroxyl groups. 2. Organic tracer compounds from monoterpenes. *Environ. Sci. Technol.* 39, 5661-5673.

<sup>127</sup> Kleindienst T. E., et al. (2004). Determination of secondary organic aerosol products from the photooxidation of toluene and their implications in ambient PM<sub>2.5</sub>. *J. Atmos. Chem.* 47, 70-100.

<sup>128</sup> Pye, H.O.T., et al. (2013). Epoxide pathways improve model predictions of isoprene markers and reveal key role of acidity in aerosol formation. *Environ. Sci. Technol.* 47(19), 11056-11064.

<sup>129</sup> Carlton, A.G., et al. (2010). To what extents can biogenic SOA be controlled? *Environ. Sci. Technol.* 44(9), 3376-3380.

The aromatic tracer compounds and their mass fractions have been used to estimate monthly ambient aromatic SOA concentrations from March 2004 to February 2005 in five U.S. Midwestern cities.<sup>130</sup> The annual tracer-based SOA concentration estimates were 0.15, 0.18, 0.13, 0.15, and 0.19  $\mu\text{g carbon/m}^3$  for Bondville, IL, East St. Louis, IL, Northbrook, IL, Cincinnati, OH and Detroit, MI, respectively, with the highest concentrations occurring in the summer. On average, the aromatic SOA concentrations made up 17 percent of the total SOA concentration. Thus, this work suggests that we are finding ambient PM levels on an annual basis of about 0.15  $\mu\text{g/m}^3$  associated with present toluene levels in the ambient air in these Midwest cities. Based on preliminary analysis of recent laboratory experiments, it appears the toluene tracer could also be formed during photooxidation of some of the xylenes.<sup>97</sup>

Over the past decade a variety of modeling studies have been conducted to predict ambient SOA levels. While early studies focused on the contribution of biogenic monoterpenes, additional precursors, such as sesquiterpenes, isoprene, benzene, toluene, and xylene, have been implemented in atmospheric models such as GEOS-Chem, PMCAMx, and CMAQ.<sup>112,113,116,131,132, 133,134</sup> The most generally available routine measurements available from monitoring networks for model evaluations are ambient OC or estimated OA concentrations. Without a method to attribute measured OC to different sources or precursors, identifying causes of the underestimates in modeled OC via model/measurement comparisons can be challenging. However, analysis of SOA concentrations in Pasadena and Bakersfield, California during 2010 indicate CMAQ-predicted SOA from toluene and xylene is underestimated despite overestimates of the VOC precursors.<sup>123</sup> In addition, CMAQ-predicted aromatic SOA was underestimated in the Midwest U.S. despite reasonable predictions of primary organic aerosol tracers, implying underestimated SOA yields.<sup>135</sup>

Anthropogenic emissions of  $\text{NO}_x$  and  $\text{SO}_x$  are known to modulate the abundance of SOA from oxidation of biogenic VOCs, thus allowing for additional influences of vehicle emissions on

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<sup>130</sup> Lewandowski, M., et al. (2008). Primary and secondary contributions to ambient PM in the midwestern United States. *Environ. Sci. Technol.* 42(9), 3303-3309.

<sup>131</sup> Henze, D. K., Seinfeld, J. H. (2006). Global secondary organic aerosol from isoprene oxidation. *Geophys. Res. Lett.* 33: L09812. doi:10.1029/2006GL025976.

<sup>132</sup> Henze, D. K., et al. (2008). Global modeling of secondary organic aerosol formation from aromatic hydrocarbons: high- vs. low-yield pathways. *Atmos. Chem. Phys.*, 8, 2405-2420, doi:10.5194/acp-8-2405-2008.

<sup>133</sup> Lane, T. E., Donahue, N.M. and Pandis, S.N. (2008). Simulating secondary organic aerosol formation using the volatility basis-set approach in a chemical transport model, *Atmos. Environ.*, 42, 7439-7451.

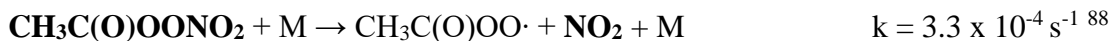
<sup>134</sup> Parikh, H.M., et al. (2011). Modeling secondary organic aerosol using a dynamic partitioning approach incorporating particle aqueous-phase chemistry, *Atmos. Environ.*, 45, 1126-1137.

<sup>135</sup> Napelenok, S. L., et al. (2014). Diagnostic air quality model evaluation of source specific primary and secondary fine particulate carbon, *Environ. Sci. Technol.*, doi: 10.1021/es4033024w.

ambient PM<sub>2.5</sub>.<sup>136,137,138,139,140,141</sup> The SOA that results from oxidation of isoprene is one of the most evaluated and constrained SOA systems in CMAQ. In CMAQ, most isoprene SOA results from acid-catalyzed reactions of later-generation isoprene oxidation products (specifically isoprene-epoxydiols or IEPOX). The original implementation was documented in Pye et al. 2013 (introduced v5.1) with parameter updates v5.2 that improved the magnitude and speciation of the resulting SOA (Pye et al. 2017). The isoprene SOA has been evaluated against speciated isoprene SOA measurements across the U.S. (Pye et al. 2013, 2-methyltetrol and organosulfates), speciated isoprene SOA from filter measurements at Look Rock, TN (Budisulistiorini et al. 2017), and isoprene SOA surrogates (specifically AMS PMF factors) at Centerville AL and Look Rock TN (Pye et al. 2017).<sup>142</sup> Constraints for the monoterpene system are only starting to be leveraged, but indicate NO<sub>x</sub> plays a major role via its regulation of oxidant abundance and by promoting oxidation via the nitrate radical resulting in high SOA yields.<sup>136,138</sup>

### 9.1.3 Ozone

As mentioned above, the addition of ethanol to fuels has been shown to contribute to PAN formation and this is one way for it to contribute therefore to ground-level ozone formation downwind of NO<sub>x</sub> sources. PAN is a reservoir and carrier of NO<sub>x</sub> and is the product of acetyl radicals reacting with NO<sub>2</sub> in the atmosphere. One source of PAN is the photooxidation of acetaldehyde, but many VOCs have the potential for forming acetyl radicals and therefore PAN or a PAN-type compound.<sup>143</sup> PAN can undergo thermal decomposition with a lifetime of approximately 1 hour at 298K or 148 days at 250K.



The reaction above shows how NO<sub>2</sub> is released in the thermal decomposition of PAN, along with a peroxy radical which can oxidize NO to NO<sub>2</sub> and form other species that convert NO to NO<sub>2</sub>

<sup>136</sup> Pye, H.O.T., et al. (2019). Anthropogenic enhancements to production of highly oxygenated molecules from autoxidation. *P. Natl. Acad. Sci.*, 116 (14), 6641-6646.

<sup>137</sup> Zhang, H. F., et al. (2018). Monoterpenes are the largest source of summertime organic aerosol in the southeastern United States. *P. Natl. Acad. Sci.*, 115 (9), 2038-2043.

<sup>138</sup> Ng, N. L., et al. (2017). Nitrate radicals and biogenic volatile organic compounds: oxidation, mechanisms, and organic aerosol. *Atmos. Chem. Phys.*, 17 (3), 2103-2162.

<sup>139</sup> Pye, H. O. T., et al. (2013). Epoxide pathways improve model predictions of isoprene markers and reveal key role of acidity in aerosol formation. *Environ. Sci. Technol.*, 47 (19), 11056-11064.

<sup>140</sup> Weber, R. J., et al. (2007). A study of secondary organic aerosol formation in the anthropogenic-influenced southeastern United States. *J. Geophys. Res. Atmos.*, 112, D13302.

<sup>141</sup> Xu, L., et al. (2015). Effects of anthropogenic emissions on aerosol formation from isoprene and monoterpenes in the southeastern United States. *P. Natl. Acad. Sci.*, 112 (1), 37-42.

<sup>142</sup> Budisulistiorini, S. H. (2017). Simulating aqueous-phase isoprene-epoxydiol (IEPOX) secondary organic aerosol production during the 2013 Southern Oxidant and Aerosol Study (SOAS). *Environ. Sci. Technol.* 51 (9), 5026-5034.

<sup>143</sup> Many aromatic hydrocarbons, particularly those present in high percentages in gasoline (toluene, m-, o-, p-xylene, and 1,3,5-, 1,2,4-trimethylbenzene), form methylglyoxal and biacetyl, which are also strong generators of acetyl radicals (Smith, D.F., T.E. Kleindienst, C.D. McIver (1999). Primary product distribution from the reaction of OH with m-, p-xylene and 1,2,4- and 1,3,5-Trimethylbenzene. *J. Atmos. Chem.*, 34: 339- 364).

through photochemical reactions, as previously shown in Section 9.1.1. NO<sub>2</sub> further photolyzes to produce ozone (O<sub>3</sub>).



The temperature sensitivity of PAN allows it to be stable enough at low temperatures to be transported long distances before decomposing to release NO<sub>2</sub>. NO<sub>2</sub> can then participate in ozone formation in regions remote from the original NO<sub>x</sub> source.<sup>144</sup> A discussion of CB6 mechanisms for ozone formation can be found in Yarwood et al. (2010).<sup>79</sup>

Another important way that ethanol fuels contribute to ozone formation is by increasing the formation of new radicals through increases in formaldehyde and acetaldehyde. The photolysis of both aldehydes results in up to two molecules of either hydroperoxy radical or methylperoxy radical, both of which oxidize NO to NO<sub>2</sub> leading to ozone formation.

#### 9.1.4 Uncertainties Associated with Chemical Mechanisms

A key source of uncertainty with respect to the air quality modeling results is the photochemical mechanisms in CMAQ. Pollutants such as ozone, PM, acetaldehyde, formaldehyde, and acrolein can be formed secondarily through atmospheric chemical processes. Since secondarily formed pollutants can result from many different reaction pathways, there are uncertainties associated with each pathway. Simplifications of chemistry must be made in order to handle reactions of thousands of chemicals in the atmosphere. Mechanisms for formation of ozone, PM, acetaldehyde, and peroxyacetyl nitrate (PAN) are discussed in the Section 9.1.1.

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<sup>144</sup> Finlayson-Pitts BJ, Pitts JN Jr. (1986). Atmospheric Chemistry: Fundamentals and Experimental Techniques, Wiley, New York.

## 9.2 Creating Ozone and PM<sub>2.5</sub> Fused Fields Based on Observations and Model Surfaces

The following data were used to create the spatial fields of ozone and PM<sub>2.5</sub> concentrations for the 2016 “with-RFS” and 2016 “pre-RFS” scenarios:

- (1) Daily 2016 “with-RFS” and 2016 “pre-RFS” modeling-based concentrations of 24-hour average PM<sub>2.5</sub> component species and maximum daily 8-hour average MDA8 ozone;
- (2) Baseline, 2016 “with-RFS” “fused surfaces” of measured and modeled air quality<sup>145</sup> representing quarterly average PM<sub>2.5</sub> component species concentrations and ozone concentrations for the seasonal average ozone metrics. These “fused surfaces” use the ambient data to adjust modeled fields to match observed data at locations of monitoring sites. Details on the methods for creating fused surfaces are provided below.

For PM<sub>2.5</sub>, daily gridded PM<sub>2.5</sub> species were processed into annual average surfaces which combine observed values with model predictions using the enhanced Voronoi Neighbor Average (eVNA) method.<sup>146,147,148</sup> These steps were performed using EPA’s software package, Software for the Modeled 8-30 Attainment Test-Community Edition (SMAT-CE)<sup>149</sup> and have been previously documented both in the user’s guide for the predecessor software<sup>150</sup> and in EPA’s modeling guidance document.<sup>151</sup> As explained above, we first create a 2016 “with-RFS” eVNA surface for each PM component species. To create the 2016 “with-RFS” eVNA surface, SMAT-CE first calculates quarterly average values (January-March; April-June; July-September; October-December) for each PM<sub>2.5</sub> component species at each monitoring site with available measured data. For this calculation we used three years of monitoring data (2015-2017).<sup>152</sup> SMAT-CE then creates an interpolated field of the quarterly-average observed data for each PM<sub>2.5</sub> component species using inverse distance squared weighting resulting in a separate 3-year average interpolated observed field for each PM<sub>2.5</sub> species and each quarter. The interpolated

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<sup>145</sup> In this analysis, a “fused surface” represents a spatial field of concentrations of a particular pollutant that was derived by applying the Enhanced Voronoi Neighbor Averaging with adjustment using modeled and measured air quality data (i.e., eVNA) technique (Ding et al. 2016).

<sup>146</sup> Gold C, Remmele P.R., Roos T., (1997). In: Algorithmic Foundation of Geographic Information Systems. In: Lecture Notes in Computer Science, Vol. 1340 (van Kereveld M, Nievergelt J, Roos T, Widmayer P, eds) Berlin, Germany: Springer-Verlag. Voronoi methods in GIS. pp. 21–35.

<sup>147</sup> US EPA (2007). Technical Report on Ozone Exposure, Risk, and Impact Assessments for Vegetation. EPA 452/R-07-002. Prepared by Abt Associates Inc. for U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards, Health and Environmental Impacts Division. Research Triangle Park, NC. ([https://www3.epa.gov/ttn/naaqs/standards/ozone/data/2007\\_01\\_environmental\\_tsd.pdf](https://www3.epa.gov/ttn/naaqs/standards/ozone/data/2007_01_environmental_tsd.pdf)).

<sup>148</sup> Ding, D., Zhu, Y., Jang, C., Lin, C., Wang, S., Fu, J., Gao, J., Deng, S., Xie, J., Qui, X. (2015). Evaluation of health benefit using BenMAP-CE with an integrated scheme of model and monitor data during Guangzhou Asian Games. *Journal of Environmental Science*. 29, 178-188.

<sup>149</sup> Software download and documentation available at <https://www.epa.gov/scram/photochemical-modeling-tools>

<sup>150</sup> Abt Associates, 2014. User’s Guide: Modeled Attainment Test Software. [https://www3.epa.gov/ttn/scram/guidance/guide/MATS\\_2-6-1\\_manual.pdf](https://www3.epa.gov/ttn/scram/guidance/guide/MATS_2-6-1_manual.pdf)

<sup>151</sup> USEPA, (2018). Modeling Guidance for Demonstrating Attainment of Air Quality Goals for Ozone, PM<sub>2.5</sub>, and Regional Haze, Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency, Research Triangle Park, NC. [https://www3.epa.gov/ttn/scram/guidance/guide/O3-PM-RH-Modeling\\_Guidance-2018.pdf](https://www3.epa.gov/ttn/scram/guidance/guide/O3-PM-RH-Modeling_Guidance-2018.pdf)

<sup>152</sup> Three years of ambient data is used to provide a more representative picture of air pollution concentrations.

observed fields are then adjusted to match the spatial gradients from the modeled data. These two steps can be calculated using Equation 9.1:

$$eVNA_{g,s,q,2016 \text{ "with-RFS"}} = \sum Weight_x Monitor_{x,s,q,2015-2017} \frac{Model_{g,s,q,2016 \text{ "with-RFS"}}}{Model_{x,s,q,2016 \text{ "with-RFS"}}}$$
 Equation 9.1

Where:

- $eVNA_{g,s,q,2016 \text{ "with-RFS"}}$  is the gradient adjusted quarterly-average eVNA value at grid-cell, g, for PM component species, s, during quarter, q for the year 2016;
- $Weight_x$  is the inverse distance weight for monitor x at the location of grid-cell, g;
- $Monitor_{x,s,q,2015-2017}$  is the 3-year (2015-2017) average of the quarterly monitored concentration for species, s, at monitor, x, during quarter, q;
- $Model_{g,s,q,2016 \text{ "with-RFS"}}$  is the 2016 “with-RFS” modeled quarterly-average concentrations of species, s, at grid cell, g, during quarter, q;
- $Model_{x,s,q,2016 \text{ "with-RFS"}}$  is the 2016 “with-RFS” modeled quarterly-average concentration of species, s, at the location of monitor, x, during quarter q.

The 2016 “with-RFS” eVNA field serves as the starting point for fused model surfaces. As described in above, to create a gridded 2016 “pre-RFS” eVNA surfaces for the 2016 “with-RFS” and 2016 “pre-RFS” scenarios, we take the ratio of the modeled 2016 “pre-RFS” quarterly average concentration to the modeled 2016 “with-RFS” concentration in each grid cell and multiply that by the corresponding 2016 “with-RFS” eVNA quarterly PM<sub>2.5</sub> component species value in that grid cell, as shown in Equation 9.2.

$$eVNA_{g,s,q,2016 \text{ "pre-RFS"}} = (eVNA_{g,s,q,2016 \text{ "with-RFS"}}) \times \frac{Model_{g,s,q,2016 \text{ "pre-RFS"}}}{Model_{g,s,q,2016 \text{ "with-RFS"}}$$
 Equation 9.2

This results in a gridded future-year projection which accounts for adjustments to match observations in the 2016 modeled data.

Particulate ammonium concentrations are impacted both by emissions of precursor ammonia gas as well as ambient concentrations of particulate sulfate and nitrate. Because of uncertainties in ammonium speciation measurements combined with sparse ammonium measurements in rural areas, the SMAT-CE default is to calculate ammonium values using the degree of sulfate neutralization (i.e., the relative molar mass of ammonium to sulfate with the assumption that all nitrate is fully neutralized). Degree of neutralization values are mainly available in urban areas while sulfate measurements are available in both urban and rural areas. Ammonium is thus calculated by multiplying the interpolated degree of neutralization value by the interpolated sulfate value at each grid-cell location which allows the ammonium fields to be informed by rural sulfate measurements in locations where no rural ammonium measurements are available.

The degree of neutralization is not permitted to exceed the maximum theoretical molar ratio of 2:1 for ammonium:sulfate. When creating the future year surface for particulate ammonium, we use the default SMAT-CE assumption that the degree of neutralization for the aerosol remains at 2016 levels.

A similar method for creating 2016 “pre-RFS” eVNA surfaces is followed for the MDA8 ozone metric with a few key differences. First, while PM<sub>2.5</sub> is split into quarterly averages and then averaged up to an annual value, we look at ozone as a summer-season average using definitions that match metrics from epidemiology studies (May-Sep for MDA8). The other main difference in the SMAT-CE calculation for ozone is that the spatial interpolation of observations uses an inverse distance weighting rather than an inverse distance squared weighting. This results in interpolated observational fields that better replicate the more gradual spatial gradients observed in ozone compared to PM<sub>2.5</sub>.

### 9.3 Air Quality Model Performance Evaluation

An operational model performance evaluation for ozone, PM<sub>2.5</sub> and its related speciated components, specific air toxics (i.e., formaldehyde, acetaldehyde, benzene, 1,3-butadiene, and acrolein), as well as nitrate and sulfate deposition, was conducted using 2016 state/local monitoring site data in order to estimate the ability of the CMAQ modeling system to replicate the base year concentrations for the 12 km Continental United States domain (Section 6.7.2, Figure 6.1). Included in this evaluation are statistical measures for model versus observed pairs that were paired in space and time on a daily or weekly basis, depending on the sampling frequency of each network (measured data). We excluded the CMAQ predictions for certain time periods with missing ozone, PM<sub>2.5</sub>, air toxic and nitrate and sulfate deposition observations from our calculations. It should be noted when comparing model and observed data that each CMAQ concentration represents a grid-cell volume-averaged value, while the ambient network measurements are made at specific locations.

Model performance statistics were calculated for several spatial scales and temporal periods. Statistics were calculated for individual monitoring sites and for each of the nine National Oceanic and Atmospheric Administration (NOAA) climate regions of the 12-km U.S. modeling domain (Figure 9.1).<sup>153</sup> The regions include the Northeast, Ohio Valley, Upper Midwest, Southeast, South, Southwest, Northern Rockies, Northwest and West<sup>154,155</sup> as were originally

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<sup>153</sup> NOAA, National Centers for Environmental Information scientists have identified nine climatically consistent regions within the contiguous U.S. <https://www.ncdc.noaa.gov/monitoring-references/maps/us-climate-regions.php>

<sup>154</sup> The nine climate regions are defined by States where: Northeast includes CT, DE, ME, MA, MD, NH, NJ, NY, PA, RI, and VT; Ohio Valley includes IL, IN, KY, MO, OH, TN, and WV; Upper Midwest includes IA, MI, MN, and WI; Southeast includes AL, FL, GA, NC, SC, and VA; South includes AR, KS, LA, MS, OK, and TX; Southwest includes AZ, CO, NM, and UT; Northern Rockies includes MT, NE, ND, SD, WY; Northwest includes ID, OR, and WA; and West includes CA and NV.

<sup>155</sup> Note most monitoring sites in the West region are located in California, therefore statistics for the West will be mostly representative of California ozone air quality.

identified in Karl and Koss (1984).<sup>156</sup> The statistics for each site and climate region were calculated by season (“Winter” is defined as average of December, January, and February; “Spring” is defined as average of March, April, and May; “Summer” is defined as average of June, July, and August; and “Fall” is defined as average of September, October, and November). For 8-hour daily maximum ozone, we also calculated performance statistics by region for the May through September ozone season.<sup>157</sup> In addition to the performance statistics, we prepared several graphical presentations of model performance. These graphical presentations include regional maps which show the mean bias, mean error, normalized mean bias and normalized mean error calculated for each season at individual monitoring sites. The full model performance evaluation can be found in a memo with the online supplemental materials.<sup>158</sup>

### U.S. Climate Regions

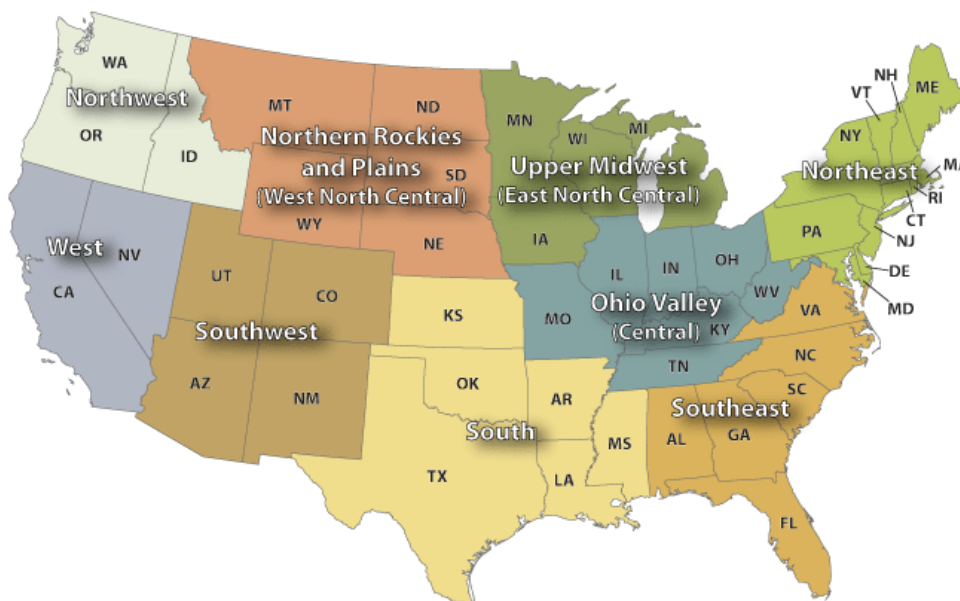


Figure 9.1 NOAA Nine Climate Regions<sup>159</sup>

<sup>156</sup> Karl, T. R. and Koss, W. J. (1984). "Regional and National Monthly, Seasonal, and Annual Temperature Weighted by Area, 1895-1983." Historical Climatology Series 4-3, National Climatic Data Center, Asheville, NC, 38 pp.

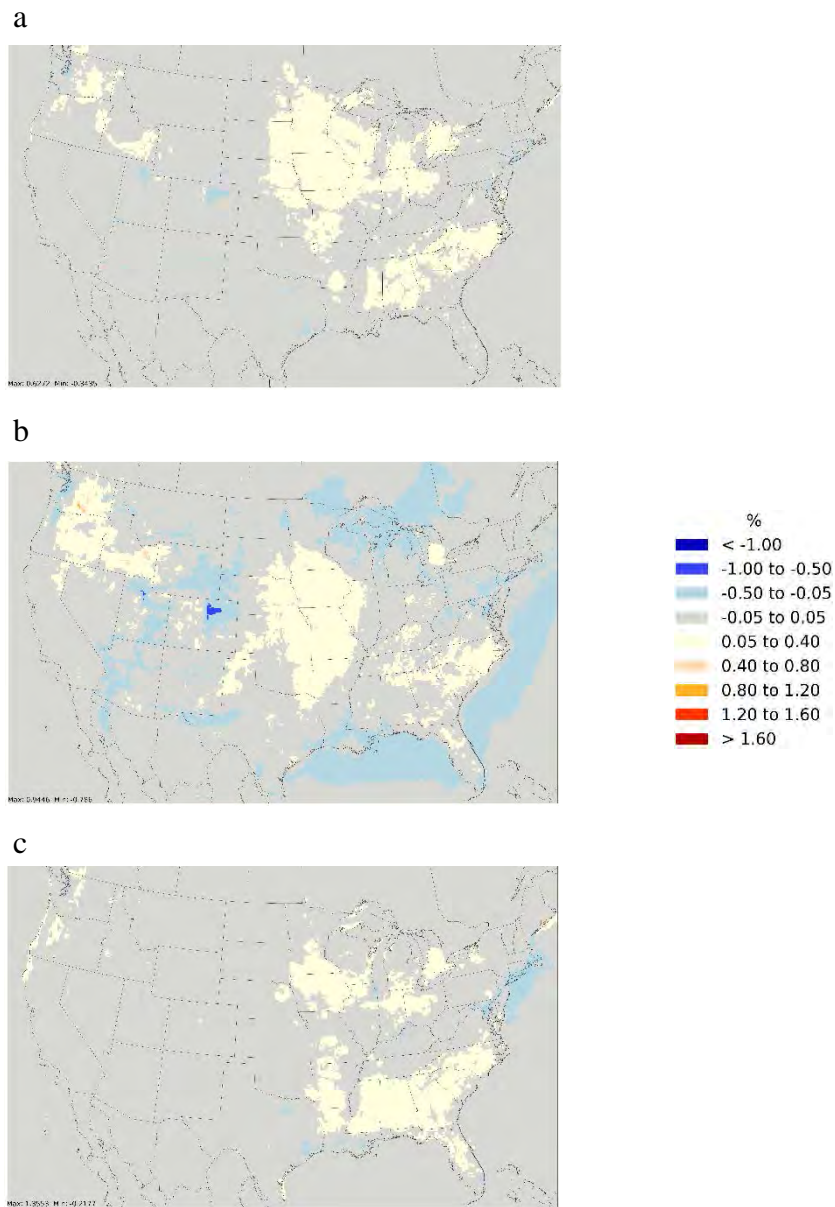
<sup>157</sup> In calculating the ozone season statistics, we limited the data to those observed and predicted pairs with observations that exceeded 60 ppb in order to focus on concentrations at the upper portion of the distribution of values.

<sup>158</sup> <https://www.epa.gov/renewable-fuel-standard-program/anti-backsliding-determination-and-study>

<sup>159</sup> Source: <https://www.ncdc.noaa.gov/monitoring-references/maps/us-climate-regions.php#references>

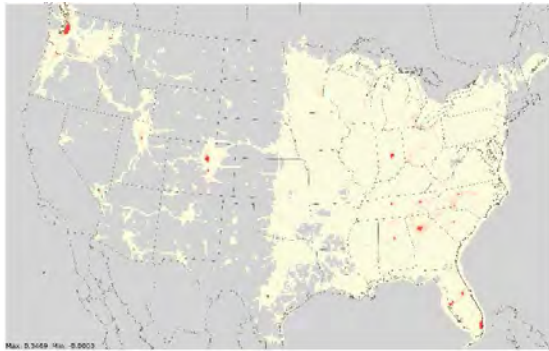


## 9.4 Monthly Air Quality Difference Maps

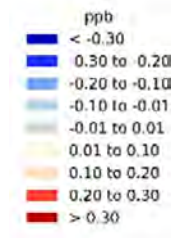
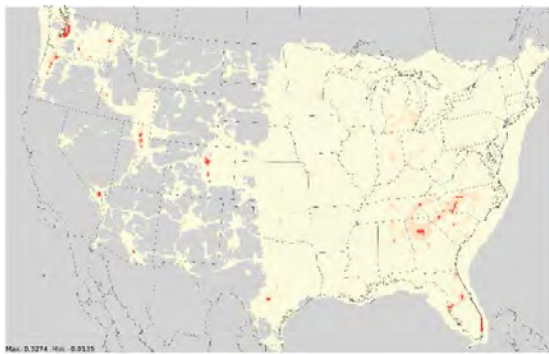


**Figure 9.2 Percent difference in (a) annual average, (b) January average, and (c) July average  $PM_{2.5}$  concentrations between “pre-RFS” and “with-RFS” scenarios for 2016**

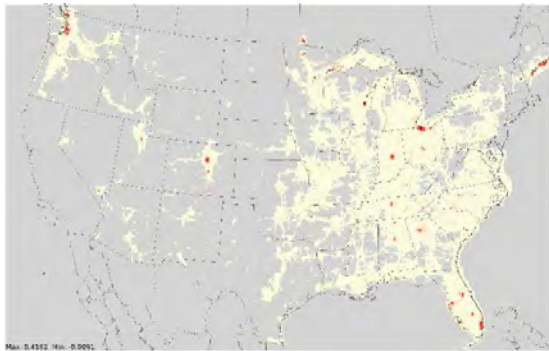
a



b

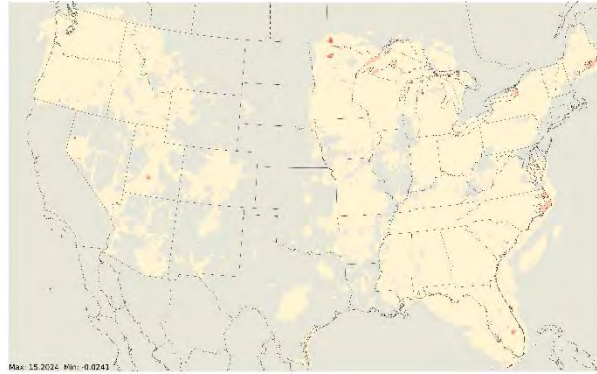


c

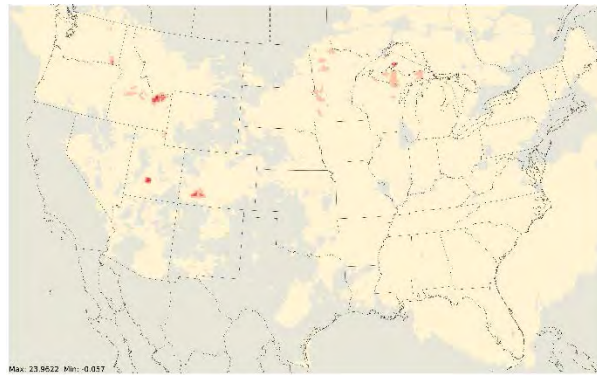


**Figure 9.3 Absolute difference in (a) annual average, (b) January average, and (c) July average NO<sub>2</sub> concentrations between “pre-RFS” and “with-RFS” scenarios for 2016**

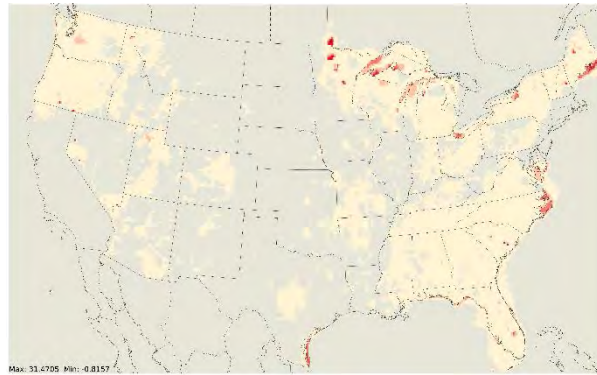
a



b

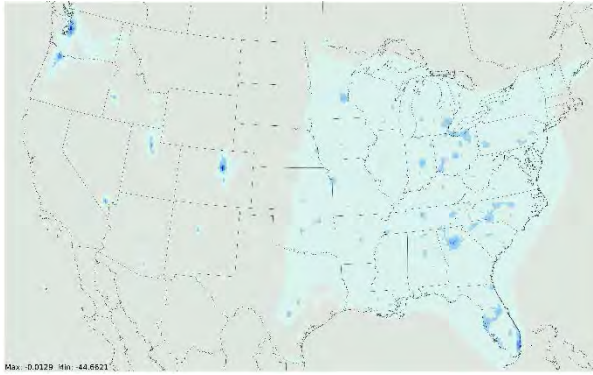


c

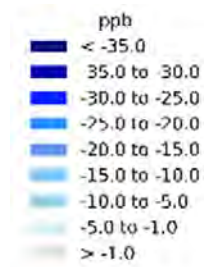


**Figure 9.4 Percent difference in (a) annual average, (b) January average, and (c) July average NO<sub>2</sub> concentrations between “pre-RFS” and “with-RFS” scenarios for 2016**

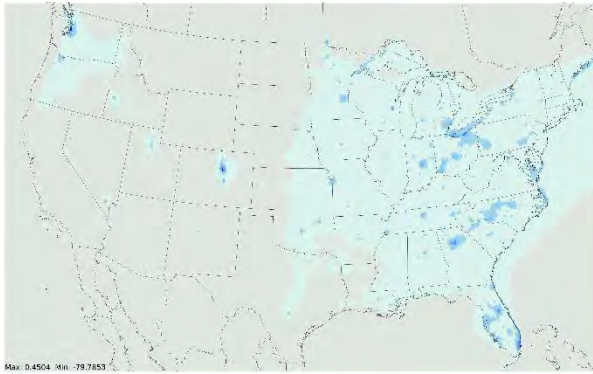
a



b



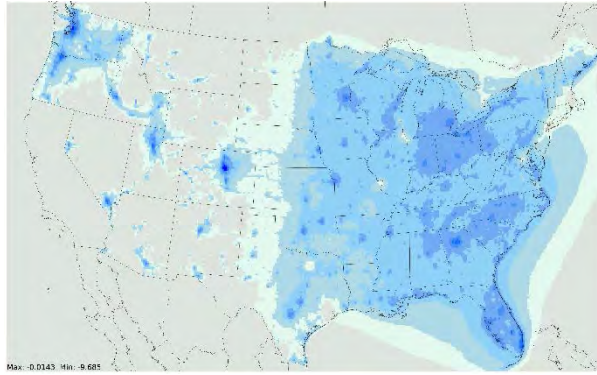
c



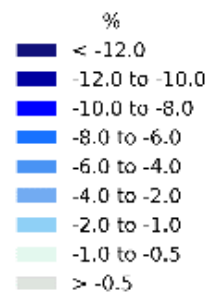
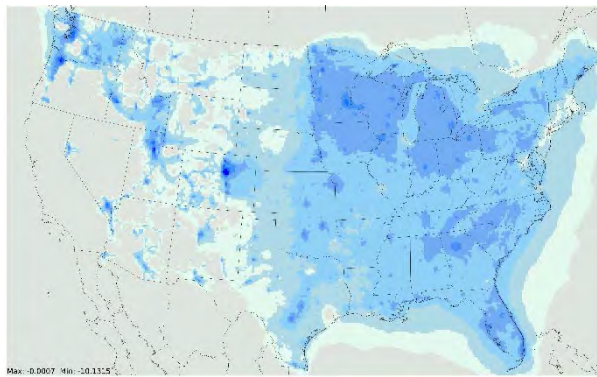
**Figure 9.5 Absolute difference in (a) annual average, (b) January average, and (c) July average CO concentrations between “pre-RFS” and “with-RFS” scenarios for 2016**



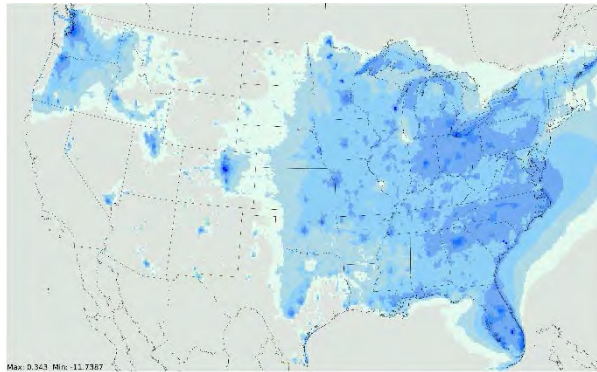
a



b

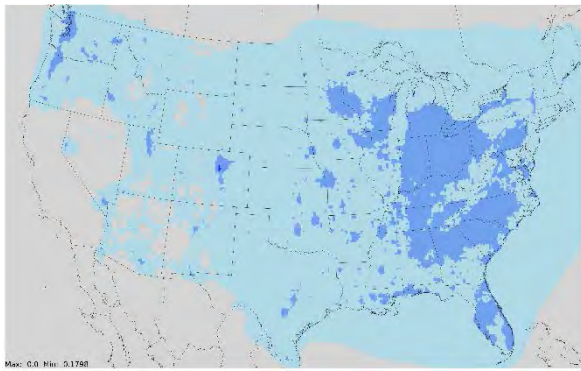


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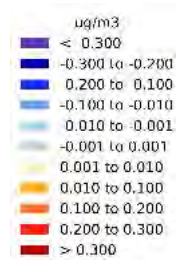
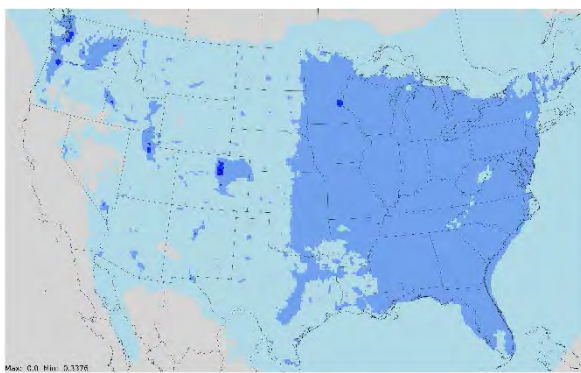


**Figure 9.6 Percent difference in (a) annual average, (b) January average, and (c) July average CO concentrations between “pre-RFS” and “with-RFS” scenarios for 2016**

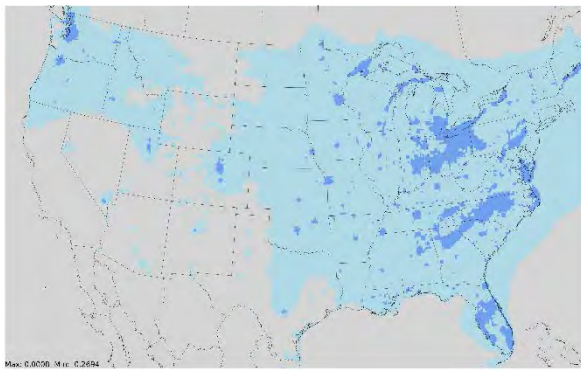
a



b

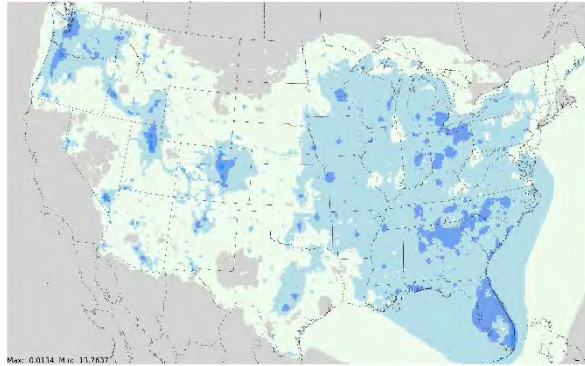


c

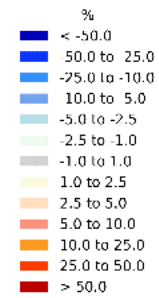
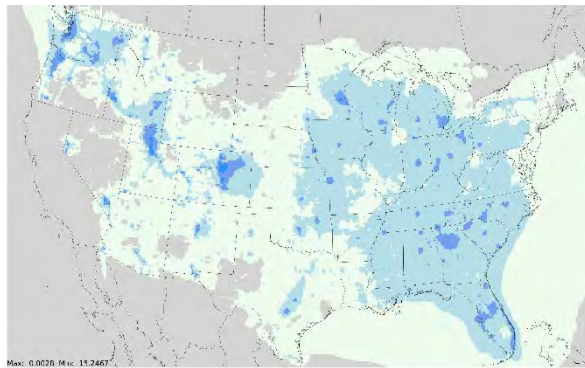


**Figure 9.7 Absolute difference in (a) annual average, (b) January average, and (c) July average benzene concentrations between “pre-RFS” and “with-RFS” scenarios for 2016**

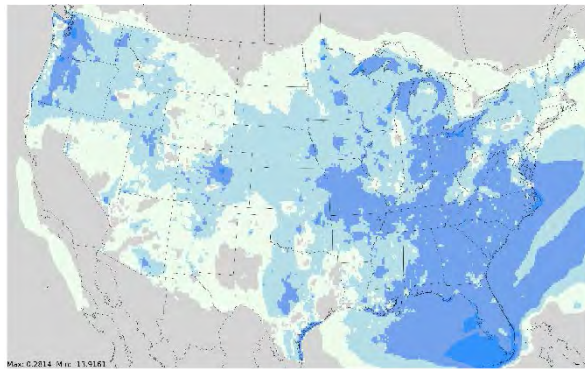
a



b



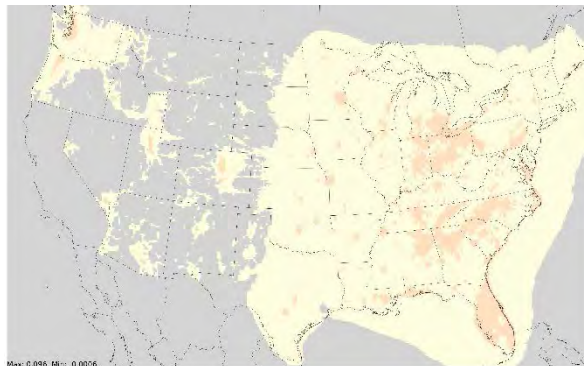
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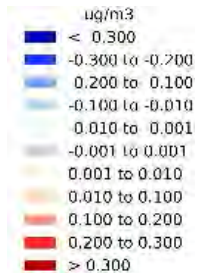
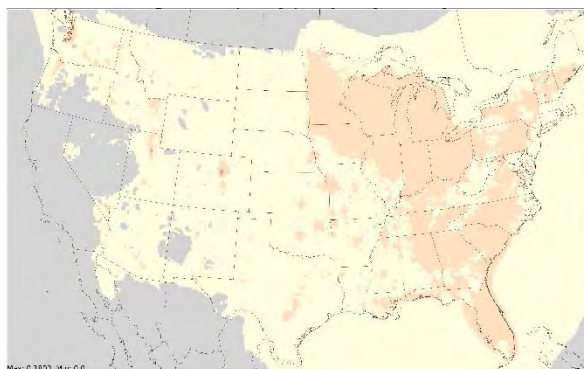
**Figure 9.8 Percent difference in (a) annual average, (b) January average, and (c) July average benzene concentrations between “pre-RFS” and “with-RFS” scenarios for 2016**



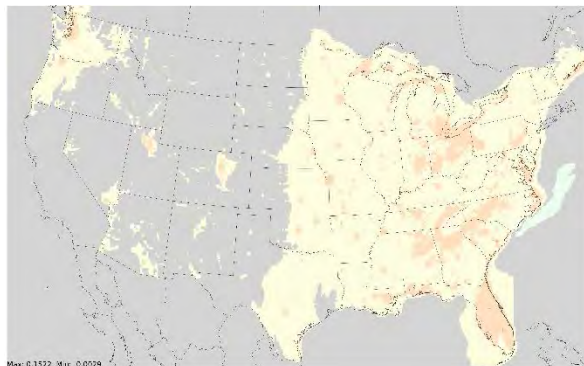
a



b

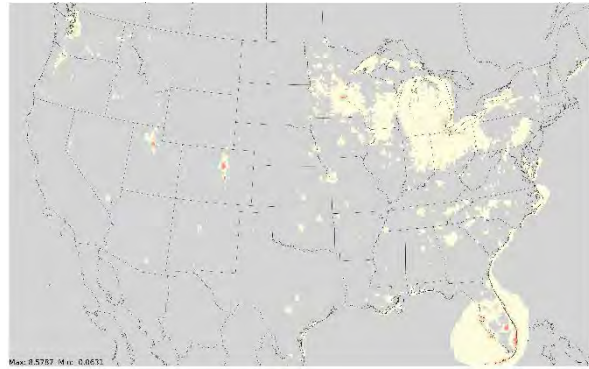


c

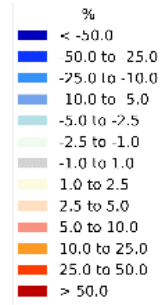
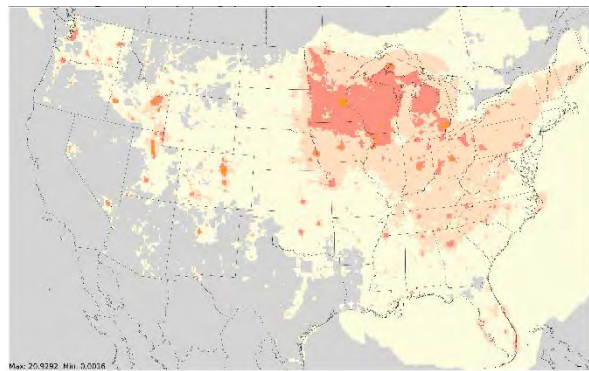


**Figure 9.9 Absolute difference in (a) annual average, (b) January average, and (c) July average acetaldehyde concentrations between “pre-RFS” and “with-RFS” scenarios for 2016**

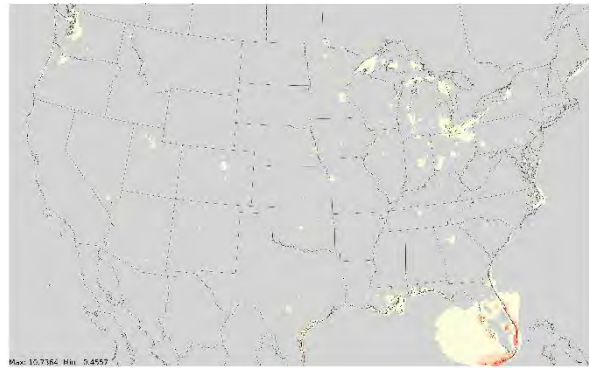
a



b

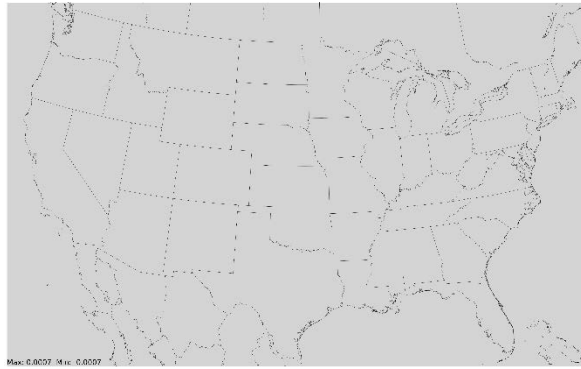


c

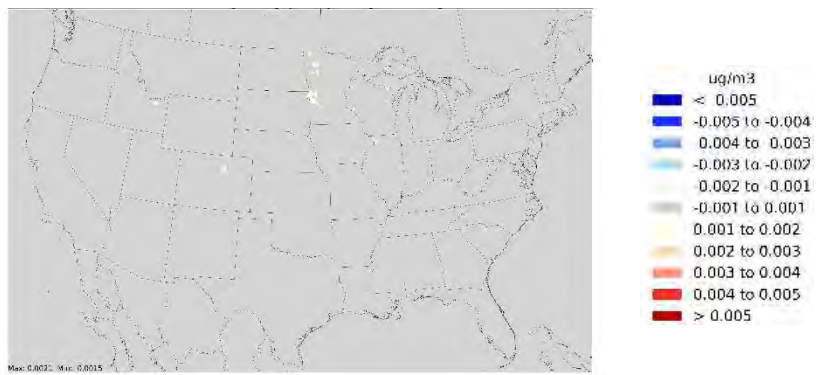


**Figure 9.10 Percent difference in (a) annual average, (b) January average, and (c) July average acetaldehyde concentrations between “pre-RFS” and “with-RFS” Scenarios for 2016**

a



b

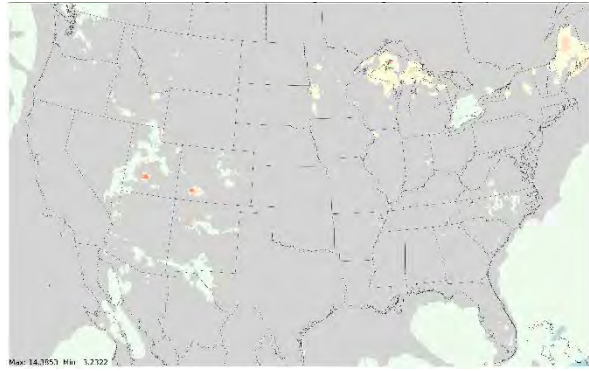


c

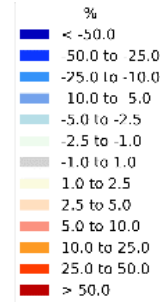
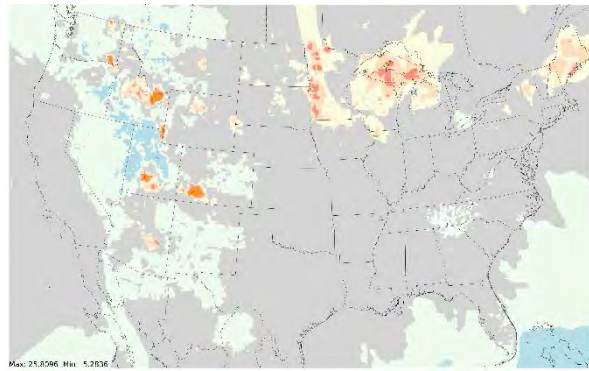


**Figure 9.11 Absolute difference in (a) annual average, (b) January average, and (c) July average acrolein concentrations between “pre-RFS” and “with-RFS” scenarios for 2016**

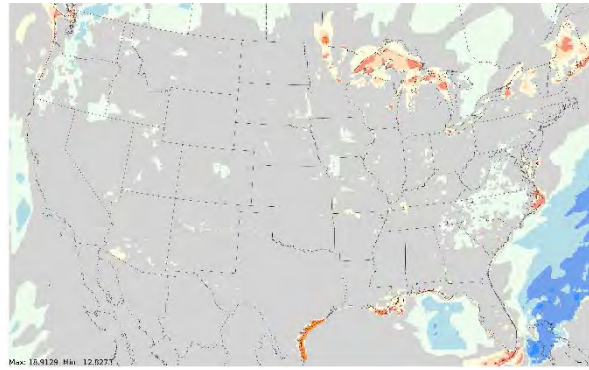
a



b

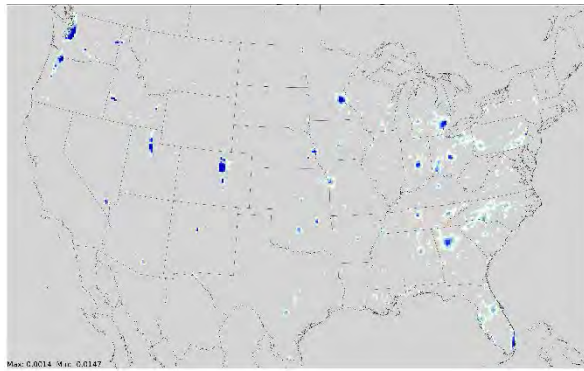


c

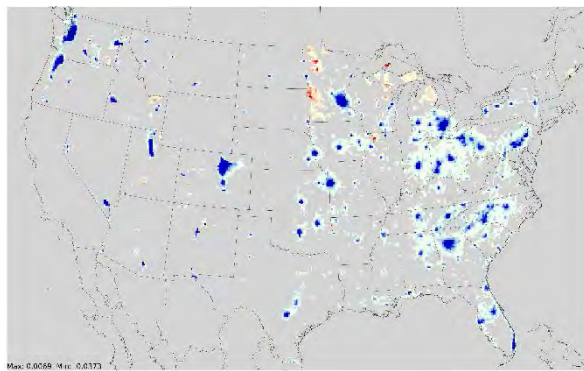


**Figure 9.12 Percent difference in (a) annual average, (b) January average, and (c) July average acrolein concentrations between “pre-RFS” and “with-RFS” scenarios for 2016**

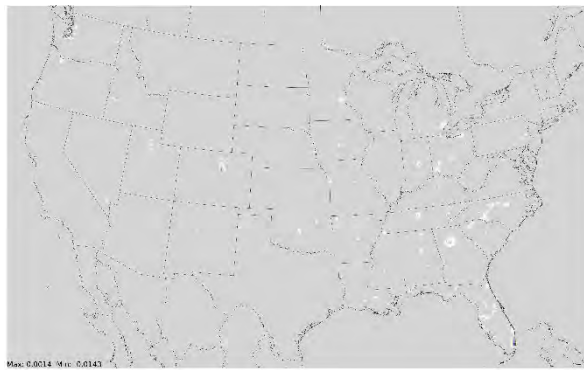
a



b



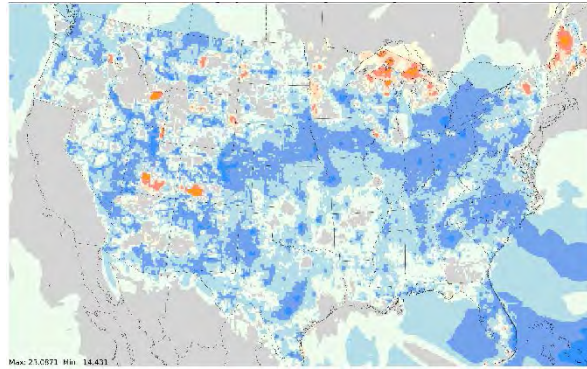
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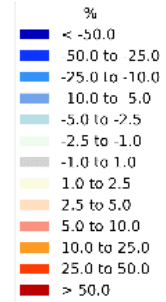
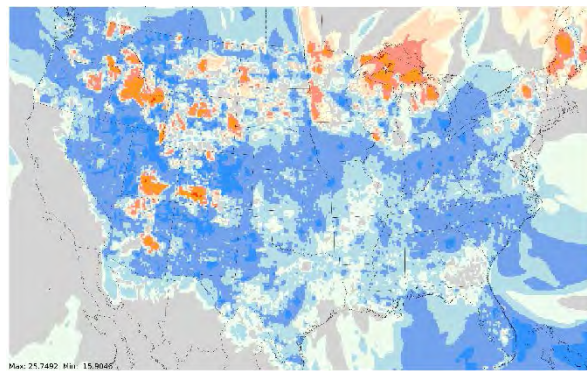
**Figure 9.13 Absolute difference in (a) annual average, (b) January average, and (c) July average 1,3-butadiene concentrations between “pre-RFS” and “with-RFS” scenarios for 2016**



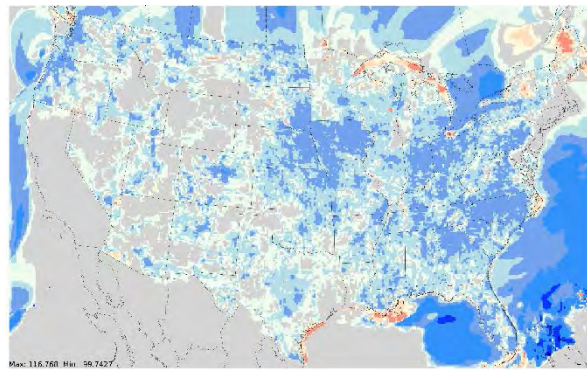
a



b

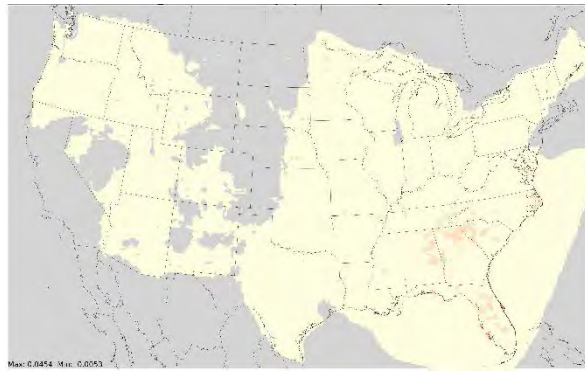


c

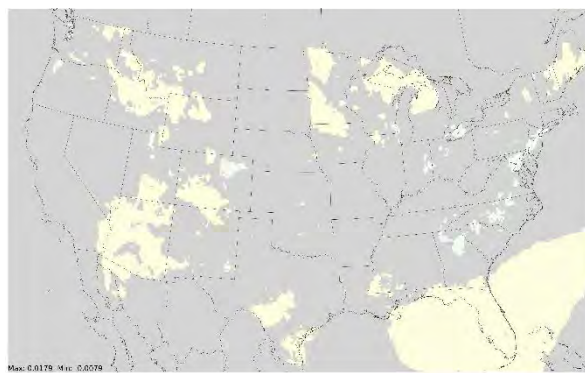


**Figure 9.14 Percent difference in (a) annual average, (b) January average, and (c) July average 1,3-butadiene concentrations between “pre-RFS” and “with-RFS” scenarios for 2016**

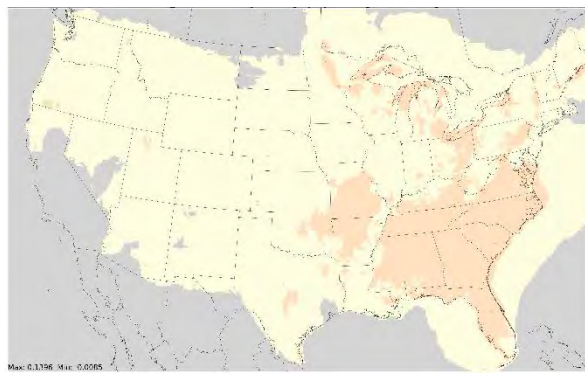
a



b



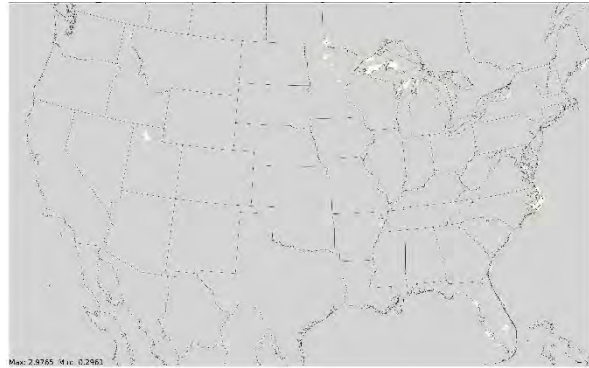
c



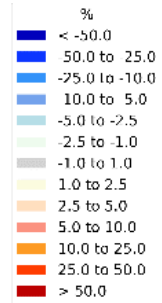
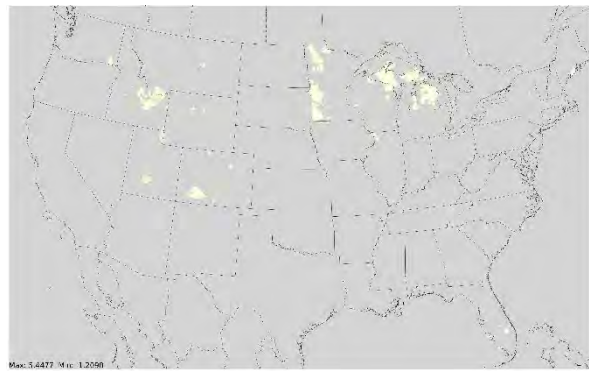
**Figure 9.15 Absolute difference in (a) annual average, (b) January average, and (c) July average formaldehyde concentrations between “pre-RFS” and “with-RFS” scenarios for 2016**



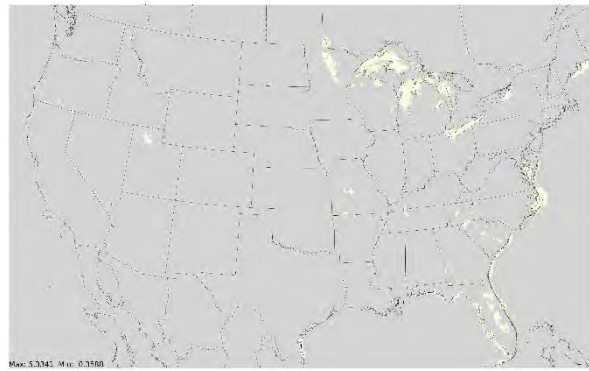
a



b

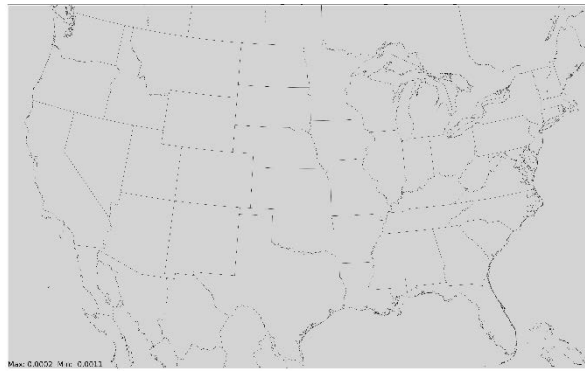


c

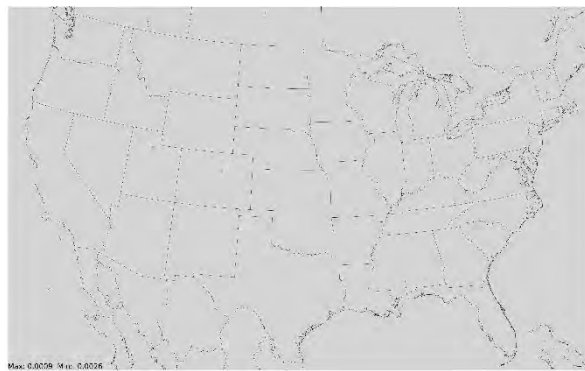


**Figure 9.16 Percent difference in (a) annual average, (b) January average, and (c) July average formaldehyde concentrations between “pre-RFS” and “with-RFS” scenarios for 2016**

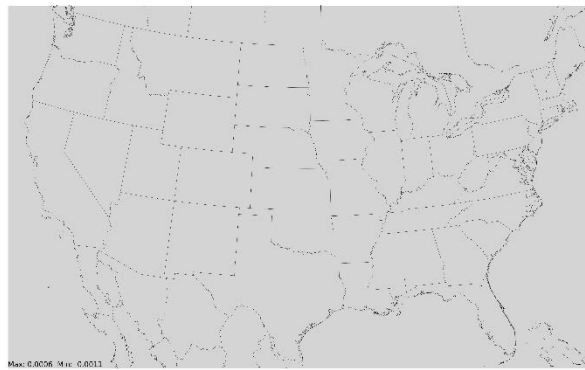
a



b

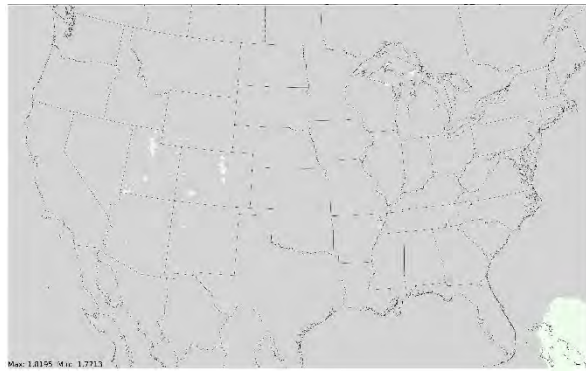


c

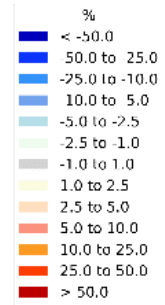
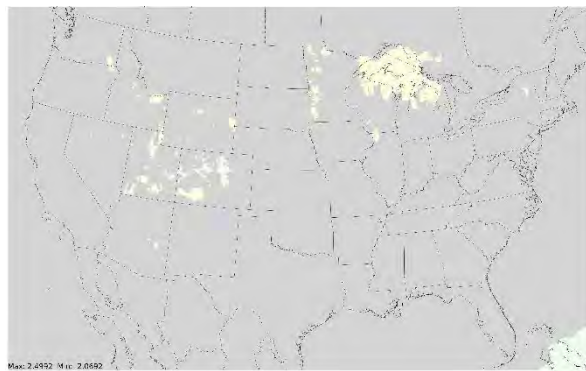


**Figure 9.17 Absolute difference in (a) annual average, (b) January average, and (c) July average naphthalene concentrations between “pre-RFS” and “with-RFS” scenarios for 2016**

a



b



c



**Figure 9.18 Percent difference in (a) annual average, (b) January average, and (c) July average naphthalene concentrations between “pre-RFS” and “with-RFS” scenarios for 2016**

## 9.5 National Emission Inventories for Criteria and Toxic Pollutants

In addition to the detailed county-level emission inventories generated for air quality modeling as described in Sections 6.2 and 6.3, national emission inventories were developed as a quality check, using the pre-aggregation feature of MOVES2014b at the national level.

In this approach, a single MOVES run can be performed to estimate the emission from the entire United States (50 states, excluding Puerto Rico and Virgin Islands). Input variables such as temperatures, fuel properties, effects of the Inspection and Maintenance (I/M) programs, and activity data including vehicle age distributions, speed distributions, road type distributions across the country are pre-aggregated and approximated by a single set of national average values. This approach is simpler but coarser compared to the approach used to develop the detailed inventories needed for air quality modeling.

The national emission inventories in calendar year 2016 for the “with-RFS” and “pre-RFS” cases are provided for both onroad vehicles and nonroad equipments along with their totals in Table 9-1.

The national-level emission inventory changes between the “pre-RFS” and the “with-RFS” cases as summarized in Table 9-1 show the trends generally similar to those for the national (CONUS, excluding California) total emissions summarized in Table 6.1 based on detailed emission inventories for air quality modeling.

**Table 9-1 Comparison of national emission inventory for onroad vehicles and nonroad equipments (tons per year) for the "with-RFS" and "pre-RFS" cases in calendar year 2016**

Pollutant	Sector	"with-RFS"	"pre-RFS"	diff	% diff
NO <sub>x</sub>	Total	5,268,765	5,130,752	138,014	2.7%
	Gasoline - onroad	1,854,796	1,770,048	84,748	4.8%
	Gasoline - nonroad	196,087	151,763	44,324	29.2%
	Diesel - onroad	2,315,858	2,306,916	8,941	0.4%
	Diesel - nonroad	851,516	851,516	0	0.0%
VOC	Total	3,056,755	2,969,017	87,739	3.0%
	Gasoline - onroad	1,645,135	1,570,525	74,609	4.8%
	Gasoline - nonroad	1,092,235	1,072,828	19,407	1.8%
	Diesel - onroad	230,086	236,364	-6,278	-2.7%
	Diesel - nonroad	76,799	76,799	0	0.0%
PM <sub>10</sub>	Total	381,585	384,543	-2,958	-0.8%
	Gasoline - onroad	133,357	132,666	692	0.5%
	Gasoline - nonroad	42,753	42,753	0	0.0%
	Diesel - onroad	138,201	141,850	-3,650	-2.6%
	Diesel - nonroad	65,085	65,085	0	0.0%

PM <sub>2.5</sub>	Total	245,220		247,965	-2,746	-1.1%
	Gasoline - onroad	46,735		46,123	612	1.3%
	Gasoline - nonroad	39,333		39,333	0	0.0%
	Diesel - onroad	94,094		97,452	-3,358	-3.4%
	Diesel - nonroad	63,133		63,133	0	0.0%
SO <sub>2</sub>	Total	29,600		28,978	622	2.1%
	Gasoline - onroad	22,784		22,162	622	2.8%
	Gasoline - nonroad	1,007		1,007	0	0.0%
	Diesel - onroad	4,546		4,546	0	0.0%
	Diesel - nonroad	1,148		1,148	0	0.0%
CO	Total	30,994,245		34,219,868	-3,225,623	-9.4%
	Gasoline - onroad	18,091,224		19,008,086	-916,862	-4.8%
	Gasoline - nonroad	11,311,391		13,597,070	-2,285,679	-16.8%
	Diesel - onroad	895,826		918,908	-23,082	-2.5%
	Diesel - nonroad	395,127		395,127	0	0.0%
Acetaldehyde	Total	35,697		27,596	8,101	29.4%
	Gasoline - onroad	15,359		8,605	6,754	78.5%
	Gasoline - nonroad	4,111		2,541	1,570	61.8%
	Diesel - onroad	9,103		9,326	-223	-2.4%
	Diesel - nonroad	6,810		6,810	0	0.0%
Acrolein	Total	4,549		4,463	87	1.9%
	Gasoline - onroad	925		854	71	8.3%
	Gasoline - nonroad	359		302	57	19.0%
	Diesel - onroad	1,570		1,611	-42	-2.6%
	Diesel - nonroad	1,653		1,653	0	0.0%
Benzene	Total	74,640		84,871	-10,231	-12.1%
	Gasoline - onroad	43,186		48,352	-5,166	-10.7%
	Gasoline - nonroad	26,715		31,731	-5,016	-15.8%
	Diesel - onroad	1,965		2,014	-49	-2.4%
	Diesel - nonroad	2,748		2,748	0	0.0%
1,3-Butadiene	Total	12,048		12,973	-925	-7.1%
	Gasoline - onroad	6,768		7,594	-825	-10.9%
	Gasoline - nonroad	4,597		4,678	-81	-1.7%
	Diesel - onroad	536		554	-18	-3.3%
	Diesel - nonroad	142		142	0	0.0%
Formaldehyde	Total	63,957		63,912	45	0.1%
	Gasoline - onroad	12,537		11,781	756	6.4%
	Gasoline - nonroad	7,156		7,376	-220	-3.0%

	Diesel - onroad	22,831		23,322	-491	-2.1%
	Diesel - nonroad	19,115		19,115	0	0.0%
Naphthalene	Total	6,439		6,529	-90	-1.4%
	Gasoline - onroad	2,208		2,223	-14	-0.6%
	Gasoline - nonroad	1,682		1,701	-18	-1.1%
	Diesel - onroad	2,260		2,316	-57	-2.5%
	Diesel - nonroad	288		288	0	0.0%