

## Chapter III: Environmental Impact

### A. Inventory Impacts of Tier 2/Sulfur

Today's proposal, if adopted, would reduce NO<sub>x</sub>, VOC, particulate, SO<sub>x</sub>, carbon monoxide, and hazardous air pollutant emissions from cars and light trucks by lowering the VOC, NO<sub>x</sub>, and PM emission standards for these vehicles and requiring that gasoline sulfur levels be reduced. Over time, the projected benefits of today's proposal would grow as vehicles meeting the new standards replace older, higher-emitting vehicles and as total VMT continues to grow. The results of our analysis of light-duty inventory levels with and without today's action are presented and discussed for each pollutant in the following sections. In all cases, "without Tier 2/Sulfur" refers to continuation of National LEV on in-use fuel as currently specified; sulfur levels for Conventional Gasoline are estimated at 330 ppm, summertime Phase 2 RFG levels are estimated at 150 ppm (i.e., baseline case). "With Tier 2/Sulfur" refers to implementation of a 30 ppm sulfur standard nationwide in 2004 and the phase-in of NO<sub>x</sub>, VOC, and PM standards proposed under today's action (i.e., control case).<sup>a</sup>

For this proposal, EPA developed new inventory projections for the United States excluding California, Alaska, and Hawaii.<sup>b</sup> These inventory projections can be divided into three major types of sources for the purpose of describing the methodologies used: stationary and area sources, nonroad mobile sources, and highway motor vehicles. To assess air quality need and the impact of today's proposal on urban areas, separate inventory analyses were also performed on four high ozone cities: New York, Chicago, Atlanta and Charlotte. Inventory estimates for each city were developed using the same data sources as the 47-state inventory discussed below, except where noted. Comprehensive inventories (47-state and four city) are presented in Appendix A with and without Tier 2/sulfur control, in 2005 (47-state only), 2007, 2010, 2015, 2020 and 2030.

These 47-state inventory projections are described more fully in this section. These projections differ in some respects from the inventory projections used for the ozone analyses

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<sup>a</sup>Today's proposal includes a provision for the averaging, banking and trading (ABT) of sulfur levels which would allow average sulfur levels to be higher than 30 ppm in 2004/2005 in exchange for sulfur control prior to 2004 (See Section IV.C.3.c.i of the preamble for a detailed discussion of this program). We expect that overall emission reductions from the ABT program between 2001 and 2005 would be consistent with implementation of 30 ppm in 2004 without prior sulfur reduction, and hence assumed the latter schedule for the control case results presented here.

<sup>b</sup>The 47-state region comprised of the U.S. minus California, Alaska and Hawaii is interchangeably referred to as "nationwide" throughout this section. Although excluded from this analysis, emission reductions will be realized in each of these states. Today's action applies fully to Alaska, Hawaii, and U.S. territories; California, although subject to a separate vehicle and fuel control program, will benefit from lower-emitting Federal vehicles migrating to and/or traveling within the state, as well as California vehicles operating on cleaner non-California fuel.

## **Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999**

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described in Section B.1. and the inventory projections used for the benefit/cost analyses described in Chapter VII. The methods used to develop the inventory projections presented in this section are described below. Subsequently, the differences between those methods and the methods used to develop the inventories used for the ozone analyses and benefit/cost analyses are described.

The 47-state inventory projections include updated emission estimates for the stationary, area, nonroad, and highway mobile source sectors.<sup>1</sup> For stationary and area sources, we relied on a set of 47-state projection inventories developed for this analysis by E.H. Pechan and Associates. Pechan used state inventories developed for the Regional Ozone Transport Rule (ROTR, 63 FR 57356, October 27, 1998) and Trends inventories for the non-OTAG states to develop a base year stationary and area source inventory. For sources not covered by emissions caps, the total emissions were grown using BEA-based growth factors. Emissions for large electric utilities were held constant at projected 2005 levels, consistent with emissions cap requirements in OTAG states and projections of shifts in types of fuel used in other states.

For nonroad mobile sources (except locomotives, aircraft, and commercial marine), we developed 47-state emission inventories using EPA's Draft NONROAD emissions model. This model is a significant update in data sources and methodologies compared to the NEVES inventories which have been the basis for nonroad emission estimates since 1992. Although NONROAD has only been released in draft form, the emissions estimation data and methods incorporated in it represent our most recent analysis of nonroad emission levels. For this reason, we chose to use the draft NONROAD model to develop our nonroad mobile source emission estimates used to evaluate the impact of the Tier 2/Sulfur proposal on emission inventories. The methods and data used in NONROAD are also consistent with the methods and data used in recent EPA proposed and final rules on nonroad engine standards and the nonroad emissions projections used here reflect all final and proposed standards for nonroad engines and equipment. Growth estimates in NONROAD are based on a linear projection of historical populations of nonroad equipment.

Because NONROAD does not yet include locomotives, aircraft, or commercial marine vessels, we had to use different sources to project emission inventories for these sources. Estimates of projected locomotive emissions were based on estimates in EPA's Final Rule on locomotive standards, adjusted to reflect the 47-state basis of the inventory described above. Commercial marine emissions were based on estimates in EPA's proposed commercial marine rule. Aircraft emissions estimates were based on Trends estimates adjusted to reflect a 47-state basis and grown using FAA growth estimates.

The most critical piece of our 47-state inventory analysis is the light-duty on-highway vehicle inventory. We are in the process of updating the on-highway mobile source emission factor models MOBILE (NO<sub>x</sub>, VOC and CO) and PART (PM and SO<sub>x</sub>), and the latest versions of these models (MOBILE6 and PART6) are not yet available. However, many of the modified inputs and assumptions which will be used in these models have at least been developed in draft form; thus, we were able to develop an up-to-date assessment of light-duty vehicle and truck

emission inventory for today's proposal using a model which incorporated available elements which have or will be proposed as part of the MOBILE6 and PART6 models. This model, referred to as the Tier 2 Model<sup>c</sup>, reflects updated assessments of in-use emission deterioration and off-cycle emissions, fuel sulfur impacts, and the increase in truck sales relative to cars. The model also reflects existing national and local motor vehicle control programs including National LEV (NLEV), Supplemental Federal Test Procedure (SFTP), On-Board Diagnostics (OBD), reformulated gasoline (RFG) and Inspection/Maintenance (I/M) programs. We used this model to develop baseline emission estimates assuming that NLEV program continued in perpetuity (i.e., that there would be no Tier 2/Sulfur standards implemented) and to develop emission estimates for various control scenarios.<sup>2</sup>

The 47-state nonexhaust VOC emission inventory was developed using MOBILE5b, since MOBILE6 estimates of evaporative emissions were not available at the time of the analysis. However, we incorporated available elements of MOBILE6 where possible, including mileage accumulation, VMT mix, and age distribution.<sup>3</sup> A modified version of MOBILE5b was also developed to estimate the benefits of today's proposed evaporative standards.<sup>4</sup>

47-state inventory estimates for heavy-duty gasoline, heavy-duty diesel and motorcycles also incorporated available aspects of MOBILE6 and PART6, including new base emission rates, defeat device emissions for heavy-duty diesel vehicles, off-cycle emissions, mileage accumulation and age distribution. New standards recently finalized for heavy-duty diesel vehicles were accounted for in these inventories, as were standards expected to be proposed for heavy-duty gasoline vehicles.<sup>5,6</sup>

To generate inventory projections, we needed to combine the on-highway emission factors generated using the Tier 2 Model or other means described above with estimates of on-highway vehicle miles traveled (VMT). For our 47-state inventory analysis, VMT estimates through 2010 were based on EPA's Trends Report through 2010. Beyond 2010, we developed VMT estimates for light-duty cars and trucks based on current trends in VMT growth as reported by NHTSA. From 2010 through 2015, we project that 47-state light-duty VMT will grow at a rate of 2.1 percent per year compounded; beyond 2015, we estimate VMT growth will be reduced to a linear 2.1 percent per year (i.e., 2.1 percent of 2015 VMT added incrementally in successive years). Projected 47-state VMT levels for heavy-duty gasoline and diesel vehicles were developed based on data from EPA's Trends Report.

Consistent with EPA's Trends Report, the 47-state inventory estimates were developed

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<sup>c</sup> The Tier 2 Model is the next generation of Modified MOBILE5b (T2AT), the inventory model used in the Tier 2 Study. Since the study, the model has been transferred to a Microsoft Excel platform, updated extensively and expanded to include SO<sub>x</sub> and PM emissions. The development of this model and generation of light-duty inventory results presented in this section are outlined in the technical report "Development of Light-Duty Emissions Inventory Estimates in the Notice of Proposed Rulemaking for Tier 2 and Sulfur Standards" contained in Docket No. A-97-10. The Tier 2 Model is being made available in concurrence with the publication of today's proposal.

## **Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999**

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on the basis of annual tons emitted. Annual VMT estimates were used in conjunction with emission factors which reflected seasonal fuel control (i.e., low sulfur RFG in the summer only). Because of limitations in the Tier 2 Model, however, seasonal temperature adjustments were not made. Thus, the annual tonnages developed for the 47-state inventories are annual tonnages assuming summertime temperatures year-round. The effect of using summertime conditions to estimate year-round emissions is to understate annual NO<sub>x</sub> emissions by about seven percent. Estimates of NO<sub>x</sub> emission reductions from changes in standards will be similarly underestimated. EPA does not consider this small error to be material.

The urban (four-city) analysis was performed on the basis of summertime (May through September) emissions. Stationary and area source inventories were provided by E.H. Pechan and Associates and were based on state inventories developed for the Regional Ozone Transport Rule (ROTR). Emissions from nonroad equipment were estimated using the NONROAD model, which includes the capability to allocate emissions to the county level. For nonroad equipment not included in the NONROAD model (locomotives, aircraft, and commercial marine) we did not have enough information to directly allocate the 47 inventories described above down to the county level for these urban areas. However, E.H. Pechan has calculated national and local inventories for these categories and areas using older assumptions about future emissions standards. We used those older inventories to calculate the proportion of national emissions from locomotives, aircraft, and commercial marine engines in the four urban areas. We then applied those proportions to the our newer national inventories for the three categories to estimate emissions for locomotives, aircraft, and commercial marine engines in the four urban areas using our latest assumptions about the effects of new standards.

Summertime VMT estimates for each area used in generation of OTAG inventories were provided by Pechan for 1995 and 2007; in order to more closely match localized VMT growth trends, the values were linearly interpolated between these years, and extrapolated linearly beyond 2007. Emission factors for highway vehicles were derived using the same methods described above for the 47-state inventories, but with local specific inputs, such as I/M programs or reformulated gasoline, where applicable.

The emission inventories used for the ozone analyses described in Section B.1. of this chapter and the benefit/cost analyses described in Chapter VII were developed prior to the 47-state inventory described in this section. The ozone analysis and benefit/cost analysis inventories differ from one another and from the 47-state inventories in several respects. It should be noted, however, that we used the emission inventory analyses described in this section to determine the change in emissions from the proposed Tier 2/Sulfur standards for both of these analyses.

The inventories used for the ozone modeling are described more fully in Section B.1. To develop the car and light truck baseline inventories (the inventories that would result if the Tier 2/Sulfur proposal were not adopted), we used the car and light truck inventories developed for the ROTR; these inventories were based on MOBILE5 inputs and emission factors. To estimate the change in emissions from cars and light trucks that would occur if the proposed Tier 2/Sulfur standards were implemented, we used the same methods used to develop the 47-state emission

inventories (as described in this section). The inventories for highway heavy-duty engine emissions and nonroad emissions were based on the emission modeling tools that were available to the Ozone Transport Assessment Group (OTAG) and were used during that process and the subsequent rulemaking process that resulted in promulgation of the ROTR. The highway heavy-duty engine emissions were based on MOBILE5 inputs and emission factors; the nonroad emissions were based on NEVES inputs.

The benefit/cost analyses described in Chapter VII used an even earlier set of estimates for highway, nonroad, and stationary and area source emissions that was developed before the ROTR was proposed or promulgated. These estimates were developed using the emission modeling tools available at the time the Regulatory Impact Analyses for the revised ozone and PM NAAQS rules were developed.<sup>7</sup> The inventories used in the benefit/cost analyses are described more fully in Chapter VII.

### **1. NO<sub>x</sub>**

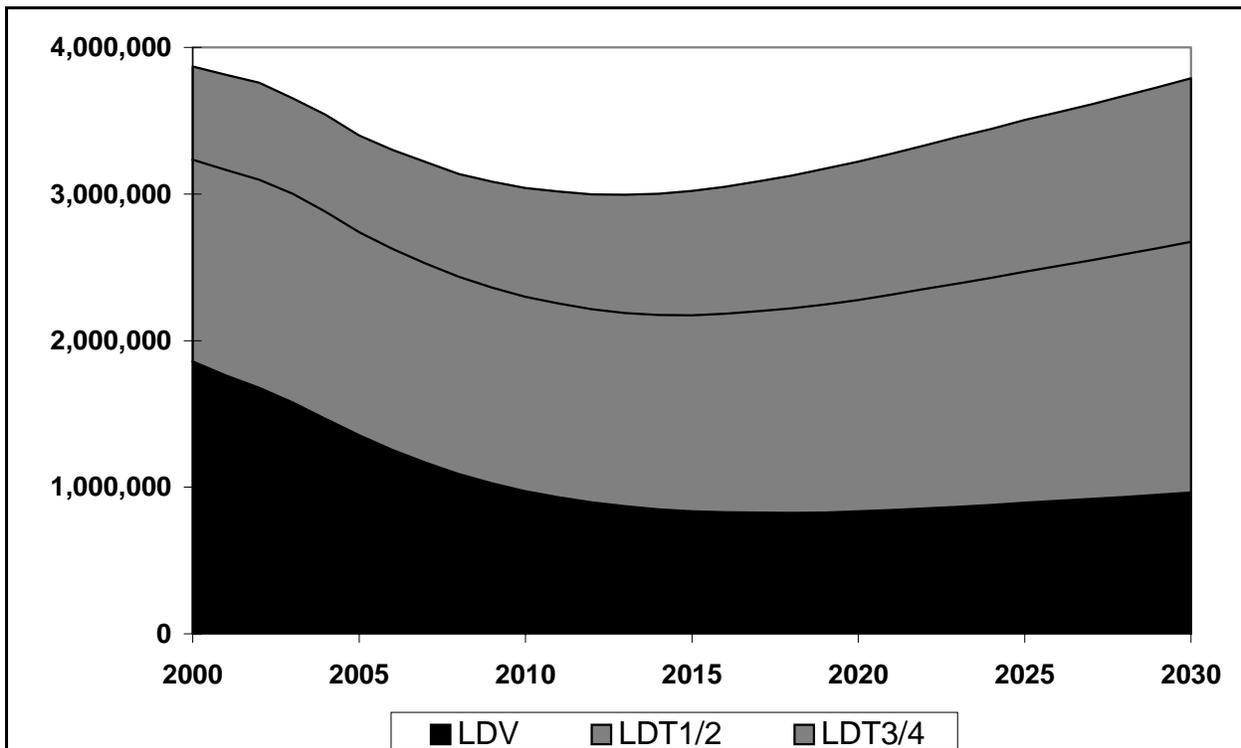
#### **a. Light-Duty NO<sub>x</sub> Trends Without Tier 2/Sulfur**

Total NO<sub>x</sub> emissions produced annually in the 47 states by cars and trucks without Tier 2/Sulfur controls are shown in Table III-1 and Figure III-1, broken down by relative contribution of cars (light-duty vehicles, or LDVs), LDT1s and 2s (light pickup trucks, minivans and most sport utility vehicles), and LDT3s and 4s (heavier pickup trucks and sport utility vehicles). As shown, total light-duty emissions decline from approximately 3.9 million tons to 3.0 million tons between 2000 and 2010 due to turnover of Tier 1 and NLEV vehicles and the phase in of off-cycle standards (SFTP). By 2014, however, the effect of these control programs begins to be offset by increases in overall VMT, in conjunction with the shift of VMT from cars to trucks. Light-duty emissions increase to 3.2 million tons by 2020 and 3.8 million tons by 2030, such that the gains from the Tier 1, NLEV and SFTP control programs are almost completely eradicated by VMT growth.

**Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999**

**Table III-1. 47-State Light Duty NOx Emissions Without Tier 2/Sulfur (Annual Tons)**

Year	Light-Duty Emissions	Contribution by Vehicle Class		
		LDV	LDT1/2	LDT3/4
2000	3,869,383	48.0%	35.6%	16.4%
2004	3,539,655	41.4%	39.9%	18.7%
2007	3,218,530	36.3%	42.2%	21.5%
2010	3,041,639	32.0%	43.6%	24.4%
2015	3,020,806	27.7%	44.3%	28.0%
2020	3,221,151	25.9%	44.8%	29.3%
2030	3,790,840	25.4%	45.1%	29.5%



**Figure III-1. 47-State Light-Duty NOx Emissions Without Tier 2/Sulfur (Annual Tons)**

The impact of steady truck growth on overall light-duty NOx emissions is clearly

demonstrated in the preceding figure. In 2000, we project that trucks will produce 50 percent of overall NOx emissions. Over the next 30 years, trucks will grow to dominate light-duty NOx emissions due to the combined effects of sales migration, higher mileage accumulation rates, longer lifespan, and more relaxed emission standards relative to LDVs. By 2010, we project trucks will make up two-thirds of light-duty NOx emissions; by 2020, nearly three-quarters of all light-duty NOx emissions will be produced by trucks. As shown in Figure III-1, the decrease in overall light-duty emission levels is due solely to reductions in LDV emissions. The benefits from Tier 1, NLEV and SFTP are not as pronounced for trucks, and are offset almost immediately by growth in truck VMT. As a result, truck emissions are stable through 2010 and begin increasing steadily beyond this as VMT growth overtakes the gains from existing control programs.

The emission trends for the four urban areas we analyzed show similar behavior. Although the presence of localized control programs (I/M and in some cases, RFG) do serve to delay the upturn in light-duty emissions, they are not sufficient to counteract the effects of VMT growth. As shown in Table III-2, light-duty emissions decrease steadily in each city through 2010. Emission trends beyond 2010 depend on the rate of VMT growth in each city. In New York, which is projected to have relatively low VMT growth, emissions continue to decrease steadily through 2015 before leveling off and then turning upwards by 2017. In Chicago, Atlanta, and Charlotte, emissions begin to level off by 2010. Emissions start to increase in 2017 in Chicago, 2015 in Atlanta, and 2013 in Charlotte. For the latter two cities, emissions increase at a rapid rate beyond these years. We project that Atlanta's emission reductions achieved from programs currently in place will be almost fully offset by rapid VMT growth by 2030, while we project Charlotte's rapid VMT growth to cause emissions in 2030 to be over 10 percent higher than in 2000.

**Table III-2. Four-City Light-Duty NOx Emissions Without Tier 2/Sulfur (Summer Tons)**

<i>Year</i>	<i>New York</i>	<i>Chicago</i>	<i>Atlanta</i>	<i>Charlotte</i>
2000	78,287	37,037	33,267	4,714
2004	66,857	32,314	30,912	4,526
2007	57,753	28,399	28,313	4,230
2010	51,811	25,958	26,846	4,081
2015	47,634	24,440	26,384	4,109
2020	48,033	25,080	27,721	4,402
2030	52,280	28,165	32,018	5,239

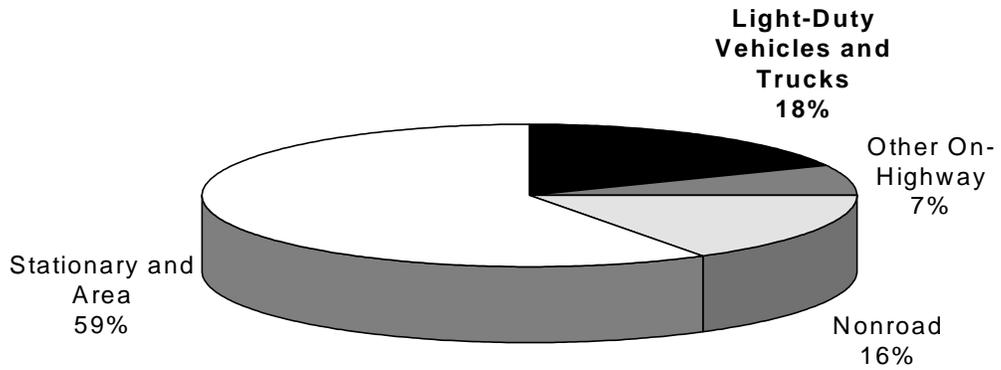
Figures III-2 and III-3 show our projections of the contribution of light-duty vehicles and

## **Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999**

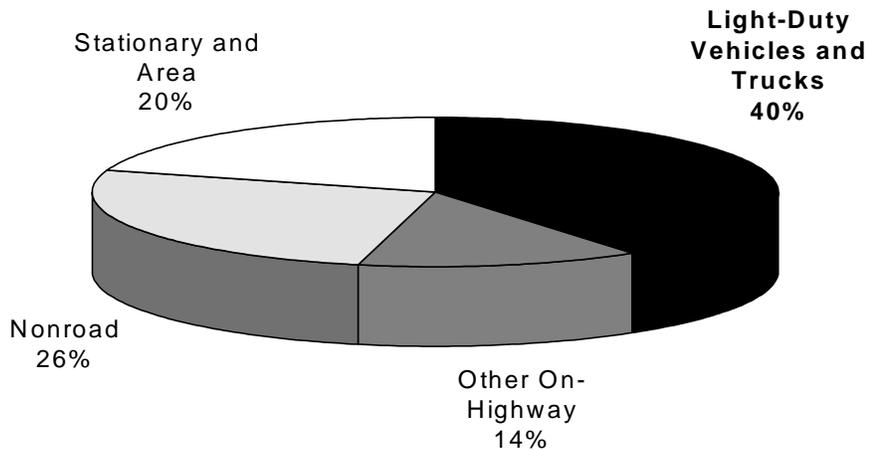
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trucks to the total NOx inventory (i.e., NOx emission from all sources, including stationary, area, nonroad) in the 47 states and in Atlanta, in 2020. Table III-3 shows this same contribution across the 47 states and all four cities from 2007 through 2030. Across the 47 states, cars and trucks produce nearly one-fifth of total NOx emissions across all years. In urban areas, however, this contribution can be significantly higher. Atlanta provides the most striking example of this; we project that roughly 40 percent of all NOx emissions will be produced by cars and trucks through 2030. While less than Atlanta, the light-duty contribution in New York is significantly higher than the national estimates; from 2007 through 2030, we project that 27 to 29 percent of all emissions in this area will be produced by light-duty cars and trucks. We estimate the contribution in Chicago and Charlotte to be slightly higher but comparable to the 47-state estimate of one-fifth of the total NOx inventory.

Light-duty NOx contribution in urban areas is generally higher than the 47-state region because of the increased concentration of VMT, in conjunction with the decreased prevalence of significant NOx contributors which are largely in non-urban areas (primarily utilities and agricultural nonroad sources). We expect that this trend will be consistent across many high-ozone urban areas.



**Figure III-2: Breakdown of Total 2020 47 State NOx Inventory Without Tier 2**



**Figure III-3. Breakdown of Total 2020 Atlanta NOx Inventory Without Tier 2**

## Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999

**Table III-3. Light-Duty Contribution to Total NOx Inventory Without Tier 2/Sulfur**

<i>Year</i>	<i>47-state</i>	<i>New York</i>	<i>Chicago</i>	<i>Atlanta</i>	<i>Charlotte</i>
2007	17%	29%	19%	38%	18%
2010	17%	28%	18%	38%	19%
2015	17%	27%	18%	39%	19%
2020	18%	27%	19%	40%	21%
2030	20%	28%	19%	42%	22%

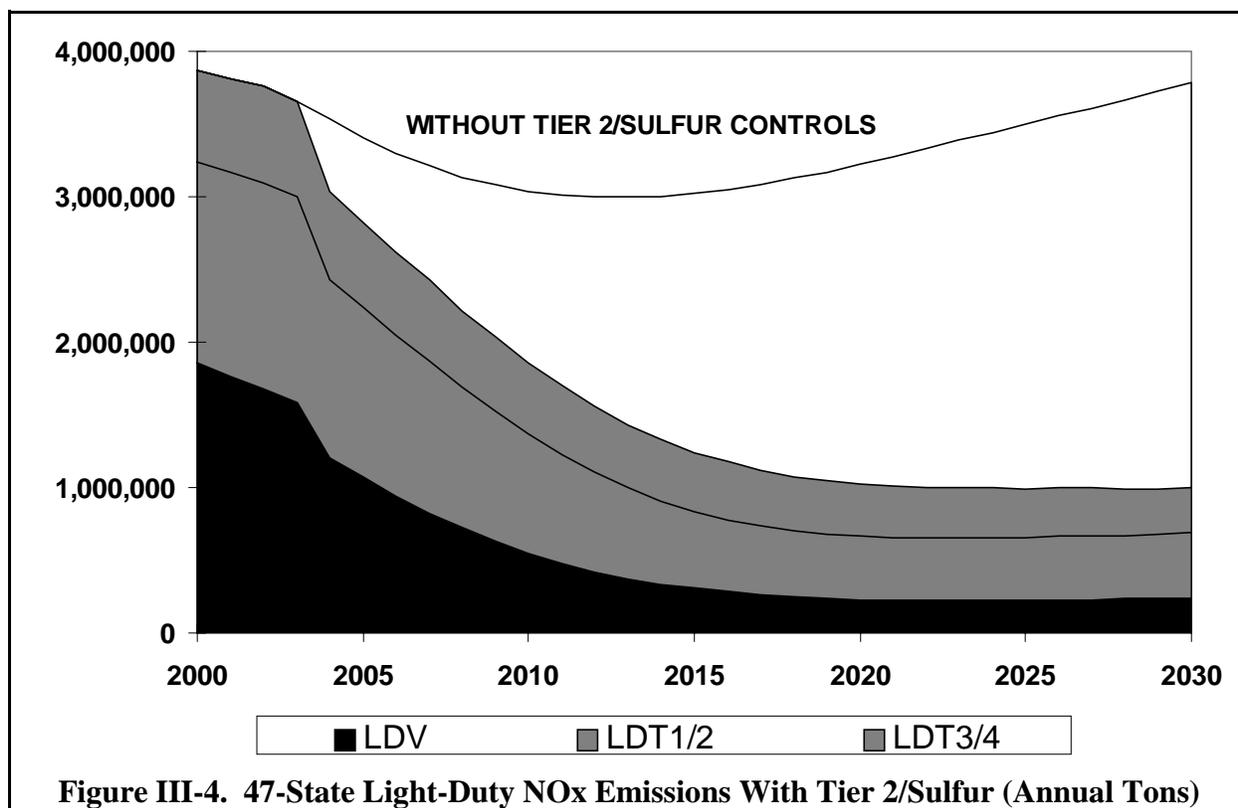
### b. NOx Reductions Due To Tier 2/Sulfur

Today's proposal would provide substantial reductions in NOx emissions from cars and trucks. The implementation of low sulfur fuel in 2004 would afford an immediate drop in NOx emissions, while the phase-in of tighter vehicle standards would continue to reduce emissions over time, serving to mitigate through at least 2030 the projected upward trend in light-duty NOx emissions that would occur with no control. Table III-4 contains annual tons of NOx we project would be reduced by today's proposal, encompassing benefits of low sulfur fuel and the introduction of Tier 2 light-duty vehicle and light-duty truck standards. Figure III-4 shows annual 47-state light-duty NOx emissions with implementation of the Tier 2/Sulfur program, broken down by LDV, LDT1/2 and LDT3/4 categories.

**Table III-4. 47-State Light-Duty NOx Reductions Due To Tier 2/Sulfur (Annual Tons)**

<i>Year</i>	<i>Light-Duty Emissions Without Tier 2/Sulfur</i>	<i>Light-Duty Emissions With Tier 2/Sulfur</i>	<i>Emissions Reduced</i>	<i>Percent Reduction in Baseline Inventory</i>	
				<i>Light-Duty</i>	<i>All Sources*</i>
2004	3,539,655	3,037,144	502,511	14%	-
2007	3,218,530	2,422,796	795,734	25%	4%
2010	3,041,639	1,859,316	1,182,323	39%	7%
2015	3,020,806	1,241,925	1,778,881	59%	10%
2020	3,221,151	1,023,038	2,198,113	68%	12%
2030	3,790,840	1,004,495	2,786,345	74%	15%

\* Includes emission reductions from Heavy-Duty Gasoline Vehicles due to sulfur control



As shown, the implementation of reduced sulfur levels in 2004 would result in an immediate benefit of over one-half million tons, a 14 percent drop in uncontrolled 2004 light-duty emissions; this is the equivalent of emissions produced by over 26 million pre-Tier 2 cars and trucks.<sup>d,8</sup> In 2004, nearly all of the benefits would be due to reduced emissions from Tier 0, Tier 1 and NLEV vehicles.

After 2004, emissions are reduced further as the fleet turns over to predominantly Tier 2 vehicles operating on low sulfur fuel, versus NLEVs and Tier 1 trucks operating on current in-use sulfur levels. By 2020, the projected benefit represents a two-thirds reduction in 2020 light-duty emissions without Tier 2/Sulfur, equivalent to the emissions from over 166 million pre-Tier 2 cars and trucks. NOx emissions from all sources would be reduced by 12 percent.

We project that light-duty emissions would continue to decrease beyond 2020, reversing the upward emissions trend in the baseline case brought on by VMT growth. By 2030, essentially the entire fleet would consist of Tier 2 vehicles. The projected benefit of 2.8 million tons in this year represents a nearly three-quarters reduction in 2030 light-duty emissions without Tier 2/Sulfur, equivalent to the emissions from 213 million pre-Tier 2 cars and trucks. These emission reductions would amount to 15 percent of total man-made NOx emissions in that year in the absence of today's proposal.

<sup>d</sup>i.e., vehicles that would be on the road in the absence of Tier 2/Sulfur control.

## Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999

The estimated percentage reductions in total inventory presented in Table III-4 include benefits that would be realized on heavy-duty gasoline vehicles due to sulfur control. We estimate these heavy-duty emission reductions to be on the order of approximately 30,000 tons per year for every year starting in 2004, as shown in Appendix A.

NOx reductions due to today's proposal would be of a similar scope in urban areas. Table III-5 shows NOx emissions reduced due to Tier 2/Sulfur control, and light-duty highway vehicle emissions remaining, for each of the four cities. Table III-6 presents these reductions in terms of the percentage of baseline light-duty and total inventory reduced.

**Table III-5. Four-City Light-Duty NOx Emissions With Tier 2/Sulfur (Summer Tons)**

<i>Year</i>	<i>New York</i>		<i>Chicago</i>		<i>Atlanta</i>		<i>Charlotte</i>	
	<i>Reduced</i>	<i>Remain</i>	<i>Reduced</i>	<i>Remain</i>	<i>Reduced</i>	<i>Remain</i>	<i>Reduced</i>	<i>Remain</i>
2004	7,368	59,489	3,062	29,252	4,550	26,362	666	3,860
2007	12,139	45,614	5,546	22,853	7,346	20,967	1,098	3,133
2010	18,432	33,380	8,915	17,043	10,975	15,871	1,668	2,413
2015	27,544	20,089	14,020	10,421	16,483	9,901	2,567	1,542
2020	33,177	14,857	17,296	7,784	20,188	7,534	3,206	1,196
2030	39,488	12,792	21,259	6,906	25,160	6,857	4,117	1,122

**Table III-6. Percent Reduction From Light-Duty and Total Baseline NOx Emissions in Four Cities**

Year	New York		Chicago		Atlanta		Charlotte	
	Light-Duty	All Sources*						
2004	11%	-	9%	-	15%	-	15%	-
2007	21%	6%	20%	4%	26%	10%	26%	5%
2010	36%	10%	34%	6%	41%	16%	41%	8%
2015	58%	16%	57%	10%	62%	25%	62%	12%
2020	69%	19%	69%	13%	73%	30%	73%	15%
2030	76%	21%	75%	15%	79%	33%	79%	18%

\* Includes emission reductions from Heavy-Duty Gasoline Vehicles due to sulfur control

The magnitude of reductions in urban areas reflect those nationally. An immediate reduction in light-duty emission would result from sulfur control, even in RFG areas (New York and Chicago). Over one-third of baseline light-duty emissions would be reduced by 2010 in each city. Light-duty emissions would be reduced by roughly 70 percent in 2020 and over 75 percent in 2030. Reductions in Atlanta and Charlotte are consistently larger in percentage terms than in New York and Chicago because they are not RFG areas; emission reductions in non-RFG urban areas would be particularly large since these areas would tend to have higher-sulfur fuel than RFG areas in the absence of today's proposal. We project that emissions would continue to decrease through at least 2028 in all four cities, indicating that today's program would be successful in reducing light-duty NOx emissions in the face of high VMT growth rates.

The impact on total inventory would also be significant, particularly in New York and Atlanta. By 2020, we project that the total NOx inventory would be reduced by nearly one-fifth in New York and one-third in Atlanta due to Tier 2/Sulfur control.

Concurrently, we project that the light-duty contribution to total NOx emissions would drop significantly. Figures III-5 and III-6 show our 2020 projections of this contribution in the 47 states and in Atlanta with Tier 2/Sulfur control. Table III-7 shows this same contribution across the 47 states and all four cities from 2007 through 2030. In 2020, we project that the light-duty contribution would drop to seven percent nationally, from 18 percent without Tier 2/Sulfur control. This trend is similar across the four cities, depending on the level of contribution without Tier 2/Sulfur control. We project that with Tier 2/Sulfur control, car and truck emissions would contribute 10 percent of total emissions in New York (down from 27 percent), seven percent in Chicago and Charlotte (down from 19 percent and 21 percent), and 16 percent in Atlanta (down from 40 percent) in 2020.

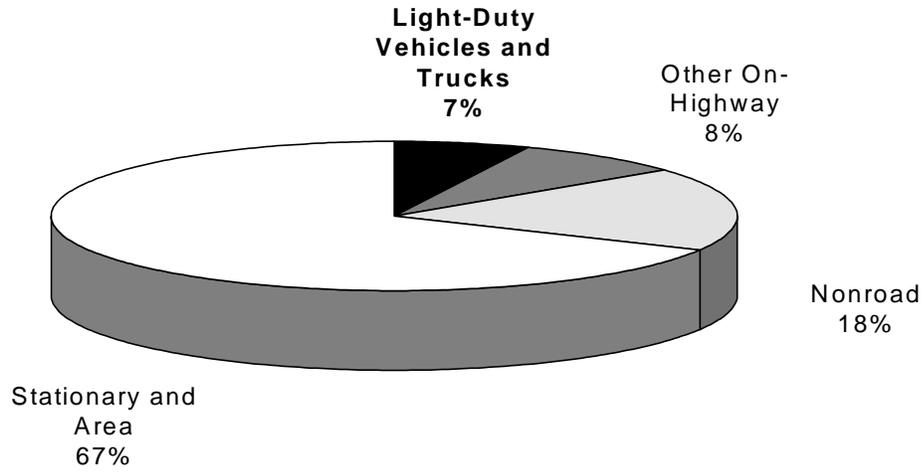


Figure III-5. Breakdown of Total 2020 47-State NOx Inventory With Tier 2/Sulfur

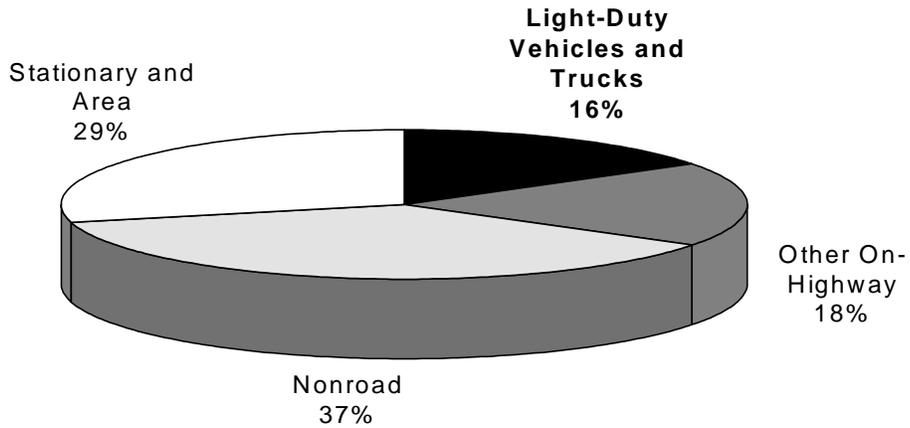


Figure III-6. Breakdown of Total 2020 Atlanta NOx Inventory With Tier 2/Sulfur

**Table III-7. Light-Duty Contribution to Total NOx Inventory With Tier 2/Sulfur**

<i>Year</i>	<i>47 State</i>	<i>New York</i>	<i>Chicago</i>	<i>Atlanta</i>	<i>Charlotte</i>
2007	14%	24%	16%	31%	14%
2010	11%	20%	13%	27%	12%
2015	8%	14%	9%	20%	8%
2020	7%	10%	7%	16%	7%
2030	6%	9%	6%	13%	6%

**c. NOx Emission Reductions From Other Options**

We developed 47-state light-duty emission inventory projections for three alternative vehicle/fuel control options to allow comparison with the emission reductions projected to result from today's action. These alternative options are:

- 1) Car and truck emission standards and implementation schedule as proposed in today's action in conjunction with sulfur control proposed to EPA by the American Petroleum Institute (API) and National Petroleum Refiners Association (NPRA). Under this plan, sulfur would be reduced in 2004 to 150 ppm in the eastern half of the U.S., referred to as the "API NOx Control Region", and 300 ppm in the remainder of the 49-state region.<sup>e</sup>
- 2) Option (1) above with implementation of a "rebuttable" element of the API/NPRA proposal in which sulfur would be reduced to 30 ppm in 2010 in the API NOx Control Region, while the remainder of the country remains at 300 ppm.
- 3) Sulfur control as proposed in today's action in conjunction with the default Tier 2 car and truck emission standards contained in the Clean Air Act. Under this alternative, LDVs and LDT1s would be required to meet full useful life emission standards of 0.125 g/mi NMHC and 0.20 g/mi NOx, assumed for this analysis to follow the implementation schedule for Tier 2 standards contained in today's proposal. LDT2s would be subject to California's applicable LEV I standards in 2004, while LDT3s and LDT4s would remain at Tier 1 levels.

For Options 1 and 2, the effects of sulfur irreversibility were accounted for using the methodology described in detail in Appendix B. In short, all cars and trucks complying with the Supplemental Federal Test Procedure (SFTP) were assigned an irreversibility effect of 50

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<sup>e</sup>The API/NPRA fuel proposal is discussed in detail in the Preamble, Section IV.C.1

## Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999

percent, meaning that vehicles within the API NOx Control Region exposed to higher sulfur levels outside the region would experience a permanent degradation in emissions performance equivalent to the average of emissions generated on fuel in and outside of the API Region. It was assumed that at any given time 25 percent of cars and trucks in the API NOx Control Region fleet would have traveled outside of the region, and hence been exposed to higher sulfur fuel.<sup>f</sup>

47-state light-duty NOx emissions projected for these options are shown in Tables III-8, in comparison with today's proposal. Table III-9 provide a direct comparison to today's proposal in terms of shortfall (i.e., emission reductions "lost" by the three options compared to today's proposal) and total benefits relative to the Tier 2/Sulfur proposal.

**Table III-8. 47-State Light-Duty NOx Reductions From Alternative Control Options (Annual Tons)<sup>g</sup>**

<i>Option:</i>	<i>Today's Proposal</i>	<i>1</i>	<i>2</i>	<i>3</i>
<i>Vehicle Program:</i>	<i>Proposed Tier 2</i>	<i>Proposed Tier 2</i>	<i>Proposed Tier 2</i>	<i>Clean Air Act Default</i>
<i>Fuel Program:</i>	<i>Proposed Tier 2</i>	<i>API No 30 ppm</i>	<i>API 30 ppm API Region 2010</i>	<i>Proposed Tier 2</i>
2007	795,733	397,886	397,886	611,020
2010	1,182,323	750,100	1,020,812	740,258
2020	2,198,113	1,713,531	2,000,129	1,026,690

<sup>f</sup>The baseline emission inventory estimates presented here do not account for sulfur irreversibility effects in RFG areas. Although vehicles in these areas will likely experience irreversibility effects due to exposure to higher sulfur levels during winter months, the overall impact on baseline emissions are expected to be small because a) LDT2/3/4s are less sensitive to sulfur under NLEV than expected under the standards proposed in today's action, and b) vehicles operating on summertime RFG make up a relatively small portion (less than 15%) of annual VMT in the 47-state region. Accounting for this effect would serve to increase the estimated benefits of today's proposal.

<sup>g</sup>Although not shown, Options 1 and 2 will also increase emissions from heavy-duty gasoline vehicles relative to today's action due to higher sulfur levels.

**Table III-9. NOx Reduction Shortfall From Alternative Control Options Relative to Today’s Proposal**

<i>Option:</i>	<i>1</i>		<i>2</i>		<i>3</i>	
<i>Vehicle Program:</i>	<i>Proposed Tier 2</i>		<i>Proposed Tier 2</i>		<i>Clean Air Act Default</i>	
<i>Fuel Program:</i>	<i>API No 30 ppm</i>		<i>API 30 ppm API Region 2010</i>		<i>Proposed Tier 2</i>	
<i>Year</i>	<i>Shortfall (Annual Tons)</i>	<i>Benefit Relative to Tier 2/Sulfur Proposal</i>	<i>Shortfall (Annual Tons)</i>	<i>Benefit Relative to Tier 2/Sulfur Proposal</i>	<i>Shortfall (Annual Tons)</i>	<i>Benefit Relative to Tier 2/Sulfur Proposal</i>
2007	397,848	50%	397,848	50%	184,713	77%
2010	432,223	63%	162,012	86%	442,066	63%
2020	484,582	78%	197,984	91%	1,171,423	47%

**2. VOC**

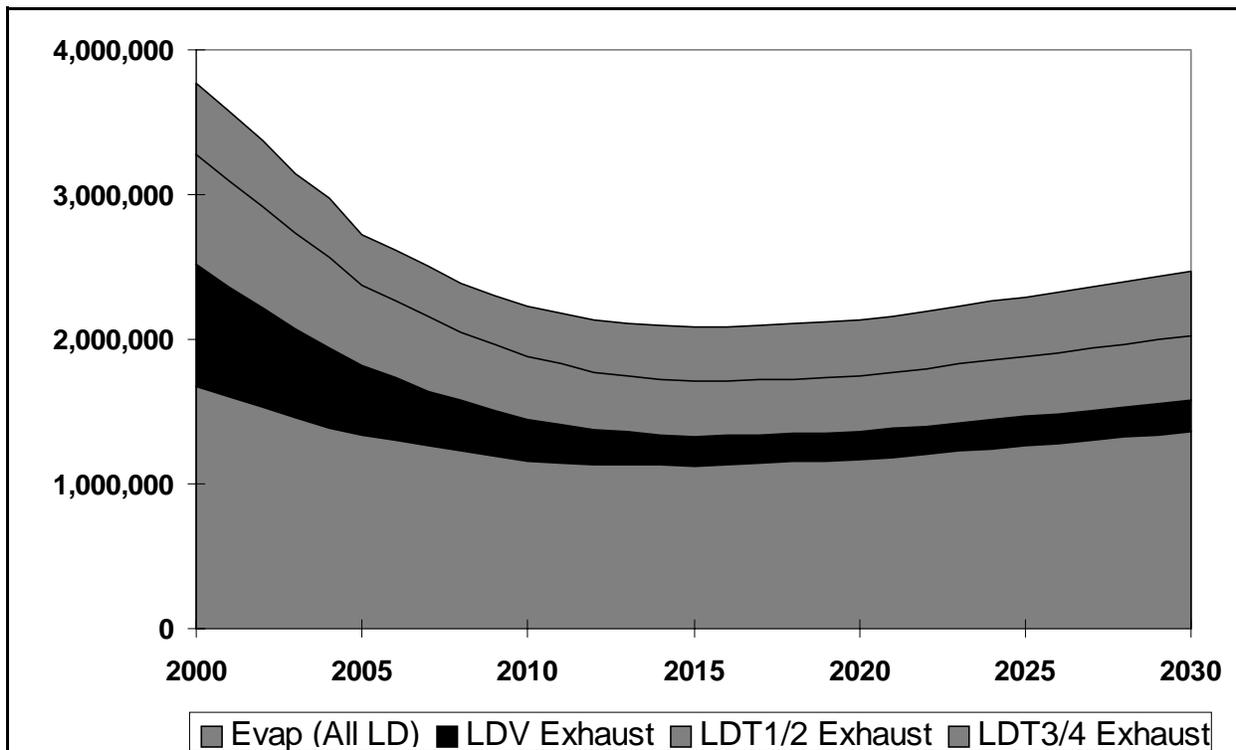
**a. Light-Duty VOC Trends Without Tier 2/Sulfur**

Total VOC emissions produced nationwide by cars and trucks without Tier 2/Sulfur control are shown in Table III-10 and Figure III-7, broken down by relative contribution of evaporative emissions (across all cars and trucks), and exhaust emissions for LDVs, LDT1/2s and LDT3/4s. We project VOC emissions from light-duty vehicles will decline from approximately 3.8 million tons to 2.0 million tons between 2000 and 2015 as the fleet becomes increasingly dominated by cars and trucks complying with NLEV, Enhanced Evaporative control and SFTP requirements. Beginning in 2016, however, light-duty VOC emissions are projected to begin an upward trend due to VMT and vehicle fleet growth, increasing to 2.1 million tons by 2020 and 2.5 million tons by 2030.

**Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999**

**Table III-10. 47-State Light-Duty VOC Emissions Without Tier 2/Sulfur (Annual Tons)**

Year	Light-Duty Emissions	Contribution by Emission Source / Vehicle Class			
		Evaporative (All LDV/LDT)	Exhaust		
			LDV	LDT1/2	LDT3/4
2000	3,771,569	44.3%	22.6%	20.1%	13.0%
2004	2,969,912	46.8%	18.3%	21.1%	13.7%
2007	2,503,855	50.4%	15.3%	20.4%	14.0%
2010	2,227,593	52.1%	12.6%	19.8%	15.5%
2015	2,080,284	54.1%	9.9%	18.2%	17.8%
2020	2,132,070	54.7%	9.1%	18.0%	18.2%
2030	2,475,783	54.8%	8.8%	18.1%	18.2%



**Figure III-7. 47-State Light-Duty VOC Emissions Without Tier 2/Sulfur (Annual Tons)**

Although evaporative emissions are projected to grow to over half of the light-duty inventory, exhaust emissions from trucks play an increasingly significant role in shaping the overall VOC trend. In 2000, we project that trucks will produce approximately 60 percent of exhaust VOC emissions; by 2020, trucks account for 80 percent of these emissions, while overall emissions produced by trucks increase steadily. The benefits from Tier 1, NLEV and SFTP are not as pronounced for trucks relative to cars, and are offset almost immediately by growth in truck VMT. As a result, exhaust VOC emissions from trucks see only modest initial reductions due to these programs before being offset by VMT growth.

The emission trends for the four urban areas we analyzed show similar behavior to the national trends. As shown in Table III-11, light-duty emissions decrease steadily in each city through 2010. In all cities, the decline in emissions due to existing vehicle standards essentially ends by 2020, after which VOC emissions are projected to increase if today’s proposal is not adopted.

**Table III-11. Four-City Light-Duty VOC Emissions Without Tier 2/Sulfur (Summer Tons)**

<i>Year</i>	<i>New York</i>	<i>Chicago</i>	<i>Atlanta</i>	<i>Charlotte</i>
2000	56,782	27,145	28,791	4,080
2004	40,063	19,768	22,166	3,245
2007	30,609	15,404	18,139	2,710
2010	25,732	13,151	15,869	2,412
2015	22,062	11,386	14,239	2,217
2020	21,124	11,061	14,195	2,254
2030	22,744	12,264	16,149	2,642

Figures III-8 and III-9 show our projections of the contribution of light-duty vehicles and trucks to the total anthropogenic (i.e., human-caused) 2020 VOC inventory in the 47 states and in Atlanta. Table III-12 shows this same contribution across the 47 states and all four cities from 2007 through 2030. Nationally, cars and trucks produce nearly one-fifth of total VOC emissions in 2007; this percentage declines subsequent years before stabilizing at 14 percent by 2015 and increasing after 2020. The light-duty contribution in New York, Chicago, and Charlotte are slightly lower than the national average, but significantly higher in Atlanta, where we project that one-fourth of all VOC emissions will be produced by cars and trucks in 2020.

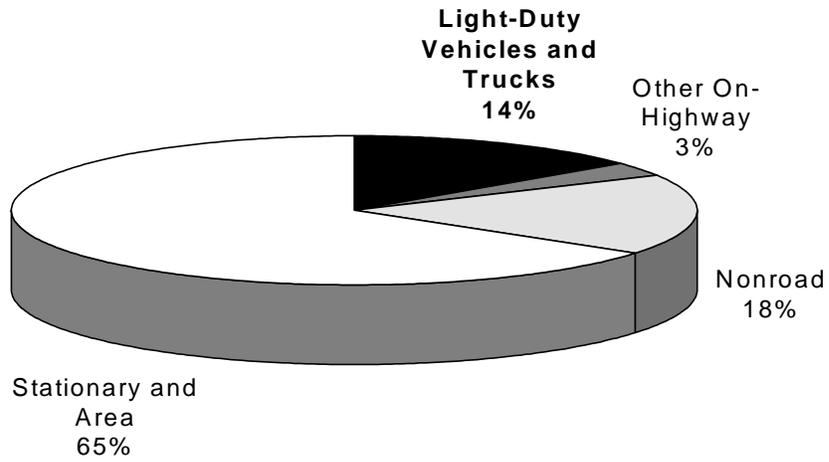


Figure III-8. Breakdown of Total 2020 47-State VOC Inventory Without Tier 2/Sulfur

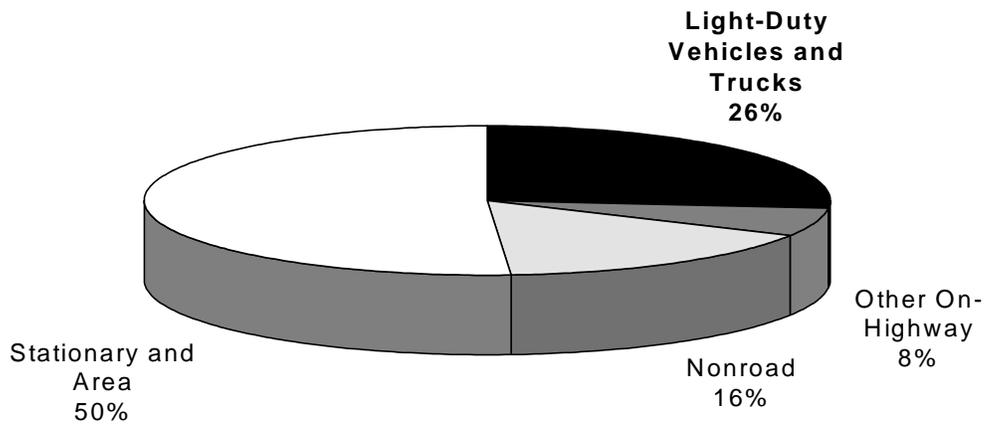


Figure III-9. Breakdown of Total 2020 Atlanta VOC Inventory Without Tier 2/Sulfur

**Table III-12. Light-Duty Contribution to Total VOC Inventory Without Tier 2/Sulfur**

<i>Year</i>	<i>47 State</i>	<i>New York</i>	<i>Chicago</i>	<i>Atlanta</i>	<i>Charlotte</i>
2007	18%	15%	12%	33%	15%
2010	16%	13%	11%	31%	14%
2015	14%	11%	10%	28%	12%
2020	14%	10%	9%	26%	12%
2030	15%	10%	9%	26%	12%

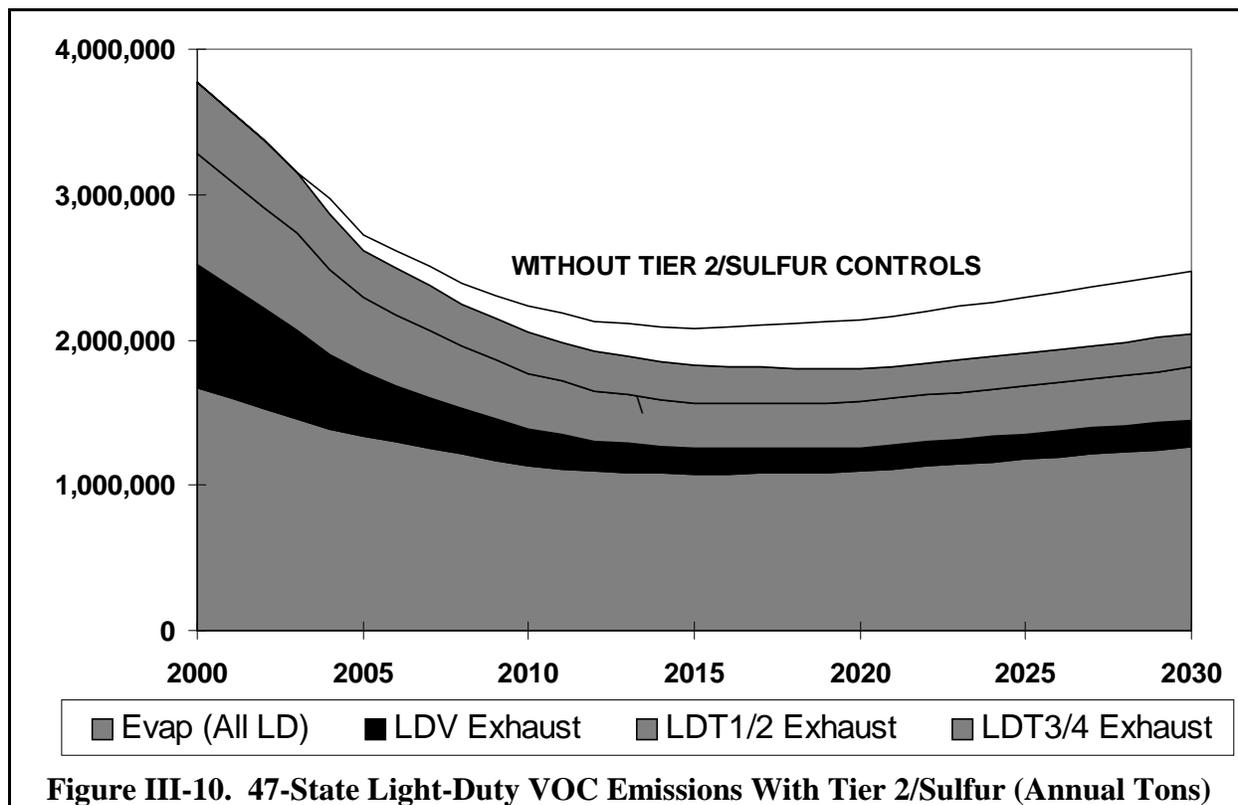
**b. VOC Reductions Due To Tier 2/Sulfur**

Table III-13 contains annual nationwide tons of VOC we project would be reduced due to today’s proposal, encompassing the effects of low sulfur fuel and the introduction of Tier 2 light-duty vehicle and light-duty truck standards for both exhaust and evaporative emissions. Figure III-10 shows projected 47-state emissions with Tier 2/Sulfur control, broken down by light-duty evaporative emissions and exhaust emissions from LDVs, LDT1/2s and LDT3/4s.

**Table III-13. 47-State Light-Duty VOC Reductions Due to Tier 2/Sulfur (Annual Tons)**

<i>Year</i>	<i>Light-Duty Emissions Without Tier 2/Sulfur</i>	<i>Light-Duty Emissions With Tier 2/Sulfur</i>	<i>Emissions Reduced</i>	<i>Percent Reduction in Baseline Inventory</i>	
				<i>Light-Duty</i>	<i>All Sources*</i>
2004	2,969,912	2,865,843	104,069	4%	-
2007	2,503,855	2,372,427	131,428	5%	1.0%
2010	2,227,593	2,050,465	177,128	8%	1.3%
2015	2,080,284	1,821,904	258,380	12%	1.8%
2020	2,132,070	1,800,394	331,676	16%	2.3%
2030	2,475,783	2,039,802	435,981	18%	2.7%

\* Includes emission reductions from Heavy-Duty Gasoline Vehicles due to sulfur control



We project that lower sulfur levels in 2004 would reduce light-duty emissions four percent, due almost entirely to reduced emission from Tier 0, Tier 1 and NLEV vehicles; this is the equivalent of emissions produced by 6.5 million pre-Tier 2 cars and trucks. After 2004, the introduction of LDT2s, LDT3s, and LDT4s complying with the Tier 2 NMOG standard and operating on low sulfur fuel reduce emission further. By 2020, baseline light-duty VOC emissions are reduced 16 percent due to Tier 2/Sulfur control, the equivalent of emissions from 38 million pre-Tier 2 cars and trucks. This represents a 2.3 percent reduction of the total anthropogenic VOC inventory. With Tier 2/Sulfur, we project that the upturn in light-duty VOC emissions will begin in 2021, five years later than the baseline case.

In addition to emission benefits on light-duty vehicles and trucks, we project that heavy-duty gasoline vehicles would decrease emissions by approximately 7,000 tons per year beginning in 2004, growing to 12,000 tons in 2030. These reductions are shown in Appendix A, and are included in the estimates of mobile source and all source percent reduction contained in Table III-13.

Tables III-14 and III-15 show VOC reductions in the four cities in both tonnage and percentage terms; the percentage reductions are expressed relative to light-duty emissions and total anthropogenic emissions if today's proposal were not adopted. VOC reductions would be larger in these areas in percentage terms than is the average throughout the 47 states. In 2020, we

project that 23 percent of light-duty VOC emissions would be reduced in these cities, versus 16 percent for the 47-state region. This difference is driven by the presence of I/M in each area and RFG in some of these areas. As modeled, vehicles with malfunctioning emission control systems do not realize the full benefit of the proposed Tier 2 vehicle standards. With I/M, it is assumed that a good portion of these vehicles are identified and repaired, thus increasing the relative benefit of the Tier 2/Sulfur program.<sup>h</sup> The reductions in total anthropogenic VOC inventory are generally consistent with the 47-state results, although in Atlanta the reductions are larger; by 2020, we project that 6.1 percent of Atlanta’s total VOC emission would be reduced by today’s action, versus 2.3 percent nationally.

**Table III-14. Four-City Light-Duty VOC Reductions Due To Tier 2/Sulfur (Summer Tons)**

<i>Year</i>	<i>New York</i>		<i>Chicago</i>		<i>Atlanta</i>		<i>Charlotte</i>	
	<i>Reduced</i>	<i>Remain</i>	<i>Reduced</i>	<i>Remain</i>	<i>Reduced</i>	<i>Remain</i>	<i>Reduced</i>	<i>Remain</i>
2004	1,282	38,781	611	19,157	1,110	21,056	163	3,083
2007	1,870	28,739	913	14,490	1,462	16,677	218	2,492
2010	2,675	23,057	1,333	11,818	1,860	14,009	283	2,130
2015	3,919	18,143	2,007	9,380	2,592	11,648	404	1,814
2020	4,919	16,205	2,568	8,493	3,240	10,955	515	1,740
2030	6,031	16,713	3,247	9,017	4,184	11,965	685	1,958

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<sup>h</sup>The approach used for developing I/M benefits for Tier 2 vehicles is discussed in detail in the technical report “Development of Light-Duty Emission Inventory Estimates in the Notice of Proposed Rulemaking for Tier 2 and Sulfur Standards”

**Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999**

**Table III-15. Percent Reduction From Light-Duty and Total Baseline VOC Emissions in Four Cities**

<i>Year</i>	<i>New York</i>		<i>Chicago</i>		<i>Atlanta</i>		<i>Charlotte</i>	
	<i>Light-Duty</i>	<i>All Sources*</i>						
2004	3%	-	3%	-	5%	-	5%	-
2007	6%	0.9 %	6%	0.8%	8%	2.8%	8%	1.3%
2010	10%	1.4%	10%	1.1%	12%	3.7%	12%	1.7%
2015	18%	2.0%	18%	1.7%	18%	5.1%	18%	2.3%
2020	23%	2.4%	23%	2.1%	23%	6.1%	23%	2.8%
2030	27%	2.8%	26%	2.4%	26%	6.9%	26%	3.3%

\* Includes emission reductions from Heavy-Duty Gasoline Vehicles due to sulfur control

Figures III-11 and III-12 show the contribution of light-duty cars and trucks to total 2020 VOC inventory in the 47 states and in Atlanta with Tier 2/Sulfur control. Table III-16 shows this same contribution across the 47 states and all four cities from 2007 through 2030. In 2020, the light-duty contribution would drop to 12 percent nationally, from 14 percent without Tier 2/Sulfur control. This trend would be similar across the four cities, depending on the level of light-duty contribution without Tier 2/Sulfur control. We project that with Tier 2/Sulfur control, car and truck emissions would contribute eight percent of total emissions in New York (down from 10 percent), seven percent in Chicago (down from nine percent), ten percent in Charlotte (down from 12 percent), and 22 percent in Atlanta (down from 26 percent).

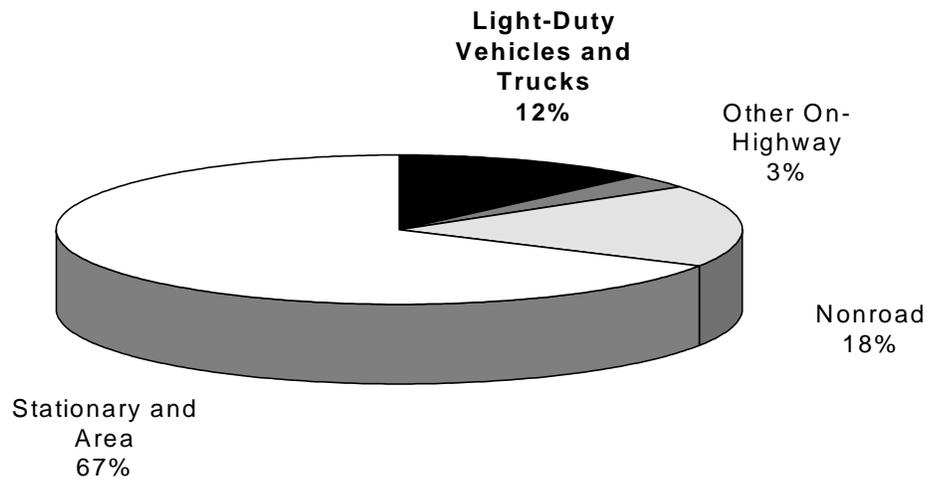


Figure III-11. Breakdown of Total 2020 47 State VOC Inventory With Tier 2/Sulfur

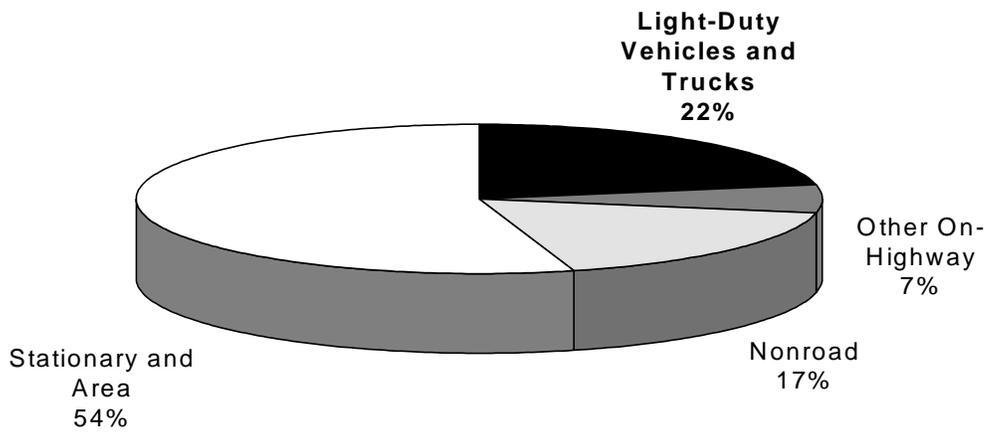


Figure III-12. Breakdown of Total 2020 Atlanta VOC Inventory With Tier 2/Sulfur

**Table III-16. Light-Duty Contribution to Total VOC Inventory With Tier 2/Sulfur**

<i>Year</i>	<i>47 State</i>	<i>New York</i>	<i>Chicago</i>	<i>Atlanta</i>	<i>Charlotte</i>
2007	17%	14%	12%	32%	14%
2010	15%	12%	10%	28%	12%
2015	13%	9%	8%	24%	10%
2020	12%	8%	7%	22%	10%
2030	13%	8%	7%	21%	10%

**c. VOC Emission Reductions From Other Options**

VOC reductions for the three alternative options discussed under Section III.A.1.c above are shown in Tables III-17 and III-18, in comparison to reductions projected from today's proposal. It is assumed for this analysis that the evaporative controls contained in today's action would be included in each option.

**Table III-17. 47-State Light-Duty VOC Reductions From Alternative Control Options (Annual Tons)**

<i>Option:</i>	<i>Today's Proposal</i>	<i>1</i>	<i>2</i>	<i>3</i>
<i>Vehicle Program:</i>	<i>Proposed Tier 2</i>	<i>Proposed Tier 2</i>	<i>Proposed Tier 2</i>	<i>Clean Air Act Default</i>
<i>Fuel Program:</i>	<i>Proposed Tier 2</i>	<i>API No 30 ppm</i>	<i>API 30 ppm API Region 2010</i>	<i>Proposed Tier 2</i>
2007	131,428	74,331	74,331	101,706
2010	177,128	118,809	155,750	107,955
2020	331,676	264,220	305,361	131,552

**Table III-18. VOC Reduction Shortfall From Alternative Control Options Relative to Today’s Action**

<i>Scenario:</i>	<i>1</i>		<i>2</i>		<i>3</i>	
<i>Vehicle Program:</i>	<i>Proposed Tier 2</i>		<i>Proposed Tier 2</i>		<i>Clean Air Act Default</i>	
<i>Fuel Program:</i>	<i>API No 30 ppm</i>		<i>API 30 ppm API Region 2010</i>		<i>Proposed Tier 2</i>	
<i>Year</i>	<i>Shortfall (Annual Tons)</i>	<i>Benefit Relative to Tier 2/Sulfur Proposal</i>	<i>Shortfall (Annual Tons)</i>	<i>Benefit Relative to Tier 2/Sulfur Proposal</i>	<i>Shortfall (Annual Tons)</i>	<i>Benefit Relative to Tier 2/Sulfur Proposal</i>
2007	57,097	57%	57,097	57%	29,722	77%
2010	58,319	67%	21,378	88%	69,173	61%
2020	67,456	80%	26,315	92%	200,123	40%

**3. SOx**

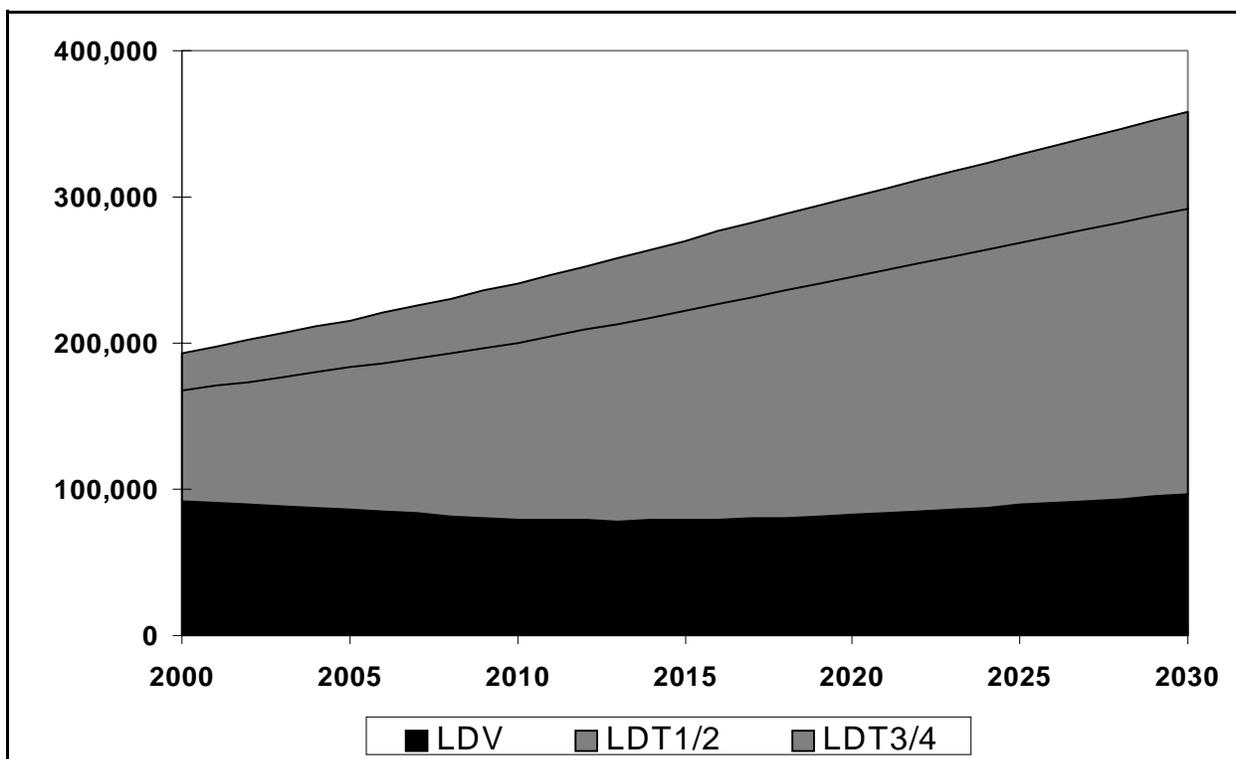
**a. Light-Duty SOx Trends Without Sulfur Control**

Gaseous SOx emissions are formed by the combustion of fuel sulfur, and hence depend entirely on the level of sulfur in the fuel. SOx emissions without sulfur control are shown in Table III-19 and Figure III-13, broken down by LDV, LDT1/2 and LDT3/4. As shown, we project that SOx emission levels will increase unabated through 2030 in conjunction with VMT growth in the absence of any action to reduce fuel sulfur levels. In 2000, we project light-duty vehicles and trucks will emit 194,000 tons of SOx; by 2020, this level is projected to be nearly 300,000 tons, an increase of 55 percent.

**Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999**

**Table III-19. 47-State SOx Emissions Without Sulfur Control (Annual Tons)**

Year	Emissions From All Sources	Light-Duty Emissions	Light-Duty Contribution to All Sources	Contribution by Vehicle Class		
				LDV	LDT1/2	LDT3/4
2000	-	193,467	-	48%	39%	13%
2004	-	211,072	-	41%	44%	15%
2005	18,045,277	215,659	1.2%	40%	45%	15%
2010	18,350,974	240,694	1.3%	33%	50%	17%
2015	18,773,428	270,174	1.4%	29%	53%	18%
2020	19,161,564	299,959	1.6%	28%	54%	18%
2030	20,099,769	357,611	1.8%	27%	55%	18%



**Figure III-13. 47-State Light-Duty SOx Emissions Without Sulfur Control (Annual Tons)**

Trucks, primarily LDT1s and LDT2s, are responsible for the steady increase in light-duty SOx emissions. While LDV SOx emissions are relatively stable, SOx emissions from trucks (and hence the contribution to light-duty inventory produced by trucks) are projected to increase

steadily. In 2000, trucks account for roughly half of light-duty SOx emissions, growing to over 70 percent by 2020.

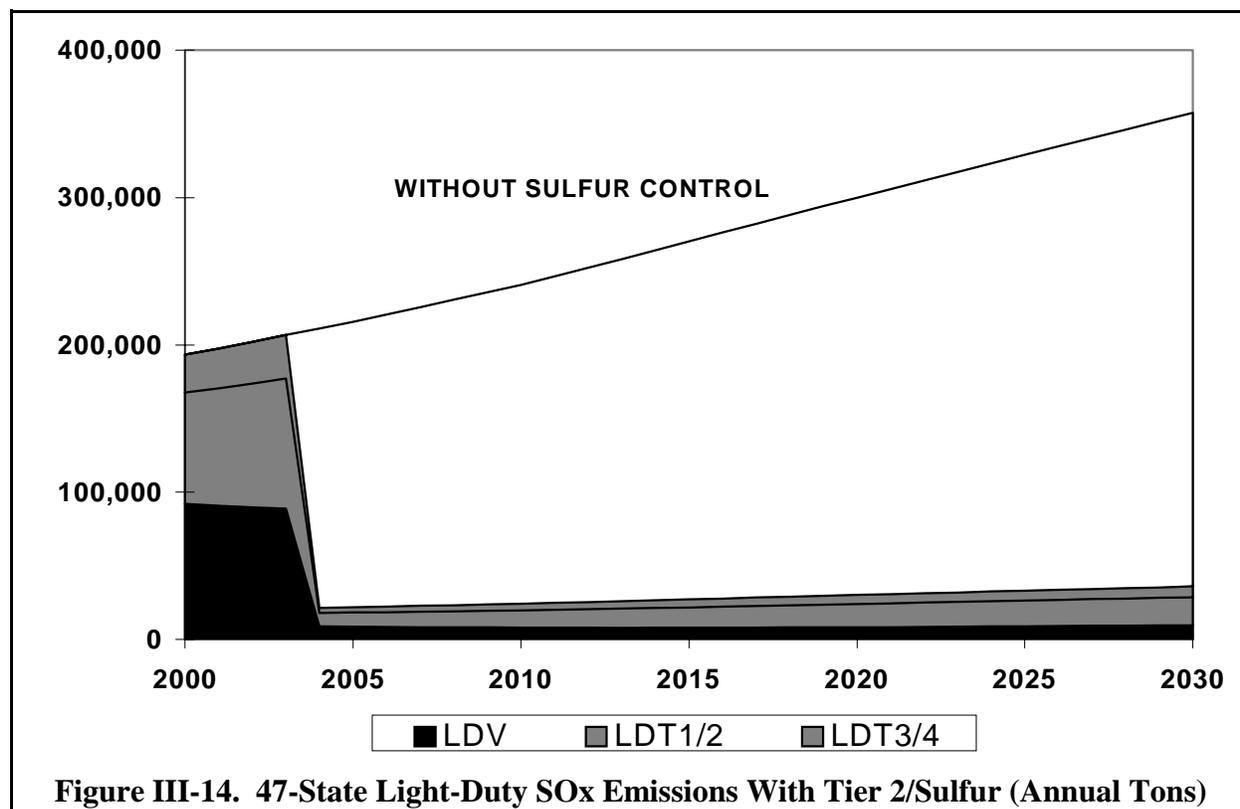
**b. SOx Reductions Due To Sulfur Control**

We project that today's proposal would immediately and substantially reduce SOx emissions from cars and trucks once its fuel sulfur provisions take effect. Table III-20 contains annual nationwide tons of gaseous SOx we project will be reduced from light-duty vehicles and trucks due to sulfur control. Figure III-14 shows SOx emissions after sulfur control, broken down by LDV, LDT1/2 and LDT3/4.

**Table III-20. 47-State Light-Duty SOx Reductions Due To Sulfur Control (Annual Tons)**

<i>Year</i>	<i>Light-Duty Emissions Without Sulfur Control</i>	<i>Light-Duty Emissions With Sulfur Control</i>	<i>Emissions Reduced</i>	<i>Percent Reduction in Baseline Inventory</i>	
				<i>Light-Duty</i>	<i>All Sources*</i>
2004	211,072	21,426	189,646	90%	-
2005	215,659	21,899	193,760	90%	1.3%
2010	240,694	24,257	216,437	90%	1.4%
2015	270,174	27,210	242,964	90%	1.5%
2020	299,959	30,203	269,756	90%	1.6%
2030	357,611	36,002	321,609	90%	1.8%

\* Includes reductions from Heavy-Duty Gasoline Vehicles, Motorcycles and Nonroad Sources



As shown, a 90 percent reduction in light-duty SOx emissions would be realized beginning in 2004. This relative reduction remains constant beyond 2004, since SOx emissions are not reduced further as new Tier 2 VOC, NOx, and PM standards are phased in. The absolute level of emission reductions would become larger with time, however, due to VMT growth.

SOx emission reductions will also occur from heavy-duty gasoline vehicles and motorcycles due to sulfur control; we estimate this reduction to be approximately 10,000 tons in 2005, growing to 14,000 tons by 2020. In addition, emissions from all gasoline-powered nonroad equipment would be reduced due to sulfur control. Based on our NONROAD model, we estimate this benefit would be approximately 25,000 tons per year on average between 2005 and 2020. These reductions, shown in Appendix A, are included in the percent reductions from all sources in Table III-20.

#### 4. Particulate Matter

Trends in particulate matter emissions will depend very strongly on the prevalence of diesel vehicles in the light-duty fleet. Currently, diesels make up a very small portion (less than one percent) of overall car and truck sales. However, sharp increases in diesel sales are a

reasonable possibility given the focus on diesel technology for improving fuel economy under the Partnership for a New Generation of Vehicles (PNGV). Thus, we assessed PM emissions under two sales scenarios: a “no growth” scenario, for which current diesel sales trends were assumed to continue, and an “increased growth” scenario, for which diesels grow to 50 percent of light-duty truck sales by 2010. The effects of Tier 2/Sulfur control were assessed for both scenarios. The results presented here are for direct exhaust PM, comprising carbonaceous PM and sulfate emitted directly from the tailpipe and a subset of Total PM (which also includes direct non-exhaust PM from tire and brake wear, and indirect PM caused by secondary reactions to emitted NO<sub>x</sub> and SO<sub>x</sub> in the atmosphere). Direct PM<sub>2.5</sub> and PM<sub>10</sub> emissions are presented separately for the “no growth” scenario.

**a. “No Growth” Diesel Sales Scenario**

*i. Light-Duty Direct Exhaust PM<sub>2.5</sub> Trends Without Tier 2/Sulfur*

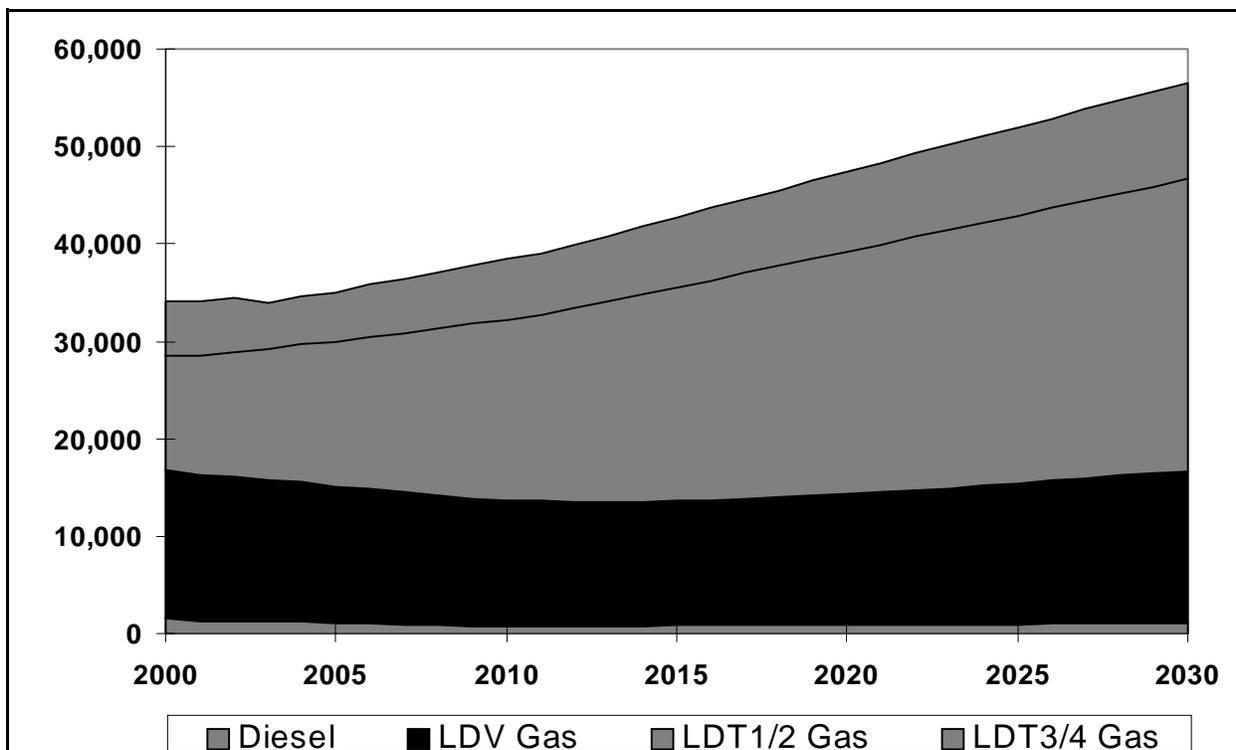
In general, gasoline vehicles emit PM at rates much lower than their diesel counterparts. Under the no growth scenario, direct PM emissions are driven largely by sulfate emissions from gasoline vehicles, which depend primarily on gasoline fuel sulfur level. Without Tier 2/Sulfur control, these emissions increase at a steady rate in conjunction with VMT, as shown in Tables III-21 and Figure III-15. In 2005, we project that approximately 35,000 tons will be emitted annually by light-duty cars and trucks. This level is projected to exceed 47,000 tons in 2020 and 56,000 tons in 2030.

**Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999**

**Table III-21. 47 State Light-Duty Direct Exhaust PM<sub>2.5</sub> Emissions Without Tier 2/Sulfur  
No Growth in Diesel Sales  
(Annual Tons)**

Year	Emissions From All Sources*	Light-Duty Exhaust Emissions	Light-Duty Contribution to All Sources	Contribution by Fuel Type / Vehicle Class			
				Diesel LDV/LDT	Gas LDV	Gas LDT1/2	Gas LDT3/4
2000	-	34,072	-	5%	45%	34%	17%
2004	-	34,612	-	3%	42%	41%	14%
2005	2,071,897	35,051	1.7%	3%	40%	42%	15%
2010	2,108,058	38,409	1.8%	2%	34%	48%	16%
2015	2,217,074	42,724	1.9%	2%	30%	51%	17%
2020	2,318,805	47,397	2.0%	2%	28%	53%	17%
2030	2,544,434	56,505	2.2%	2%	28%	53%	17%

\* Excludes natural and miscellaneous sources (e.g., fugitive dust), but includes indirect sources such as tire and brake wear.



**Figure III-15. 47-State Light-Duty Direct Exhaust PM<sub>2.5</sub> Emissions Without Tier 2/Sulfur - No Diesel Growth (Annual Tons)**

As expected, the diesel contribution to overall emissions in the no growth scenario is relatively small. Rather, gasoline trucks (primarily LDT1s and LDT2s) are responsible for the steady increase in PM emissions. Under this scenario, we project the contribution of gasoline trucks to light-duty PM<sub>2.5</sub> inventory to grow to 70 percent by 2020.

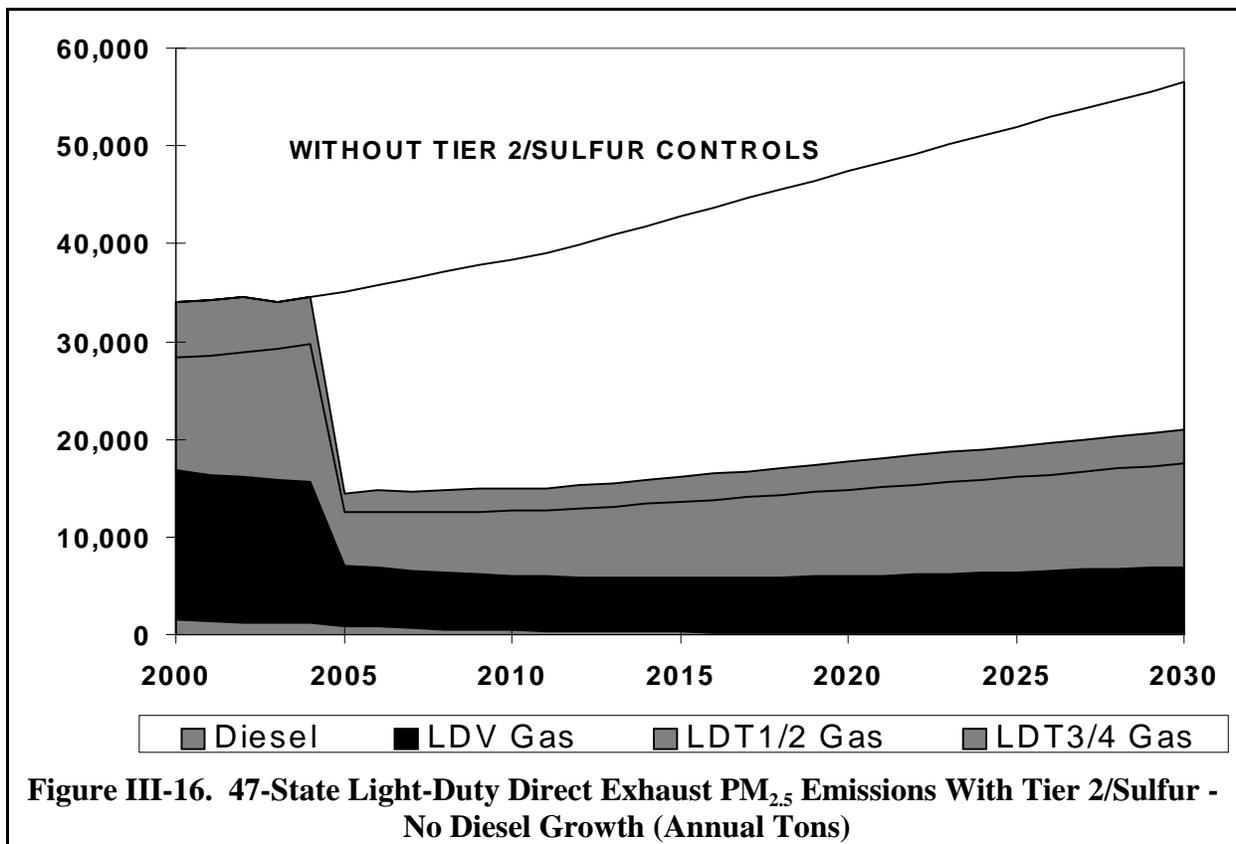
ii. *Direct Exhaust PM<sub>2.5</sub> Reductions Due To Tier 2/Sulfur Control*

Under the no growth scenario, today's proposal would provide an immediate and substantive reduction in direct PM emissions from cars and trucks, due primary to sulfur control. Table III-22 contains annual nationwide tons of direct exhaust PM<sub>2.5</sub> we project would be reduced from light-duty vehicles and trucks due to Tier 2/Sulfur control. Figure III-16 shows PM<sub>2.5</sub> emissions after Tier 2/Sulfur control broken down by diesel (all light-duty cars and trucks) and gasoline LDV, LDT1/2 and LDT3/4.

**Table III-22. 47-State Light-Duty Direct Exhaust PM<sub>2.5</sub> Reductions Due To Tier 2/Sulfur No Growth in Diesel Sales (Annual Tons)**

Year	Light-Duty Emissions Without Tier 2/Sulfur	Light-Duty Emissions With Tier 2/Sulfur	Emissions Reduced	Percent Reduction in Baseline Inventory	
				Light-Duty	All Sources*
2004	34,612	14,703	19,909	58%	-
2005	35,051	14,509	20,542	59%	1.0%
2010	38,409	14,999	23,410	61%	1.1%
2015	42,724	16,129	26,595	62%	1.2%
2020	47,397	17,690	29,707	63%	1.3%
2030	56,505	20,956	35,549	63%	1.4%

\* Includes emission reductions from Heavy-Duty Gasoline Vehicles due to sulfur control



Reductions from gasoline vehicles would result almost entirely from sulfur control, rather than the proposed PM<sub>2.5</sub> exhaust standards. PM<sub>2.5</sub> emissions on current technology gasoline vehicles are much lower than diesel vehicles, and gasoline vehicle emissions are not expected to be reduced in response to the PM<sub>2.5</sub> standards contained in today's proposal. As such, we project that an immediate emission reduction of 58 percent from baseline levels would be realized due to sulfur control, increasing to 63 percent by 2020.

In addition to light-duty PM benefits, sulfur control would reduce PM<sub>2.5</sub> emissions from heavy-duty gasoline vehicles. We estimate these benefits would be approximately 700 tons per year beginning in 2004, increasing to 1,000 tons by 2020. Across all sources, we project Tier 2/Sulfur control would reduce direct PM<sub>2.5</sub> from all non-natural sources by about one percent.

*iii. Direct Exhaust PM<sub>10</sub> Reductions Due To Tier 2/Sulfur Control*

Direct exhaust PM<sub>10</sub> emissions exhibit similar trends to PM<sub>2.5</sub>, and are thus shown here only for the no growth diesel case; PM<sub>10</sub> emissions with and without Tier 2/Sulfur control are shown in Table III-23.

**Table III-23. 47-State Light-Duty PM<sub>10</sub> Emissions With and Without Tier 2/Sulfur Control  
No Growth in Diesel Sales  
(Annual Tons)**

Year	Emissions From All Sources Without Tier 2/Sulfur**	Light-Duty Exhaust Emissions Without Tier 2/Sulfur	Light-Duty Contribution to All Sources	Light-Duty Exhaust Emissions With Tier 2/Sulfur	Emissions Reduced	Percent Reduction in Baseline Inventory	
						Light-Duty	All Sources*
2004	-	37,323	-	15,861	21,462	58%	-
2005	2,985,623	37,794	1.3%	15,649	22,145	59%	0.8%
2010	3,060,154	41,412	1.4%	16,173	25,239	61%	0.9%
2015	3,207,687	46,064	1.4%	17,390	28,674	62%	0.9%
2020	3,345,810	51,102	1.5%	19,071	32,031	63%	1.0%
2030	3,659,928	60,922	1.7%	22,591	38,331	63%	1.1%

\* Includes emission reductions from Heavy-Duty Gasoline Vehicles due to sulfur control

\*\* Excludes natural and miscellaneous sources (e.g., fugitive dust), but includes indirect sources such as tire and brake wear.

#### **b. “Increased Growth” Sales Scenario**

The “increased growth” scenario was developed with the intent of analyzing an upper bound for diesel growth. We developed this scenario by assuming that the percent of diesels making up total light-duty truck sales increase to five percent in 2001, adding five percentage points per subsequent year until diesels represent 50 percent of light-duty truck sales in 2010; beyond 2010, the diesel engine share of the light truck market was assumed to stay at 50 percent. Within the period of diesel sales growth, we assumed that light duty truck classes were “converted” to diesels in a sequential manner starting with the heaviest trucks; i.e., LDT4s became diesels first, then LDT3s, etc. This methodology resulted in the diesel sales penetrations shown in Table III-24.

**Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999**

**Table III-24. Diesel LDT Sales Penetration Under Increased Growth Scenario**

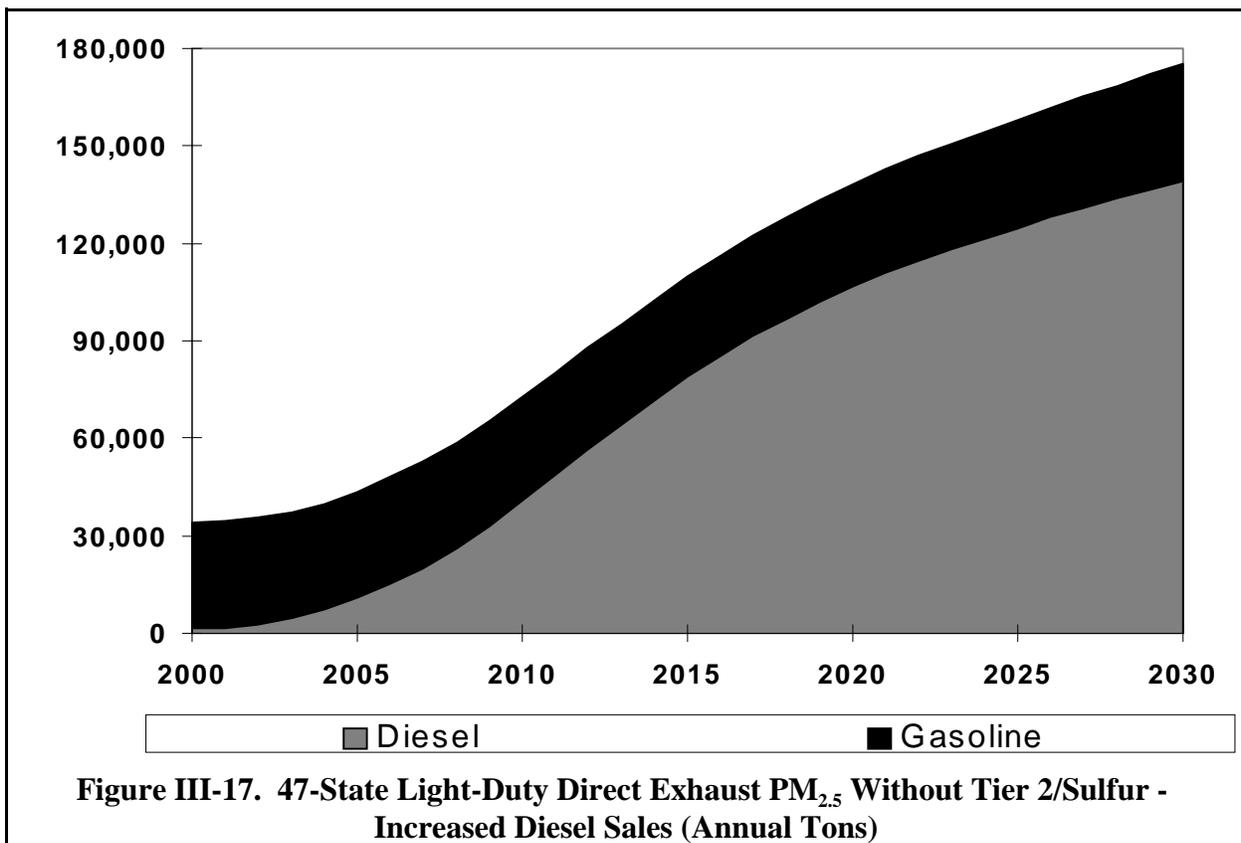
<i>Model Year</i>	<i>Diesel Sales Penetration</i>			
	<i>All LDT</i>	<i>LDT2</i>	<i>LDT3</i>	<i>LDT4</i>
2001	5%	0%	0%	63%
2002	10%	0%	12%	100%
2003	15%	0%	41%	100%
2004	20%	0%	71%	100%
2005	25%	0%	100%	100%
2006	30%	9%	100%	100%
2007	35%	18%	100%	100%
2008	40%	26%	100%	100%
2009	45%	35%	100%	100%
2010 and later	50%	44%	100%	100%

*i. Light-Duty Direct Exhaust PM<sub>2.5</sub> Trends Without Tier 2/Sulfur*

Our projections for light-duty direct exhaust PM<sub>2.5</sub> under the increased diesel sales scenario are down in Table III-25 and Figure III-17. As expected, this scenario is projected to result in dramatic increases in light-duty PM<sub>2.5</sub> emissions. 2005 baseline emissions are approximately 43,000 tons, 23 percent higher than the 35,000 tons projected in the no growth diesel case from Table III-21. However, by 2020, we project this scenario would result in direct PM emissions of 138,000 tons, nearly three times the emissions projected for the no growth scenario in the same year.

**Table III-25. 47 State Light-Duty Direct Exhaust PM<sub>2.5</sub> Emissions Without Tier 2/Sulfur Increased Diesel Growth Scenario (Annual Tons)**

Year	Light-Duty Emissions Without Tier 2/Sulfur	Contribution by Fuel Type	
		Diesel LDV/LDT	Gasoline LDV/LDT
2000	34,072	5%	95%
2004	39,932	19%	81%
2005	43,439	25%	75%
2010	72,626	56%	44%
2015	109,622	72%	28%
2020	138,177	77%	23%
2030	175,068	80%	20%



## Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999

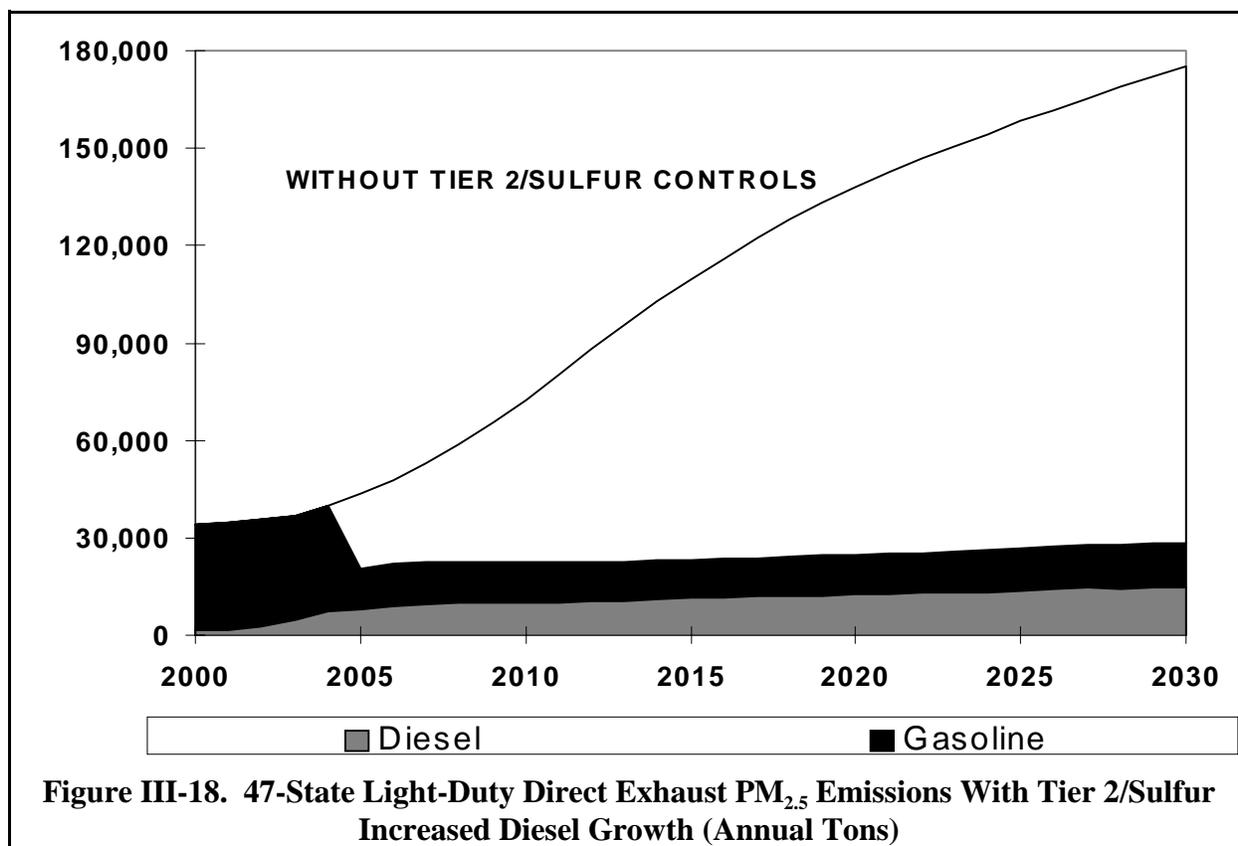
As shown, the rapid growth of diesels in conjunction with high per-vehicle PM emissions from diesels drive overall direct PM emissions under this scenario. In 2005, we project diesels would already account for 25 percent of all light-duty emissions. Diesel contribution grows to over 50 percent by 2010 and over 75 percent by 2020.

### ii. Direct Exhaust PM<sub>2.5</sub> Reductions Due To Tier 2/Sulfur

Tier 2/Sulfur control would effectively neutralize excess PM emissions generated under our increased diesel penetration scenario. Table III-26 contains reductions in direct exhaust PM<sub>2.5</sub> emissions due to Tier 2/Sulfur standards for the increased diesel sales penetration case. Figure III-18 shows these emissions with Tier 2/Sulfur control, broken down by diesel and gasoline.

**Table III-26. 47-State Light-Duty Direct Exhaust PM<sub>2.5</sub> Reductions Due To Tier 2/Sulfur Increased Diesel Growth Scenario (Annual Tons)**

Year	Light-Duty Emissions Without Tier 2/Sulfur	Light-Duty Emissions With Tier 2/Sulfur	Emissions Reduced	Percent Reduction in Baseline Inventory	
				Light-Duty	All Sources*
2004	39,932	19,700	20,232	51%	-
2005	43,439	20,696	22,743	52%	1.1%
2010	72,626	22,542	50,084	69%	2.4%
2015	109,622	23,275	86,347	79%	3.8%
2020	138,177	24,754	113,423	82%	4.7%
2030	175,068	28,393	146,675	84%	5.6%



In 2005, the fleet would still be comprised primarily of gasoline vehicles under this scenario; thus, significant benefits from gasoline sulfur control would be realized immediately, as with the no growth case. The rapid growth of diesel market penetration in conjunction with implementation of the proposed Tier 2 PM standards would result in a diesel fleet comprised almost exclusively of vehicles compliant with Tier 2. Thus, a large share of the baseline inventory would be reduced very soon after implementation of the Tier 2/Sulfur standards. In 2010, nearly 70 percent of baseline light-duty exhaust PM<sub>2.5</sub> inventory is reduced; by 2020, we project 82 percent of baseline emissions would be reduced. Today's proposal would serve to mitigate the large increases in direct PM emissions which would occur without control due to increased growth in diesel penetration, effectively stabilizing these emissions through at least 2030.

## B. Air Quality Measures

This section describes the analyses performed to evaluate the impact of the Tier 2/Sulfur proposal on ozone and visibility levels, as discussed in Section III of the preamble. These analyses were performed using different emission inventories, control assumptions, ozone and visibility models, and analysis years than the air quality modeling we conducted for the benefit/cost analysis described in Chapter VII. As a result, the ozone and visibility modeling results presented in Section III of the preamble and described more fully in this section are not

## **Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999**

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directly comparable to the ozone and visibility modeling results used for the benefit/cost analysis.

This section does not discuss the impact of the Tier 2/Sulfur proposal on PM levels, since the PM air quality modeling we performed was conducted for the benefit/cost analysis described in Chapter VII. This section also does not discuss the impact of the Tier 2/Sulfur proposal on CO levels since we have not quantified the proposal's impact on CO emission levels at this time.

### **1. Ozone**

Current air quality with respect to ozone can be expressed in terms of formal designation of attainment or nonattainment of the 1-hour standard (there is as yet no such designation for the 8-hour standard) and in terms of measured ambient design values (defined below) for both the 1-hour and 8-hour standards. Estimates of the ozone impact of today's proposal and the expected future ozone concentrations after its implementation can be obtained by modeling a base case (before control measures) and a control case (after control measures). The outputs of these and other model runs are used in combination with measured design values to project future design values. Other metrics described in this section are also used to compare one model run to another. The structure of this section is as follows:

- Subsection B.1.a. presents the data supporting the discussion in the preamble of current nonattainment status, including an explanation of design values.
- Subsection B.1.b. describes in general terms the ozone modeling that we used to estimate the effects of Tier 2/Sulfur controls and the way we used that modeling to estimate the resulting design values.
- Subsection B.1.c. explains the "rollback method," used to estimate future design values based on measured historical ozone levels and ozone modeling results.
- Subsection B.1.d. describes the ozone modeling simulations used to evaluate the impact of Tier 2/Sulfur controls on future ozone levels.
- Subsection B.1.e. presents the results of two ozone simulations that were used to explore the relative effects of NO<sub>x</sub> and VOC controls on ozone levels.
- Subsection B.1.f. describes two ozone simulations that were used to estimate the effects of today's proposal on ozone levels.

#### **a. Measures of Current Attainment and Non-attainment**

Measures of attainment and non-attainment consist of both the formal attainment and nonattainment designations and the most recent set of ambient design values, which are based on measurements from 1995 to 1997. Formal attainment/nonattainment status applies only to the 1-hour standard, since such designations have not yet been made for the 8-hour standard.

Outside of California, the 1990 census showed 72 million people living in areas that were formally designated as non-attainment for the 1-hour standard as of August 10, 1998. The

individual areas, their populations, and their nonattainment classifications are listed in Table C-1 in Appendix C.

### *Design values*

An ozone design value is the concentration or average of concentrations that determines whether a monitoring site meets the NAAQS for ozone. Because of the way they are defined, design values can only be determined for three-year monitoring periods. We estimate the design values, and therefore the attainment effects, resulting from control programs by using a combination of modeling results and measured design values. Air quality model runs for a base year and a future year are used to determine the relative change in ozone levels produced by the controls that would be implemented between the base and future years. This relative change is used to adjust the measured historical design values in the region being analyzed, as described in detail below.<sup>i</sup>

A 1-hour design value is the fourth highest daily maximum 1-hour average ozone concentration measured over a three-year period at a given monitor. An 8-hour design value is the three-year average of the annual fourth highest daily maximum 8-hour average ozone concentration at a given monitor. The full details of these determinations (including accounting for missing values and other complexities) are given in Appendices H and I of 40 CFR Part 50. As discussed in these appendices, design values are truncated to whole ppb. Due to the precision with which the standards are expressed (0.12 ppm for the 1-hour, 0.08 ppm for the 8-hour), nonattainment of the 1-hour standard is defined as a design value greater than or equal to 125 ppb and nonattainment of the 8-hour standard is defined as a design value greater than or equal to 85 ppb.

For the 1-hour standard, the design value for a county is the highest design value of the monitors within that county. Typically, there is one or zero monitors per county. If a county does not contain an ozone monitor, it cannot have a design value. For most of our analyses, county design values are consolidated where possible into design values for metropolitan areas. The design value for a metropolitan area is the highest design value among the included counties. Counties that are not in metropolitan areas are listed separately. For the purposes of the analyses described in this section, we have assumed that the definition of county and metropolitan area design values for the 8-hour standard will be the same as for the 1-hour standard. It should be noted, however, that we have not yet determined how county and metropolitan area design values will be defined for the 8-hour standard.

To simulate the air quality effects of today's proposal, design values are estimated or

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<sup>i</sup> The procedure described in this paragraph can also be used to isolate the effects of a particular control program on design values. To do so, one needs to apply the procedure twice: once for a future year without applying the control program of interest, and once for a future year with the control program of interest applied. The impact of the control program on ozone design values is given by the difference between the design values calculated for the two different cases.

## **Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999**

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projected from measured 1995-1997 county design values by the method described in Subsection B.1.c. Projected design values are determined only for counties that have measured design values. The measured 1995-1997 design values that exceeded the 1-hour standard in metropolitan areas and rural counties in the 37 states that participated in OTAG are shown in Table C-2 in Appendix C. Similar measured design values for the 8-hour standard are shown in Table C-3.

### **b. General Description of Ozone Modeling in the OTAG Domain to Estimate the Effect of Tier 2/Sulfur Controls**

We used the considerable development work done for the ROTR as a foundation to estimate the impact of the proposed Tier 2/Sulfur controls on ozone levels in the OTAG domain. A method for estimating the design values that result from a given control scenario was also developed for the ROTR and has been extended to estimate the effects of Tier 2/Sulfur controls. Further details of the modeling work are presented below and in a technical memorandum to Docket A-97-10, "Photochemical Air Quality Simulations in Support of Tier 2/Sulfur," by Harvey Michaels.

#### *The basic method of using modeling*

The modeling methodology requires running two simulations, a base case (without Tier 2/Sulfur controls) and a control case (with Tier 2/Sulfur controls). The effects of the control program are then evaluated by comparing the modeling results of the control case with those of the base case. The base case for our Tier 2/Sulfur ozone analysis is the 2007 post-ROTR scenario. We used two versions of this base case; the first was published with the ROTR's Supplemental Notice of Proposed Rulemaking (SNPR) and used the SNPR emission inventories, while the second used the ROTR Final Inventory (September 1998), which was updated based on public comments. Table III-27 indicates which of our Tier 2/Sulfur simulations correspond to which base case.

Emissions inventories and meteorology were developed for four historical ozone episodes, each about 10 days long, from 1988, 1991, 1993, and 1995. When the photochemical grid model was judged to satisfactorily reproduce the historical episodes, the meteorology was retained and emission inventories for the base and control cases were substituted for the historical emission inventories. The base and control cases were then run for all four episodes.

The model output is hourly average ozone concentrations in all grid cells of the modeling domain for all hours of the simulation. A large number of different metrics have been developed to compare the ozone concentrations in the base case with those in the control case. One of the most useful of these, because of its relationship to measured design values used to determine attainment and nonattainment, is projected design values. Design values were projected using

the modeling results from three episodes: 1991, 1993, and 1995.<sup>j</sup> The calculation of projected design values is described in the next section.

To facilitate the ozone modeling, the emission reduction due to Tier 2/Sulfur controls was expressed as a percentage reduction from the 2007 post-ROTR emission inventory for all highway mobile sources. The procedure used to do this is described in a technical memo to Docket No. A-97-10 (“Methodology for Developing Inventory Reductions Used in Ozone Modeling,” by John Koupal). These percentage reductions were applied everywhere in the modeling domain to all on-highway emissions in the base case. The proposed Tier 2/Sulfur program would achieve almost all of its emission reductions from cars and light trucks, but converting these reductions to a percentage of all on-highway emissions greatly streamlined the process of modeling the proposed Tier 2/Sulfur controls.

The standard ozone metrics applied to the modeling results are relatively simple and self-explanatory. For example, “Grid Cell Days Above the Standard” is a count of all the grid cells on all simulation days (except for 2 or 3 startup days in each episode) that the daily maximum ozone concentration (either 1- or 8-hour average, depending on the specific metric) exceeded the standard. The “rollback method” of projecting design values is considerably more complicated, because it uses both measured design values and simulations. This method is described below.

### **c. The “Rollback Method” for Estimating Design Values Resulting from Control Measures**

Because of the way they are defined, design values can only be determined for three-year monitoring periods. We estimate the design values resulting from a given control program by beginning with the measured design values and then using two model runs to determine the relative change produced by the control program. The first model run is the base year case and uses an emissions inventory that corresponds to the measured design values. For 1995-1997 design values, we used the 1995/96 Base Year emissions inventory. The second model run is the control case and employs the inventories for which we are projecting resulting design values. We projected design values for three control cases: 2007 ROTR, 2007 Tier 2/Sulfur (OMS4) and 2020 Tier 2/Sulfur (OMS3). The relative change between the base year case and control case is used to adjust the measured design values, as described in this section. This process, called the “rollback method,” was used in the ROTR rulemaking and is more fully described in the document: “Procedures for Estimating the Impact of OTAG Strategy Run 5 on Attainment of the

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<sup>j</sup> This method, and the reasons why 1991, 1993, and 1995 were used, are discussed in “Procedures for Estimating the Impact of the OTAG Strategy Run 5 on Attainment of the 8-Hr Ozone NAAQS.” Draft: October 1997. Staff Report, Air Quality Modeling Group, Emissions, Monitoring and Analysis Division, Office of Air Quality Planning and Standards, U.S. EPA. EPA Air Docket A-96-56, II-A-24.

## Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999

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8-Hr Ozone NAAQS”, Staff Report, Draft October 1997, (EPA Air Docket A-96-56, II-A-24).<sup>k</sup>

There are three ozone episodes that are used in the modeling. Each corresponds to the meteorological conditions of a historical air pollution episode and is approximately 10 days long. The actual episodes occurred in 1991, 1993, and 1995. When we refer to running those three episodes for the ROTR in 2007, for example, we mean using the meteorology of those three episode with the emissions inventory we expect in 2007 following the ROTR.

The following procedure, referred to as the rollback method, was used to estimate the effects of control strategies on 1-hr and 8-hr ozone design values. Note that, except for Step 1(a), the procedures for treating 1-hr and the 8-hr design values are the same. The base year case refers to the 1995/96 Base Year inventory, which corresponds to the 1995-1997 period used to determine measured design values. The control case refers to one of the three cases for which we are projecting design values: either the 2007 post-ROTR scenario or the 2007 Tier 2/Sulfur case or the 2020 Tier 2/Sulfur case.

Step 1: Calculate ambient design values

- (a) For each monitor in a county determine the monitor-specific 1-hr design values by taking the 4th highest daily maximum value from ozone data collected at the monitoring site for the period 1995-1997. For determining an 8-hr design value, calculate the 3-year average of the 4th highest daily maximum 8-hr value in each year at the monitor.
- (b) Select the highest design value from all monitors within the county as the county-specific design value.

Step 2: Generate model predictions for three OTAG episodes (July 1991, 1993 and 1995) for the base year case and for the control case.

- (a) The base year case model predictions reflect emissions levels in the 1995-1997 time period.
- (b) The control case model predictions reflect a future year control scenario.

Step 3: Calculate an adjustment factor for each grid cell

Notes:

- (1) The adjustment factor is based on the percent difference in ozone predictions between the base year case and the control case. These factors will be used in

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<sup>k</sup> In keeping with Appendices H and I of 40 CFR Part 50, projected design values are truncated to whole ppb. Nonattainment of the 1-hour standard is defined as a design value greater than or equal to 125 ppb. Nonattainment of the 8-hour standard is defined as a design value greater than or equal to 85 ppb.

Step 5 to "rollback" ambient design values to reflect the impacts of the control case.

(2) Step 3 must be followed separately for the base year case and the control case.

For each grid cell:

- (a) Calculate daily maximum ozone concentrations for every day simulated (excluding the ramp up days for each episode) for the three OTAG episodes identified in Step 2. The ramp up days are one and two for the 1993 episode and one, two, and three for the other episodes.
- (b) For each episode select the 1st, 2nd, and 3rd highest daily maximum values.
- (c) For each of these "ranks" (i.e., 1st, 2nd, and 3rd ranked values), average the concentrations across the episodes (e.g., sum all 1st ranked values and divide by number of episodes). This yields an average value for each rank (i.e., the average of the highest, the average of the 2nd highest, and the average of the 3rd highest concentrations).
- (d) For each of the average ranks, calculate the ratio of the control case to the base year case. This yields a set of three ratios, one for each rank.
- (e) Average these three ratios to get the Adjustment Factor, which is multiplied times the 1995-1997 design value for a given grid cell to get the new design value for that grid cell.

Step 4: Assign a unique grid cell adjustment factor to each individual county

- (a) The cell with the largest portion of its area in the county is assigned to that county. If more than one cell is completely contained in the county, the cell with the highest base year case value is assigned to the county. The 1990 Base Year OTAG model predictions were used in those cases where it was necessary to choose among multiple grid cells for assigning a grid cell to a county.
- (b) The step of assigning a unique grid cell to each county yields the county-specific adjustment factor. Note that only one grid cell is assigned to a county. Thus, there is no spatial averaging or spatial weighting of adjustment factors using multiple grid cells in determining the county-specific factors.

Step 5: Calculate the rollback ambient design value

- (a) This step adjusts the ambient design values in each county to reflect the ozone reductions estimated to result from the control case.

## Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999

- (b) Multiply the county-specific ambient design value from Step 1 times the county-specific adjustment factor from Step 4.

### d. Specific Simulations Used to Evaluate Tier 2/Sulfur

Four simulations were performed in support of the Tier 2/Sulfur rulemaking. They are listed in the following table with the percentage reduction in highway vehicle emissions that were modeled for each run.

**Table III-27. Percent Reductions for Tier 2/Sulfur Ozone Modeling Runs**

<i>Run</i>	<i>% Reduction from Base Case</i>		<i>Base Case</i>
	<i>NO<sub>x</sub></i>	<i>VOC</i>	
OMS1	0%	30.3%	2007 Post-ROTR published with SNPR
OMS2	54.2%	0%	
OMS3	50.2%	10.5%	2007 Post-ROTR Final Inventory, September 1998
OMS4	18.5%	4.3%	

OMS1 and OMS2 were intended to explore the relative effect on ozone of VOC and NO<sub>x</sub> reductions. OMS3 was intended to model the effect of Tier 2/Sulfur in 2020, when it would affect a large portion of the fleet. OMS4 was intended to model Tier 2/Sulfur in 2007, which is an important year for ozone attainment and is also a year for which a large body of ROTR-related modeling results are available.

As mentioned previously, the percentage reductions in Table III-27 are those that when applied to the whole highway mobile source fleet will reproduce the reductions expected to result from Tier 2/Sulfur controls. The emissions modeling used to obtain the percent reductions from the base case are described in a technical memo to the Docket A-97-10 from John Koupal, titled "Methodology for Developing Inventory Reductions Used in Ozone Modeling."

### e. Results of the NO<sub>x</sub>-only and VOC-only Runs (OMS1 and OMS2)

While today's proposal decreases both NO<sub>x</sub> and VOC, NO<sub>x</sub> is decreased preferentially because it has a far greater effect on ozone. Most areas are NO<sub>x</sub>-limited—their ozone concentrations respond more to decreases in NO<sub>x</sub> than to decreases in VOCs. Only a few, highly localized areas are VOC-limited. For this reason, the Ozone Transport Assessment Group reached a broad consensus that regional ozone reductions in the eastern U.S. are best accomplished by reducing NO<sub>x</sub>. This consensus is reflected in the ROTR, which only reduces

NO<sub>x</sub>.

We have demonstrated that this conclusion is still valid even after the large NO<sub>x</sub> reductions resulting from the ROTR are taken into account. The demonstration used two OTAG-domain modeling runs that simulated the separate effects of mobile source VOC and NO<sub>x</sub> reductions. OMS2 simulated a 54 percent reduction in highway mobile source NO<sub>x</sub> and OMS1 a 30 percent reduction in highway mobile source VOC.<sup>1</sup>

The results of the OMS1 and OMS2 runs demonstrate that mobile source NO<sub>x</sub> reductions are much more effective at reducing ozone than are mobile source VOC reductions. The number of grid cell days on which the daily maximum 1-hour average ozone concentration exceeded 124 ppb fell 46% for the NO<sub>x</sub> reductions but only 2% for the VOC reductions. The number of grid cell days on which the daily maximum 8-hour average ozone concentration exceeded 84 ppb fell 40% for the NO<sub>x</sub> reductions but only 1% for the VOC reductions.

### **f. Details of the Tier 2/Sulfur Ozone Modeling Runs (OMS3 and OMS4)**

The results of our modeling of Tier 2/Sulfur have been summarized in the preamble. In this section, we discuss the detailed methods and results that were not covered there. The design-value results for all counties are listed in Appendix C.

As stated above, OMS3 and OMS4 were intended to model Tier 2/Sulfur in 2020 and 2007, respectively. For these two runs, the emission inventory for the base case, to which they are compared, is the ROTR budget case. This base case inventory uses the OTAG nonroad, highway heavy-duty, and highway light-duty emission estimates, as updated for the ROTR final rule based on public comment as of September 1998. This inventory is a newer version than the base case inventory used for the OMS1 and OMS2 runs. For both base case inventories, the nonroad inventories are based on the NEVES study, and the highway mobile source inventories are based on MOBILE5 emission factors, vehicle distributions, and mileage accumulation patterns. The fact that these inventories do not reflect the more recent information incorporated in the emission inventory analyses presented in Section A. of this chapter creates some uncertainty as to the absolute values of design values and which counties are in attainment or nonattainment under the base case.

Emissions and ozone levels were modeled for 2007 and 2020. The 2007 case is straightforward because we produced a full emission inventory for 2007 for the ROTR. For 2020, we assumed that total emissions (under ROTR plus current vehicle standards and fleet turnover) would be the same as in 2007, i.e., that emission reductions from fleet turnover and emission increases from growth in all sectors balance each other. This assumption is not exactly correct, but is close. Our best estimate is that without Tier 2/Sulfur NO<sub>x</sub> emissions from all

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<sup>1</sup> For comparison, we estimate that today's proposal will actually reduce mobile source NO<sub>x</sub> 50.0 percent and VOC 10.2 percent in 2020.

## Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999

human sources actually would be about 3% lower in 2020, and VOC emissions would be about 5% higher. The details of these estimates are in Section A of this chapter, "Inventory Impacts of Tier 2/Sulfur."

A relatively minor caveat is that the Tier 2/Sulfur NO<sub>x</sub> and VOC reductions were distributed evenly over all highway mobile sources, which have a different spatial distribution from light-duty vehicles.

After OMS3 and OMS4 had been run and design values calculated by the rollback method, our proposal was refined, resulting in slightly different percent reductions from the base case. These two sets of percent reductions are shown in Table III-28.

**Table III-28. Percentage Reductions from the 2007 Post-ROTR Inventory of NO<sub>x</sub> and NMHC for OMS3 (2020), OMS4 (2007), and for Today's Proposal in 2007, 2010, and 2020.**

<i>Year</i>	<i>OMS3 and OMS4</i>		<i>Today's Proposal</i>	
	<i>NMHC</i>	<i>NO<sub>x</sub></i>	<i>NMHC</i>	<i>NO<sub>x</sub></i>
2007	4.3%	18.5%	4.0%	18.1%
2010	—	—	5.4%	26.9%
2020	10.5%	50.2%	10.2%	50.0%

Because OMS3 and OMS4 were so close to today's proposal, we obtained the design values for today's proposal by linearly interpolating or extrapolating slightly based on the differences in NO<sub>x</sub> alone. Interpolations and extrapolations were also used to determine design values in 2010, which was not modeled, and to estimate design values in 2010 without Tier 2/Sulfur. The percentage reduction in total highway NO<sub>x</sub> emissions between the 2007 baseline and 2010 without Tier 2/Sulfur was 4.3%.

For discussing the effects of today's proposal on one- and eight-hour design values, we projected county design values using the rollback method for three modeling runs: 2007 post ROTR, OMS3, and OMS4. All other projected design values have been linearly interpolated based on NO<sub>x</sub>. As we have discussed previously, the primary effect on ozone has been produced by NO<sub>x</sub>.

All measured and projected county design values are in Appendix C. In addition, the preamble indicates counts of metropolitan areas and rural counties whose measured or projected design values meet various criteria with respect to the one and eight-hour standards. Appendix C contains the lists of these metropolitan areas and counties together with their design values and populations.

2. Visibility/Regional Haze

The Northern Front Range Air Quality Study (NFRAQS) report collected numerous ambient PM<sub>2.5</sub> samples in various areas around Denver, including urban areas such as Welby and rural areas such as Brighton, during the winter of 1997. The samples were analyzed for their composition, including the contribution of carbon-based, sulfate, nitrate, and crustal matter particles to each sample. The results of that analysis are summarized in Table III-29.

**Table III-29. NFRAQS Compositional Analysis of PM<sub>2.5</sub> Samples**

<i>Site</i>	<i>Carbon-based PM<sub>2.5</sub></i>	<i>Sulfate-based PM<sub>2.5</sub></i>	<i>Nitrate-based PM<sub>2.5</sub></i>	<i>Crustal Matter PM<sub>2.5</sub></i>
Welby	49%	10%	25%	16%
Brighton	42%	15%	32%	11%

The study used a variety of techniques to determine how much of the carbon-based, sulfate, and nitrate PM found in the PM<sub>2.5</sub> samples came from gasoline vehicles. Organic tracer compounds were used to determine how much of the carbonaceous PM<sub>2.5</sub> came from gasoline vehicles and to separate the contribution of normal emitting vehicles and higher emitting vehicles. A combination of inventory analysis, dispersion modeling, atmospheric chemistry, and analysis of compositional variation over time were used to determine the contribution of gasoline vehicles to sulfate and nitrate PM<sub>2.5</sub>. The study reported the following average percentages of sulfates and nitrates coming from gasoline vehicles. The proportion of each type of PM<sub>2.5</sub> determined to come from gasoline vehicles is shown in Table III-30.

**Table III-30. Percentage of PM<sub>2.5</sub> Coming from Gasoline Vehicles**

<i>Site</i>	<i>Carbon-Based</i>	<i>Sulfate-Based</i>	<i>Nitrate-Based</i>
Welby	57%	20%	36%
Brighton	62%	14%	38%

From these two sets of numbers, one can calculate the contribution of each type of PM<sub>2.5</sub> from gasoline vehicles to total PM<sub>2.5</sub>, as shown in the middle three columns of Table III-34. The results can be summed to derive the contribution of gasoline vehicles to total PM<sub>2.5</sub>, as shown in the last column in Table III-31.

**Table III-31. Percentage of Total PM<sub>2.5</sub> From Gasoline Vehicles**

<i>Site</i>	<i>Carbon-Based</i>	<i>Sulfate-Based</i>	<i>Nitrate-Based</i>	<i>Total</i>
Welby	28%	2%	9%	39%
Brighton	26%	2%	12%	40%

This section presents the analytic basis for the preamble discussion of the impact of mobile sources on visibility impairment in the U.S. In this context, “visibility impairment” refers to the reduction in the distance that one can see as the result of air pollution. As discussed in the preamble, fine particles suspended in the atmosphere are the primary cause of visibility impairment.

As discussed in the preamble, the Grand Canyon Visibility Transport Commission examined visibility impairment on the Colorado Plateau. Figures II-4 and II-5 in the Commission’s June 10, 1996 report titled “Recommendations for Improving Western Vistas” contain estimates for the contribution of 11 different sources to the man-made visibility impairment at Hopi Point. Figure II-4 is for annual average light extinction<sup>m</sup> and Figure II-5 for the worst days. Each figure gives estimates for 1990, 2000, 2010, and 2040. In 2000, for both annual average and worst days, the contribution from “Mobile” to light extinction is about 10 percent. EPA understands this category to consist of highway vehicles only, since there is a separate category for “Non Road Diesel.” Furthermore, the “Mobile” category must exclude dust caused by highway vehicle travel since there is a separate category for “Road Dust.” The road dust category is estimated to be responsible for about 30 percent of light extinction at Hopi Point.

It is generally recognized that the traditionally-used emission factors and transport assumptions for road dust have considerable uncertainty. Therefore, the contribution of road dust may be overstated in these figures. If light extinction from highway vehicles is expressed as a percentage of all light extinction not attributable to road dust, the highway vehicle contribution is 14 percent. Hence efforts to reduce highway vehicle emissions that cause light extinction can contribute significantly to improved visibility on the Colorado Plateau.

The benefit/cost analysis in Chapter VII includes the visibility-related economic benefits that would result from implementation of the Tier 2/Sulfur proposal.

### **C. Air Toxics**

This section summarizes our analysis of the impact of the proposed Tier 2/Sulfur standards on emissions of and exposure to air toxics. Section C.1. reviews the effects of selected air toxics emissions on human health. Section C.2. describes our analysis of air toxics emissions

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<sup>m</sup> Light extinction is a measure of visibility impairment.

and exposure and the effect that the proposed Tier 2/Sulfur standards may have on air toxics emissions and exposure.

### 1. Health Effects

Our assessment of motor vehicle toxics focused on the following compounds with cancer potency estimates that have or could have significant emissions from cars and light trucks: benzene, 1,3-butadiene, formaldehyde, acetaldehyde, and diesel PM. It should be noted, however, that the EPA does not have an official quantitative estimate of diesel emissions potency at present. The current estimate is still draft, as discussed below. A brief summary of health effects information on these compounds follows. The information in this section is based on our preliminary study of motor vehicle toxics emissions. The study will be peer reviewed in the near future. We anticipate updating our estimates once the study completes peer review.

#### a. Benzene

Benzene is an aromatic hydrocarbon which is present as a gas in both exhaust and evaporative emissions from motor vehicles. Benzene in the exhaust, expressed as a percentage of total organic gases (TOG), varies depending on control technology (e.g., type of catalyst) and the levels of benzene and aromatics in the fuel, but is generally about three to five percent. The benzene fraction of evaporative emissions depends on control technology (i.e., fuel injector or carburetor) and fuel composition (e.g., benzene level and Reid Vapor Pressure, or RVP) and is generally about one percent.

The EPA has recently reconfirmed that benzene is a known human carcinogen by all routes of exposure.<sup>9</sup> Respiration is the major source of human exposure. At least half of this exposure is by way of gasoline vapors and automotive emissions (EPA 1998a). Long-term exposure to high levels of benzene in air has been shown to cause cancer of the tissues that form white blood cells. Among these are acute nonlymphocytic<sup>n</sup> leukemia, chronic lymphocytic leukemia and possibly multiple myeloma (primary malignant tumors in the bone marrow),

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<sup>n</sup>Leukemia is a blood disease in which the white blood cells are abnormal in type or number. Leukemia may be divided into nonlymphocytic (granulocytic) leukemias and lymphocytic leukemias. Nonlymphocytic leukemia generally involves the types of white blood cells (leukocytes) that are involved in engulfing, killing, and digesting bacteria and other parasites (phagocytosis) as well as releasing chemicals involved in allergic and immune responses. This type of leukemia may also involve erythroblastic cell types (immature red blood cells). Lymphocytic leukemia involves the lymphocyte type of white blood cells that are responsible for the immune responses. Both nonlymphocytic and lymphocytic leukemia may, in turn, be separated into acute (rapid and fatal) and chronic (lingering, lasting) forms. For example; in acute myeloid leukemia (AML) there is diminished production of normal red blood cells (erythrocytes), granulocytes, and platelets (control clotting) which leads to death by anemia, infection, or hemorrhage. These events can be rapid. In chronic myeloid leukemia (CML) the leukemic cells retain the ability to differentiate (i.e., be responsive to stimulatory factors) and perform function; later there is a loss of the ability to respond.

## Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999

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although the evidence for the latter has decreased with more recent studies.<sup>10,11</sup> Leukemias, lymphomas, and other tumor types have been observed in experimental animals that have been exposed to benzene by inhalation or oral administration (EPA 1985, Clement 1991). Exposure to benzene and/or its metabolites has also been linked with genetic changes in humans and animals<sup>12</sup> and increased proliferation of mouse bone marrow cells.<sup>13</sup> Furthermore, the occurrence of certain chromosomal changes in individuals with known exposure to benzene may serve as a marker for those at risk for contracting leukemia.<sup>14</sup>

The latest assessment by EPA places the excess risk of developing acute nonlymphocytic leukemia at  $2.2 \times 10^{-6}$  to  $7.7 \times 10^{-6}/\mu\text{g}/\text{m}^3$ . In other words, there is a risk of two to eight excess acute nonlymphocytic leukemia cases in one million people exposed to  $1\mu\text{g}/\text{m}^3$  benzene over a lifetime (70 years). These numbers represent the maximum likelihood (MLE) estimate of risk, not an upper confidence limit (UCL).

A number of adverse noncancer health effects, blood disorders such as preleukemia and aplastic anemia, have also been associated with low-dose, long-term exposure to benzene (EPA 1985, Clement 1991, <sup>15</sup>). People with long-term exposure to benzene may experience harmful effects on the blood-forming tissues, especially the bone marrow. These effects can disrupt normal blood production and cause a decrease in important blood components, such as red blood cells and blood platelets, leading to anemia (a reduction in the number of red blood cells), leukopenia (a reduction in the number of white blood cells), or thrombocytopenia (a reduction in the number of blood platelets, thus reducing the ability for blood to clot). Chronic inhalation exposure to benzene in humans and animals results in pancytopenia<sup>o</sup>, a condition characterized by decreased numbers of circulating erythrocytes (red blood cells), leukocytes (white blood cells), and thrombocytes (blood platelets).<sup>16,17</sup> Individuals that develop pancytopenia and have continued exposure to benzene may develop aplastic anemia,<sup>p</sup> whereas others exhibit both pancytopenia and bone marrow hyperplasia (excessive cell formation), a condition that may indicate a preleukemic state.<sup>18,19</sup> The most sensitive noncancer effect observed in humans is the depression of absolute lymphocyte counts in the circulating blood.<sup>20</sup> A draft reference concentration (RfC) has been developed for benzene. The reference concentration (RfC) is an estimate of a continuous inhalation exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious noncancer effects during a lifetime; these estimates frequently have uncertainty levels that span perhaps an order of magnitude. The benzene RfC is  $9\mu\text{g}/\text{m}^3$ , which means that long-term exposures to benzene

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<sup>o</sup>Pancytopenia is the reduction in the number of all three major types of blood cells (erythrocytes, or red blood cells, thrombocytes, or platelets, and leukocytes, or white blood cells). In adults, all three major types of blood cells are produced in the bone marrow of the vertebra, sternum, ribs, and pelvis. The bone marrow contains immature cells, known as multipotent myeloid stem cells, that later differentiate into the various mature blood cells. Pancytopenia results from a reduction in the ability of the red bone marrow to produce adequate numbers of these mature blood cells.

<sup>p</sup> Aplastic anemia is a more severe blood disease and occurs when the bone marrow ceases to function, i.e., these stem cells never reach maturity. The depression in bone marrow function occurs in two stages - hyperplasia, or increased synthesis of blood cell elements, followed by hypoplasia, or decreased synthesis. As the disease progresses, the bone marrow decreases functioning. This myeloplastic dysplasia (formation of abnormal tissue) without acute leukemia is known as preleukemia. The aplastic anemia can progress to AML (acute myelogenous leukemia).

should be kept below  $9 \mu\text{g}/\text{m}^3$  to avoid appreciable risks of these non-cancer effects.<sup>21</sup>

### **b. 1,3-Butadiene**

1,3-Butadiene is formed in vehicle exhaust by the incomplete combustion of the fuel. It is not present in vehicle evaporative and refueling emissions, because it is not present in any appreciable amount in gasoline. 1,3-Butadiene accounts for 0.4 to 1.0 percent of total exhaust TOG, depending on control technology and fuel composition.

EPA recently prepared a draft assessment that would determine sufficient evidence exists to consider 1,3-butadiene a known human carcinogen.<sup>22</sup> However, the Environmental Health Committee of EPA's Scientific Advisory Board (SAB), in reviewing the draft document, issued a majority opinion that 1,3-butadiene should instead be classified as a probable human carcinogen.<sup>23</sup> In the draft EPA assessment, the MLE estimate of a lifetime extra cancer risk from continuous 1,3-butadiene exposure is about  $3.9 \times 10^{-6}/\mu\text{g}/\text{m}^3$ . In other words, it is estimated that approximately 4 persons in one million exposed to  $1 \mu\text{g}/\text{m}^3$  1,3-butadiene continuously for their lifetime (85 years in this case) would develop cancer as a result of their exposure. Lower exposures are expected to result in risks that are lower.

1,3-Butadiene also causes a variety of reproductive and developmental effects in mice and rats (no human data) when exposed to long-term, low doses of butadiene (EPA 1998c). The most sensitive effect was reduced litter size at birth and at weaning. These effects were observed in studies in which male mice exposed to 1,3-butadiene were mated with unexposed females. In humans, such an effect might manifest itself as an increased risk of spontaneous abortions, miscarriages, still births, or very early deaths. Long-term exposures to 1,3-butadiene should be kept below its reference concentration of  $0.33 \mu\text{g}/\text{m}^3$  to avoid appreciable risks of these reproductive and developmental effects (EPA 1998c).

### **c. Formaldehyde**

Formaldehyde is the most prevalent aldehyde in vehicle exhaust. It is formed from incomplete combustion of both gasoline and diesel fuel and accounts for one to four percent of total exhaust TOG emissions, depending on control technology and fuel composition. It is not found in evaporative emissions.

Formaldehyde exhibits extremely complex atmospheric behavior.<sup>24</sup> It is present in emissions and is also formed by the atmospheric oxidation of virtually all organic species, including biogenic (produced by a living organism) hydrocarbons. Mobile sources contribute both primary formaldehyde (emitted directly from motor vehicles) and secondary formaldehyde (formed from photooxidation of other VOCs emitted from vehicles). The mobile source contribution is difficult to quantify, but it appears that at least 30 percent of formaldehyde in the ambient air may be attributable to motor vehicles (EPA 1993a).

## Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999

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EPA has classified formaldehyde as a probable human carcinogen<sup>25</sup> based on limited evidence for carcinogenicity in humans and sufficient evidence of carcinogenicity in animal studies, rats, mice, hamsters, and monkeys. Epidemiological studies in occupationally exposed workers suggest that long-term inhalation of formaldehyde may be associated with tumors of the nasopharyngeal cavity (generally the area at the back of the mouth near the nose), nasal cavity, and sinus (Clement 1991, EPA 1993a). Studies in experimental animals provide sufficient evidence that long-term inhalation exposure to formaldehyde causes an increase in the incidence of squamous (epithelial) cell carcinomas (tumors) of the nasal cavity (Clement 1991, EPA 1993a, EPA 1987). The distribution of nasal tumors in rats suggests that not only regional exposure but also local tissue susceptibility may be important for the distribution of formaldehyde-induced tumors (Clement 1991, EPA 1993a). Research has demonstrated that formaldehyde produces mutagenic activity in cell cultures.

The MLE estimate of a lifetime extra cancer risk from continuous formaldehyde exposure is about  $1.3 \times 10^{-6}/\mu\text{g}/\text{m}^3$ . In other words, it is estimated that approximately 1 person in one million exposed to  $1 \mu\text{g}/\text{m}^3$  formaldehyde continuously for their lifetime (70 years) would develop cancer as a result of their exposure. Lower exposures are expected to result in risks that are lower.

Formaldehyde exposure also causes a range of noncancer health effects. At low concentrations (0.05-2.0 ppm), irritation of the eyes (tearing of the eyes and increased blinking) and mucous membranes is the principal effect observed in humans. At exposure to 1-11 ppm, other human upper respiratory effects associated with acute formaldehyde exposure include a dry or sore throat, and a tingling sensation of the nose. Sensitive individuals may experience these effects at lower concentrations. Forty percent of formaldehyde-producing factory workers reported nasal symptoms such as rhinitis (inflammation of the nasal membrane), nasal obstruction, and nasal discharge following chronic exposure.<sup>26</sup> In persons with bronchial asthma, the upper respiratory irritation caused by formaldehyde can precipitate an acute asthmatic attack, sometimes at concentrations below 5 ppm;<sup>27</sup> formaldehyde exposure may also cause bronchial asthma-like symptoms in nonasthmatics.<sup>28,29</sup> However, it is unclear whether asthmatics are more sensitive than nonasthmatics to formaldehyde's effects.<sup>30</sup>

Immune stimulation may occur following formaldehyde exposure, although conclusive evidence is not available. Also, little is known about formaldehyde's effect on the central nervous system. Several animal inhalation studies have been conducted to assess the developmental toxicity of formaldehyde: The only exposure-related effect noted was decreased maternal body weight gain at the high-exposure level but no adverse effects on reproductive outcome of the fetuses that could be attributed to treatment were noted. An inhalation reference concentration (RfC), below which long-term exposures would not pose appreciable non-cancer health risks, is not available for formaldehyde at this time.

### d. Acetaldehyde

Acetaldehyde is a saturated aldehyde that is found in vehicle exhaust and is formed as a result of incomplete combustion of both gasoline and diesel fuel. It is not a component of evaporative emissions. Acetaldehyde comprises 0.4 to 1.0 percent of exhaust TOG, depending on control technology and fuel composition.

The atmospheric chemistry of acetaldehyde is similar in many respects to that of formaldehyde (Ligocki et al., 1991, <sup>31</sup>). Like formaldehyde, it can be both produced and destroyed by atmospheric chemical transformation, so mobile sources contribute to ambient acetaldehyde levels both by their primary emissions and by secondary formation resulting from their VOC emissions. Data from emission inventories and atmospheric modeling indicate that roughly 40 percent of the acetaldehyde in ambient air may be attributable to mobile sources.

Acetaldehyde emissions are classified as a probable human carcinogen. The MLE estimate of a lifetime extra cancer risk from continuous acetaldehyde exposure is about  $0.78 \times 10^{-6}/\mu\text{g}/\text{m}^3$ . In other words, it is estimated that less than 1 person in one million exposed to  $1 \mu\text{g}/\text{m}^3$  acetaldehyde continuously for their lifetime (70 years) would develop cancer as a result of their exposure.

Non-cancer effects in studies with rats and mice showed acetaldehyde to be moderately toxic by the inhalation, oral, and intravenous routes.<sup>32,33,34</sup> The primary acute effect of exposure to acetaldehyde vapors is irritation of the eyes, skin, and respiratory tract. At high concentrations, irritation and pulmonary effects can occur, which could facilitate the uptake of other contaminants. Little research exists that addresses the effects of inhalation of acetaldehyde on reproductive and developmental effects. The *in vitro* and *in vivo* studies provide evidence to suggest that acetaldehyde may be the causative factor in birth defects observed in fetal alcohol syndrome, though evidence is very limited linking these effects to inhalation exposure. Long-term exposures should be kept below the reference concentration of  $9 \mu\text{g}/\text{m}^3$  to avoid appreciable risk of these non-cancer health effects.<sup>35</sup>

### **e. Diesel Particulate Matter**

The particulate matter (PM) from diesel exhaust typically consists of a solid core, composed mainly of elemental carbon, which has a coating of various organic and inorganic compounds. The diameter of diesel particles is very small with typically 75-95 percent of the particle mass having a diameter smaller than  $1.0 \mu\text{m}$ . The characteristically small particle size increases the likelihood that the particles and the attached compounds will reach and lodge in the deepest and more sensitive areas of the human lung. Both the diesel particle and the attached compounds may be influential in contributing to a potential for human health hazard from long term exposure.

The heavy-duty highway and off-road diesel engines, as a group, account for most of the diesel particulate emissions currently released into ambient air.<sup>36</sup> Diesel particulate matter is mainly attributable to the incomplete combustion of fuel hydrocarbons, though some may be due

## Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999

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to engine oil or other fuel components.

In two human studies on railroad workers, and one on truckers, occupationally exposed to diesel exhaust (EPA 1998d), it was observed that long-term inhalation of diesel exhaust produced an excess risk of lung cancer. Taken together, the human studies show a positive association between diesel exhaust exposure and lung cancer. Studies in experimental animals provide additional evidence that long-term inhalation exposure to high levels of diesel particulate may pose a significant cancer risk. Research has demonstrated that exposure to high diesel exhaust levels causes an increase in lung tumors in two strains of rats and two strains of mice (EPA 1998d). Also, as a result of extensive studies, the direct-acting mutagenic activity of both the particle and gaseous fractions of diesel exhaust has been shown (EPA 1998d).

EPA's draft Diesel Health Assessment identifies both lung cancer as well as several other adverse respiratory health effects including respiratory tract irritation, immunological changes, and changes in lung function, as possible concerns for long term exposure to diesel exhaust. The evidence in both cases comes from the studies involving occupational exposures and or high exposure animal studies mentioned above, and the Health Assessment, when completed, will recommend how the data should be interpreted for lower environmental levels of exposure. The draft Health Assessment is currently being revised to address comments from a peer review panel of the Clean Air Science Advisory Committee. Based on human epidemiology studies, the draft MLE estimate of a lifetime extra cancer risk from continuous diesel exhaust particulate exposure ranges from  $3.0 \times 10^{-4}$  to  $1.0 \times 10^{-3}/\mu\text{g}/\text{m}^3$ . In other words, it is estimated that approximately 300 to 1000 persons in one million exposed to  $1 \mu\text{g}/\text{m}^3$  diesel exhaust particulate continuously for their lifetime (70 years) would develop cancer as a result of their exposure.

The California Air Resources Board has identified diesel exhaust PM as a "toxic air contaminant" under the state's air toxics program, based on the information available on cancer and non-cancer health effects. California is in the process of determining the need for, and appropriate degree of, control measures for diesel exhaust particulate matter. Note that California limited its finding to diesel particulate matter, as opposed to diesel exhaust. EPA's assessment activities of diesel exhaust PM are coincident with, but independent from, California's evaluation.

Particulates (i.e., particulate matter, PM) are a prominent part of diesel exhaust and play a role in contributing to total ambient PM, especially  $\text{PM}_{2.5}$  (PM less than  $2.5 \mu\text{m}$  in diameter). This means that EPA's new National Ambient Air Quality Standard for  $\text{PM}_{2.5}$  provides another health-based reference point. Diesel exhaust particles may pose a particularly serious health risk since more than 75 percent of the particles can be less than  $1 \mu\text{m}$  and the smaller diesel particles can be inhaled and deposited deeper in the lung. Diesel particles also have a large surface area per unit mass and carry a coating of organic compounds with them which may contribute to the health effects observed from particles. At the present time, EPA believes that for many people, keeping long term exposures to diesel particulate matter at or below  $5 \mu\text{g}/\text{m}^3$  provides an adequate margin of safety for the noncancer respiratory hazards.<sup>37</sup>

## **2. Assessment of Emissions and Exposure**

In 1993, EPA released the "Motor Vehicle-Related Air Toxics Study" to meet the requirements of Section 202(l)(1) of the Clean Air Act, which required EPA to complete a study of the need for, and feasibility of, controlling emissions of toxic air pollutants associated with motor vehicles and motor vehicle fuels (EPA 1993a). In 1998, EPA updated the emissions and exposure analyses done for this study to account for new information<sup>38,39</sup>. Base scenarios for 1990, 1996, 2007, and 2020 were included in the assessment, as well as several control scenarios in 2007 and 2020. Toxic emissions and exposure were modeled for the following urban areas: Chicago, Denver, Houston, Minneapolis, New York, Philadelphia, Phoenix, Spokane, and St. Louis. Results for these urban areas were extrapolated nationwide. As mentioned previously, EPA has assessed emissions, and exposure from the following air toxics: benzene, formaldehyde, acetaldehyde, 1,3-butadiene, and diesel particulate matter. An assessment of the cancer and non-cancer effects of mobile source emissions of these compounds has not yet been completed as part of the updated analyses.

This subsection describes the analysis we have conducted to update our 1993 study. Subsection C.2.a. discusses the emission modeling conducted for mobile source gaseous air toxics (including both exhaust and nonexhaust air toxics) and diesel PM. Subsection C.2.b. describes how we calculated nationwide air toxic emissions for our baseline scenario, which assumed continuation of the National Low Emission Vehicle program indefinitely. Subsection C.2.c. describes our analysis of air toxics exposure for our baseline scenario. Subsection C.2.d. describes our analysis of the effects of various vehicle and fuel control scenarios on air toxics emissions and exposure. It also describes how we used those analyses to estimate the effect of the proposed Tier 2/Sulfur standards on air toxics emissions. This subsection also reviews our analysis of the potential impact of increased diesel engine use in light trucks on diesel PM emissions and exposure.

### **a. Emissions Modeling**

#### *i. Gaseous Air Toxics Emissions Modeling*

In these analyses, emissions of benzene, formaldehyde, acetaldehyde, and 1,3-butadiene were estimated using a toxic emission factor model, MOBTOX5b. This model is based on a modified version of MOBILE5b, which estimates emissions of regulated pollutants, and essentially applies toxic fractions to TOG estimates. The TOG basic emission rates used in this modeling were similar, but not identical, to the rates used for previous modeling studies. The model accounted for differences in toxic fractions between technology groups, driving cycles, and normal versus high emitters. Impacts of fuel formulations were also addressed in the modeling. Motor vehicle toxic emissions were modeled for the following urban areas: Chicago, Denver, Houston, Minneapolis, New York, Philadelphia, Phoenix, Spokane, and St. Louis.

#### *Exhaust Emissions*

## Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999

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Analysis of speciation data from 1990 technology light-duty gasoline vehicles done for the EPA Complex Model for Reformulated Gasoline showed that the fraction of toxic emissions relative to TOG differs among eight technology groups within the Complex Model as well as between normal emitters and high emitters.<sup>40</sup> This difference is especially significant for 1,3-butadiene; its TOG fraction is about three times larger for high emitters than for normal emitters. If this difference is not taken into account, the impact of I/M programs and fleet turnover to vehicles with lower deterioration rates will be underestimated. Thus, the input format for exhaust toxic adjustment factors in MOBTOX5b was structured to allow input of high and normal emitter toxic emission rates for a given “target” fuel. These toxic emission rates were then weighted to come up with a composite toxic emission factor, based on a distribution of normal and high emitters. This distribution is not supplied directly by the MOBILE model. Instead, this distribution was determined from the fleet average TOG emission rate on baseline fuel as determined by MOBILE and average normal and high TOG emission rates on baseline fuel derived from the Complex Model. Essentially, “toxic-TOG curves” were developed that plot the target fuel toxic emission rate against the base fuel TOG emission rate.

To construct these curves, the distribution of normal and high emitters was determined in the following manner for each model year. A TOG gram per mile emission rate for normal emitters (TOG-N) and a TOG emission rate for high emitters (TOG-H) on baseline fuel were input into MOBTOX5b. TOG-N from newer technology light-duty gasoline vehicles and trucks were obtained from an unconsolidated version of the Complex Model, which provides output for normal emitters in each of eight technology groups. The Complex Model provides estimates for mass of exhaust VOC, which is TOG minus the mass of methane and ethane. TOG was estimated by applying a conversion factor which accounts for the mass of these compounds. The conversion factor was derived by analysis of weight percent emissions of methane and ethane from available speciation data. Based on the distribution of technology groups in given model year, the individual TOG estimates were weighted appropriately to obtain a composite estimate for all normal emitters. Since the unconsolidated model’s TOG-N emission rates are applicable only to Tier 0 light duty vehicles, they had to be adjusted for Tier 1 and later vehicles. This adjustment was performed by multiplying the unconsolidated model results by the ratio of the emission standard for these later vehicles to the Tier 0 emission standard. TOG-H was also obtained from the unconsolidated version of the Complex Model. TOG-H was assumed to be the same for all Tier 0 and later vehicles.

For benzene, 1,3-butadiene, formaldehyde, and acetaldehyde, milligram per mile toxic emission rates for normal and high emitters running on a given fuel formulation were also entered into MOBTOX5b, using output from the unconsolidated version of the Complex Model.

An example of the data file format is provided in Table III-32. Using the information in the data file, an overall FTP toxic emission rate for each vehicle class in a given model year is calculated. This overall rate takes into account the distribution of normal and high emitters by calculating the slope and intercept of a straight line (the “toxic-TOG” curve), where the FTP toxic emission rates for a vehicle class in a given model year are a linear function of the baseline fuel TOG emission rate:

$$TOX_{\text{Fit, Fuel A, FTP}} = A + B * TOG_{\text{Baseline fuel, FTP}} \quad (1)$$

A and B are determined as follows:

$$A = (TOG-H * TOX-N - TOG-N * TOX-H) / (TOG-H - TOG-N) \quad (2)$$

$$B = (TOX-H - TOX-N) / (TOG-H - TOG-N) \quad (3)$$

where:

TOX-N = toxic emission rate for normal emitters derived from the Complex Model

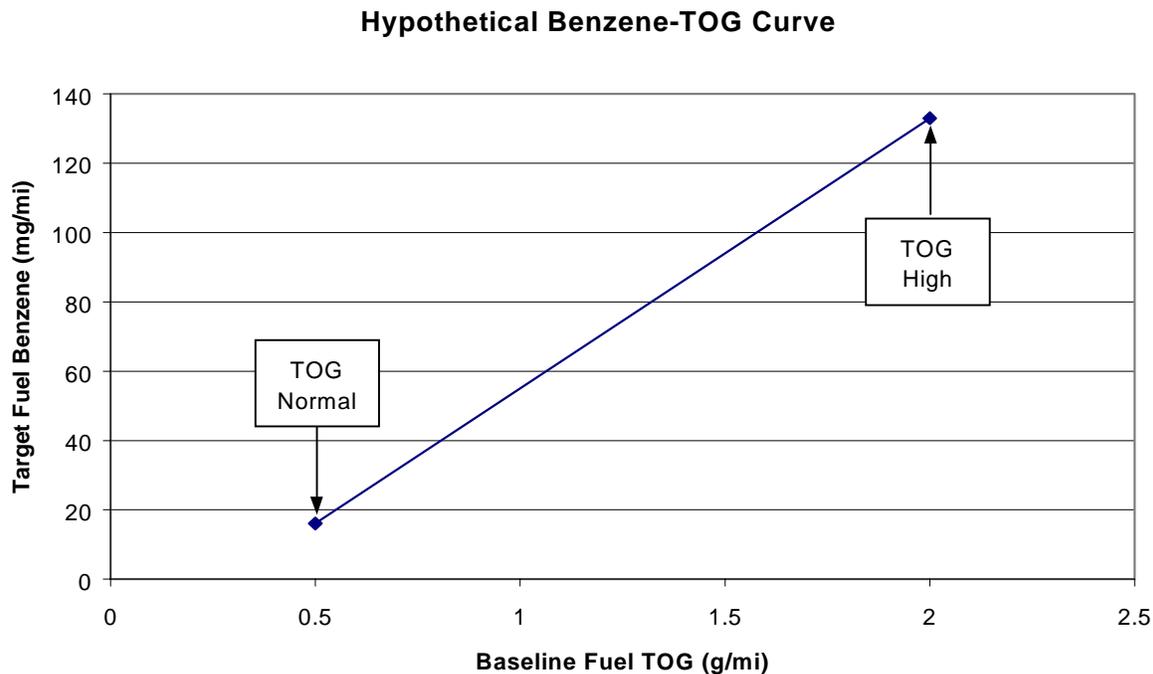
TOX-H = toxic emission rate for high emitters derived from the Complex Model

**Table III-32. Example of Data File Format for Toxic Adjustment Factors**

IV	MYA	MYB	TOG-N	TOG-H	BZ-N	BZ-H	AC-N	AC-H	FR-N	FR-H	BD-N	BD-H
1	1965	1974	0.000	10.00	0.00	276.93	0.00	109.72	0.00	224.28	0.00	93.15
1	1975	1980	0.000	10.00	0.00	263.61	0.00	108.70	0.00	173.41	0.00	44.57
1	1981	1987	0.640	4.03	28.63	113.23	5.07	32.89	7.16	44.59	2.14	25.84
1	1988	1999	0.570	4.03	17.49	116.45	4.02	28.65	5.67	36.68	2.04	30.82

IV = vehicle class, MYA = initial model year, MYB = final model year, TOG-N = TOG for normal emitters running on baseline fuel in g/mi, TOG-H = TOG for high emitters on baseline fuel in g/mi, BZ = benzene in mg/mi for vehicles running on fuel A, AC = acetaldehyde in mg/mi on fuel A, FR = formaldehyde in mg/mi on fuel A, BD = 1,3-butadiene in mg/mi on fuel A

These relationships can be thought of graphically, as illustrated in Figure III-19, below.



**Figure III-19. Example Plot of Target Fuel Benzene Versus Baseline Fuel TOG under FTP Conditions**

An issue related to the above methodology is whether the linear assumption is valid for baseline TOG values above the high emitter point and below the normal emitter point. This is particularly relevant in cases where A and B values are determined from Tier 0 vehicles (e.g., the Complex model), but the results are applied to Tier 1 and LEV-category vehicles. For the simple example presented above, negative benzene emissions are estimated for the target fuel when the baseline fleet-average TOG emission rate falls below 0.295 g/mi. Thus, for fleet-average emission rates below (and above) the normal (and high) emitter values, a different methodology was needed. In those cases, it was assumed that the toxic emission rate was the same on a fractional basis. For the example above, the benzene emission rate for a baseline TOG value of 0.1 g/mi would be calculated as follows:

$$BZ_{(TOG=0.1 \text{ g/mi})} = 0.1 \text{ g/mi} * (16 \text{ mg/mi BZ} / 0.5 \text{ g/mi TOG}) = 3.2 \text{ mg/mi}$$

This has the effect of forcing the toxic-TOG curve from the normal-emitter point back through the origin and thus avoids negative toxic emission rate estimates for Tier 1 and LEV-category vehicles. The same approach is used in cases where the fleet-average baseline TOG emission rate is above the high emitter point.

For non-light duty vehicle classes and older technology light-duty vehicles, such as non-catalyst and oxidation catalyst vehicles, adequate toxic emissions data were not available to

distinguish between emission rates of normal and high emitters. In such cases, the toxic fraction was assumed to be constant.

Next, aggressive driving corrections were applied to the FTP toxic emission rates for light duty vehicles. These corrections were provided in an external data file and were multiplicative in form. Several recent studies suggest that toxic fractions of TOG differ between FTP and aggressive driving conditions<sup>41,42,43</sup>. Thus, another adjustment to the toxic emission rates was applied to take into account this difference in toxic fractions. This adjustment took the form of the ratio of the toxic mass fraction over the unified cycle (FTP and off-cycle) to the toxic mass fraction over the FTP. The adjustment was obtained from an analysis of unpublished CARB data as described in Sierra Research et al. (Sierra 1998). The toxic emission rate under the unified cycle (FTP and off-cycle) was calculated in the model as follows:

$$TOX_{UC} = TOX_{FTP} * ADJ_{Aggressive\ Driving} * ADJ_{TOX\ UC/FTP} \quad (4)$$

where

$TOX_{UC}$  = Unified Cycle toxic emission rate

$TOX_{FTP}$  = FTP toxic emission rate

$ADJ_{Aggressive\ Driving}$  = Adjustment to TOG emissions for aggressive driving

$ADJ_{TOX\ UC/FTP}$  = Adjustment for difference in toxic mass fraction over the UC versus FTP

Next, toxic emission rates were adjusted in the model to take into account air conditioning effects on light duty vehicles. In the absence of data, we assumed that FTP-based toxic fractions will apply to the increased TOG mass as a result of air conditioning usage. Thus, the increase in TOG mass as a result of air conditioner usage was estimated from model year-specific corrections for air conditioner use on TOG emissions. The corrections were provided in an external data file in the model. The model calculates the increase in toxic emissions as a result of air conditioner use as follows:

$$TOX_{A/C} = TOX_{FTP} * ADJ_{A/C} \quad (5)$$

where

$TOX_{A/C}$  = increase in toxic emissions as a result of air conditioning usage

$TOX_{FTP}$  = toxic emission rate under FTP conditions

$ADJ_{A/C}$  = Air conditioner usage adjustment for TOG

This result was then added to the  $TOX_{UC}$  estimate for an overall in-use toxic emission rate. MOBTOX5b then applies temperature, speed, humidity and load corrections.

*Evaporative, Refueling, Running Loss, and Resting Loss Emissions*

MOBTOX5b estimated evaporative, refueling, running loss, and resting loss toxic emissions for benzene. (1,3-Butadiene, formaldehyde, and acetaldehyde are not found in fuel and hence are not found in nonexhaust emissions. Because their nonexhaust emissions are zero,

## **Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999**

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they were not included in the portions of MOBTOX5b used to estimate nonexhaust emissions.) Benzene fractions of total hydrocarbons were entered in an external data file. Separate fractions were entered for hot soak, diurnal, refueling, running loss, and resting loss. Toxic fractions for evaporative, refueling and running loss benzene from gasoline vehicles were obtained from the Complex Model (EPA 1994). The Complex Model does not estimate resting loss emissions. EPA assumed that the benzene fractions of diurnal and resting loss emissions were the same.

### *ii. Diesel PM Emissions Modeling*

To estimate diesel PM emissions, we used EPA's PART5 model. PART5 is similar in structure and function to the MOBILE series of models. It calculates exhaust and non-exhaust (e.g., road dust) particulate emissions for each vehicle class included in the MOBILE models. Only primary exhaust PM emission rates from diesel vehicles were included in these analyses since cancer potencies are not available for PM emissions such as tire and brake wear or for secondary PM formed through transformation of diesel engine emissions of SO<sub>x</sub>, NO<sub>x</sub>, and VOC. A particle size cut-off of 10 μm was specified in the model inputs since essentially all primary exhaust PM from diesel engines is smaller than 10 μm.

### **b. Nationwide Toxic Emissions Estimates – Baseline Scenario**

Nationwide urban emission estimates were developed by mapping each county in the United States to one of the modeled urban areas, based primarily on geographic considerations (Sierra, 1998). The resulting county level emission rates were weighted by VMT estimates to come up with average nationwide rates. Average nationwide emission rates for baseline scenarios in 1990, 1996, 2007, and 2020 are given in Table III-33. The baseline scenario assumed implementation of NLEV standards (0.09 g/mi) for light-duty gasoline vehicles and light duty trucks under 6000 lbs. gross vehicle weighting, Tier 1 standards for light-duty trucks over 6000 lbs., and a mix of conventional gasoline and Phase 2 reformulated gasoline with no additional sulfur control.

**Table III-33. Average Nationwide Highway Vehicle Toxic Emission Rates (mg/mi) In 1990, 1996, 2007, and 2020, for Baseline Scenarios.**

<i>Toxic</i>	<i>CY 1990</i>	<i>CY 1996</i>	<i>CY2007</i>	<i>CY2020</i>
Benzene	126	62	26	16
Acetaldehyde	19	14	6	3
Formaldehyde	61	35	14	8
1,3-Butadiene	17	9	3	2
Diesel PM	93	62	23	17

A number of rough approximations had to be made due to the small number of cities actually modeled. For instance, most of the South was mapped to Houston, a reformulated gasoline area, even though most Southern cities do not require reformulated gasoline. Also, all of California was mapped to Phoenix, which does not take into account the California LEV program. EPA plans to perform additional modeling prior to the final rulemaking to improve the national estimate. Despite these limitations, however, the nationwide exposure estimates should provide reasonable approximations.

**c. Exposure – Baseline Scenario**

Exposure modeling was done for 1990 using the Hazardous Air Pollutant Exposure Model for Mobile Sources, Version-3, or HAPEM-MS3.<sup>44</sup> This model uses CO as a tracer for toxics. Since most ambient CO comes from cars and light trucks, we believe CO exposure is a reasonable surrogate for exposure to other motor vehicle emissions, including toxics emissions. The HAPEM model links human activity patterns with ambient CO concentration to arrive at average exposure estimates for 22 different demographic groups (e.g., outdoor workers, children 0 to 17, working men 18 to 44, women 65+, etc.) and for the total population. The model simulates the movement of individuals between home and work and through a number of different microenvironments. The CO concentration in each microenvironment is determined by multiplying ambient concentration by a microenvironmental factor.

With the 1990 CO exposure estimates generated by HAPEM model for each urban area, EPA determined the fraction of exposure that was a result of on-road motor vehicle emissions. This calculation was accomplished by scaling the exposure estimates (which reflect exposure to total ambient CO) by the fraction of the 1990 CO emissions inventory from on-road motor vehicles, determined from the EPA Emission Trends database.<sup>45</sup> Nationwide urban CO exposure from on-road motor vehicles was estimated by first calculating a population-weighted average CO exposure for the nine modeled areas. This number was adjusted by applying a ratio of

## Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999

population-weighted annual average CO for urban areas in the entire country versus average ambient CO concentration for the modeled areas. To estimate rural exposure, the urban estimate was scaled downward using rough estimates of urban versus rural exposure from the 1993 *Motor Vehicle-Related Air Toxics Study* (EPA 1993a).

Modeled onroad CO exposure for 1990 was divided by 1990 CO grams per mile emission estimates to create a conversion factor. The conversion factor was applied to modeled toxic emission estimates (in grams per mile terms) to determine exposure to onroad toxic emissions, as shown in Equation 6:

$$\text{TOX}_{\text{Exposure}(\mu\text{g}/\text{m}^3)} = [\text{CO}_{\text{Exposure}(\mu\text{g}/\text{m}^3)} / \text{CO}_{\text{EF}(\text{g}/\text{mi})}]_{1990} \times \text{TOX}_{\text{EF}(\text{g}/\text{mi})} \quad (6)$$

where TOX reflects one of the six toxic pollutants considered in this study.

The exposure estimates for calendar years 1996, 2007, and 2020 were adjusted for VMT growth relative to 1990. 1,3-Butadiene exposure was adjusted for atmospheric transformation. The multiplicative factors used were 0.44 for summer, 0.70 for spring and fall, and 0.96 for winter.<sup>46</sup> In contrast, estimated exposure to formaldehyde and acetaldehyde was based on direct emissions. For these pollutants, removal of direct emissions in the afternoon was assumed to be offset by secondary formation. Table III-34 presents annual average exposure estimates for the entire population. Estimates were also developed for outdoor workers, and children 0 - 17 years of age. Exposure among outdoor workers was higher than for the entire population, and among children it was slightly lower.

**Table III-34. Average Nationwide Highway Vehicle Toxic Exposure ( $\mu\text{g}/\text{m}^3$ )  
In 1990, 1996, 2007, and 2020, for Baseline Scenarios.**

<i>Toxic</i>	<i>CY 1990</i>	<i>CY 1996</i>	<i>CY2007</i>	<i>CY2020</i>
Benzene	1.35	0.90	0.46	0.36
Acetaldehyde	0.20	0.20	0.10	0.08
Formaldehyde	0.66	0.51	0.26	0.20
1,3-Butadiene	0.14	0.09	0.04	0.04
Diesel PM	0.99	0.89	0.42	0.40

It should be noted that recent California-EPA studies estimated a population-weighted average outdoor diesel exhaust  $\text{PM}_{10}$  (particulate matter < 10  $\mu\text{m}$ ) exposure for 1995.<sup>47</sup> California also estimated indoor and total exposure concentrations for 1995. The 1995 indoor and total air exposure concentrations were estimated to be 1.47  $\mu\text{g}/\text{m}^3$  and 1.54  $\mu\text{g}/\text{m}^3$ , respectively. This estimate compares to the estimated annual average nationwide diesel PM

1996 exposure estimate of  $0.89 \mu\text{g}/\text{m}^3$  in Table III-37. The difference may be due to differences in estimates of emission rates, exposure patterns, the concentration of diesel vehicle traffic, or the spatial distribution of diesel engine emissions.

### **d. Impact of Potential Vehicle and Fuel Controls**

The following control scenarios for 2007 and 2020 were assessed:

- base fuels and emissions with NLEV and a 40 ppm sulfur standard.
- NLEV, 40 ppm sulfur, and 0.055 NMHC standard in 2004 for light duty gasoline vehicles and trucks.
- NLEV, 40 ppm sulfur, 0.055 NMHC standard, and light duty diesel trucks 50 percent of light duty truck sales in 2004 (phased in starting in 2001)

Although none of these scenarios represent the standards actually being proposed, the assessment shows that VOC emission reductions would reduce the health risk posed by many of the hazardous air pollutants emitted by light-duty vehicles and trucks beyond what was projected under baseline conditions. Estimates of the impact of VOC reductions from a 0.055 gram per mile NMHC standard for the full useful life of the vehicle, combined with a 40 ppm sulfur standard, on toxics emissions and exposure, are provided in Tables III-35 through III-39. Actual reductions under the standards being proposed would be smaller, since the VOC emission standards being proposed are less stringent. Under the proposed standards, VOC emissions would be about 20 percent larger than under the 0.055 NMHC/40 ppm sulfur scenario modeled; thus a similar difference would be expected for gaseous toxics emissions and exposure.<sup>9</sup> Table III-39 presents gaseous toxics exposure under the proposed standards, assuming the impact on toxics exposure is equivalent to the impact on VOC emissions.

The 1998 revision to the 1993 *Motor Vehicle-Related Air Toxics Study* also evaluated the potential increase in diesel PM emissions and exposure due to increased use of diesel engines in light trucks. Diesel engines are used in a very small portion of the cars and light-duty trucks in service today. However, engine and vehicle manufacturers have projected that diesel engines are likely to be used in an increasing share of light trucks. Some manufacturers have announced capital investments to build such engines. The 1998 study evaluated the potential increase in diesel PM emissions and exposure associated with introducing more diesel engines into the light-duty fleet, absent any action by EPA to mitigate those risks. An extreme case was modeled, with light duty diesel trucks accounting for 50 percent of light-duty truck sales in 2004, phased in starting in 2001.

The impact of such increased diesel penetration on emissions and exposure are provided

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<sup>9</sup>The difference in toxic reductions would not be exactly the same, since fleet average toxic emissions are affected differently by such factors the distribution of normal and high emitters, and the mix of vehicle control technologies.

## **Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999**

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in Tables III-35 through III-39. Based on the exposure estimates for 2020, the potential nationwide cancer risk from diesel particulate matter would increase by 137 percent under this scenario. Beyond 2020, the health risks would be even greater for two reasons. First, the proportion of light trucks equipped with diesel engines would continue to increase as the older, gasoline-powered trucks are replaced by a mix of gasoline and diesel trucks. Second, continued growth in the total number of miles driven would increase diesel PM emissions.

It should be noted that this increase in diesel sales is more rapid than the increase in diesel sales analyzed for its effect on direct and secondary PM levels, which assumes that diesel engines do not reach 50 percent of light truck sales until 2010. However, both analyses assume that diesel engines' share of light truck sales eventually reach the same level, and the two analyses' estimates of the total number of diesel trucks on the road tend to converge after 2010. Under this more gradual phase-in schedule, the increase in nationwide cancer risk would be slightly lower, about 128 percent. This estimate was developed by adjusting the estimated potential increase in risk for the more rapid phase-in to reflect the approximately four percent decrease in projected diesel PM emissions in 2020 that would result from the more gradual phase-in schedule.

Under both phase-in scenarios, we have estimated that the proposed Tier 2 standards for PM emissions from light-duty gasoline vehicles and trucks would reduce the potential increase in diesel PM cancer risk from cars and light trucks by over 85 percent. The potential number of cancers avoided would be even larger in future years as the proportion of diesel-powered light-duty trucks, and the number of miles they are driven, increased.

**Table III-35. Average Nationwide Highway Vehicle Toxic Emission Rates (mg/mi) in 2007, for Various Scenarios**

<i>Toxic</i>	<i>No New Controls Scenario</i>	<i>40 ppm Sulfur Scenario</i>	<i>0.055 NMHC Standard w/40 ppm Sulfur Scenario</i>	<i>0.055 NMHC Standard, 40 ppm Gasoline Sulfur, &amp; High Diesel Sales Scenario (50% of 2004 Sales)</i>
Benzene	25.54	24.43	23.44	20.89
Acetaldehyde	5.54	5.43	5.29	5.23
Formaldehyde	14.30	14.38	14.04	14.15
1,3-Butadiene	3.26	2.96	2.86	2.82
Diesel PM	23.36	23.36	23.36	38.69

**Table III-36. Average Nationwide Highway Vehicle Toxic Emission Rates (mg/mi) in 2020, for Various Scenarios**

<i>Toxic</i>	<i>No New Controls Scenario</i>	<i>40 ppm Sulfur Scenario</i>	<i>0.055 NMHC Standard w/40 ppm Sulfur Scenario</i>	<i>0.055 NMHC Standard, 40 ppm Gasoline Sulfur, &amp; High Diesel Sales Scenario (50% of 2004 Sales)</i>
Benzene	15.61	14.68	11.36	9.02
Acetaldehyde	3.39	3.29	2.82	2.77
Formaldehyde	8.42	8.45	7.29	7.45
1,3-Butadiene	2.29	2.04	1.69	1.53
Diesel PM	17.42	17.42	17.42	41.29

**Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999**

**Table III-37. Average Nationwide Highway Vehicle Toxic Exposures for the Entire Population ( $\mu\text{g}/\text{m}^3$ ) in 2007, for Various Scenarios**

<i>Toxic</i>	<i>No New Controls Scenario</i>	<i>40 ppm Sulfur Scenario</i>	<i>0.055 NMHC Standard w/40 ppm Sulfur Scenario</i>	<i>0.055 NMHC Standard, 40 ppm Gasoline Sulfur, &amp; High Diesel Sales Scenario (50% of 2004 Sales)</i>
Benzene	0.46	0.44	0.42	0.37
Acetaldehyde	0.10	0.10	0.095	0.094
Formaldehyde	0.26	0.26	0.26	0.26
1,3-Butadiene	0.044	0.040	0.038	0.038
Diesel PM	0.42	0.42	0.42	0.70

**Table III-38. Average Nationwide Highway Vehicle Toxic Exposures for the Entire Population ( $\mu\text{g}/\text{m}^3$ ) in 2020, for Various Scenarios**

<i>Toxic</i>	<i>No Tier 2/Sulfur Scenario</i>	<i>40 ppm Sulfur Scenario</i>	<i>0.055 NMHC Standard w/40 ppm Sulfur Scenario</i>	<i>0.055 NMHC Standard, 40 ppm Gasoline Sulfur, &amp; High Diesel Sales Scenario (50% of 2004 Sales)</i>
Benzene	0.36	0.34	0.26	0.21
Acetaldehyde	0.078	0.075	0.064	0.064
Formaldehyde	0.20	0.20	0.17	0.17
1,3-Butadiene	0.039	0.035	0.029	0.026
Diesel PM	0.40	0.40	0.40	0.96

**Table III-39. Average Nationwide Highway Vehicle Gaseous Toxic Exposures for the Entire Population ( $\mu\text{g}/\text{m}^3$ ) in 2020, under proposed Tier 2 standards.**

<i>Toxic</i>	<i>Exposure (<math>\mu\text{g}/\text{m}^3</math>)</i>
Benzene	0.25
Acetaldehyde	0.077
Formaldehyde	0.21
1,3-Butadiene	0.031

**e. Limitations**

The analysis referenced above was conducted by Sierra Research for the Office of Mobile Sources. It will undergo formal scientific peer review in the near future. Once that review is complete and the peer review comments are addressed, OMS expects to conduct a formal risk assessment on health risk of toxic emissions from mobile sources for the final rule.

## **Tier 2/Sulfur Draft Regulatory Impact Analysis - April 1999**

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