Development of Emission Rates for Light-Duty Vehicles in the Motor Vehicle Emissions Simulator (MOVES2009)

Draft Report

United States
Environmental Protection
Agency

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

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1. Criteria Pollutant Emissions from Light-Duty Gasoline Vehicles (THC, CO, NOx)

This chapter describes the technical development of emission rates for criteria pollutants (HC, CO, NOx) from light-duty vehicles for use in the draft MOVES model .

Section 1 describes the structure of the MOVES emissionRateByAge table, as it applies to criteria-pollutant emissions from gasoline-fueled light-duty vehicles. Section 1.5 describes the development of emission rates for vehicles manufactured prior to model year 2000. Sub-sections 1.5.1 and 1.5.2 describe the process of data selection and quality assurance. Rates were generated either directly from available data (sub-section 1.5.3) or by development and application of statistical "hole-filling" models (subsection 1.5.4). These rates were derived using data from the Phoenix I/M program and represent rates characteristic of a program with features similar to those in the Phoenix program.

Because steps 1.5.3 and 1.5.4 relied on data collected on IM240 and IM147 cycles, we thought it appropriate to evaluate the extrapolation with power to high levels beyond those covered by the IM cycles. The development and application of adjustments to rates in operating modes at high power is discussed in sub-section 1.5.5.

In MOVES terminology, pollutants are emitted by "sources" via one or more "processes." Within processes, emissions may vary by operating mode, as well as by age Group.

The relevant processes are exhaust emissions of total hydrocarbons (THC), carbon monoxide (CO) and oxides of nitrogen (NOx), during running operation (running exhaust) (pollutantID = 1,2,3, respectively). The pollutant process is running exhaust emissions (process 01). Thus, the pertinent values of polProcessID are 101, 201 and 301, respectively.

For these pollutant processes, the meanBaseRate is expressed in units of g/SHO, where SHO denotes "source-hours operating."

1.1 Emissions Sources (sourceBinID)

For these pollutant processes emissions sources include light-duty vehicles (cars and trucks). The corresponding sourceBins are defined as shown in Table 1 - 1. Note that the engine-size and weight-class attributes are not used, as they were for energy consumption. Unlike fuel or energy consumption, these parameters are assumed not to influence emissions, since light-duty vehicles are required to meet applicable standards irrespective of size and weight.

Table 1 - 1. Construction of sourceBins for Running-Exhaust Emissions for running-exhaust emissions from light-duty vehicles

Parameter	MOVES Database Attribute	Values
Fuel type	fuelTypeID	Gasoline = 01 Ethanol = 05
Engine Technology	engtechid	Conventional = 01
Regulatory Class	regClassID	LDV = 20 LDT = 30
Model-Year group	shortModYrGroupID	<as shown where?>
Engine Size Class	engSizeID	<not used>
Vehicle Test Weight	weightClassID	<not used>

1.2 Age Groups (ageGroupID)

To account for emissions deterioration, MOVES estimates emission rates for vehicles in a series of age ranges, identified as age groups (ageGroupID). Seven groups are used, as follows: 0-3, 4-5, 6-7, 8-9, 10-14, 15-19, and 20+ years. The values of the attribute ageGroupID for these classes are 3, 405, 607, 809, 1014, 1519, and 2099, respectively. These groups assume that the most rapid change in emissions as vehicles age occurs between 4 and 10 years.

1.3 Operating Modes (opModeID)

For running emissions, the key concept underlying the definition of operating modes is "vehicle-specific power" (VSP, P_{V}). This parameter represents the tractive power exerted by a vehicle to move itself and its cargo or passengers[1]. It is estimated in terms of a vehicle's speed and weight, as shown in Equation 1 - 1.

$$P_{\mathrm{V},t} = \frac{Av_t + Bv_t^2 + Cv_t^3 + mv_t a_t}{m} \qquad \textbf{1 - 1}$$

In this form, VSP ($P_{\mathrm{V},t}$, kW/tonne) is estimated in terms of vehicles':

- speed at time t (v_t, m/sec),
- acceleration a_t (m/sec^2) ,
- - mass m (tonne) (usually referred to as "weight, "),
- - track-road load coefficients A, B and C[3], representing rolling resistance, rotational resistance and aerodynamic drag, in units of kW-sec/m, kW-sec^2/m^2 and kW-sec^3/m^3, respectively.

Note that this version of the equation does not include the term accounting for effects of road grade, because the data used in this analysis was measured on chassis dynamometers.

On the basis of VSP, speed and acceleration, a total of 23 operating modes are defined for running-exhaust processes (Table 1 - 2). Aside from deceleration/braking, which is defined in terms of acceleration, and idle, which is defined in terms of speed alone, the remaining 21 modes are defined in terms of VSP within broad speed classes. Two of the modes represent “coasting,” where VSP < 0. and the remainder represent “cruise/acceleration,” with VSP ranging from 0 to over 30 kW/tonne. For reference, each mode is identified by a numeric label, the “opModeID.”

Table 1 - 2. Definition of the MOVES Operating Mode Attribute for Motor Vehicles (opModeID).

Operating Mode	Operating Mode Description	Vehicle-Specific Power (VSP_t, kW/tonne)	Vehicle Speed (v_t, mi/hr)	Vehicle Acceleration (a, mi/hr-sec)
0	Deceleration/Braking			$a_t \leq -2.0$ OR ($a_t < -1.0$ AND $a_{t-1} < -1.0$ AND $a_{t-2} < -1.0$)
1	Idle		$-1.0 \leq v_t < 1.0$	
11	Coast	$VSP_t < 0$	$0 \leq v_t < 25$	
12	Cruise/Acceleration	$0 \leq VSP_t < 3$	$0 \leq v_t < 25$	
13	Cruise/Acceleration	$3 \leq VSP_t < 6$	$0 \leq v_t < 25$	
14	Cruise/Acceleration	$6 \leq VSP_t < 9$	$0 \leq v_t < 25$	
15	Cruise/Acceleration	$9 \leq VSP_t < 12$	$0 \leq v_t < 25$	
16	Cruise/Acceleration	$12 \leq VSP_t$	$0 \leq v_t < 25$	
21	Coast	$VSP_t < 0$	$25 \leq v_t < 50$	
22	Cruise/Acceleration	$0 \leq VSP_t < 3$	$25 \leq v_t < 50$	
23	Cruise/Acceleration	$3 \leq VSP_t < 6$	$25 \leq v_t < 50$	
24	Cruise/Acceleration	$6 \leq VSP_t < 9$	$25 \leq v_t < 50$	
25	Cruise/Acceleration	$9 \leq VSP_t < 12$	$25 \leq v_t < 50$	
27	Cruise/Acceleration	$12 \leq VSP < 18$	$25 \leq v_t < 50$	
28	Cruise/Acceleration	$18 \leq VSP < 24$	$25 \leq v_t < 50$	
29	Cruise/Acceleration	$24 \leq VSP < 30$	$25 \leq v_t < 50$	
30	Cruise/Acceleration	$30 \leq VSP$	$25 \leq v_t < 50$	
33	Cruise/Acceleration	$VSP_t < 6$	$50 \leq v_t$	
35	Cruise/Acceleration	$6 \leq VSP_t < 12$	$50 \leq v_t$	
37	Cruise/Acceleration	$12 \leq VSP < 18$	$50 \leq v_t$	
38	Cruise/Acceleration	$18 \leq VSP < 24$	$50 \leq v_t$	
39	Cruise/Acceleration	$24 \leq VSP < 30$	$50 \leq v_t$	
40	Cruise/Acceleration	$30 \leq VSP$	$50 \leq v_t$	

1.4 Scope

In estimation of energy consumption for MOVES2004, it was possible to combine data from various sources without regard for the places of residence for various vehicles. In contrast, when turning attention to the criteria pollutants, it was clear that it would be essential to know with a high degree of confidence whether vehicles had been subject to inspection-and-maintenance (I/M) requirements at the time of measurement. After reviewing data sources, it became clear that the amounts of data collected within I/M areas vastly exceeded those collected in non-I/M areas. We also concluded that I/M programs themselves could provide a large and valuable source of data. In consideration of the demanding analytic tasks posed by the ambitious MOVES design, we elected to estimate rates for vehicles in I/M areas first, as the “base-line” or “default” condition. Following construction of a set of rates representing I/M conditions, the plan was to estimate rates for non-I/M areas relative to those in I/M areas. This approach is an inversion of that used in MOBILE, in which “non-I/M” is that “default condition” relative to which “I/M” emissions are calculated during a model run.

In addition, the rates described below represent emissions on the FTP temperature range (68 – 86 °F), to provide a baseline against which temperature adjustments would be applied during model runs.

1.5. Emission-Rate development: Subgroup 1 (Model years through 2000)

1.5.1 Data Sources

For emissions data to be eligible for use in MOVES development, several requirements were imposed:

- To derive rates for operating modes, it was essential to acquire data measured on transient tests.
- Data had to be measured at a frequency of approximately 1 Hz., e.g., continuous or “second-by-second” measurements.
- To make allowance for application of temperature adjustments (developed separately), it was necessary to know the temperature at the time of test.

1.5.1.1 Vehicle Descriptors

In addition to the requirements listed above, complete descriptive information for vehicles was required. Vehicle parameters required for incorporation into MOVES are shown in Table 1 - 3.

Table 1 - 3. Required Vehicle Parameters.

Parameter	Units	Purpose
VIN		Verify MY or other parameters
Fuel type		
Make		
Model		
Model year		Assign sourceBinID, calculate age-at-test
Vehicle class		Assign sourceBinID
GVWR	lb	Distinguish trucks from LDV
Track road-load power	hp	Calculate track road-load coefficients *A*, *B* and *C*

1.5.1.1.1 Track Road-Load Coefficients: Light-Duty Vehicles

For light-duty vehicles, we calculated the track load coefficients from the "track road load power at 50 mph" (TRLP, hp), based on Equation 1 - 2.

$$
\begin{aligned}
A &= \mathrm{PF}_A \cdot \left(\frac{\mathrm{TRLHP} \cdot c_1}{v_{50} \cdot c_2} \right) \\
B &= \mathrm{PF}_B \cdot \left(\frac{\mathrm{TRLHP} \cdot c_1}{\left(v_{50} \cdot c_2\right)^2} \right) \\
C &= \mathrm{PF}_C \cdot \left(\frac{\mathrm{TRLHP} \cdot c_1}{\left(v_{50} \cdot c_2\right)^3} \right)
\end{aligned}
\qquad \textbf{1 - 2}
$$

where:

PF_A = default power fraction for coefficient *A* at 50 mi/hr (0.35),
PF_B = default power fraction for coefficient *B* at 50 mi/hr (0.10),
PF_C = default power fraction for coefficient *C* at 50 mi/hr (0.55),
c_1 = a constant, converting TRLP from hp to kW (0.74570 kW/hp),
v_{50} = a constant vehicle velocity (50 mi/hr),

c_2 = a constant, converting mi/hr to m/sec (0.447 m·hr/mi·sec)).

In the process of performing these calculations, we converted from english to metric units, in order to obtain values of the track road-load coefficients in SI units, as listed above. Values of TRLP were obtained from the Sierra I/M Look-up Table.[2]

1.5.1.2 Test Descriptors

In addition, a set of descriptive information was required for sets of emissions measurements on specific vehicles. Essential items for use in MOVES are listed in Table 1 - 4.

Table 1 - 4. Required Test Parameters

Parameter	Units	Purpose
Date		Determine vehicle age at test
Time of day		Establish sequence of replicates
Ambient temperature	°F	Identify tests on target temperature range
Test Number		Identify 1st and subsequent replicates
Test duration	sec	Verify full-duration of tests
Test result	pass/fail	Assign tests to correct result stratum
Test weight	lb	Calculate vehicle-specific power

1.5.1.3 Candidate Data Sources

In addition to the parameters listed in Table 1 - 3 and Table 1 - 4, datasets with historic depth and large sample sizes were highly desirable, to characterize the high variability typical of exhaust emissions as well as trends against age.

Various datasets were available, representing several million vehicles when taken together (Table 1 - 5). In some cases they could be combined as broadly comparable pairs representing I/M and non-I/M conditions. Likely candidates were subjected to a high degree of scrutiny and quality-assurance, after which some were excluded from further consideration for specific reasons.

Table 1 - 5. Datasets considered for use in Estimating Light-duty Runnning Emissions.

Dynamometer		Remote-Sensing (RSD)	
I/M	non-I/M	I/M	non-I/M
AZ (Phoenix)		AZ Phoenix	
IL (Chicago)		IL (Chicago)	
MO (St. Louis)		MO St. Louis	
British Columbia			
CO (Denver)			
Indiana			
Ohio			
Wisconsin			
NY (New York)			
	Other MSOD		
		Maryland/N Virginia	VA (Richmond)
		CA (Los Angeles)	
		TX (Houston)	
		GA (Atlanta)	GA (Augusta/Macon)
			NE (Omaha)
			OK (Tulsa)

Several remote-sensing datasets received consideration. However, we elected not to use remote-sensing data directly to estimate rates, for several reasons: (1) For the most part, the RSD datasets on hand had very restricted model-year by age coverage (historic depth), which limited their usefulness in assigning deterioration. (2) The measurement of hydrocarbons by RSD is highly uncertain. The instruments are known to underestimate the concentrations of many VOCs relative to flame-ionization detectors. In inventory estimation, a multiplicative adjustment of 2.0-2.2 is often applied to allow comparison to HC measurements by other methods.[3] (3) In MOVES, emissions are expressed in terms of mass rates (mass/time). While fuel-specific rates (mass emissions/mass fuel) can be estimated readily from remote-sensing data[4], mass rates cannot be calculated without an independently estimated CO_2 mass rate. It followed that RSD would not provide rates for any MY×Age combinations where dynamometer data were not available. In these cases, RSD would be dependent on and to some extent redundant with dynamometer data. (4) Because remote-sensing measurements are typically sited to catch vehicles operating under light to moderate acceleration, results can describe emissions only selected cruise/acceleration operating modes. However, RSD cannot provide measurements for coasting, decel/braking or idle modes. For these reasons we reserved the RSD for secondary roles, such as verification of results obtained from dynamometer data.

Table 1 - 6. Characteristics of Candidate Datasets

	Chicago	**Phoenix**	**NYIPA**	**St. Louis**
Type	Enhanced	Enhanced	Basic/Enhanced	Enhanced
Network	Centralized	Centralized	De-centralized	Centralized
Exempt MY	4 most recent	4 most recent	2 most recent	2 most recent
Collects random sample?	YES	YES	n/a	NO
Program Tests	Idle, IM240, OBD-II	Idle/SS, IM240, IM147, OBD-II	IM240	IM240
Fast-pass/Fast-fail?	YES	YES	n/a	YES
Test type (for random sample)	IM240	IM240, IM147	IM240	n/a
Available CY	2000-2004	1995-99 2002-2004	1999-2002	2002-2005
Size (no. tests)	8,900	62,500	8,100	

Dynamometer datasets that received serious consideration are described below and summarized in Table 1 - 6.

Metropolitan Chicago. We acquired data collected over four calendar years (2000-04) in Chicago's centralized enhanced program. In addition to routine program tests, the program performed IM240 tests on two random vehicle samples. One is the "back-to-back" random sample. This sample is relatively small (n ~ 9,000 tests), but valuable because each selected vehicle received two full-duration IM240 tests in rapid succession, obviating concerns about conditioning. A second is the "full-duration" random sample, in which selected vehicles received a single full-duration IM240. This sample is much larger (n > 800,000) but less valuable due to the lack of replication. Despite its size, the full-duration sample has no more historic depth than the back-to-back sample, and thus sheds little additional light on age trends in emissions. Both samples were simple random samples, indicating that in the use of the data, users must assume that the samples are self-weighting with respect to characteristics such as high emissions, passing/failing test results, etc.

St. Louis. Another large program dataset is available from the program in St. Louis. While a large sample of program tests is available, this program differed from the others in that no random evaluation sample was available. Because vehicles were allowed to “fast-pass” their routine tests, results contained many partial duration tests (31 – 240 seconds). At the same time, the lack of replication raised concerns about conditioning. Partial duration was a concern in itself in that the representation of passing vehicles declined with increasing test duration, and also because it compounded the issue of conditioning. In addition, while OBD-equipped vehicles failing a scan received IM240s, those passing their scans did not. Because addressing the interwoven issues of inadequate conditioning, “fast-pass bias” and “OBD-screen bias” proved impractical, we excluded this dataset from further consideration.

Phoenix. At the outset, the random samples from the Phoenix program appeared attractive in that they had over twice the historic depth of any other dataset, with model-year × age coverage spanning 11 calendar years. Usage of these samples is somewhat complicated by the fact that no random samples were collected for two years (2000-01) and by the fact that the sample design employed changed in the middle of the ten-year period. During the first four years, a simple “2% random sample” was employed. During the last four years, a stratified design was introduced which sampled passing and failing vehicles independently and at different rates. In the stratified sample, failures were over-sampled relative to passing vehicles. Thus, using these data to estimate representative rates and to combine them with the 2% sample, assumed to be self-weighting, required reconstruction of the actual stratified sampling rates, as described below.

New York Instrumentation/Protocol Assessment (NYIPA). This dataset differs from the others in that while it was collected within an I/M area in New York City, it is not an I/M program dataset as such. It is, rather, a large-scale research program designed to establish correlation between the IM240 and an alternative transient test. It is not entirely clear whether it can be considered a random sample, in part because estimation of representative averages was not a primary goal of the study. All data that we accessed and used was measured on full-duration IM240s during a four-year period. There was a high degree of replication in the conduction of tests, allowing fully-conditioned operation to be isolated by exclusion of the initial test in a series of replicates. While these data played a prominent role in development of energy consumption rates for MOVES2004, the four-year duration of the program limits its usefulness in analysis of age trends for criteria pollutants.

1.5.2 Data Processing and Quality-assurance

We performed several quality-assurance steps to avoid known biases and issues in using I/M data to estimate mean emissions. One source of error, “inadequate conditioning” can occur when vehicles idle for long periods while waiting in line. To ensure that measurements used reflected fully-conditioned vehicles we excluded either portions of tests or entire tests, depending on test type and the availability of replicates. If back-to-back replication was performed, we discarded the first test in a series of replicates. If replication was not performed, we excluded the first 120 seconds of tests (for IM240s only).

Another problem occurs when calculation of fuel economy for tests yields values implausible enough to indicate that measurements of one or more exhaust constituents are invalid. To identify and exclude such tests, we identified tests with outlying measurements for fuel economy, after grouping vehicles by vehicle make, model-year and displacement.

An issue in some continuous or second-by-second datasets is that cases occur in which the emissions time-series appears to be "frozen" or saturated at some level, not responding to changes in power. We found that the occurrence of such problems was more or less evenly distributed among the fleet regardless of age or model year, and that severe instances were rare. We excluded tests in which 25% or more of the measurements were "frozen."

For a modal analysis assuming that emissions respond to power on short time scales,
It is critical that the emissions time-series be aligned to the power time-series. Consequently, we examined alignment for all tests. As necessary, we re-aligned emissions time series to those for VSP by maximizing correlation coefficients, using parametric Pearson coefficients for CO_2 and NO_x, and non-parametric Spearman coefficients for CO and THC.

1.5.2.1 Sample-design reconstruction (Phoenix only)

For data collected in Phoenix during CY 2002-05, we constructed sampling weights to allow use of the tests to develop representative means. The program implemented a stratified sampling strategy, in which failing vehicles were sampled at higher rates than passing vehicles.
It is thus necessary to reconstruct the sample design to appropriately weight failing and passing vehicles in subsequent analyses. After selection into the random sample, vehicles were assigned to the "failing" or "passing" strata based on the result of their routine program test, with the specific test depending on model year, as shown in Figure 1 - 1. Within both strata, sample vehicles then received three replicate IM147 tests.

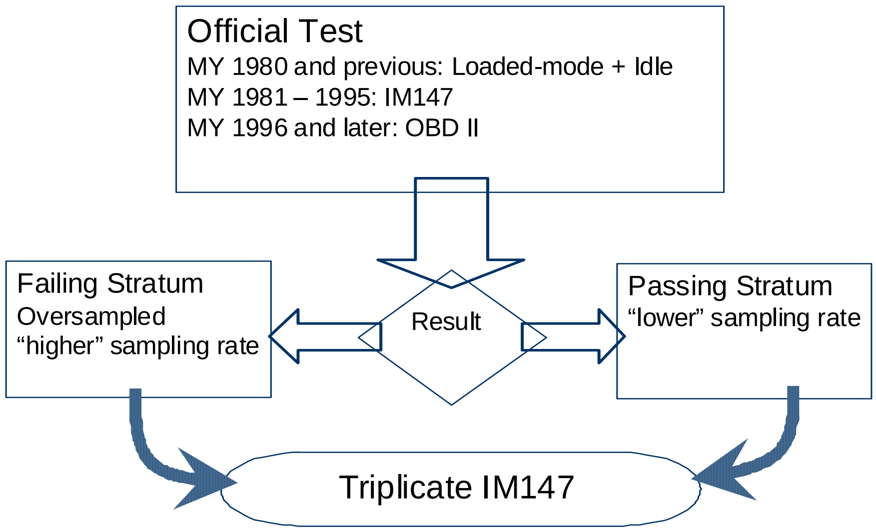
Based on test records, reconstructing sampling rates simply involved dividing the numbers of sampled vehicles by the total numbers of vehicles tested, by model year and calendar year, for failing (f) and passing (p) strata, as shown in Equation 1 - 3.

$$f_{\mathrm{f,MY,CY}} = \frac{n_{\mathrm{f,MY,CY}}}{N_{\mathrm{f,MY,CY}}} \qquad f_{\mathrm{p,MY,CY}} = \frac{n_{\mathrm{p,MY,CY}}}{N_{\mathrm{p,MY,CY}}} \tag{1 - 3}$$

Corresponding sampling weights indicate the numbers of vehicles in the general fleet represented by each sample vehicle. They were derived as the reciprocals of the sampling fractions, as shown in Equation 1 - 4.

$$w_{\mathrm{f,MY,CY}} = \frac{1}{f_{\mathrm{f,MY,CY}}} \qquad w_{\mathrm{p,MY,CY}} = \frac{1}{f_{\mathrm{p,MY,CY}}} \tag{1 - 4}$$

Figure 1 - 1. Stratified Sampling as applied in selection of the Random Evaluation Sample in the Phoenix I/M Program (CY 2002-05)



1.5.3 Source selection

After excluding the St. Louis dataset, and comparing the Phoenix, Chicago and NY datasets, analysis, we elected to rely on the Phoenix dataset for purposes of rate estimation and to use the other datasets, including selected remote-sensing data, for purposes of comparison. This course was chosen for several reasons.

For our purposes, the greater historic depth of the Phoenix data was a tremendous advantage. It was the only set deep enough to allow direct and independent assessment of deterioration. The limited depth of the other datasets would have meant that the subset of calendar years that could be covered by pooled data would have been relatively limited. Only a single calendar year, 2002, is covered by all three datasets. Several years would be covered by two out of three. Calendar 1999 is covered by Phoenix and NY; 2000 and 2001 would have been covered by NY and Chicago, and 2003 and 2004 by Chicago and Phoenix. The remaining years, 1996-98 and 2005 could have been covered only by Phoenix in any case.

In addition, pooling the three datasets would have involved several difficult technical issues. Table 1 - 6 shows that the datasets were of strongly differing sizes. Thus, if the datasets were pooled without any type of relative weighting, Phoenix would have exerted much stronger influence than the others in most shared calendar years. To rectify disparities in influence by assigning the different datasets similar or proportional influence would have required development of some sort of a weighting scheme, but a rational basis for such relative weighting is not immediately apparent.

The question of pooling is further complicated by the fact that use of the Phoenix data collected in CY 2002 to 2005 requires use of sampling weights for passing and failing tests (as discussed), whereas the Chicago and NYIPA datasets are assumed to be self-weighting. Again, no rational basis for incorporating weighted and self-weighted tests from various programs in the same CY was immediately apparent.

1.5.4 Methods

1.5.4.1 Data Driven Rates

Where data was present, the approach was simple. We calculated means and other summary statistics for each combination of source and operating mode (i.e., table cell). We classified the data by regulatory class (LDV, LDT), model-year group, age group and operating mode (Table 1). The model-year groups used are shown in Table 7, along with corresponding samples of passing and failing tests.

Table 1 - 7. Test sample sizes for the Phoenix random-evaluation sample.

Model-year group[b]	**LDV**		**LDT**	
	fail[a]	*pass*	*fail*	*pass*
1981-82	562	539	340	495
1983-85	1,776	2,078	1,124	1,606
1980-89	3,542	6,420	1,745	3,698
1990-93	2,897	8,457	1,152	4,629
1994-95	997	4,422	703	3,668
1996-98	1,330	3,773		
1996			526	1,196
1997-98			858	2,320
1999-2000	176	753	136	624
Total	11,285	26,478	6,589	18,254

[a] Note that 'failure' can indicate failure for CO, HC or NOx, as applicable.
[b] Note that these are the model-year groups used for analysis; NOT the model-year groups used in the MOVES database.

We calculated means and other summary statistics for each combination of sourceBinID, ageGroupID and opModeID. For simplicity, we will refer to a specific combination of sourceBinID, and opModeID as a "*cell*," to be denoted by label '*h*'.

1.5.4.1.1 Rates: Calculation of weighted means

The emission rate (meanBaseRate) in each cell is a (E_h) simple weighted mean

$$E_h = \frac{\sum_{i=1}^{n_{\text{test}}} w_i R_{i,t}}{\sum_{i=1}^{n_{\text{test}}} w_i} \qquad \textbf{1 - 5}$$

where w_i is a sampling weight for each vehicle in the cell, as described above, and $R_{i,t}$ is the "second-by-second" emission rate in the cell for a given vehicle at a given second t.

1.5.4.1.2 Estimation of Uncertainties for Cell Means:

To estimate sampling error for each cell, we calculated standard-errors by weighted variance components. In estimating variances for cell means, we treated the data within cells as effective cluster samples, rather than simple random samples. This approach reflects the structure of the data, which is composed of sets of multiple measurements collected on individual vehicles. Thus, measurements on a specific vehicle are less independent of other measurements on the same vehicle than of measurements on other vehicles. Accordingly, means and variances for individual vehicle tests were calculated to allow derivation of between-test and within-test variance components. These components were used in turn to calculate the variance of the mean for each cell, using the appropriate degrees of freedom to reflect between-vehicle variability[5]. To enable estimation of variances under this approach, we calculated a set of summary statistics, as listed below:

Test mean (E_i): the arithmetic mean of all measurements in a given test on a specific vehicle in a given cell.

Test sample size (n_h), the number of individual vehicle tests represented in a cell.

Measurement sample size (n_i): the number of measurements in a cell representing an individual test on an individual vehicle.

Cell sample size ($n_{h,i}$): the number of individual measurements in a cell, where each count represents a measurement collected at an approximate frequency of 1.0 Hz, (i.e., "second-by-second).

Test variance (s_i^2): the variance of measurements for each vehicle test represented in a cell, calculated as the average squared deviation of measurements for a test about the mean for that test. Thus, we calculated a separate test variance for each test in each cell.

Weighted Between-Test variance component (s_{b}^2): the component of total variance due to variability among tests in a cell, or stated differently, the weighted variance of the test means about the cell mean, calculated as

$$s_{\mathrm{b}}^2 = \frac{\sum_{i=1}^{n_i} w_i \left(E_i - E_h\right)^2}{\sum_{i=1}^{n_i} w_i - 1} \quad \textbf{1 - 6}$$

Weighted Within-Test Variance Component (s_{w}^2): the variance component due to variability within tests, or the variance of measurements within individual tests ($R_{i,t}$) about their respective test means, calculated in terms of the test variances, weighted and summed over all tests in the cell:

$$s_{\mathrm{w}}^2 = \frac{\sum_{i=1}^{n_h} w_i \left(n_i - 1\right) s_i^2}{\left(\sum_{i=1}^{n_h} w_i\right)\left(n_{h,i} - n_h\right)} \quad \textbf{1 - 7}$$

Variance of the cell mean ($s_{\bar{E}}^2$): this parameter represents the uncertainty in the cell mean, and is calculated as the sum of the between-vehicle and within-test variance components, with each divided by the appropriate degrees of freedom.

$$s_{E_h}^2 = \frac{s_{\mathrm{b}}^2}{n_h} + \frac{s_{\mathrm{w}}^2}{n_{h,i}} \quad \textbf{1 - 8}$$

Coefficient-of-Variation of the Mean (CV_{Eh}): this parameter gives a relative measure of the uncertainty in the cell mean, allowing comparisons among cells. It is calculated as the ratio of the cell standard error to the associated cell mean

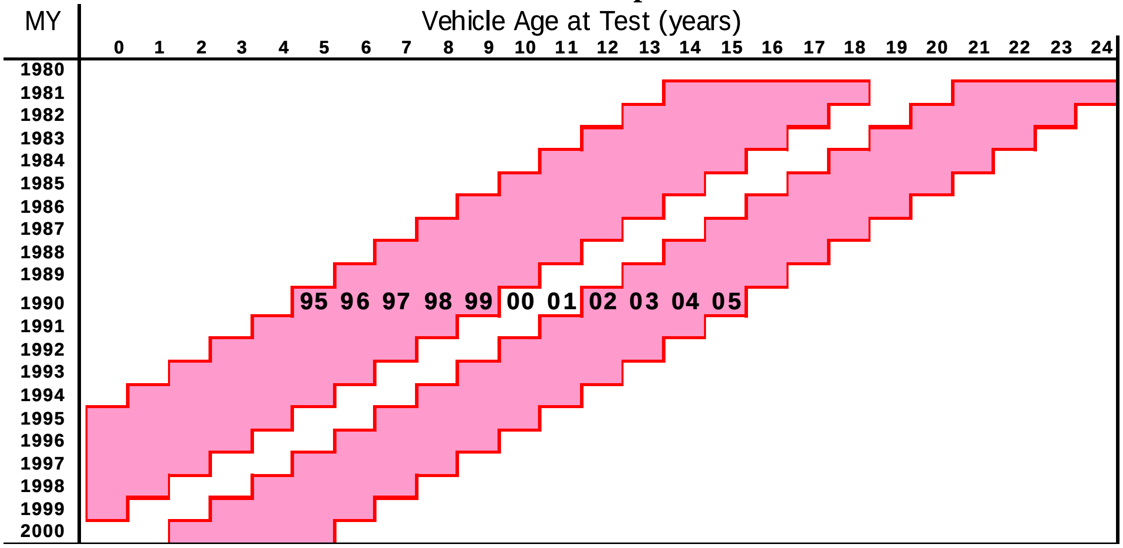
$$\mathrm{CV}_{E_h} = \frac{\sqrt{s_{E_h}^2}}{E_h} \quad \textbf{1 - 9}$$

Note that the term CV_E is synonymous with the term “relative standard error” (RSE).

1.5.4.2 Model-generated Rates (hole-filling)

Following averaging of the data, it was necessary to impute rates for cells for which no data was available, i.e., “holes.” Empty cells occur for age Groups not covered by available data (Figure 1 - 2). In the figure, “age holes” are represented by unshaded areas. Filling in these un-shaded areas required “hind-casting” emissions for younger vehicles for older model years, as “forecasting” deterioration of aging vehicles for more recent model years. Empty cells occur as well in high-power operating modes not covered by the IM147 or IM240, meaning operating modes with power greater than about 24 kW/tonne.

Figure 1 - 2. Model-year by Age Structure of the Phoenix I/M Random Evaluation Sample.



1.5.4.2.2 Rates

To estimate rates in empty cells (holes), we constructed statistical models of emissions data to extrapolate trends in VSP and age. For this purpose, we generated a series of models based on the MOVES operating-mode/ageGroup structure.

As a preliminary step, data were averaged for each test within a set of classes for VSP and speed. We averaged emissions by model-year-group, regClass, age, VSP class, speed class and test. Classes for VSP followed intervals of 3.0 kW/tonne (e.g., 0-3, 3-6, … 27-30, 30+). Speed classes followed those used for the MOVES operating modes (e.g., 1-25 mph, 25-50 mph, 50+ mph). The resulting dataset had a single mean for each test in each 6-way cell. The purpose for this averaging was to give the resulting statistical model an appropriate number of degrees of freedom for each of the class variables, i.e., the d.f. would be determined by the number of tests rather than the number of individual "second-by-second" measurements. Note that the matrix used for this purpose was finer than that represented in Table 1 - 1.

We fit separate models in three groups of operating modes. For all operating modes except brake/deceleration and idle, we fit one model that incorporates VSP. We call this group "coast/cruise/acceleration." For braking/deceleration and idle, we fit two additional models not incorporating VSP, as these modes are not defined in terms of VSP (Table 1). Overall, we fit three models for each combination of LDV and LDT, for the model-year groups shown in Table 7, giving a total of 60 models.

Before fitting a model, we drew a sample of vehicle tests in each model-year group (*n* = 1,200 to 3,500, see Table 8). This sampling was performed to fit model on smaller amount of data that a standard desktop computer could handle. The sample was stratified by test result and age with allocation proportional to that of the total sample.

Table 1 - 8. Sample sizes for statistical modeling, by regulatory class and test result

Model-year group	**LDV**		**LDT**	
	fail	*pass*	*fail*	*pass*
1980 & earlier				
1981-82				
1983-85				
1980-89				
1990-93				
1994-95				
1996-98	663	1,738		
1996			346	854
1997-8			671	1,730
1999-2000	247	954		

Each model included two sub-models, one to estimate means and one to estimate variances, as described below.

1.5.4.2.2.1 Coast/Cruise/Acceleration

Means model

For the means sub-model, the dependent variable was the natural logarithm of emissions

$$\ln E_h = \beta_0 + \beta_1 P_{\mathrm{V}} + \beta_2 P_{\mathrm{V}}^2 + \beta_3 P_{\mathrm{V}}^3 + \beta_4 a + \beta_5 \mathrm{v} + \beta_6 P_{\mathrm{V}} \mathrm{v} + \gamma_7 t_i + \varepsilon \qquad \textbf{1 - 10}$$

where :

- $\ln E_h$ = natural-logarithm transform of emissions (in cell h),
- P_{V}, P_{V}^2, P_{V}^3 = first-, second- and third-order terms for vehicle-specific power (kW/tonne),
- a = vehicle age at time of test (years),
- s = speed class (1 -25 mph, 25-50 mph and 50+ mph),
- t = test identifier (random factor)
- ε = random or residual error

- β = regression coefficients for the intercept and fixed factors p, a and s.
- γ = regression coefficients for the random factor *test*.

The model includes first-, second- and third-order terms in P_{V} to describe curvature in the power trend, e.g., enrichment for CO and the corresponding decline in NOx at high power. The age term gives an ln-linear trend in age. The speed-class term allows for a modified intercept in each speed class, whereas the power/speed-class interaction allows slightly different power slopes in each speed class. The random factor term for test fits a random intercept for each test, which does not strongly affect the mean estimates but does affect the estimation of uncertainties in the coefficients.

After fitting models, we performed basic diagnostics. We plotted residuals against the two continuous predictors, VSP and age. We checked the normality of residuals across the range of VSP and age, and we plotted predicted vs. actual values.

Variances model

The purpose of this sub-model was to model the variance of $\ln E_h$, i.e., the logarithmic variance s_l^2, in terms VSP and age. To obtain a dataset of replicate variance estimates, we drew sets of replicate test samples. Each replicate was stratified in the same manner as the larger samples (**Table 1 - 8**). To get replicate variances, we calculated ln-variance for each replicate within the VSP/age matrix described above.

Models were fit on set of replicate variances thus obtained. The dependent variable was logarithmic variance

$$s_l^2 = \alpha_0 + \alpha_1 a + \alpha_2 P_{\mathrm{V}} + \alpha_3 P_{\mathrm{V}} a + \varepsilon \qquad \textbf{1 - 11}$$

where p and a are VSP and age, as above, and α are regression coefficients. After fitting we examined similar diagnostics as for the means model.

Model application

Application of the model was simple. The first step was to construct a cell matrix including all emission rates to be calculated, as shown in **Table 1 - 9**.

Table 1 - 9. Construction of emission-rate matrix for light-duty gasoline vehicles

	Count	Category	MOVES Database attribute
	1	Fuel (gasoline)	fuelTypeID = 01
×	2	Regulatory Classes (LDV, LDT)	regClassID = 20, 30
×	10	Model-year groups[1]	
×	21	Operating modes	opModeID = 11-16, 21-30, 33-40
×	7	Age Groups	ageGroupID = 3, 405, 607, 809, 1014, 1519, 2099
×	3	Pollutant processes (running HC, CO, NOx)	polProcessID = 101, 201, 301
=	9,660	TOTAL cells	

Next, we constructed a vector of coefficients for the means sub-model (**β)** and merged it into the cell matrix.

$$\boldsymbol{\beta} = \lfloor \beta_0 \ \beta_1 \ \beta_2 \ \beta_3 \ \beta_4 \ \beta_{5(0-25)} \ \beta_{45(25-50)} \ \beta_{5(50+)} \ \beta_6 \ \rfloor \qquad \textbf{1 - 12}$$

Then, for each table cell, we constructed a vector of predictors ($\mathbf{X}_h$). Equation 13 shows an example for an operating mode in the 1 – 25 mph speed class, e.g., the value for the 1-25 mph class is 1 and the values for the 25-50 and 50+ speed classes are 0. To supply values for VSP (P_V) and age group (a), cell midpoints were calculated and applied as shown in Table 10.

$$\mathbf{X}_h = \lfloor 1 \ \ P_V \ \ P_V{}^2 \ P_V{}^3 \ a \ 1 \ \ 0 \ \ 0 \ \ P_V \rfloor \qquad \textbf{1 - 13}$$

Table 1 - 10. Values of VSP used to apply statistical models

opModeID	**Range**	**Midpoint**
11, 21	< 0	-2.0
12, 22	0 - 3	-2.5
13, 23	3 - 6	4.5
14, 24	6 - 9	7.5
15, 25	9 - 12	10.5
16	12 +	14.5
27,37	12 - 18	15.0
28,38	18 - 24	21.0
29,39	24 - 30	27.0
30	30 +	34.0
40	30 +	32.0
33	< 6	0.5
35	6 - 12	9.0

The final step was to multiply coefficient and predictor vectors, which gives an estimated logarithmic mean ($\ln E_h$) for each cell h.

$$\ln E_h = \mathbf{X}'_h \boldsymbol{\beta} \qquad \textbf{1 - 14}$$

The application of the variances model is similar, expect that the vectors have four rather than nine terms

$$\boldsymbol{\alpha} = \left[\alpha_0 \ \alpha_1 \ \alpha_2 \ \alpha_3 \ \right] \qquad \textbf{1 - 15}$$

$$\mathbf{X}_h = \left[1 \quad P_{\mathrm{V}} \quad a \quad P_{\mathrm{V}} a\right] \qquad \textbf{1 - 16}$$

Thus, the modeled logarithmic variance in each cell is given by

$$s_{l,h}^2 = \mathbf{X}_h \boldsymbol{\alpha} \qquad \textbf{1 - 17}$$

In some model-year groups, it was not always possible to develop plausible estimates for the age slope β_4, because the data did not cover a wide enough range of calendar years. For example, in the 99-00 model-year group, the available data represented young vehicles without sufficient coverage of older vehicles. We considered it reasonable to adapt the age slope for the 96-98 model-year group for LDV, and the 1997-98 model-year group for LDT.

In the groups 83-85 and 81-82, the data covered vehicles at ages of 10 years and older but not at younger ages. Simply deriving a slope from the available data would give values that were much too low, resulting in very high emissions for young vehicles. In these cases we considered it more reasonable to adopt an age slope from a subsequent model year group. When making this assumption, it is necessary to recalculate the intercept, based on the assumed slope and the earliest available data point.

Intercepts were recalculated by rearranging Equation 1 - 10 to evaluate the model in operating mode 24, using the age slope from the previous model-year group (β_4*) an estimate of ln-emissions from the available dataset at the earliest available age ($\ln E_{a*}$) at age a*. In operating mode 24, the midpoint of the VSP range (6-9) is 7.5 kW/tonne and the speed class is 25-50 mph.

$$\beta_0^* = \ln E_{a*} - 7.5\beta_1 - 7.5^2\beta_2 - 7.5^3\beta_3 - \beta_4 a * - \beta_{5(25-50)} - 7.5\beta_6 \qquad 1 - 18$$

On a case by case basis, age slopes were adopted from earlier or later model-year groups. In a similar way, ln-variance models or estimates could be adopted from earlier or later model years.

1.5.4.2.2.2 Braking/Deceleration

Means model

We derived models similar to those one used for coast/cruise/acceleration. For these operating modes, the models were much simpler, in that they did not include VSP or the speed classes used to define the coast/cruise/accel operating modes. Thus, emissions were predicted solely in terms of age, although random intercepts were fit for each test as before:

$$\ln E_h = \beta_0 + \beta_1 a + \gamma_7 t_i + \varepsilon \qquad \textbf{1 - 19}$$

Variances model

In addition, we fit variances models for these operating modes, which were also simple functions of age.

$$s_l^2 = \alpha_0 + \alpha_1 a + \varepsilon \qquad \textbf{1 - 20}$$

Model application

In these operating modes, rates were to be modeled for a total of 840 cells. This total is calculated as in **Table 1 - 9**, except that the number of operating modes is 2, rather than 21. We set up coefficient and predictor vectors, as before.

For the means model the vectors are

$$\boldsymbol{\beta} = [\beta_0 \ \beta_1 \ ] \qquad \textbf{1 - 21}$$

and

$$\mathbf{X}_h = [1 \quad a \ ] \qquad \textbf{1 - 22}$$

respectively.

For the variances model the coefficients vector is

$$\boldsymbol{\alpha} = [\alpha_0 \ \alpha_1 \ ] \qquad \textbf{1 - 23}$$

and the predictor vector is identical to that for the means model.

As with CCA modes, we considered it reasonable in some model-year groups to adopt a slope or ln-variance from a previous or later model-year group. In model-year groups where the purpose was to hindcast rates for younger vehicles, rather than forecast rates for aging vehicles, it was again necessary to recalculate the intercept based on a borrowed age slope and an estimate of lnE_h calculated from the sample data for the youngest available age class. In this case, equation 24 is a rearrangement of Equation 1 - 19.

$$\beta_0^* = \ln E_{a*} - \beta_4 a * \qquad \textbf{1 - 24}$$

After these steps, the imputed values of lnE_h were calculated, as in Equations 1 - 14 and 1 - 17.

1.5.4.2.3 Estimation of Model Uncertainties

We estimated the uncertainty for each estimated lnE_h in each cell. During each model run, we saved the covariance matrix of the model coefficients ($\boldsymbol{s_\beta}^{\mathbf{2}}$). This matrix contains covariances of each of the nine coefficients in relation to the others, with the diagonal containing variances for each coefficient.

$$\mathbf{s}_{\boldsymbol{\beta}}^{\mathbf{2}} = \begin{bmatrix} \sigma_0^2 & . & . & . & \sigma_{0,4}^2 & . & . & . & \sigma_{0,6}^2 \\ . & \sigma_1^2 & . & . & . & . & . & . & . \\ . & . & \sigma_2^2 & . & . & . & . & . & . \\ . & . & . & \sigma_3^2 & . & . & . & . & . \\ \sigma_{4,0}^2 & . & . & . & \sigma_4^2 & . & . & . & \sigma_{0,4}^2 \\ . & . & . & . & . & \sigma_{5(0-25)}^2 & . & . & . \\ . & . & . & . & . & . & \sigma_{5(25-50)}^2 & . & . \\ . & . & . & . & . & . & . & \sigma_{5(50+)}^2 & . \\ \sigma_{6,0}^2 & . & . & . & \sigma_{6,4}^2 & . & . & . & \sigma_6^2 \end{bmatrix}. \qquad \mathbf{1\text{ - }25}$$

Using the parameter vectors $\mathbf{X}_{\boldsymbol{h}}$ and the covariance matrix $\boldsymbol{s_\beta}^{\mathbf{2}}$, the standard of error of estimation for each cell was calculated as

$$s_{\ln E_h}^2 = \mathbf{X}_h^{'} \mathbf{s}_{\boldsymbol{\beta}}^{\mathbf{2}} \mathbf{X}_h \qquad \mathbf{1\text{ - }26}$$

The standard error of estimation in each cell represents the uncertainty of the mean estimate in the cell, based on the particular values of the predictors defining the cell[6]. The pre- and post-multiplication of the covariance matrix by the parameter vectors represents the propagation of uncertainties, in which the parameters represent partial derivatives of each coefficient with respect to all others and the covariances represent the uncertainties in each coefficient in relation to itself and the others.

1.5.4.2.4 Reverse transformation

To obtain an estimated emission rate E_h in each cell, the modeled means and variances are exponentiated as follows

$$E_h \;=\; \mathrm{e}^{\ln E_h} \mathrm{e}^{0.5 s_{l,h}^2} \qquad \mathbf{1\text{ - }27}$$

The two exponential terms use the results of the means and variances sub-models, respectively (Equations 1 - 6 and 1 - 7). The left-hand “means” term represents the geometric mean, or the center of the implied log-normal distribution, whereas the right-hand “variance” term reflects the influence of the “high-emitting” vehicles representing the tail of the distribution.

The estimate of ln-variance could be obtained in several different ways. The first and preferred option was to use the modeled variance as described above. A second option was to use an estimate of variance calculated from the available sample of ln-transformed data. A third option, also based on available data, was an estimate calculated from averaged emissions data and the mean and variance of ln-transformed emissions data. This process involves reversing Equation 1 - 19 to solve for s_l^2. If the mean of emissions data is x_a and mean of ln-transformed data is x_l, then the logarithmic variance can be estimated as

$$s_l^2 = 2\ln\left(\frac{\bar{x}_a}{e^{x_l}}\right) \qquad \textbf{1 - 28}$$

In practice one of these options was selected based which most successfully provided model estimates that matched corresponding means calculated from the data sample.

The uncertainties mentioned above represent uncertainties in $\ln E_h$. Corresponding standard errors for the reverse-transformed emission rate E_h were estimated numerically by means of a Monte-Carlo process. At the outset, we generated a pseudo-random set of 100 variates of $\ln E_h$, based on a normal distribution with a mean of 0.0 and variance equal to $s_{\ln E}^2$. We applied Equation 1 - 28 to reverse-transform each variate, and then calculated the variance of the reverse-transformed variates. This result represented the variance-of-the-mean for E_h ($s_{E_h}^2$), as in Equation 1 - 8. Finally, we calculated the CV-of-the-mean (CV_{Eh}) for each modeled emission rate, as in Equation 1 - 9.

1.5.4.3 Table Construction

After compilation of the modeling results, the subset of results obtained directly from the data (Equations 1 - 4 to 1 - 9), shaded area in Figure 1 - 1) and the complete set generated through modeling (Equations 1 - 10 to 1 - 28) were merged. A final value was selected for use in the model data table. The value generated from data was retained if two criteria were met: (1) a subsample of three or more individual vehicles must be represented in a given cell ($n_h \geq 3$), and (2) the CV_{Eh} (relative standard error, RSE) of the data-driven E_h must be less than 50% ($CV_{E_h} < 0.50$). Failing these criteria, the model-generated value was substituted. For purposes of illustration, results of both methods are presented separately.

At this point, we mapped the analytic model-year groups onto the set of model-year groups used in the MOVES database. The groups used in the database are designed to mesh with heavy-duty standards and technologies, as well as those for light-duty vehicles. To achieve the mapping, we replicated records as necessary, in cases where the analytic group was broader than the database group. Both sets of groups are shown in Table 11.

Table 1 - 11 Mapping 'Analytic" Model-Year Groups onto MOVES database Model-Year Groups.

"Analytic"		"MOVES database"
LDV	***LDT***	
1981-82	1981-82	1980 and previous
1981-82	1981-82	1981-82
1983-85	1983-85	1983-84
1983-85	1983-85	1985
1986-89	1986-89	1986-87
1986-89	1986-89	1988-89
1990-93	1990-93	1990
1990-93	1990-93	1991-1993
1994-95	1994-95	1994
1994-95	1994-95	1995
1996-98	1996	1996
1996-98	1997-98	1997
1996-98	1997-98	1998
1999-2000	1999-2000	1999
1999-2000	1999-2000	2000

1.5.5 Verification and Adjustment for High-Power Operating modes

The rates described were derived from data measured on the IM240 or IM147, which are limited in terms of the ranges of speed and vehicle-specific power that they cover. Specifically, these cycles range up to about 50 mph and 24 kW/tonne for speed and VSP, respectively. Some coverage does exist outside these limits but can be sporadic and highly variable. The operating modes outside the I/M window include modes 28,29,30, 38, 39 and 40, which we'll refer to as the 'high-power' operating modes. For these modes, the statistical models described above were used to extrapolate up to about 34 kW/tonne.

Based on comments from members of the FACA MOVES Review Workgroup, we thought it advisable to give additional scrutiny to the high power extrapolation. To obtain a framework for reference, we examined a set independently measured data, collected on drive cycles more aggressive than the IM cycles, namely, the US06 and the "Modal Emissions Cycle" or "MEC." Much of the data was collected in the course of the National Cooperative Highway Research Program (NCHRP) and the remainder on selected EPA programs, all stored in OTAQ's Mobile-Source Observation Database (MSOD). Unlike the US06, which was designed specifically to capture speed and acceleration not captured by the FTP, the MEC is an engineered cycle, designed not to be representative of any specific driving pattern but rather to exercise vehicles through a wide range of speed and acceleration. Driving traces for both cycles are shown in Figures 3 and 4. Both cycles range in speed up to over 70 mph and in VSP up to and over 30 kW/tonne.

Figure 1 - 3. Example Speed Traces for the US06 and MEC Drive Cycles.

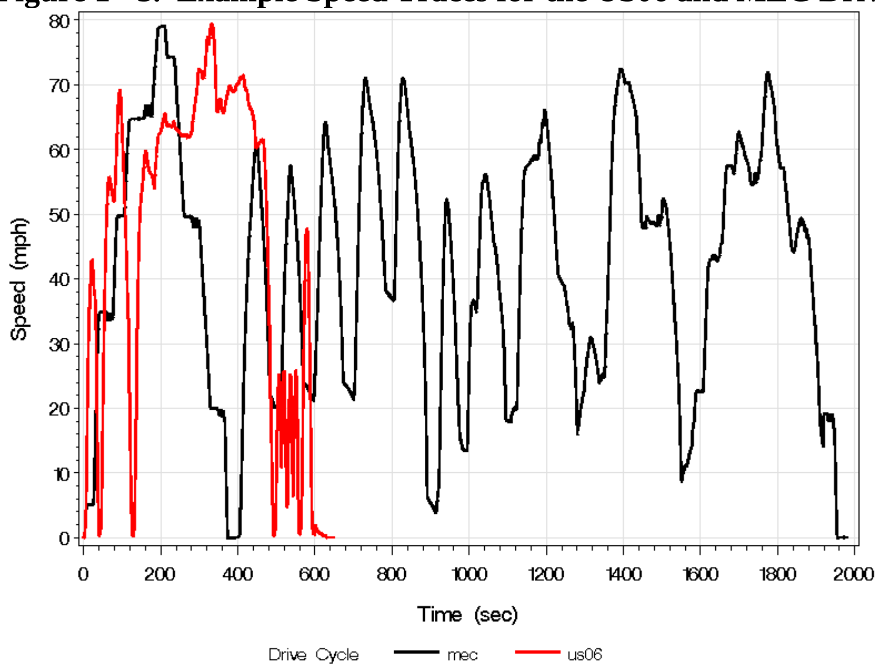


Figure 1 - 4. Example VSP Traces for the US06 and MEC Cycles

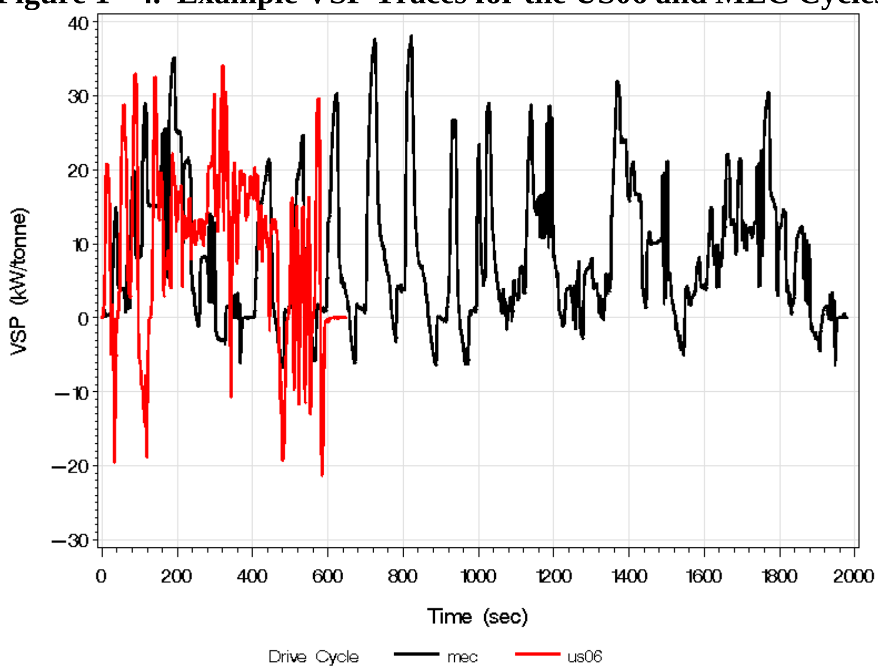


Table 12 summarizes the numbers of available tests by regulatory class, model-year group and drive cycle. had a certain number of tests, as shown in Table 12. Samples were somewhat larger for LDV for both cycles, which represented a broad range of model-years.

Table 1 - 12. Sample Sizes for US06 and MEC Samples (No. tests)

Model-year group	LDV		LDT		Total
	US06	***MEC***	***US06***	***MEC***	
1980 & earlier	4	14		6	**24**
1981-85	15	23	8	19	**65**
1986-89	21	24	13	31	**89**
1990-93	54	57	22	36	**169**
1994-95	49	45	22	30	**146**
1996-99	58	28	56	17	**159**
Total	**201**	**191**	**121**	**139**	**652**

Figure 1 - 5 to Figure 1 - 7 show trends in emissions vs. VSP for CO, HC and NOx for LDV and LDT by model year group. Both cycles were averaged and plotted as aggregates.

Figure 1 - 5. CO emissions (g/ssec) on Aggressive Cycles vs. VSP, by Regulatory Class and Model-year Group.

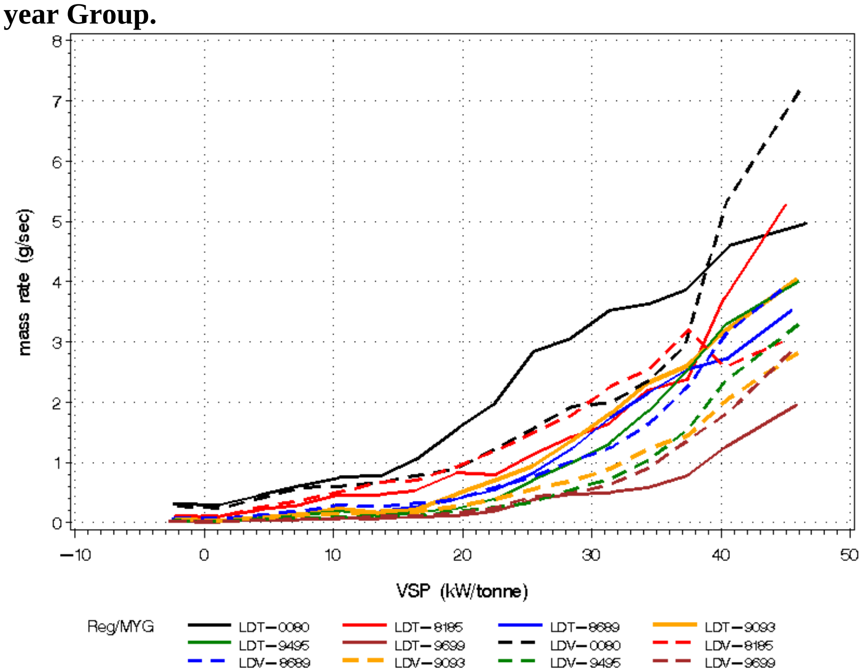


Figure 1 - 6 THC Emissions (g/sec) on aggressive Cycles vs. VSP, by Regulatory Class and Model-year Group.

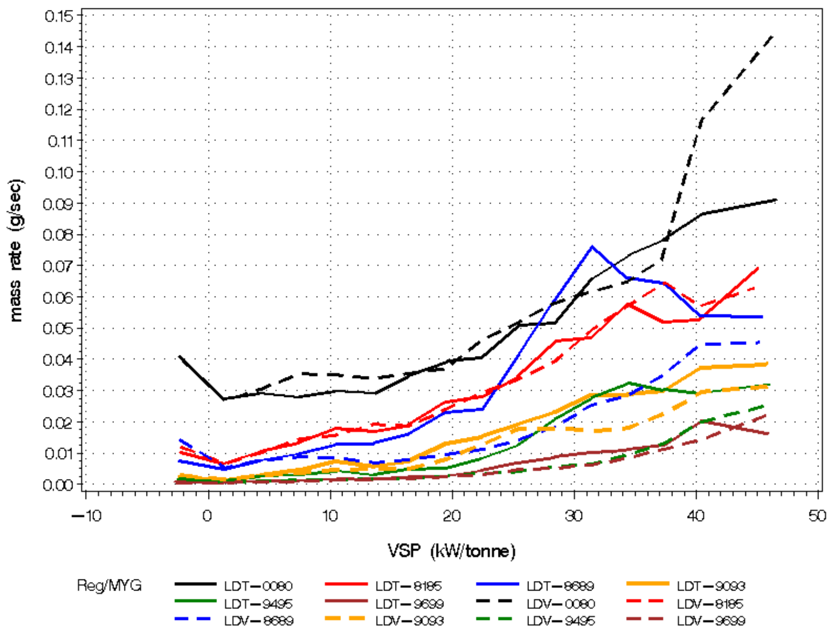
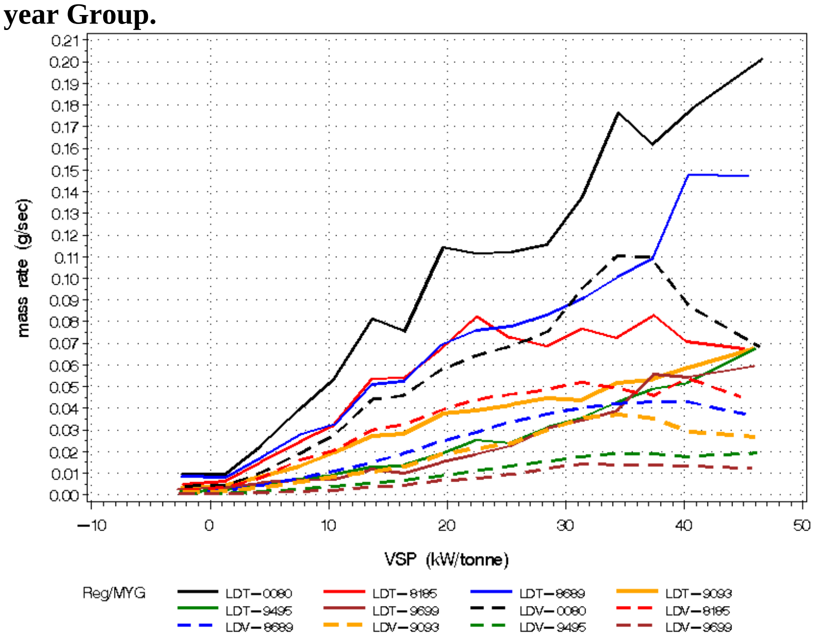


Figure 1 - 7. NOx Emissions (g/sec) on Aggressive Cycles vs. VSP by Regulatory Class and Model-year Group.



To construct a basis for reference, we averaged the data by regulatory class, model-year group and operating mode, using the model-year groups shown in Table 1 - 12. After averaging, we calculated ratios frrm high-power operating modes to a selected reference mode. Specifically, we selected two modes covered by the IM cycles (27 and 37) to serve as reference points. The midpoint VSP for each is ~15 kW/tonne. With mode 27 as a reference, we calculated ratios to modes 28, 29 and 30

$$R_{i:27} = \frac{E_{h,i}}{E_{h,27}}, \text{ for } i = 28, 29, 30 \qquad \textbf{1 - 29}$$

and with mode 37 as a reference, we calculated ratios to modes 38, 39 and 40.

$$R_{i:37} = \frac{E_{h,i}}{E_{h,37}}, \text{ for } i = 38, 39, 40 \qquad \textbf{1 - 30}$$

Calculating uncertainties in the ratios was an important step. If the ratio *R* is calculated as a numerator divided by a denominator (*N*/*D*), the variance in the ratio is propagated by summing the products of the squared partial derivatives of *R* to *N* and *D* and the variances of their respective means.

$$s_R^2 = \left(\frac{\partial R}{\partial N}\right)^2 s_N^2 + \left(\frac{\partial R}{\partial D}\right)^2 s_D^2 \qquad \textbf{1 - 31}$$

Note that this expression contains only variances and neglects potential covariances between *N* and *D*. In the case of *N/D* the partial derivatives, which express the sensitivity of the ratio to each, are simply calculated as

$$\frac{\partial R}{\partial N} = \frac{1}{D} \quad \text{and} \quad \frac{\partial R}{\partial D} = \frac{-N}{D^2} \qquad \textbf{1 - 32}$$

To apply the ratios to operating modes 28-30 and 38-40, we calculated ratio-based emissions estimates (E^R) as the products of their respective ratios and the initial rate for modes 27 or 37

$$E_{h,i}^{R} = R_{i:27} E_{h,27}^{initial}, \text{ or } E_{h,i}^{R} = R_{i:37} E_{h,37}^{initial} \qquad \textbf{1 - 33}$$

respectively, where $E_h^{initial}$ is the initial data-driven or model-generated rate calculated as previously described.

We used the variances of the ratios to calculate upper and lower confidence limits on the ratio-based rates.

$$\begin{aligned} \text{LCL}_i^R &= E_i^R - t_{0.80} s_{R,i} \\ \text{UCL}_i^R &= E_i^R + t_{0.80} s_{R,i} \end{aligned} \qquad \textbf{1 - 34}$$

In applying the confidence band as an evaluation criterion, each of the high-power operating modes *i*, the initial value $E^{initial}$ was a candidate for replacement by E^R if it fell outside the 80% confidence band of E^R, or

$$E_i^{initial} < \text{LCL}_i^R \;\; \text{OR} \;\; E_i^{initial} > \text{UCL}_i^R \qquad \textbf{1 - 35}$$

Due to volatility in the ratios, the confidence limits were quite wide in some cases. After initially calculating and evaluating a 95% confidence band, we settled on using a somewhat narrower 80% band, for the reason that it was more sensitive in identifying implausible values of $E^{initial}$, whether high or low.

We present some examples below. In the THC example (Figure 1 - 8), the initial rates fall outside the confidence intervals for the ratio-based rates for three out of six possible cases, i.e., in modes 30, 39 and 40. The resulting rate is higher for modes 30 and 40, but lower for 39. The example for CO is different (Figure 1 - 9). The initial values for modes 28-30 all fall within the confidence intervals and are thus retained. The values for 39 and 40, fall outside the band on the low side and are replaced by the ratio-based rates. Finally, in the NOx example (Figure 1 - 10), the initial rates are replaced in five out of six cases. The initial values for 28-30 and 40 all fall below the LCL, whereas that for 30 falls above the UCL.

Figure 1 - 8. THC emission rates (g/hr) vs. Operating Mode for MY-1998 LDV at ages 6-7: initial (data or statistical model) and calculated by ratio based on aggressive cycles.

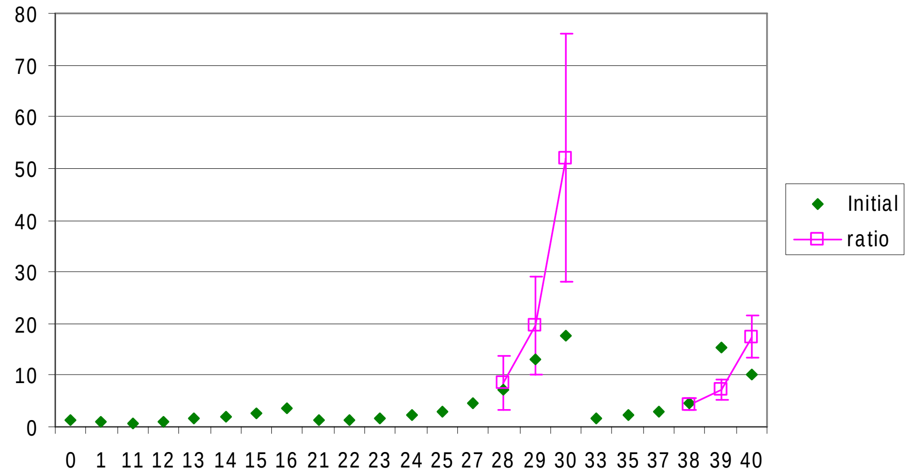


Figure 1 - 9. CO emission rates (g/hr) vs. operating mode for MY-1998 LDT at ages 6-7: initial (data or statistical model) and calculated by ratio based on aggressive cycles.

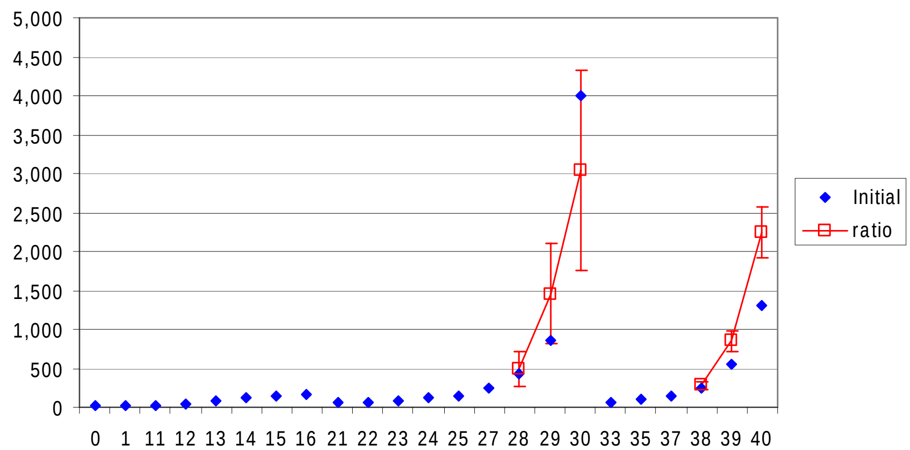
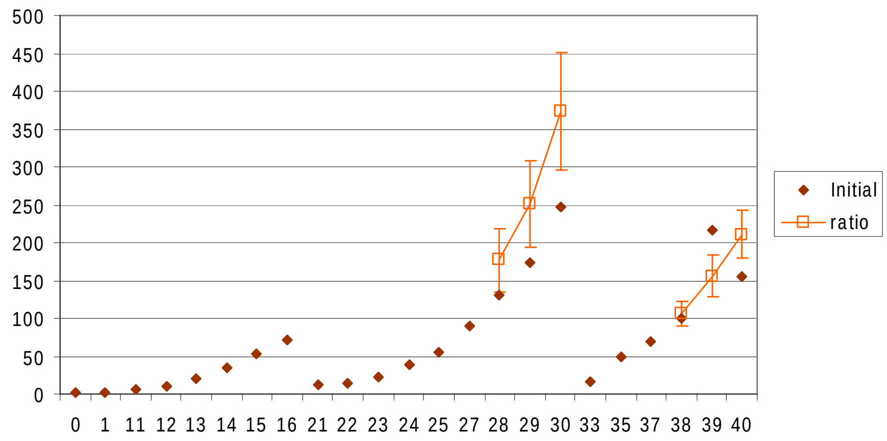


Figure 1 - 10. NOx Emission Rates (g/hr) vs. Operating Mode, for MY1995 LDV at ages 8-9: initial (data + statistical model) and calculated by ratio based on aggressive cycles.



1.5.6 Estimating Rates for non-I/M Areas

In modeling emission inventory for light-duty vehicles, it is necessary at the outset to consider the question of the influence of inspection and maintenance (I/M) programs. In this regard a fundamental difference between MOVES and MOBILE is that MOVES inverts MOBILE's approach to representing I/M. In MOBILE, the emission rates stored in the input data tables represent non-I/M conditions. During a model run, as required, emissions for I/M conditions are modeled relative to the original non-I/M rates.

In MOVES, however, two sets of rates are stored in the input table (emissionRateByAge). One set represents emissions under "I/M conditions" (meanBaseRateIM) and the other represents rates under "non-I/M conditions" (meanBaseRate). The first set, representing vehicles subject to I/M requirements, we call the "I/M reference rates". The second, representing vehicles not subject to I/M requirements, we call the "non-I/M reference rates."

For the I/M reference rates, the term "reference" is used because the rates represent a particular program, with a specific design characteristics, against which other programs with differing characteristics can be modeled. Thus, the I/M references are, strictly speaking, regional rates, and not intended to be (necessarily) nationally representative. Development of the I/M reference rates is discussed above in sections 1.5.2 and 1.5.3. As the I/M references represent Phoenix, the program characteristics implicitly reflected in them include:

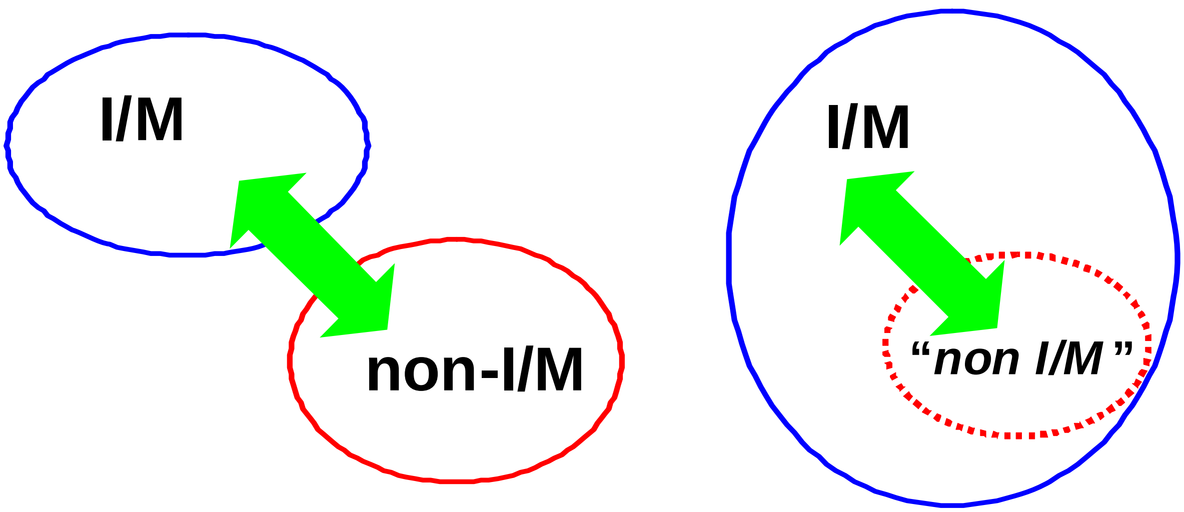
- A four-year exemption period,
- transient tailpipe tests for MY 81-95,
- OBD-II for MY 96+,
- Biennial test frequency.

In addition, the Phoenix program provides a relatively stable basis against which to represent other program designs and for application of fuel adjustments.

Our approach is to derive the non-I/M rates relative to the I/M references, by adjustment. One reason for adopting this approach is that , as mentioned, the volumes of data available in I/M areas vastly exceed those collected in non-I/M areas. An additional practical reason is that major work-intensive steps such as “hole-filling” and projection of deterioration need only be performed once.

In contrast to the I/M references, the non-I/M reference rates are designed to be nationally representative. Broadly speaking, they are intended to represent all areas in the country without I/M programs. In general, estimating the influence of I/M areas on mean emissions is not trivial, and efforts to do so commonly follow one of two broad approaches. One approach is to compare emissions for two geographic areas, one with and one without I/M (Figure 1 - 11(a)). A second and less common approach is to compare emissions between two groups of vehicles within the same I/M area, but with one group representing the main fleet ostensibly influenced by the program, and the second, far smaller, representing vehicles measured within the program but presumably not yet influenced by the program (Figure 1 - 11(b)).

Figure 1 - 11. Basic approaches to estimating differences attributable to I/M programs: (a) comparison of subsets of vehicles between two geographic areas, with and without I/M, and (b) comparison within a program area.



(a) Comparison between a program Area and a non-program area

(b) Comparison within a program Area

For convenience, we refer to the first approach as the “between-area” approach, and the second as the “within-area” approach. Neither approach attempts to measure the incremental difference attributable to a program from one cycle to the next.

The approach we adopted emphasizes the “within-area’ approach, based on a sample of vehicles “migrating” into Phoenix. To lay the basis for comparison, the primary goal was to identify a set of vehicles that had been measured by the program after moving into the Phoenix area, but that had not yet been influenced by the program. The specific criteria to identify particular migrating vehicles are presented in Table 13.

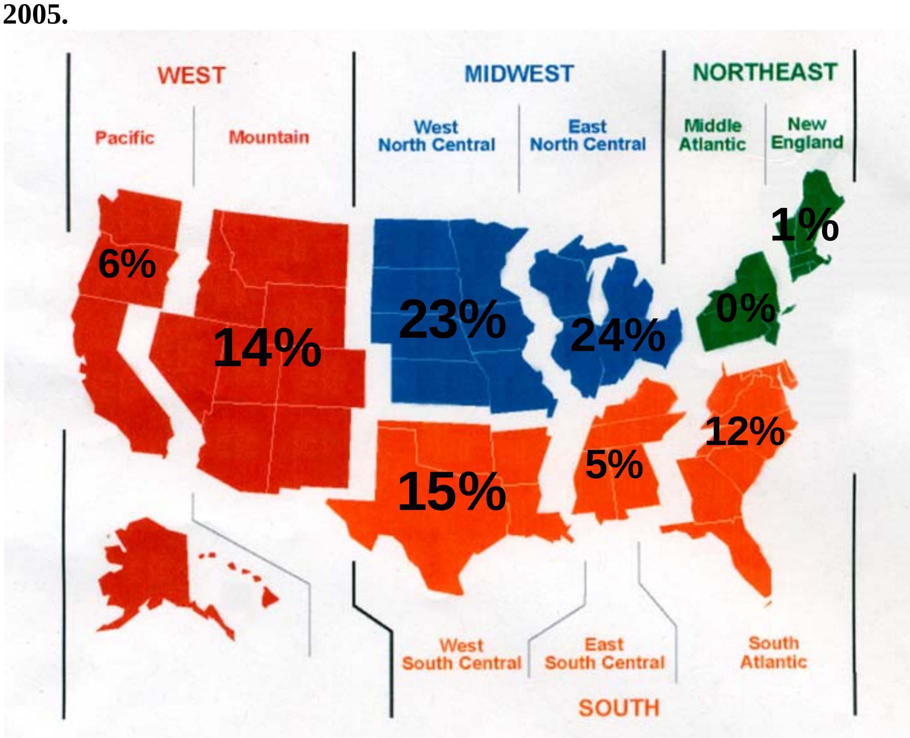
Table 1 - 13. Criteria Used to Identify Vehicles Migrating into the Phoenix Program.

logic	Criterion
	The vehicle comes from from out-of-state
OR	From a non-I/M county in AZ
AND NOT	From other I/M areas
AND	Receiving very first test in Phoenix program
AND	Selected for random evaluation sample

After applying these criteria, we identified a sample of approximately 1,400 vehicles. The origin of vehicles entering the Phoenix Area was traced by following registration histories of a set of approximately 10,000 candidate vehicles. The last registered location of vehicles was identified prior to registration in Phoenix or the vehicle's first test in the Phoenix program. Vehicles were excluded if their most recent registration location was in a state or city with an I/M program[7].

Figure 14 shows the distribution of incoming vehicles, by Census Region. Most vehicles migrating to Phoenix came from the Midwest (47%), followed by the South (32%), the West (20%) and the Northeast (1%). The low incidence from the NE may be attributable to the large number of I/M programs in that region.

Figure 1 - 12 Geographic Distribution of Vehicles Migrating into the Phoenix I/M Area, 1995 - 2005.



To assess the differences between migrating (non-I/M) and "local" (I/M) vehicles, we adopted a simple approach. We calculated ratios between means for the migrating and local groups, as shown in equation 36. We used aggregate tests, after preliminary analyses suggested that the ratios did not vary significantly by VSP. Because the sample was not large in relation to the degree of variability involved, we also aggregated tests for cars and trucks in all model years. However, we did calculate ratios separately for three broad age groups (0-4, 5-9, and 10+) years. We propagated uncertainty for these ratios as in Equations 1 - 31 and 1 - 32.

$$\text{Ratio} = \frac{\overline{E}_{\text{non-I/M}}}{\overline{E}_{\text{I/M}}} \qquad \textbf{1 - 36}$$

For purposes of verification, we compared our results to previous work. An initial and obvious comparison was to previous work based on an out-of-state fleet migrating into Phoenix that provided a model for our own analysis[7]. This previous effort, by T. Wenzel, identified a migrating fleet, and analyzed differences between it and the program fleet for vehicles in model years 1984 – 1994 measured during calendar years 1995-2001. To adapt his results for our purposes, we converted averages for migrating and program fleets into ratios as in Equation 1 - 36.

A another valuable source for comparison was remote-sensing data collected in the course of the Continuous Atlanta Fleet Evaluation (CAFE) Program[8,9]. Unlike our own analysis, this program involves a comparison between two geographic areas. The "I/M area" is the thirteen-county Atlanta area, represented by measurements for approximately 129,000 vehicles. The other (the non-I/M area) is the twelve-county non-I/M area, surrounding Atlanta, represented by measurements for approximately 28,000 vehicles. Both areas have been under a low-sulfur fuel requirement since 1999. Results used for this analysis were collected during CY 2004.
The non-I/M : I/M ratios calculated from the RSD are based on concentrations, rather than mass rates.

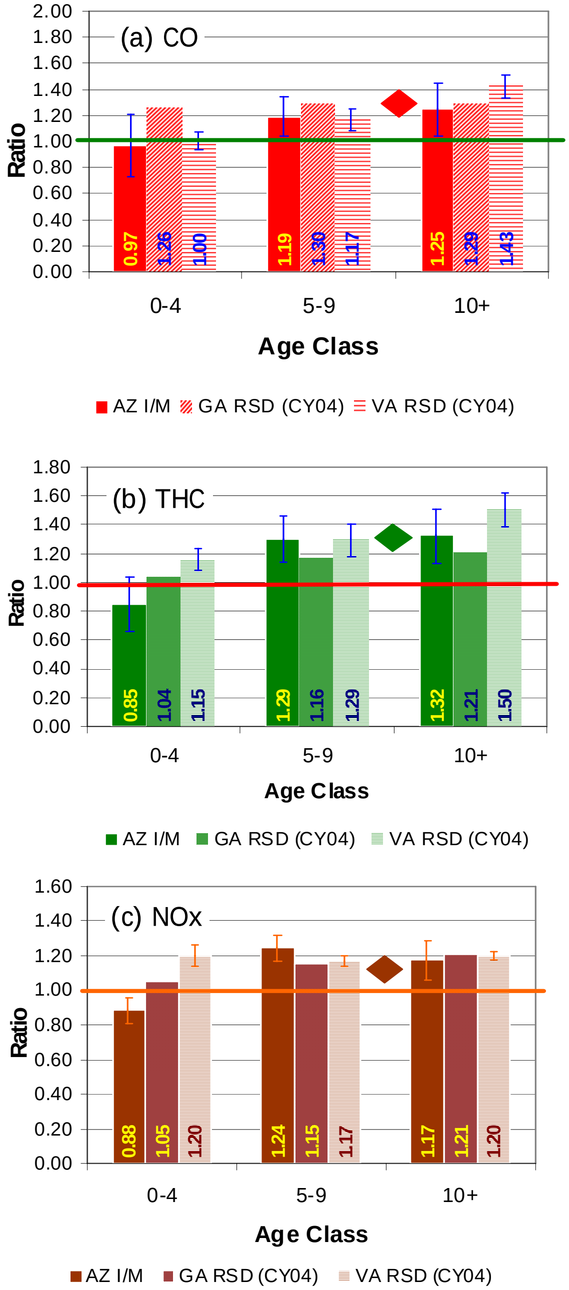
A third source was an additional remote-sensing dataset collected in N. Virginia/D.C. area
The I/M area was the "northern-Virginia" counties, and the non-I/M area was Richmond. The I/M and non-I/M areas were represented by about 94,000 and 61,000 vehicles, respectively, collected in CY 2004. In this case, the molar ratios were converted to mass rates, with use of fuel-consumption estimates derived from energy-consumption rates in MOVES2004. After this step, non-I/M : I/M ratios were calculated using the mass rates.

Results are shown in Figure 1 - 13. The charts show mean ratios for the three age groups for our migrating vehicle analysis, as well as the remote-sensing studies. The diamonds represent approximate values from Wenzel's earlier work with the Phoenix data. For our analyses (solid bars) the ratios are generally lower for the 0-4 year age Group, and larger for the 5-9 and 10+ age groups, but differences between the two older groups are small. The Atlanta results show a similar pattern for HC and NOx, but not for CO, for which the ratios are very similar for all three age groups. The Virginia results are the other hand, show increasing trends for CO and HC, but not for NOx. The ratios in Atlanta are slightly higher than those for Phoenix in the 0-4 year age group. This difference may be attributable to the shorter exemption period in Atlanta (2 years) vs. the four-year period in Phoenix, but it is not clear that these differences are statistically significant. In all three programs, ratios for the two older age classes generally appear to be statistically significant.

In interpreting the ratios derived from the Phoenix data, it is important to note that they assume full program compliance. In the migrating vehicle analysis this is the case because all emissions measurements were collected in I/M lanes. Thus, vehicle owners who evaded the program in one way or another would not be represented. On the whole, results from multiple datasets, using different methods, showed broad agreement.

If we calculate non-IM reference rates from the I/M references by ratio, with the ratios constant by model-year group and VSP, it follows that the absolute differences must increase with power. Similarly, absolute differences increase with age, for two reasons. A first reason is the same as that for VSP, that for a constant ratio, the absolute difference increases as emissions themselves increase, and on top of this, the second reason is that the ratios themselves increase with age (Figure 15). A third implication is the absolute differences would be smaller for successive model-year groups as tailpipe emissions decline with more stringent standards.

Figure 1 - 13. Non-I/M : I/M ratios for CO, HC and NOx for the Phoenix Area (this analysis) compared to remote-sensing results for Atlanta and N. Virginia, and previous work in Phoenix (diamonds).



A final practical step is to translate these results into terms corresponding to the MOVES age groups. As mentioned, the program in Phoenix has a four-year exemption period for new vehicles. However, it is not uncommon for other programs have shorter exemptions; for example, both the Atlanta and N. VA programs have two-year exemptions.

An additional factor is that the coarser age groups used for the migrating-vehicle analysis don't mesh cleanly with the MOVES age groups. It was therefore necessary to impute values to the first two MOVES age groups (0-3 and 4-5 years). We achieved this step by linearly interpolating the value for the 5-9 age Group to a value of 1.0 and 0 years of age, as shown in Figure 1 - 14. To anchor the interpolation, we associated the value of the ratio for the 5-9 year age group with the midpoint of the group (7.5 years). Then, based on a straight line interpolation, we imputed values for the 0-3 and 4-5 MOVES age groups, by taking the value on the line associated with the midpoint of each class, 1.5 and 5 years, respectively.

Figure 1 - 14. Imputation of Non-I/M Ratios for the 0-3 and 4-5 year MOVES AgeGroups by Linear Interpolation from the Midpoint of the 5-9 year Analysis Age Group.

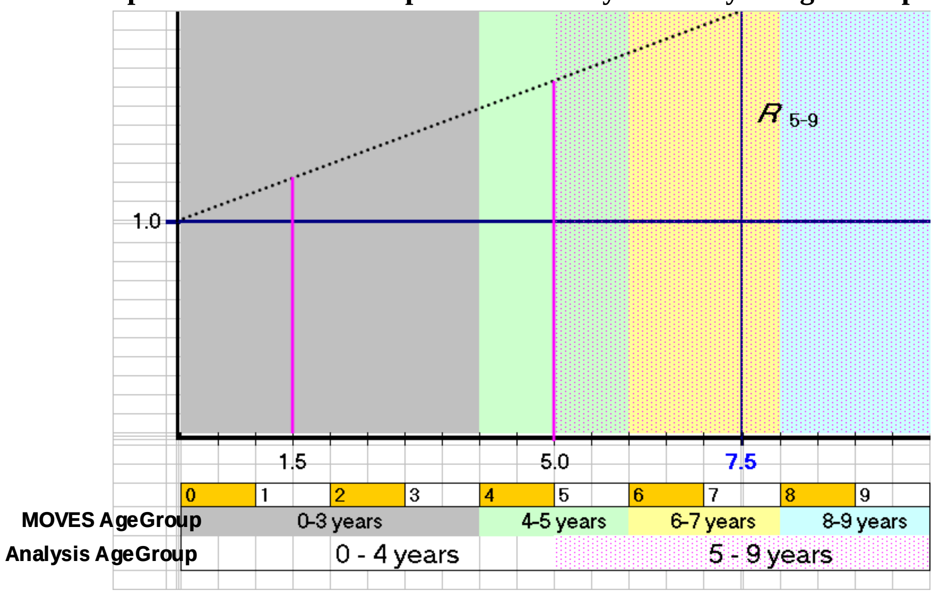
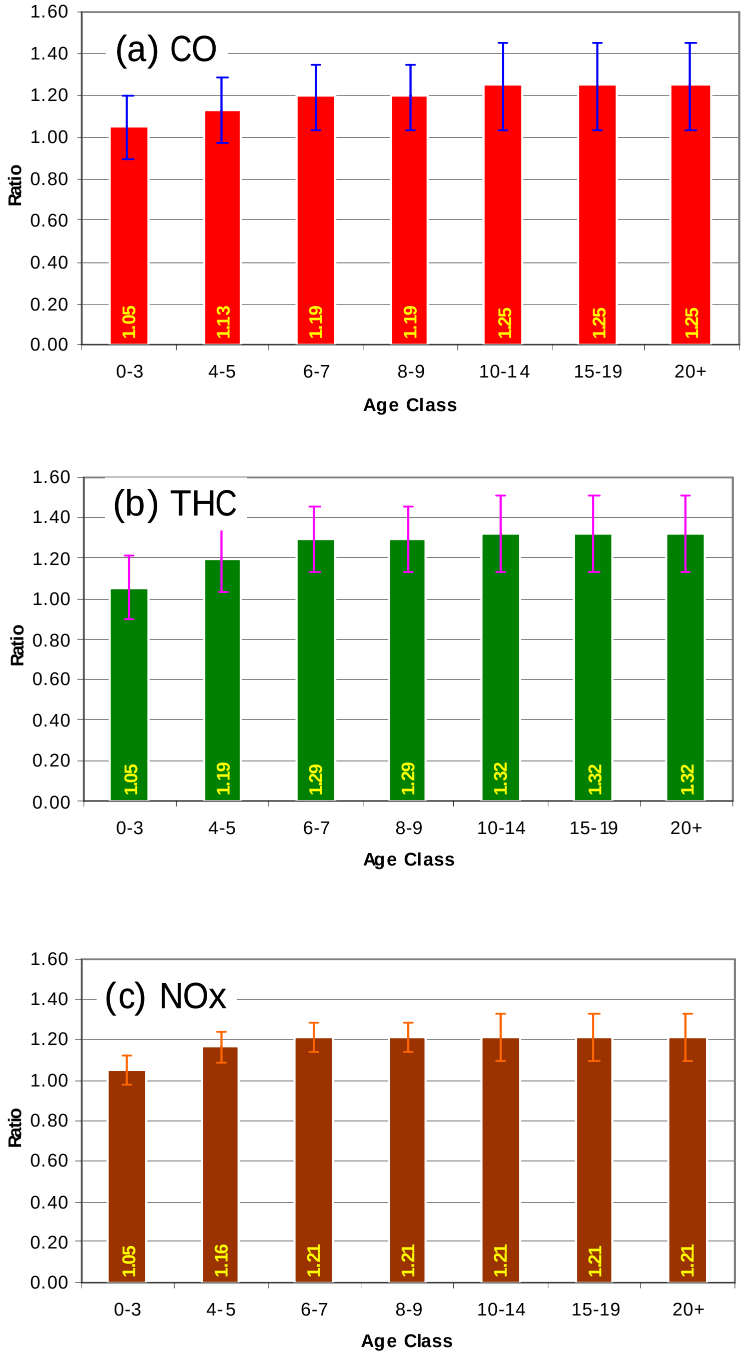


Figure 1 - 15 shows final values of the non-I/M ratios for CO, THC and NOx, with error-bars representing 95% confidence intervals. The values for each pollutant start at 5.0% and increase with age, stabilizing at maximum values at 6-10 years.

Figure 1 - 15. Final non-I/M ratios for CO, HC and NOx, by MOVES AgeGroups, with 95% confidence intervals.



The ratios shown in Figure 17 are applied to the I/M reference rates to derive non-I/M reference rates.

$$E_{h,\mathrm{non-I/M}} = \mathrm{Ratio} * E_{h,\mathrm{I/M}} \qquad \textbf{1 - 37}$$

The uncertainty in $E_{h,\text{non-I/M}}$ was calculated by propagating the uncertainty in the Ratio with that of the corresponding I/M rate $E_{h\text{I/M}}$.

$$
\begin{aligned}
s^2_{E_{h,\text{non-I/M}}} &= \left(\frac{\partial E_{h,\text{non-I/M}}}{\partial R}\right)^2 s_R^2 + \left(\frac{\partial E_{h,\text{non-I/M}}}{\partial E_{h,\text{I/M}}}\right)^2 s^2_{E_{h,\text{I/M}}} \\
s^2_{E_{h,\text{non-I/M}}} &= E^2_{h,\text{I/M}} s_R^2 + R^2 s^2_{E_{h,\text{I/M}}}
\end{aligned}
\qquad \textbf{1 - 38}
$$

Thus, for any given cell *h*, the uncertainty in the non-I/M reference rate is larger than that for the corresponding I/M reference rate, which is reasonable given the additional assumptions involved in developing the non-I/M reference rate.

Figure 1 - 16 shows an example of the reference rates by operating mode, for all three pollutants, with error bars representing 95% confidence intervals. Note that not all the modes are shown, to allow examination of differences between non-I/M and I/M rates at lower VSP. It is immediately apparent that uncertainties are considerably larger for the non-I/M rates, which reflects the uncertainties in the non-I/M:I/M ratios, in relation to the relatively small uncertainties in the I/M references derived directly from data. The very large uncertainties in high-power operating modes (28-30, 39) reflect the combined uncertainties in the high operating mode ratios (see 1.5.5) and the non-I/M ratios.

Figure 1 - 17 shows corresponding trends by age for two operating modes. The first is opmode 11, (speed = 1-25 mph, VSP <0 kW/tonne) and 27 (speed = 25-50 mph, VSP = 12-18 kW/tonne). As before, the uncertainties are visibly larger for the non-I/M rates. Trends level off at in the 10-14 year age Group. An obvious observation is that the I/M difference is much larger in the more aggressive mode (27) than in the less aggressive one (11), with the inference that I/M differences will be more strongly expressed for more aggressive than less aggressive driving.

Figure 1 - 16. Non-I/M and I/M Reference Rates by Operating Mode (Example: LDV, MY 1994, at 8-9 years of age)

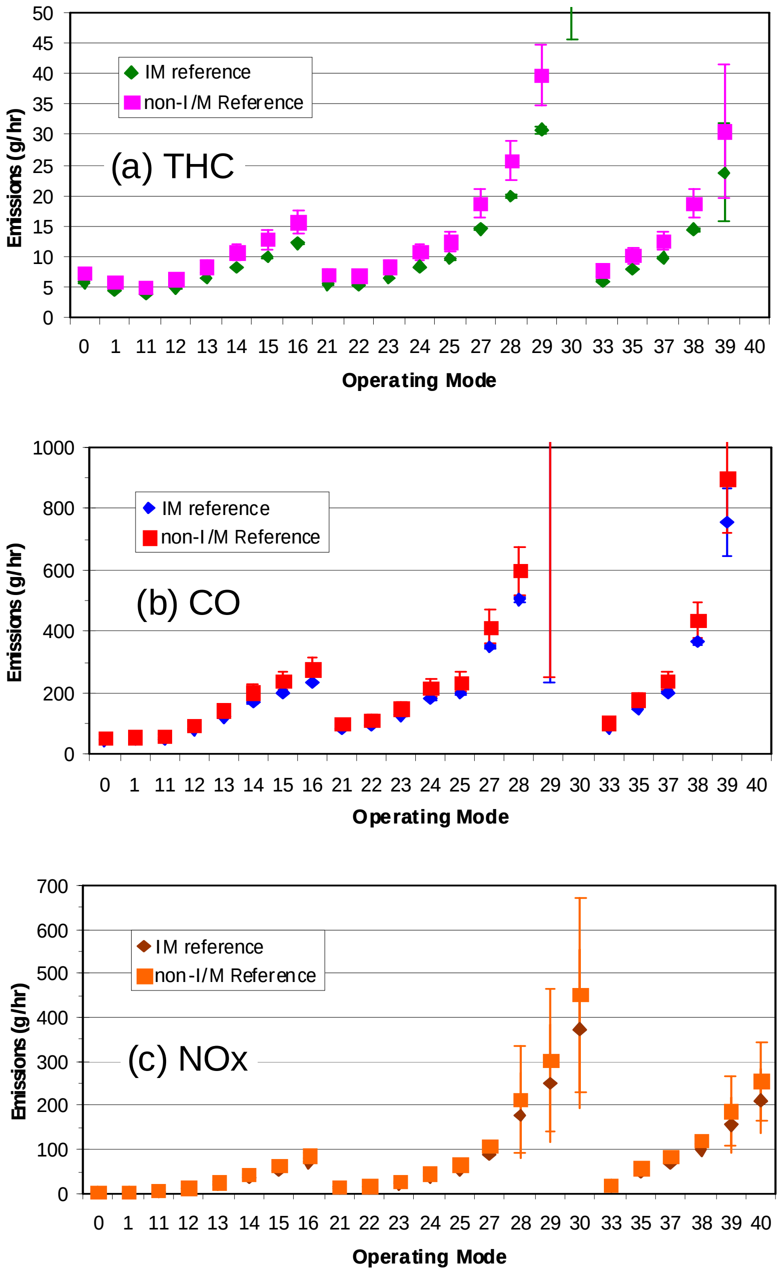
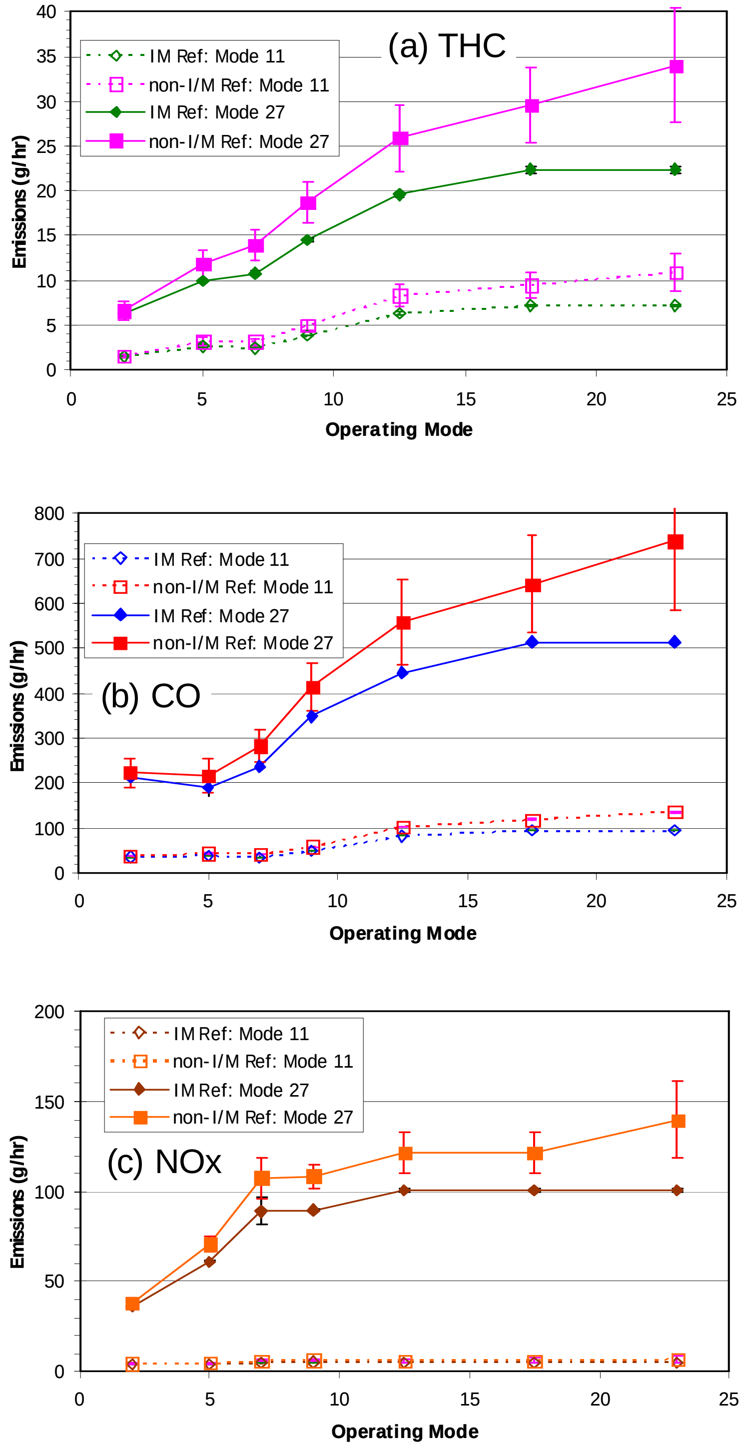


Figure 1 - 17. Non-I/M and I/M Reference Rates vs. Age for Two Operating Modes (Example: LDV, MYG 1994).



1.5.7 Stabilization of Emissions with Age

One characteristic of the data is that fleet-average emissions do not appear to increase indefinitely with age, but rather tend to stabilize at some point around 15 years of age. This behavior is visible in datasets with enough historical depth for age trends to be observable, including the Phoenix random sample and long-term remote-sensing studies[10]. Figure 1 - 18 and Figure 1 - 19 show age trends by model year for LDV and LDT, respectively. The values shown are aggregate mass rates over the IM147 expressed as g/sec for CO, THC and NOx.

1.5.7.1 I/M Reference Rates

From Figures 11 and 12, as well as Figure 1, it is clear that no data was available at ages older than 10 years of age for model years later than 1995, and that no data was available at ages older than 15 years for model years older than 1990. Thus for model years more recent than about 1995 it was necessary to project emissions for ages greater than 8-10 years.

However, it is not appropriate to simply extrapolate the statistical models past about 10-12 years. As described above, emissions were modeled as ln-linear with respect to age, which implies exponential trends for reverse-transformed values. However, exponential trends will increase indefinitely if extrapolated much beyond the range of available data, which obviously does not describe the down-turn as the trend levels off and stabilizes. To compensate for this limitation, we employed a simple approach to represent the decline and stabilization of the rates.

Figure 1 - 18. Aggregate IM147 Emissions (g/sec) for LDV, by Model year and Age, for the Phoenix Random Sample.

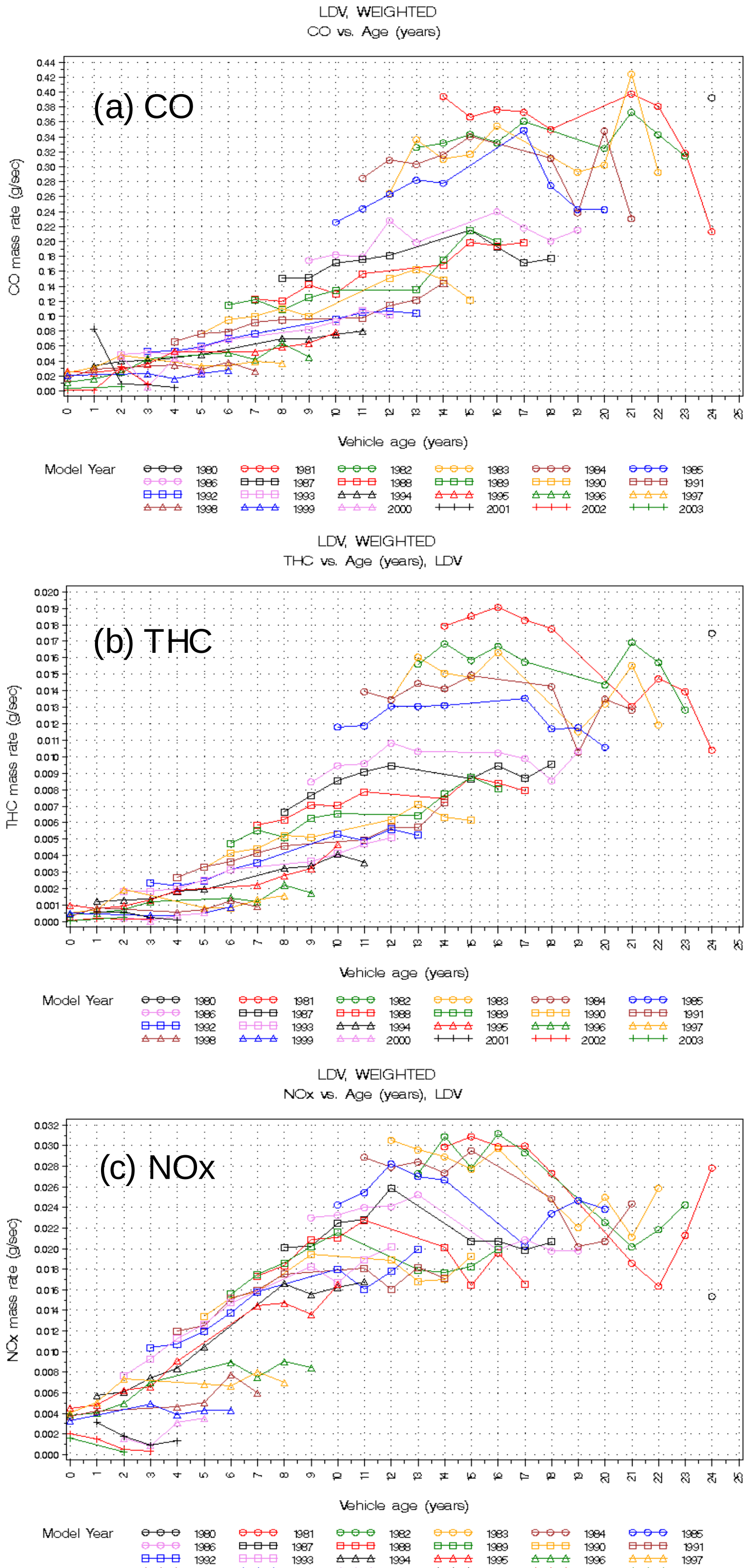
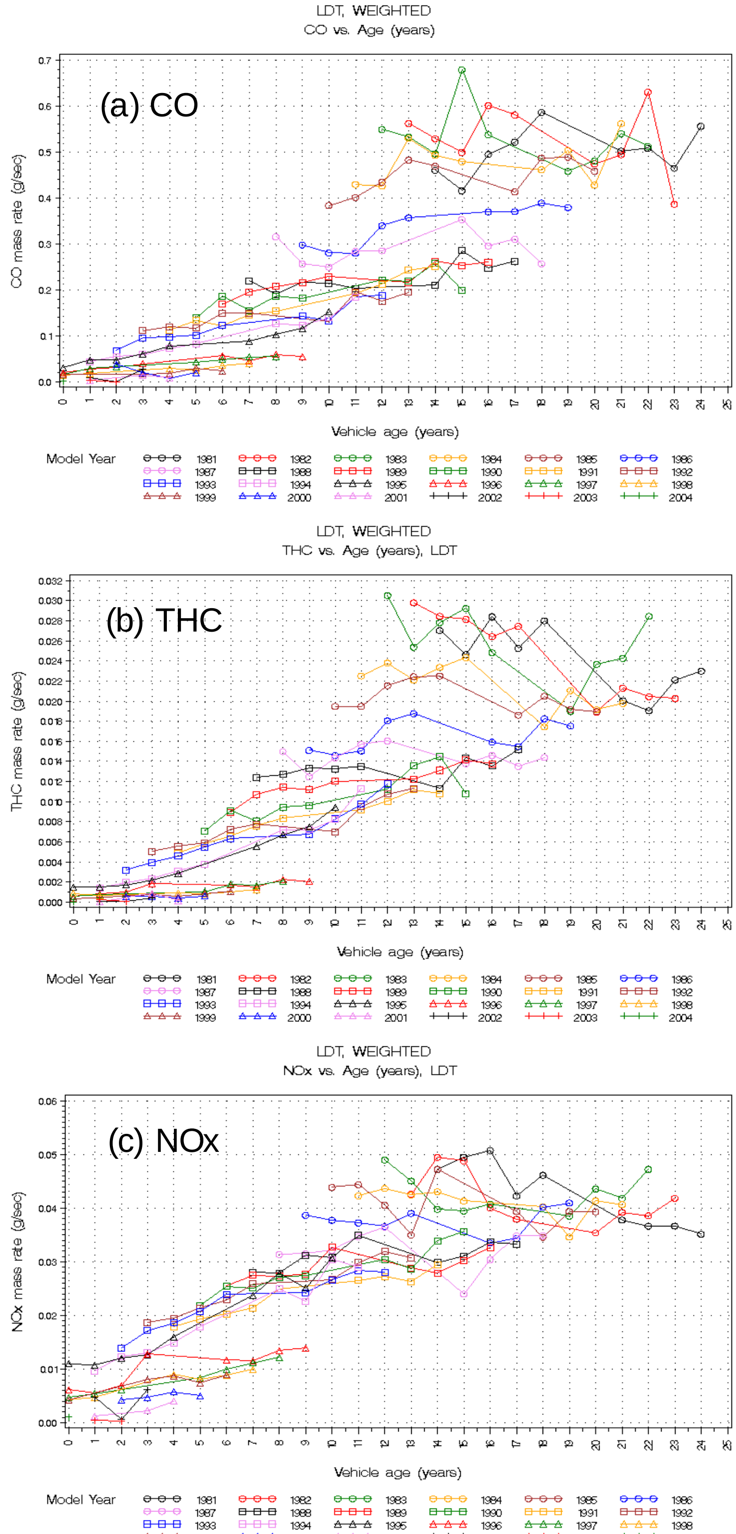


Figure 1 - 19. Aggregate IM147 Emissions (g/sec) for LDT, by Model year and Age, for the Phoenix Random Sample.



We calculated ratios of means between the 15-19 and 10-14 year age Groups for the model-year group 1986-89, which contains data for vehicles as old as 19 years. For this purpose we used Phoenix data aggregated by MOVES model-year and age groups, as shown in Table 1 - 15.
For this purpose, we used aggregated tests (g/mi). After averaging by model-year group and ageGroup, we calculated ratios of means between the 15-19 and 10-14 ageGroups.

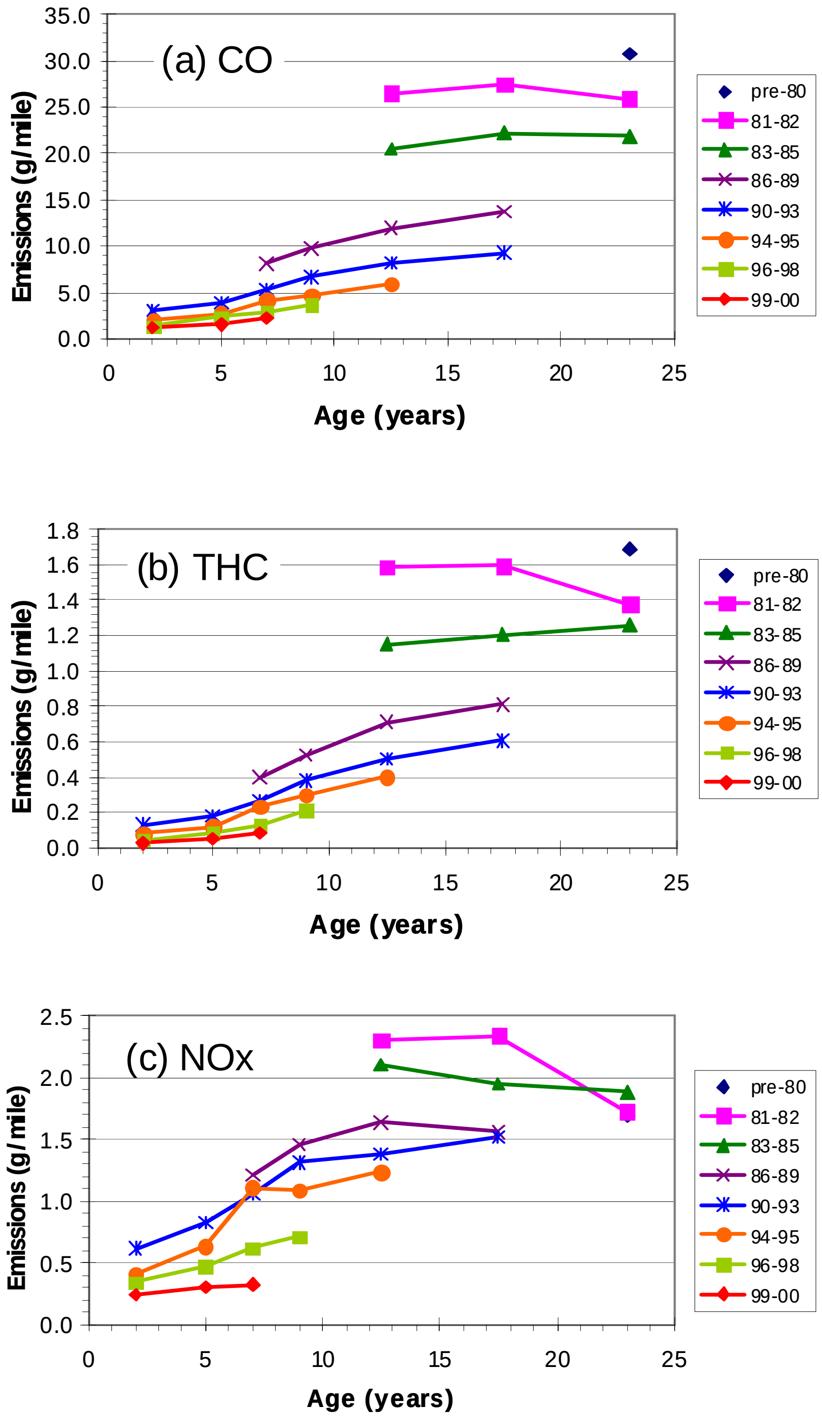
$$R_{\text{age}} = \frac{\overline{E}_{86-89,15-19}}{\overline{E}_{86-89,10-14}} \qquad \textbf{1 - 39}$$

We calculated modified rates for the 15-19 and 20+ age Groups as the product of the rate for the 10-14 ageGroup and the resulting ratio (R_{age}, **Error! Reference source not found.**). The resulting rate was the same for 15-19 and 20+. We calculated variances for the ratios, but did not propagate the uncertainty through to the final result.

Table 1 - 14. Age-Group Ratios (R_{age}) between the 15-19 and 10-14 ageGroups (MYG 1986-89)

Regulatory Class	CO	THC	NOx
LDT	1.22	1.19	1.08
LDV	1.15	1.14	1.00

Table 1 - 15. Aggregate IM147 Emissions (g/mi) by Model-Year Group and Age Group.

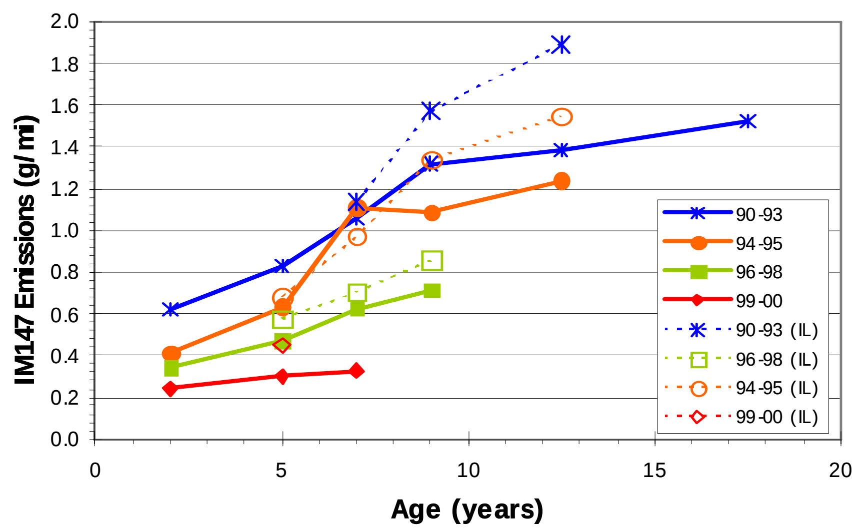


1.5.7.2 non-I/M Reference Rates

The ratios developed in 1.5.7.1 apply in I/M areas, as the underlying data was collected in the Phoenix I/M area. It is therefore plausible that the patterns observed may be specific to I/M areas The program places some pressure on high-emitting vehicles to improve their emissions, leave the fleet, leave the area, or, it could be added, evade the program in some way. However, in the absence of a program, high-emitting vehicles are not identified and owners have little incentive to repair or replace them. Thus, the question arises as to whether deterioration patterns would necessarily be identical in non-I/M as in I/M areas. Two plausible scenarios can be proposed. In the first, the pattern of deterioration followed by stabilization is similar in non-I/M as in I/M areas, but emissions stabilize at a higher level, and perhaps at a later age. In the second, emissions continue to increase in non-I/M areas, but at a slower rate after 10-15 years.

Data that sheds light on these questions are very limited, as the datasets with sufficient history are collected within I/M areas. But one possibility exists. As mentioned, we analyzed and considered data fromt the Chicago program for this project. A characteristic of this program is that vehicles were evaluated only on results for HC and CO; NOx was measured in some lanes but did not inform test results. Thus, with respect to NOx, we considered the Chicago data as a rough surrogate for a non-I/M area. It was therefore helpful to compare NOx results in Chicago to those in Phoenix, on an aggregate basis, as shown in Figure 1 - 20. In the figure, NOx emissions for the 1990-93 model-year grouip appear to be increasing at a higher rate in Chicago than in Phoenix between the ages 8 to 12.5 years. Based on this increment (15%), we assumed that emissions for all pollutants would increase by 15% between the 15-19 and 20+ year ageGroups. Note that the effects of this adjustment can be seen in the non-I/M series in **Error! Reference source not found.** above.

Figure 1 - 20. Aggregate IM147 NOx Emissions for LDV in Phoenix (unlabeled) and Chicago (IL)



1.5.8 Deterioration for Start Emissions

Because MOBILE assumed that start emissions would deteriorate, based on analyses performed at the time of its development, we thought it reasonable to include start deterioration in draft MOVES. Due to the complete lack of data in this area, we elected to model start deterioration as a function of running deterioration.

As described, start rates are defined in terms of the FTP, but the rates for running are represented by operating mode. However, the modal approach enables simulation of multiple driving patterns, as represented by test cycles, such as the Federal Test Procedure (FTP), or the US06 cycle, which represents high-speed driving in the Supplemental Federal Test Procedure (SFTP). Any driving cycle can be represented as a weighted average of the MOVES emission rates and the "operating mode distribution" for that cycle. In this case we developed an operating-mode distribution for the "hot-running" phase of the FTP (Bag 2). This phase is an 860 second long trace that represents urban driving over a 3.9 mile route after the engine has stabilized at its normal operating temperature. We estimated an operating-mode distribution using the "Physical Emission-Rate Estimator" (PERE)[2]. This distribution, shown in **Error! Reference source not found.**, represents a typical LDV, with an engine displacement of 2.73 L and test weight of 3,350 lb. Estimating the Bag 2 emissions simply involves calculation of averages of the emission rates weighted by the mode distribution.

Table 1 - 16. Operating-mode distribution for a typical LDV on the hot-stabilized phase of the FTP (Bag 2)

Operating Mode	Time in Mode (seconds)	Time in Mode (%)
0	97	11.3
1	155	18.0
11	77	8.9
12	121	14.1
13	83	9.6
14	59	6.9
15	22	2.6
16	4	0.50
21	42	4.9
22	111	12.9
23	62	7.2
24	18	2.1
25	7	0.80
27	2	0.2
28		
29		
30	1	0.1
33		
35		
37		
38		
39		
40		
Total	861	100

After calculation, the simulated FTPs were converted to ratios by dividing the value for each age Group by the value for the 0-3 year age Group.

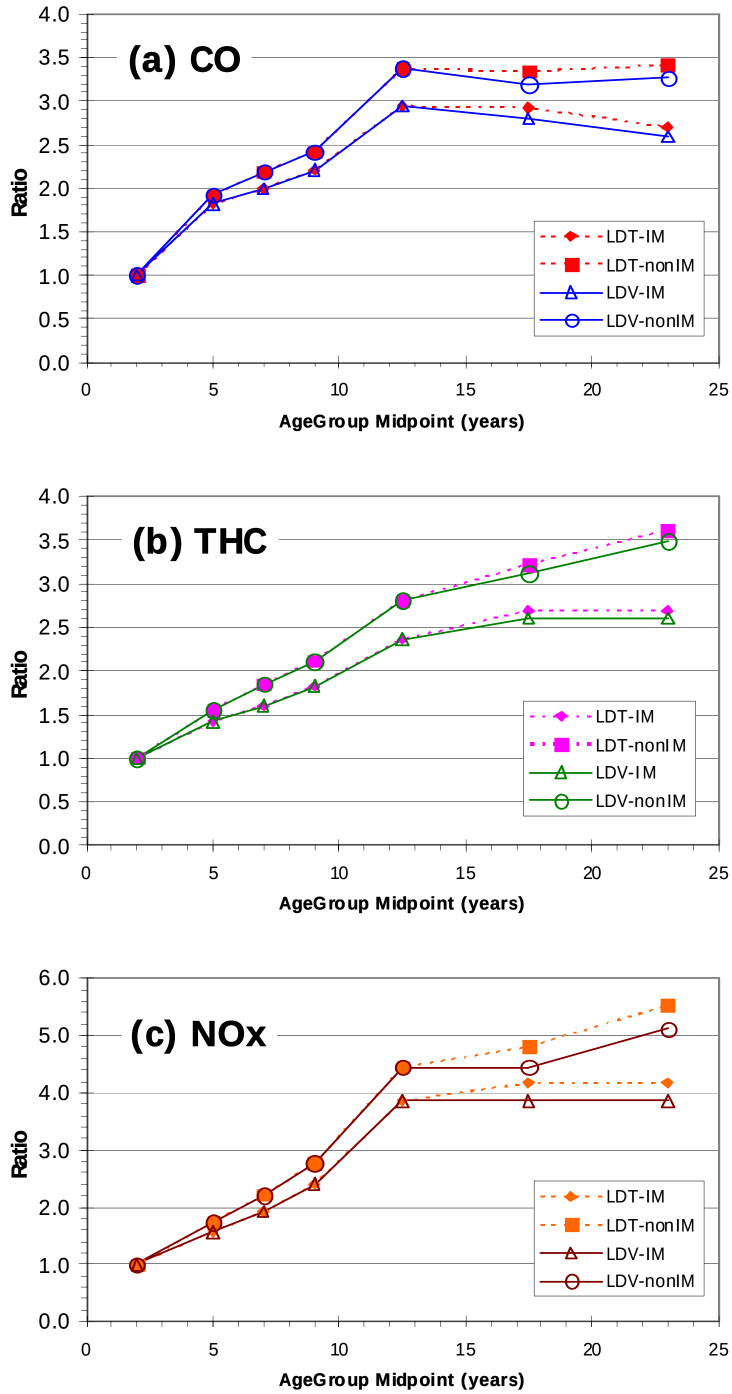
A major uncertainty in our knowledge of start emissions is whether it is reasonable to assume that start emissions would deteriorate at the same (relative) rate as running emissions. Given the lack of data in this area, we adapted assumptions applied in the MOBILE model. In MOBILE we assumed that start emissions for NOx would deteriorate at same relative rate as those for running,[11] but that HC and CO emissions would deteriorate at lower relative rates.[12] Thus, we adapted the equations for “normal” and “high” emitters applied in MOBILE and developed a set of ratios for each age Group that reduce HC and CO start deterioration relative to the running deterioration, as shown in **Error! Reference source not found.**. Final ratios after application of the relative reductions are shown in Figure 1 - 21. Deteriorated start rates are developed (in all operating modes) by multiplying the rate in the 0-3 age Group by the ratio for each successive age Group.

NOTE: if possible, these assumptions will be reviewed, and considered for revision before release of the final model.

Table 1 - 17. Ratios Expressing Relative Reduction in Start Deterioration, relative to Running Deterioration, by Age Group.

ageGroup	HC	CO
0-3	1.00	1.00
4-5	0.58	0.57
6-7	0.47	0.46
8-9	0.41	0.39
10-14	0.36	0.33
15-19	0.36	0.33
20+	0.36	0.33

Figure 1 - 21.   Example: Start Deterioration Ratios Applied to MY 2001-2021.



1.6. Emission-Rate Development: Subgroup 2 (MY 2001 and later)

1.6.1 Data Sources

Data for vehicles in model years 2001 and later was acquired from results of tests conducted under the In-Use Verification Program. This program, initiated in 2003, is run by manufacturers and administered by EPA/OTAQ through the Compliance and Innovative Strategies Division (CISD).

To verify that in-use vehicles comply with applicable emissions standards, customer-owned vehicles at differing mileage levels are tested on an as-received basis with minimal screening. Emissions are measured on the Federal Test Procedure, US06 and other cycles. The FTP is most relevant to our purposes, but the US06 is also important.

1.6.1.1 Vehicle Descriptors

In addition to the parameters listed above in Table 1 - 3. Required Vehicle Parameters., the IUVP data provides engine-family information. Using engine family, the IUVP files can be merged with certification logs by model year. The certification logs provide information on Tier level and specific emissions standards applicable to each vehicle. The Tier level refers to the body of standards to which vehicles were certified (Tier 1, NLEV, LEV-I, LEV-II), and the standards refer to specific numeric standards for HC, CO or NOx, where HC are represented by non-methane hydrocarbons (NMHC) or non-methane organic gases (NMOG), depending on Tier Level.

Table 14. Vehicle Descriptors Available in IUVP files and Certification Logs

Parameter	**Units**	**Source**		**Purpose**
		IUVP	*Cert. Log*	
VIN		Y		Verify MY or other parameters
Fuel type		Y		
Make		Y	Y	
Model		Y	Y	
Model year		Y	Y	Assign sourceBinID, calculate age-at-test
Engine Family		Y	Y	
Tier			Y	
Emissions Standard			Y	Assign Vehicle Class

Combining data from both sources allows individual test results to be properly associated with the correct Tier Level and emissions standard, which allows inference of the correct vehicle class (LDV, LDT1, LDT2, LDT3, LDT4).

1.6.2 Estimating I/M Reference Rates

The goal of this process is to represent I/M reference rates for young vehicles, i.e., the first age Group (0-3 years). The rates are estimated by Tier, model year and regulatory class. The process involves six steps.

1. *Average IUVP results* by Tier and vehicle class

2. *Develop phase-in assumptions* for MY 2001 – 2021, by Tier, vehicle class and model year

3. *Merge FTP results and Phase-in assumptions*. For running emissions, calculate weighted ratios of emissions in each model year to those for Tier 1 (MY2000). Then calculate emissions by operating mode in each model year by multiplying the MY2000 emission rates by the weighted ratio for each model year. For start emissions, use weighted average FTP starts directly.

4. *Estimate Emissions by Operating Mode*. We assumed that the emissions control at high power (outside ranges of speed and acceleration covered by the FTP) would not be as effective as at lower power (within the range of speed and acceleration covered by the FTP).

5. *Apply Deterioration* to estimate emissions for remaining six age Groups. We assume that NLEV and Tier-2 vehicles will deteriorate similarly to Tier-1 vehicles, when viewed in logarithmic terms. We therefore apply ln-linear deterioration to the rates developed in steps 1-4.

6. *Estimate non-I/M reference rates*. The rates in steps 1-6 represent I/M references. Corresponding non-I/M references are calculated by applying the ratios applied to the Tier-1 and pre-Tier-1 rates (Figure 16).

Each of these steps is described in greater detail in the sub-sections below.

1.6.2.1 Averaging IUVP Results

In using the IUVP results, "cold-start" emissions are represented as "Bag 1 – Bag 3" i.e., the mass from the cold-start phase less that from the corresponding hot-start phase. Similarly, "hot-running" emissions are represented by the "Bag 2," or the "hot-stabilized" phase, after the initial cold-start phase has conditioned the engine.

The first step is to average the IUVP results by Tier and vehicle Class. Results of this process are shown below. In the figures, note that the HC values represent non-methane hydrocarbons (NMHC) for Tier 1 and non-methane organic gases (NMOG) for NLEV and Tier 2. Figure 1 - 22 shows FTP composite results in relation to applicable certification and useful-life standards. For THC and NOx, the data show expected compliance margins in the range of 40-60% in most cases. For CO, compliance margins are even larger, ostensibly reflecting the concomitant effect of HC control on CO emissions, whereas CO standards were not stringent enough in themselves to substantially reduce emissions.

Figure 1 - 22. Composite FTP Results for Tier1, NLEV and Tier 2 Vehicles, as measured by IUVP, in relation to certification and useful-life standards.

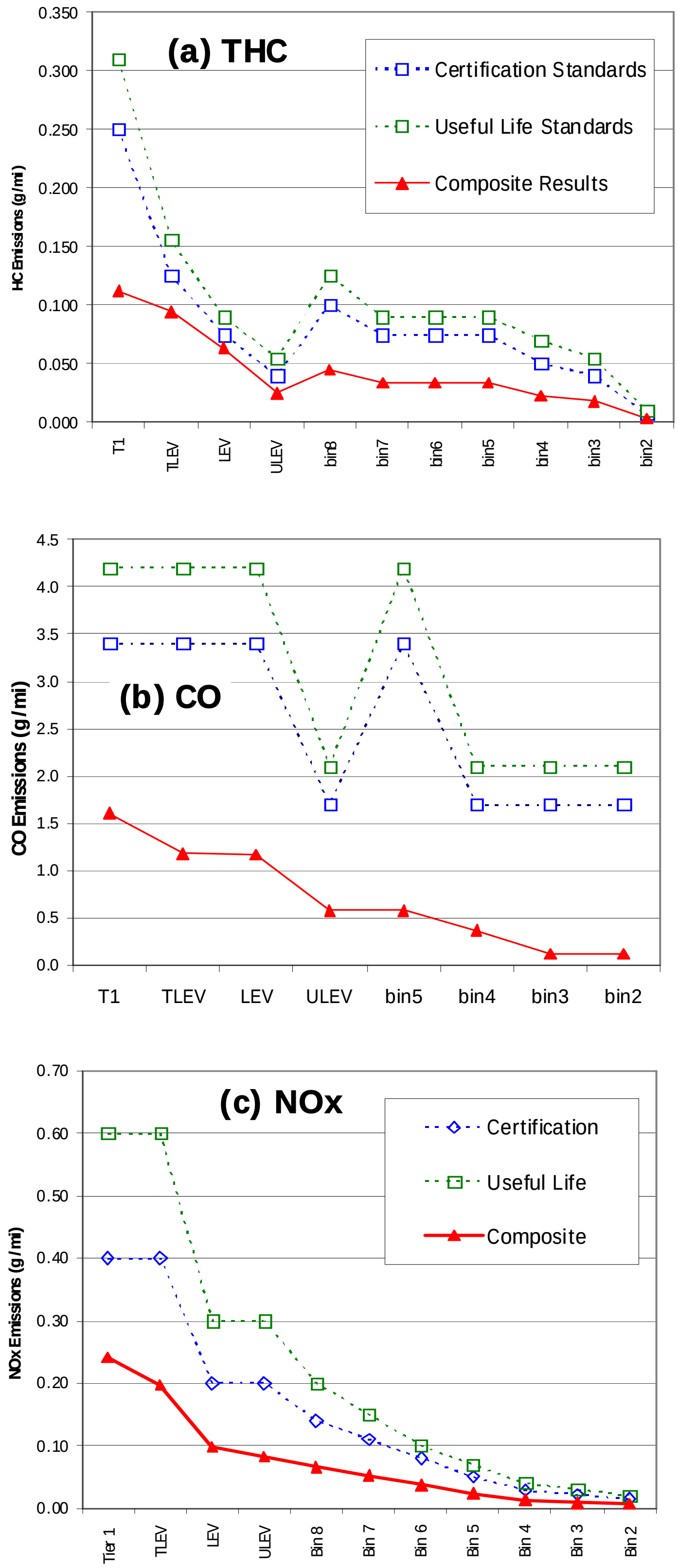


Figure 1 - 23. Cold-start (Bag 1 – Bag 3) and Hot-running (Bag 2) FTP emissions for Tier 1, NLEV and Tier 2 vehicles, as measured by IUVP (g/mi).

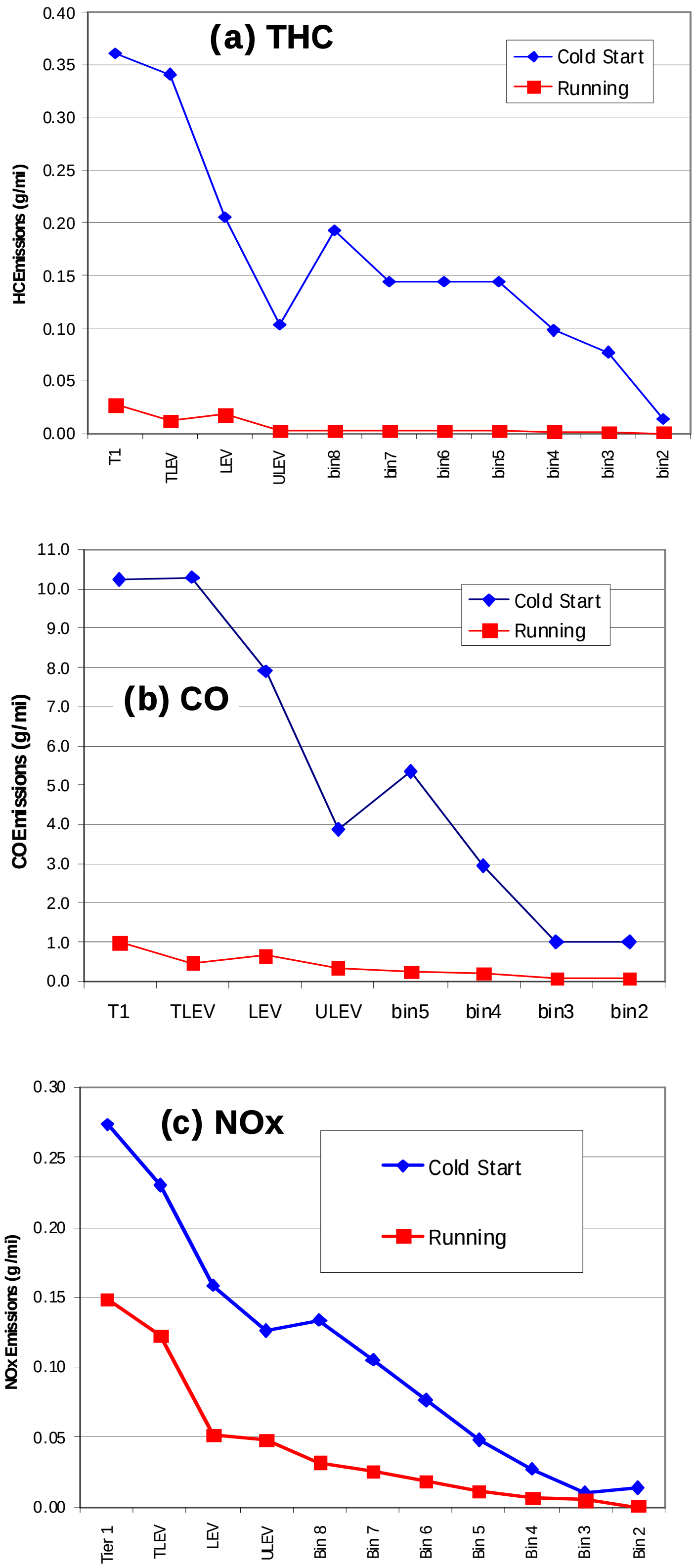


Figure 1 - 23 shows results for separate phases of the FTP, to examine differential effects of standards on start and running emissions. As mentioned, the “cold-start” emissions are represented by the difference between Bags 1 and 3, expressed as a “start rate” in g/mi. The “hot-running” emissions are represented by Bag 2 emissions, also divided by the appropriate distance to obtain an aggregate rate, in g/mi. Distinguishing start and running emissions shows that composite FTP values for HC and CO are strongly influenced by start emissions. Starts are also important for NOx, but not as markedly so. In any case, the results show that use of the composite results could give misleading results in projecting either start or running emissions.

1.6.2.2 Develop Phase-In Assumptions

For rates stored as MOVES defaults, we developed assumptions intended to apply to vehicles sold in states where Federal, rather than California standards applied. Thus, the phase-in represents the phase-in of NLEV and Tier-2 standards. To construct the default scenarios, we divided the vehicle classes into two groups. The first group includes LDV, LDT1 and LDT2, to which NLEV standards applied; the second includes LDT3 and LDT4, which transitioned directly from Tier 1 to Tier 2.

Assumptions for LDV, LDT1 and LDT2 are shown in Figure 1 - 24. For these classes, The transition between Tier 1 and NLEV is abrupt, occurring between 2000 and 2001. For the first two model years, the fleet is a mixture of “transitional LEV” (TLEV) and LEV, and entirely LEV for the following four model years. However, a three-year phase-in of Tier 2 began in 2003, and was complete by 2007, after which the Federal fleet is entirely Tier 2. The breakdown of Tier 2 Bins during the transition is shown in Figure 1 - 25. This scenario reflects a tendency for Bin 5 to dominate the fleet by 2007, followed by Bin 3, and with Bins 4 and 2 playing minor roles.

For the second group, LDT34, the transition from Tier 1 to Tier 2 lasts three years, as shown in Figure 1 - 26 and Figure 1 - 27. Like the LDV group, the transition to Tier 2 is complete in MY 2007. Tier 2 vehicles are primarily represented by Bin 8 for the first several years, after which the heavier LDTs also move into Bin 5. For both groups, we assume that all phase-in fractions are stable between MY2010 and 2021.

Figure 1 - 24. Phase-in Assumptions for Tier 1, NLEV, and Tier 2, for LDV, LDT1 and LDT2

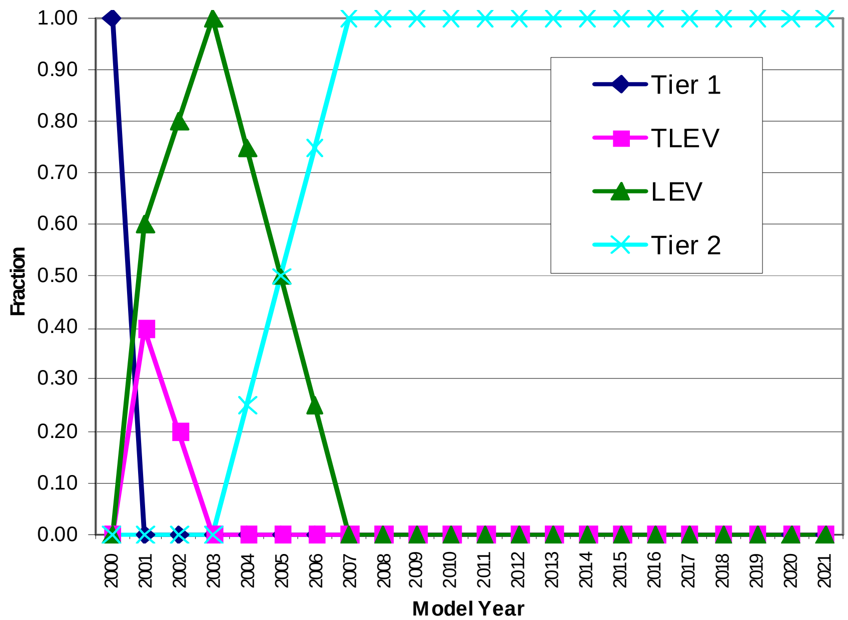


Figure 1 - 25. Phase-in Assumptions for Tier 2, by Bin, for LDV, LDT1 and LDT2.

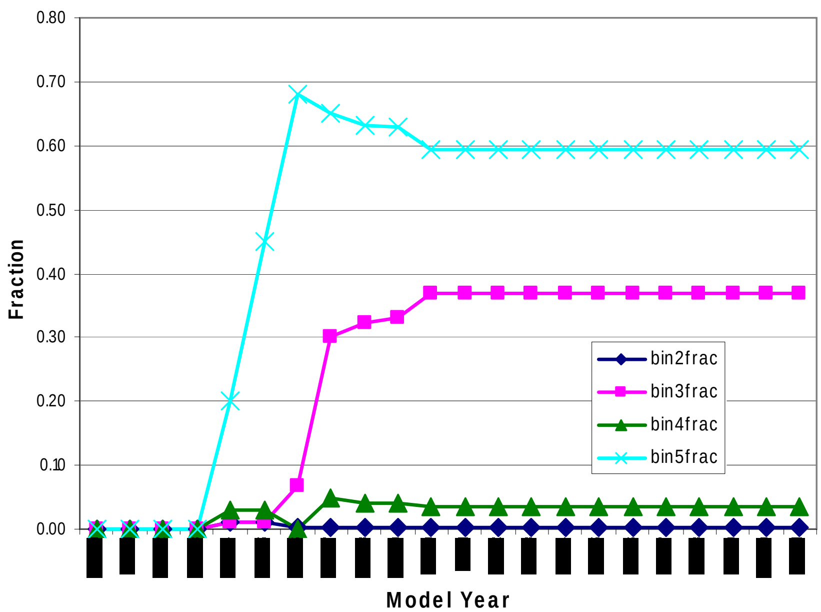


Figure 1 - 26. Phase-in Assumptions for Tier 1 and Tier 2 Vehicles, for LDT3 and LDT4.

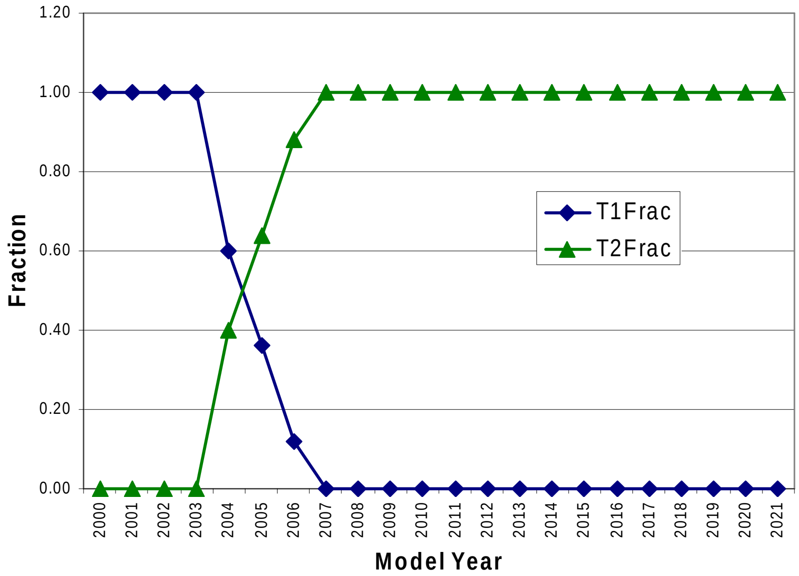
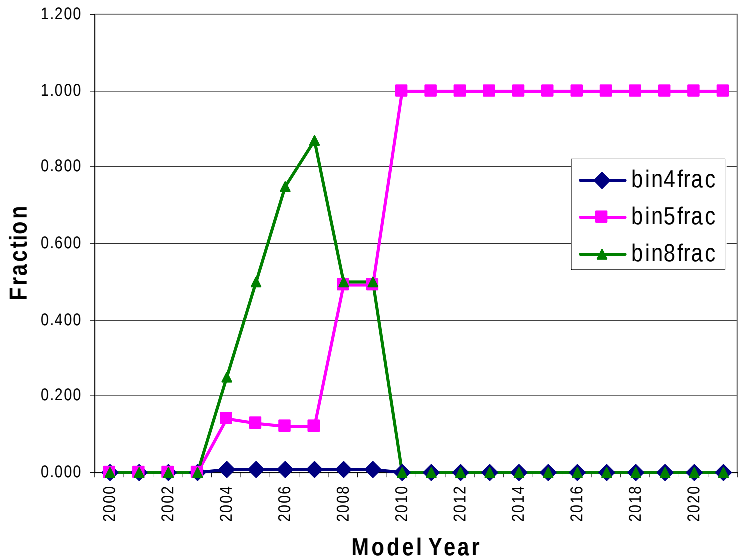


Figure 1 - 27. Phase-in Assumptions for Tier 2 Vehicles, By Bin, for LDT3 and LDT4.



1.6.2.3 Merge FTP results and phase-in Assumptions

For running emissions, the goal of this step is to calculate a weighted average of the FTP results for different tiers in each model year, with the emissions results weighted by applicable phase-in fractions. We do this step for each vehicle class separately, then we weight the four truck classes together using a set of constant fractions adapted from the MOBILE model (0.15, 0.60, 0.15, 0.10 for LDT1, 2, 3, and 4, respectively). We did not vary these fractions by model year.

Figure 1 - 28 shows an example of the Phase-in calculation for LDV NOx in four model years. The tables shows cold start and running FTP values for Tier 1, NLEV and Tier 2 standards, as well as the phase-in fractions for each standard in each model year. Start and running emissions in each model year are simply calculated as weighted averages of the emissions estimates and the phase-in fractions. The resulting weighted start estimates are used directly to represent cold-start emissions for young vehicles in each model year. For running emissions, however, the averages are not used directly; rather, each is expressed as a ratio to the corresponding Tier-1 value.

Figure 1 - 28. Example of Phase-in Calculation, LDV NOx, for four Model years.

Tier	Standard	Cold Start (g)	Running (g/mi)	Phase-In by MY 2000	2002	2005	2010
Tier 1	Tier 1	0.983	0.149	1	0	0	0
NLEV	TLEV	0.827	0.123	0	0.2	0	0
	LEV	0.569	0.052	0	0.8	0.5	0
	ULEV	0.453	0.048	0	0	0	0
Tier 2	Bin 8	0.481	0.032	0	0	0	0
	Bin 7	0.378	0.025	0	0	0	0
	Bin 6	0.275	0.018	0	0	0	0
	Bin 5	0.172	0.011	0	0	0.45	0.595
	Bin 4	0.098	0.007	0	0	0.03	0.035
	Bin 3	0.036	0.005	0	0	0.01	0.368
	Bin 2	0.049	0.00005	0	0	0.01	0.002
			Start (g)	**0.98**	**0.62**	**0.37**	**0.12**
			Running (g/mile)	0.14895	0.06577	0.0312	0.00887
			RATIO to Tier 1	***1.00***	***0.44***	***0.21***	***0.06***

Table 1 - 18 shows weighted average values for model-years 2001-2010 for simulated FTP composites, start and running emissions. The Start values, expressed as the start mass increment (g) are used directly in the MOVES emission rate table to represent cold-start emissions (operating mode 108). The composites and running emissions, expressed as rates (g/mi) are presented for comparison. For running emissions, however, the averages shown in the table are not used directly; rather, each is expressed as a ratio to the corresponding Tier-1 value, as shown in Figure 1 - 28, and in Figure 1 - 29,Figure 1 - 30 and Figure 1 - 31 below.

NOTE: We are aware of the anomalous increasing trends in running CO and HC shown in Figures 1-29 and 1-30, which are due to discontinuities in the underlying FTP results. We plan to revisit and correct this ratios before formulation of the final set of rates.

Table 1 - 18. **Weighted Average FTP Values Projected for LDT and LDV for MY 2001-2010**

regClass	MY	CO			HC			NOx		
		Composite (g/mi)	**Start (g)**	**Running (g/mi)**	**Composite (g/mi)**	**Start (g)**	**Running (g/mi)**	**Composite (g/mi)**	**Start (g)**	**Running (g/mi)**
LDT	2000	2.17	14.72	1.23	0.171	1.77	0.0587	0.298	1.34	0.184
	2001	1.81	14.15	0.848	0.140	1.45	0.0486	0.248	1.12	0.144
	2002	1.76	13.81	0.846	0.135	1.37	0.0497	0.222	1.04	0.126
	2003	1.72	13.48	0.843	0.129	1.28	0.0508	0.197	0.956	0.109
	2004	1.34	10.73	0.647	0.097	1.03	0.0337	0.144	0.726	0.0780
	2005	1.06	8.74	0.497	0.074	0.85	0.0220	0.104	0.553	0.0554
	2006	0.77	6.63	0.342	0.051	0.67	0.0102	0.0632	0.376	0.0327
	2007	0.47	4.27	0.193	0.032	0.49	0.0022	0.0286	0.206	0.0140
	2008	0.46	4.20	0.190	0.031	0.48	0.00209	0.0244	0.175	0.0120
	2009	0.46	4.18	0.189	0.031	0.47	0.00208	0.0243	0.174	0.0120
	2010	0.45	4.07	0.185	0.029	0.45	0.00197	0.0186	0.132	0.00922
LDV	2000	1.604	10.24	0.983	0.112	1.294	0.027	0.241	0.983	0.149
	2001	1.173	8.86	0.579	0.0764	0.932	0.016	0.138	0.672	0.080
	2002	1.170	8.38	0.613	0.0702	0.835	0.017	0.118	0.621	0.0662
	2003	1.167	7.91	0.647	0.064	0.738	0.018	0.0980	0.569	0.052
	2004	1.004	7.11	0.539	0.0557	0.671	0.014	0.0786	0.465	0.0415
	2005	0.857	6.46	0.435	0.0482	0.616	0.0101	0.0599	0.366	0.0312
	2006	0.693	5.68	0.325	0.0404	0.556	0.00615	0.0407	0.262	0.0208
	2007	0.430	3.92	0.180	0.0286	0.437	0.00193	0.0180	0.127	0.00899
	2008	0.420	3.83	0.176	0.0283	0.432	0.00191	0.0177	0.125	0.00886
	2009	0.418	3.80	0.175	0.0282	0.432	0.00190	0.0176	0.124	0.00884
	2010	0.402	3.65	0.169	0.0277	0.423	0.00187	0.0171	0.119	0.00863

Figure 1 - 29. Weighted Ratios for Composite, Start and Running CO Emissions, for LDT and LDV.

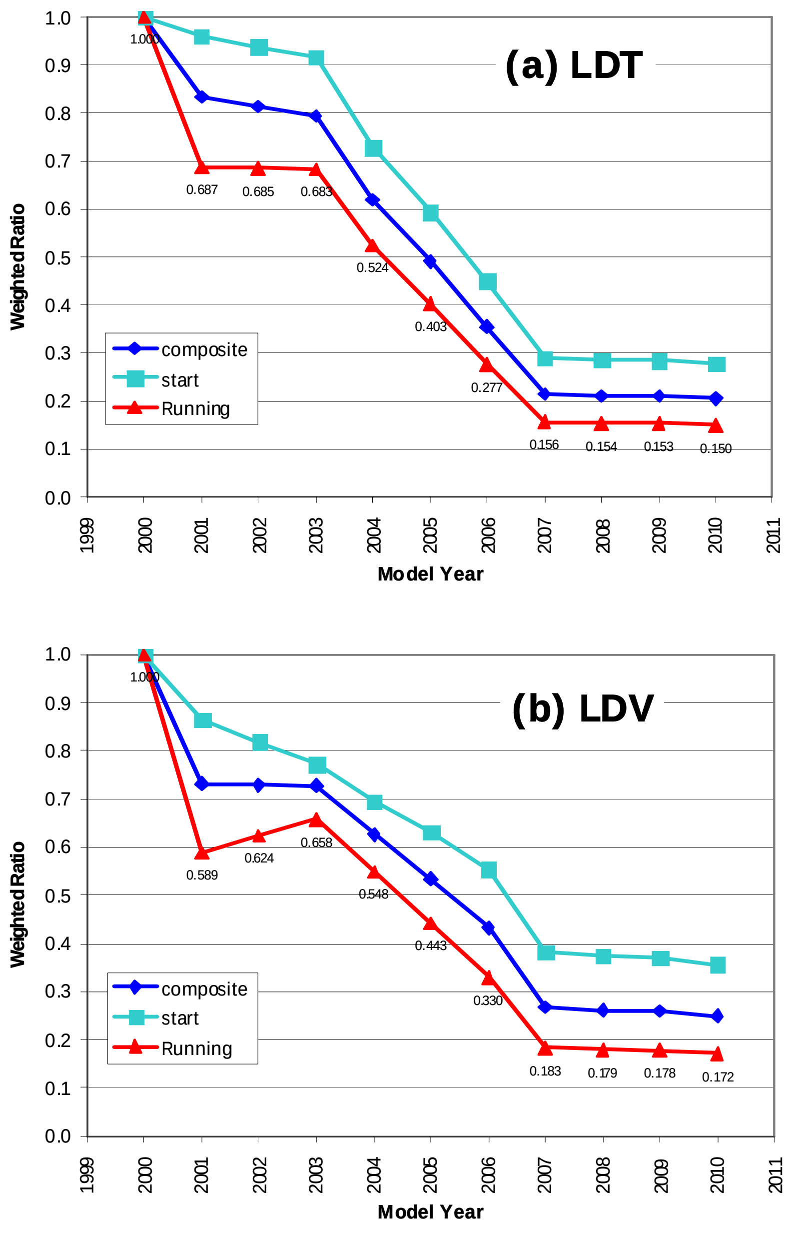


Figure 1 - 30. Weighted Ratios for Composite, Start and Running THC Emissions, for LDT and LDV.

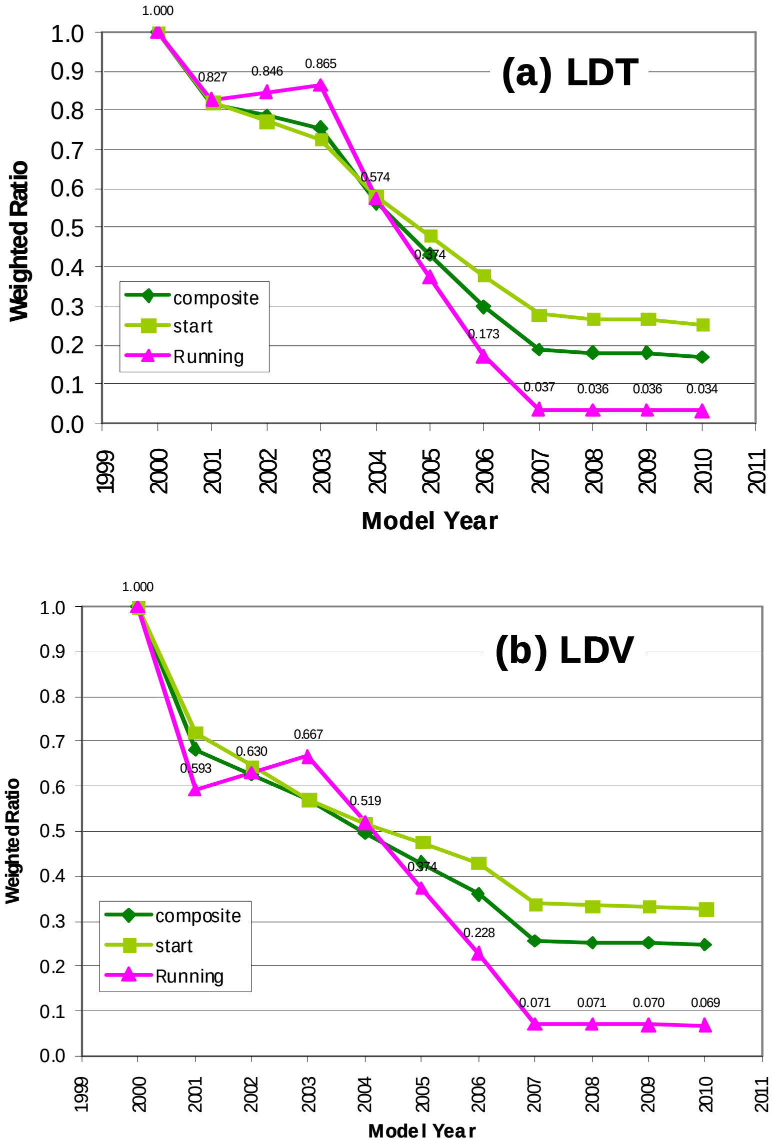
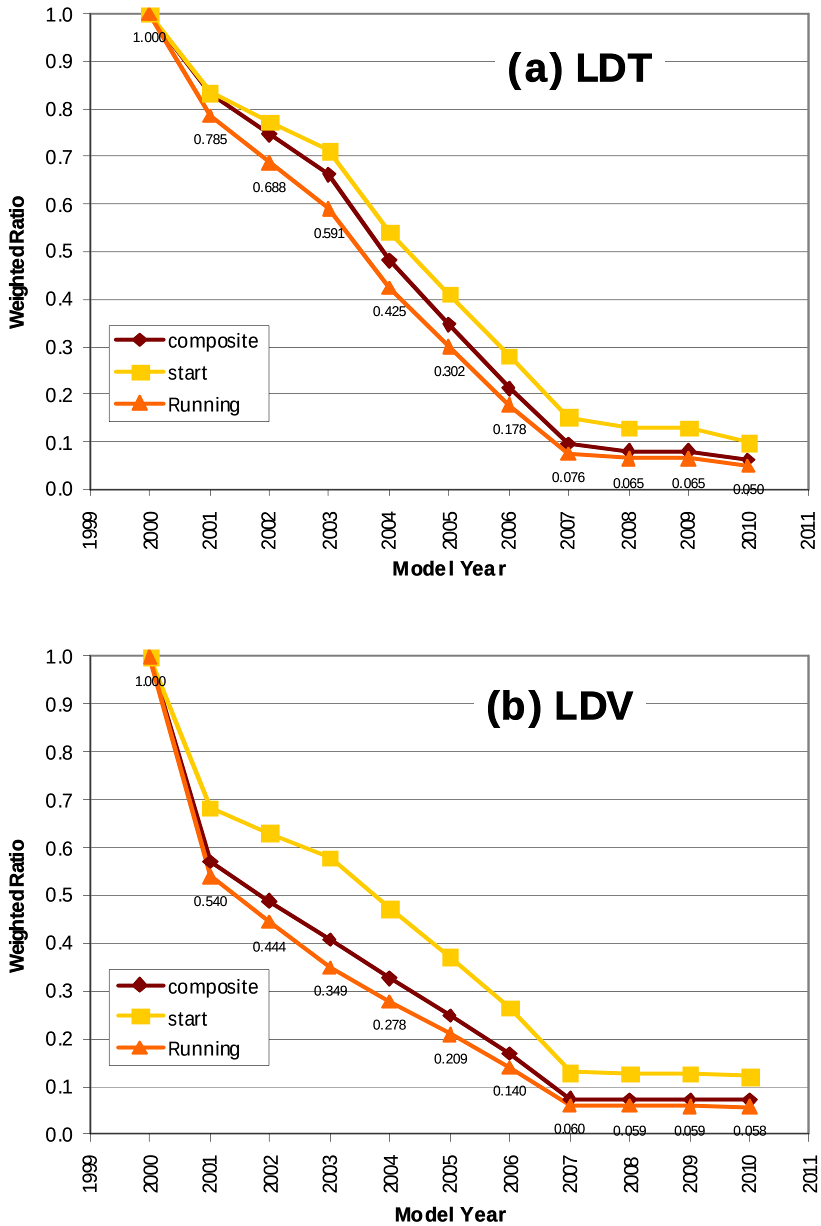


Figure 1 - 31. Weighted Ratios for Composite, Start and Running NOx Emissions, for LDT and LDV



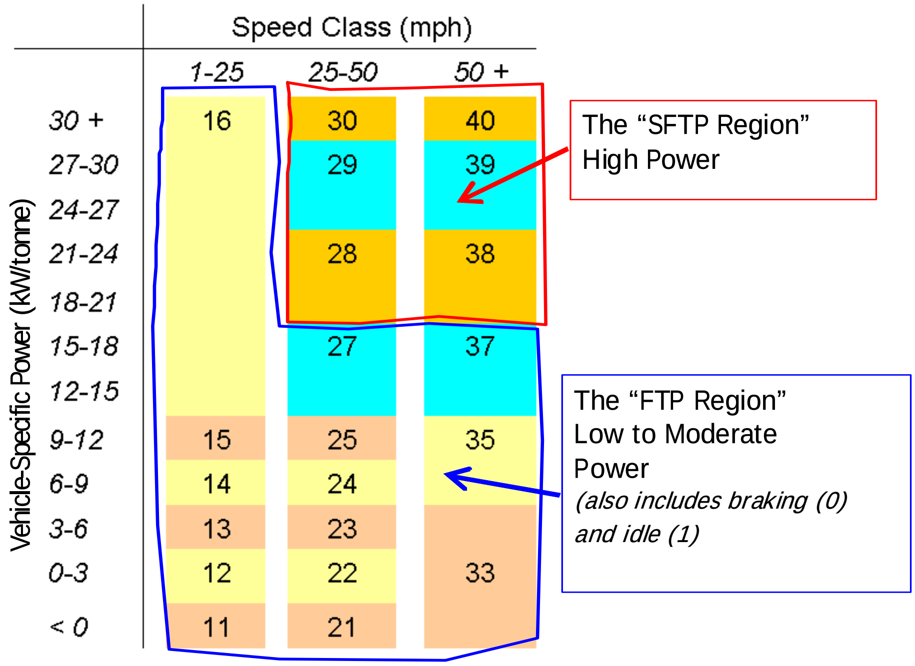
1.6.2.4 Estimating Emissions by Operating Mode

1.6.2.4.1 Running Emissions

To project emissions for NLEV and Tier-2 vehicles, we divided the operating modes for running exhaust into two groups. These groups represent the ranges of speed and power covered by the FTP standards (< ~18 kW/tonne), and the ranges covered by the SFTP standards (primarily the

US06 cycle). For convenience, we refer to these to regions as “the FTP region” and “SFTP region,” respectively (See Figure 1 - 32).

Figure 1 - 32. Operating Modes for Running Exhaust Pollutant Processes, divided into "FTP " and "SFTP" Regions.



To estimate emissions by operating mode, the approach was to multiply the emission rates for MY 2000, representing Tier 1, by a specific ratio for each model year, to represent emissions for that year. For the FTP operating modes, we applied the ratios shown in Figure 1 - 29 to Figure 1 - 31 above.

For the SFTP operating modes, we followed a different approach. At the outset, we noted that the degree of control in the FTP standards increases dramatically between MY 2000 through MY 2010, following phase-in of the Tier-2 standards. However, it was not obvious that the degree of control would increase as dramatically for the SFTP standards. Thus, in preparation of the draft rates, we adopted a conservative assumption that emissions in the SFTP region would not drop as dramatically as those in the FTP region.

It was therefore necessary to estimate a different set of ratios. However, it was not feasible to calculate ratios for the SFTP modes as described above for the FTP modes because SFTP standards do not apply to Tier 1 engines, leaving no point of reference for the ratio calculation.

So, as an initial effort for the draft, we adopted an alternative approach. Returning to the Phoenix I/M data, we pooled tests for two model-year groups, 1998-2000, representing Tier 1 vehicles not subject to SFTP requirements, and 2001-2003, representing NLEV vehicles subject to the SFTP. For each group, we calculated means for each pollutant for the SFTP operating modes (as a group), and calculated ratios between the two groups.

$$R_{\mathrm{SFTP}} = \frac{\overline{E}_{\mathrm{poll,SFTP,01-03}}}{\overline{E}_{\mathrm{poll,SFTP,98-00}}} \qquad \textbf{1 - 40}$$

The resulting ratios are 0.55, 0.35 and 0.30 for CO, THC and NOx, respectively. These values exceed those for the FTP modes for MY ~2004 and later. However, the calculation and application of these values is considered preliminary and will be reevaluated for development of the final MOVES rates.

Figure 1 - 33 and Figure 1 - 34 show application of the ratios to the FTP and SFTP operating modes in model years 2000 (the reference year), 2005, and 2010, both calculated with respect to 2000. The ratios shown in Figure 1 - 29 to Figure 1 - 31 are used for the FTP modes, and the uniform values R_{SFTP} are used for the SFTP modes. Note that the values for the SFTP modes are equal in 2005 and 2010, because the SFTP ratios are constant by model year. The results are presented on both linear and logarithmic scales. The linear plots display the differences in the high-power modes, but obscure those in the low-power modes. The logarithmic plots supplement the linear plots by making visible the relatively small differences between MY 2005 and 2010 in the lower power modes.

NOTE: the assumptions described in this section as slated for review and potential revision before releases of the final model.

Figure 1 - 33. Projected Emission Rates by Operating Mode for Age group 0-3 years, in three Model Years (LINEAR SCALE).

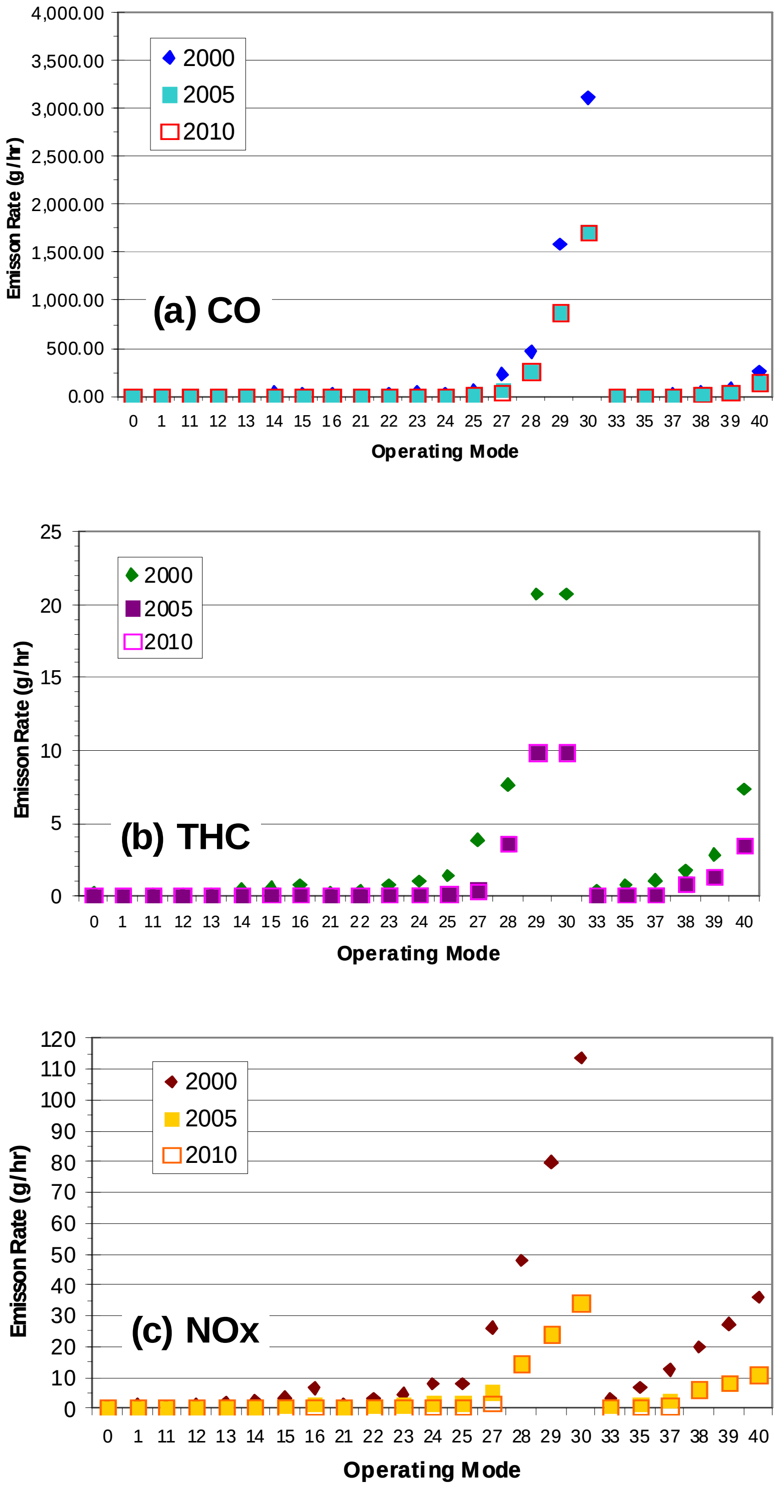
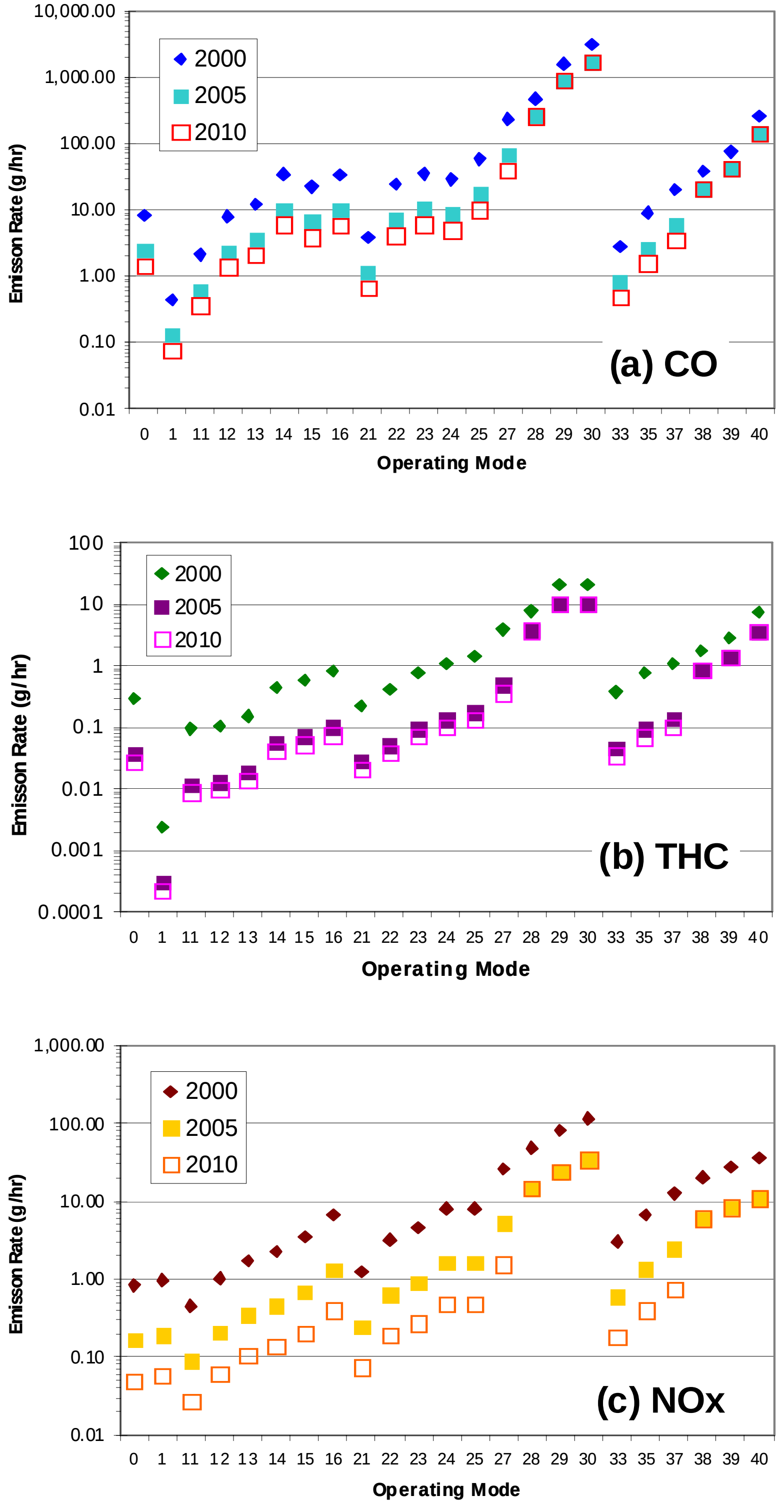


Figure 1 - 34. Projected Emission Rates by Operating Mode for Age group 0-3 years, in three Model Years (LOGARITHMIC SCALE).



1.6.2.4.2 Start emissions

As mentioned, the values for "start" shown above in Table 1 - 18 are assumed to represent cold-start emissions, denoted by opModeID 108, and defined as start emission following a soak period of 12 hours or longer (720 min). Additional operating modes for starts are defined in terms of shorter soak periods, as shown in Table 1 - 19.

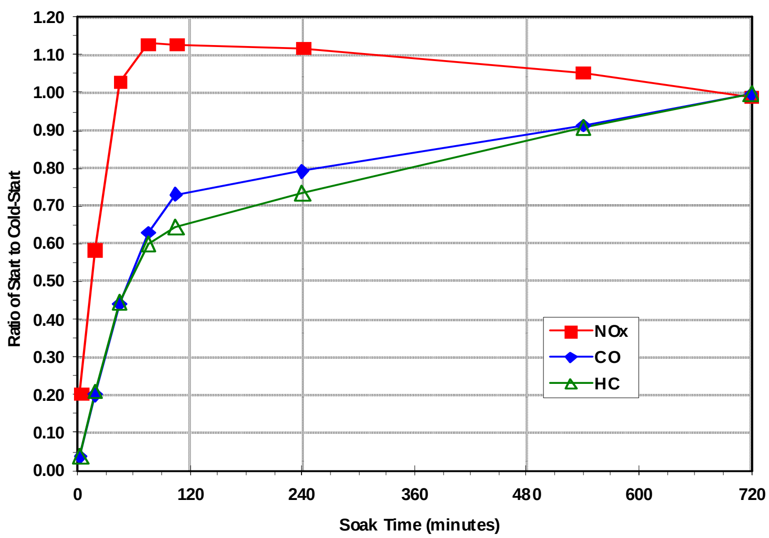
Table 1 - 19. Operating-Mode Definitions for Start Exhaust Emissions

opModeID	**Description**	**Soak Period (min)**[1]
101	"hot start"	≤ 6
102		6 - 30
103		30 – 60
104		60 – 90
105		90 – 120
106		120 – 360
107		360 - 720
108	"cold start"	≥ 720

[1] Defined in terms of *lower-bound* ≤ soak period < *upper-bound*.

To estimate start emissions for the other operating modes, we applied "soak fractions" to the "cold-start" emissions. The soak fractions were adapted from the approach applied in the MOBILE model[13]. Specifically, the part-wise regression equations used in MOBILE were evaluated at the midpoint of the soak period for each operating mode. For each mode, the start rate is the product of the cold-start rate and the corresponding soak fraction. Figure 33 shows the soak fractions for HC, CO and NOx, with each value plotted at the midpoint of the respective soak period.

Figure 1 - 35. Soak Fractions Applied to Cold-Start Emissions (opModeID = 108) to Estimate Emissions for shorter Soak Periods (operating modes 101-107).



1.6.2.5 Apply Deterioration

Based on review and analysis of the Phoenix I/M data, we assume that deterioration for different technologies is best represented by a multiplicative model, in which different technologies, represented by successive model-year groups, show similar deterioration in relative terms but markedly different deterioration in absolute terms. We implemented this approach by translating emissions for the 0-3 age Group, as calculated above, into natural logarithms and applying uniform logarithmic age trends to all model-year groups. We derived logarithmic deterioration slopes for Tier-1 vehicles (MY 1996-98) and applied them to NLEV and Tier-2 vehicles. In this process we applied the same logarithmic slope to each operating mode, which is an extension of the multiplicative deterioration assumption.

Note that we applied deterioration only to the running emissions, not to the start emissions. We carry this process out in four steps.

1.6.2.5.1 Recalculate the logarithmic mean

Starting with the values of the arithmetic mean (x_a) calculated above, we calculate a logarithmic mean (x_l), as shown in Equation 1 - 41. Note that this equation is simply a rearrangement of Equation 1 - 27.

$$\bar{x}_l = \ln \bar{x}_a - \frac{\sigma_l^2}{2} \qquad \textbf{1 - 41}$$

The values of the logarithmic variance are intended to represent values for young vehicles, as the estimates for x_a represent the 0-3 year age Group. The values applied were 1.30, 0.95 and 1.6 for CO, THC and NOx, respectively.

1.6.2.5.2 Apply a logarithmic Age slope

After estimating logarithmic means for the 0-3 age class ($x_{l,\text{0-3}}$), we estimate additional logarithmic means for successive age classes ($x_{l,\text{age}}$), by applying a linear slope in ln-space (m_l).

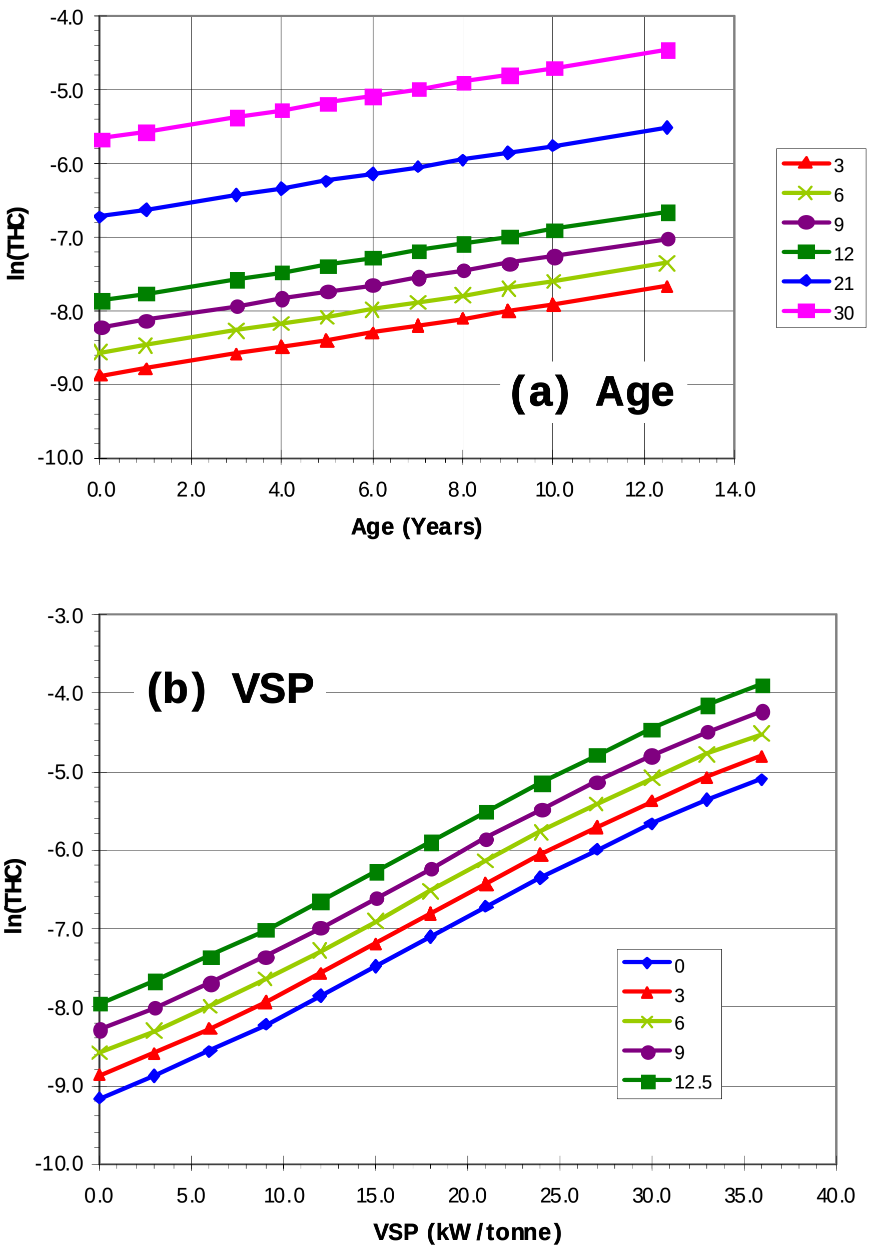
$$\bar{x}_{l,\text{age}} = \bar{x}_{l,0-3} + m_l(\text{age} - 1.5) \qquad \textbf{1 - 42}$$

The values of the logarithmic slope are adapted from values developed for the 1996-98 model – year group. The values applied were 0.18, 0.15 and 0.17 for CO, THC and NOx, respectively. When calculating the age inputs for this equation, we subtracted 1.5 years to shift the intercept to the midpoint of the 0-3 year age Group.

Figure 1 - 36 shows an example of the approach, as applied to THC from LDV in the 1996-98 model-year group. The upper plot (a) shows lnTHC vs Age, by VSP, where the VSP acts as a surrogate for operating mode. The defining characteristics of the plot are a series of parallel lines, with the gaps between the lines reflecting the magnitude of the VSP differences between them. Similarly, the lower plot shows lnTHC vs. VSP, by Age, where age acts as a surrogate for

deterioration. In this view, deterioration appears as the magnitude of the gaps between a family of similar trends against power.

Figure 1 - 36. Example of Logarithmic Deterioration Model for THC (LDV, MYG 96-98): (a) lnTHC vs. Age, by VSP level (kW/tonne), (b) lnTHC vs. VSP, by Age (yr).



1.6.2.5.3 Apply the reverse transformation

After the previous step, the values of $x_{l,\text{age}}$ were reverse-transformed, as in Equation 1 - 27. The values of the logarithmic variance used for this step were adapted from the Phoenix I/M results and are intended to represent emissions distributions for "real-world" vehicle populations, meaning that the values are higher than the value used in step 1.6.2.5.1 and may vary with age. Values of logarithmic variances for all three pollutants are shown in Table 17.

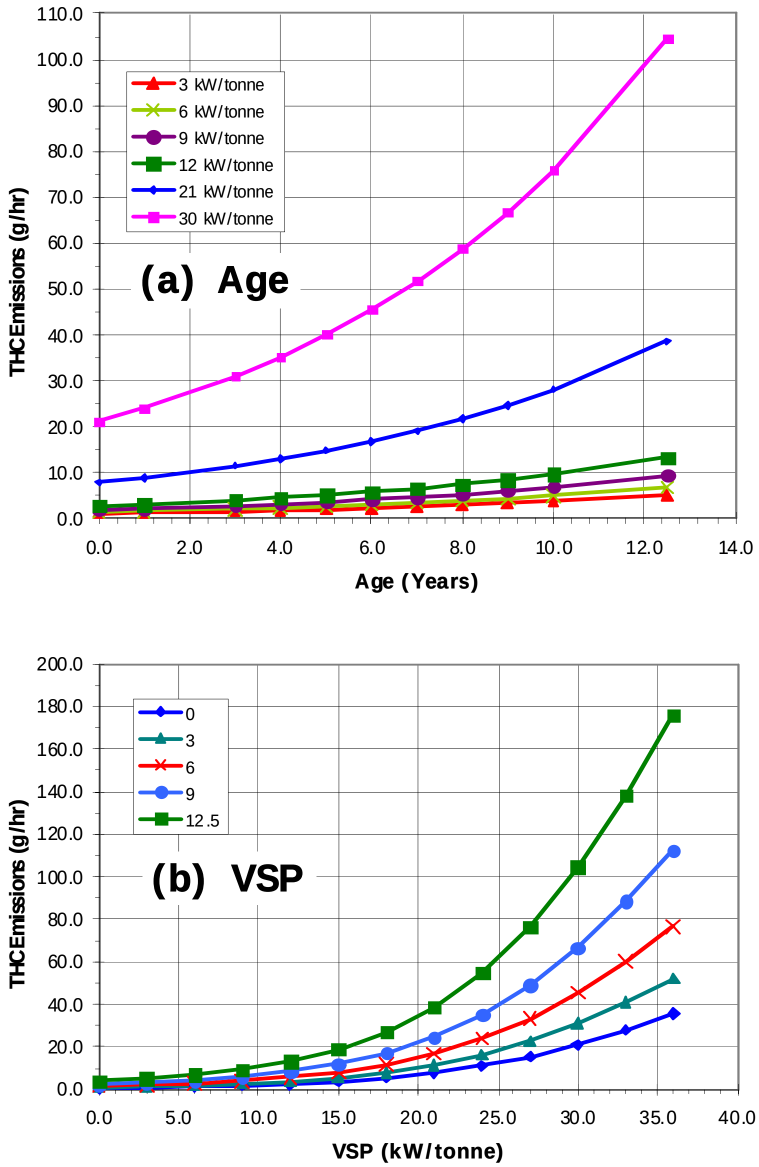
Table 1 - 20. Values of Logarithmic Variance Used to Calculate Emissions Deterioration by Reverse Transformation of Logarithmic Means.

Age Group	**Pollutant**		
	CO	**THC**	**NOx**
0-3 years	2.5	1.50	1.95
4-5	2.7	1.80	2.10
6-7	2.7	2.00	2.00
8-9	2.7	2.10	2.00
10-14	2.7	2.10	2.00

No values are presented in Table 1 - 20 for the 15-19 and 20+ year age Groups. This omission is intentional, in that we did not want to extrapolate the deterioration trend beyond the 10-14 year age Group. Extrapolation beyond this point is incorrect, as we know that emissions tend to stabilize beyond this age, while the ln-linear emissions model would project an increasingly steep and unrealistic exponential emissions trend.

Figure 1 - 37 shows the same results as Figure 1 - 36, following reverse transformation. The families of parallel logarithmic trends are replaced by corresponding "fans" of diverging exponential trends. An implication of this model is that as deterioration occurs, it is expressed more (in absolute terms) at high power. Similarly, the relationship between emissions and VSP becomes more pronounced with increasing age.

Figure 1 - 37. Example of Reverse Transformation for THC (LDV, MYG 96-98): (a) THC vs. Age, by VSP level (kW/tonne), (b) THC vs. VSP, by Age (yr).



1.6.2.6 Estimate non-I/M References

Completion of steps 1.6.2.1 – 1.6.2.6 provided a set of rates representing I/M reference rates for MY 2001-2021. As a final step, we estimated non-I/M reference rates by applying the same ratios applied to the I/M references for MY 2000 and previous, as described above.

1.7 Replication and Data-Source Identification

The rates developed as described in Sections 2 and 3 represent gasoline-fueled conventional-technology engines. For purposes of the draft version of the emissionRateByAge table, we replicated these rates to represent other fuels and technologies.

At the outset, we replicated the entire set of gasoline rates for ethanol (blends?) In addition, we replicated all the gasoline rates for the advanced engine technologies. The fuel types and engine technologies represented in the table segment are listed in **Error! Reference source not found.**.

Table 1 - 21 Fuel Types and Engine Technologies Represented for Criteria-pollutant Emissions from Light-Duty Vehicles.

Attribute	sourceBin attribute	Value	Description
Fuel type	fuelTypeID	01	Gasoline
		02	Diesel
		05	Ethanol
Engine Technology	engTechID	01	Conventional internal combustion (CIC)
		02	Advanced internal combustion (AIC)
		11	Moderate hybrid – CIC
		12	Full hybrid – CIC
		20	Hybrid – AIC
		21	Moderate hybrid – AIC
		22	Full hybrid - AIC

Throughout the process, we assigned dataSourceIDs to subgroups of rates, which identify the data and methods used for particular rates. The dataSourceIDs developed for these analyses are listed and described in Table 1 - 22.

Table 1 - 22. Descriptions of Data Sources and Methods used in Development of Criteria-Pollutant Emission Rates for Light-Duty Vehicles.

DataSourceID	Description
4400	Data driven rates: averaged from second-by-second IM240/IM147 data from Phoenix random evaluation sample, CY1995-99 and CY2002-05, on temperature range of 68-86 °F.
4427	replaces a 4400 value for opModes 28-30, calculated by ratio relative to opMode 27
4437	replaces a 4400 value for opModes 38-40, calculated by ratio relative to opMode 37
4500	imputed using statistical hole-filling models.
4527	replaces a 4500 value for opModes 28-30, calculated by ratio relative to opMode 27
4537	replaces a 4500 value for opModes 38-40, calculated by ratio relative to opMode 37
4601	calculated by ratio relative to ageGroup 10-14 (modelyeargroups 2000 and previous only, ageGroupID 15-19 and 20+ only)

4602	calculated by ratio relative to ageGroup 15-19, modelyeargroups 2000 and previous only, ageGroupID 20+ only, meanBaseRate only (corresponding meanBaseRateIM is 4601).
4800	calculated by ratio from MY2000 rates, with ratios calculated from IUVP FTP Bag-2 data, (modelyeargroups 2001 and later only ageGroup 0-3 only).
4801	calculated by applying deterioration to 4800 values, (modelyeargroups 2001 and later only, ageGroups 4-5 through 10-14)
4802	calculated by ratio relative to ageGroup 10-14 (modelyeargroups 2001 and later / ageGroupID 15-19 and 20+ only).
4803	calculated by ratio relative to ageGroup 15-19, modelyeargroups 2001 and later / ageGroupID 20+ only, meanBaseRate only (corresponding meanBaseRateIM is 4802).
4805	calculated from IUVP FTP results, as Bag 1 - Bag 3 mass (cold start, opMode 108 only, ageGroup 0-3 only).
4806	calculated by applying deterioration ratios to 4805 values (cold start, opMode 108 only, ageGroup 4-5 and older).
4807	calculated by applying soak fractions and deterioration ratios to 4805 values (opModes 101-107 only, all ageGroups).
4900	replicated from gasoline rates (fueltypeid = 1) to represent ethanol blends (fueltypeid = 5).
4901	replicated from gasoline or ethanol rates with conventional internal combustion to represent rates for advanced engine technologies.
4910	replicated from gasoline rates for all engine technologies to represent rates for tier-2 light-duty diesel engines (MY 2010 and later only).

Finally, Table 1 - 23 shows the accounting for all rates developed for light-duty criteria-pollutant emissions and included in the draft emissionRateByAge table. The leftmost four columns delineate subsets of rates by the pollutant processes included (Running, Start), and the respective fueltypes, engtechs and dataSourceIDs. The next seven “accounting” columns show the construction of subtotals corresponding to combinations of fueltype, engtech, and dataSource. The values in these columns represent numbers of groups or categories covered, i.e., two regClasses always refers to LDV and LDT.

The rates for datasourceID = 4400 – 4602 were summed as a single category, as these groups represent the outcome of a set of interrelated processes, as described in section 1.5. The count of 15 modelyeargroups includes groups through MY 2000 (see Table 1 - 11). The dataSourceIDs 4800 – 4803 represent running emissions for MY 2001+, as described in Section 1.6. The total of 21 modelyeargroups represent groups 2001 – 2021-2050. For these rows, a count of one agegroup refers to the 0-3 year ageGroup, whereas a count of four refers to the 4-5, 6-7, 8-9 and 10-14 year age Groups.

DataSourceIDs 4805 – 4807 represent start emissions for MY2001-2021. For this group, counts of 26 or 36 modelyeargroups denote MYG 1996-2021 and 1980 and earlier - 20212050, respectively. Counts of one or six ageGroups refer to the 0-3 ageGroup and the remaining six ageGroups, respectively. Counts of one or seven opModes refer to the cold-start emissions (opmode 108) or the remaining seven start modes, respectively.

The dataSourceIDs 4900 and 4901 refer to the replication of the gasoline/conventional rates for ethanol and the advanced engine technologies, respectively. The Count of 36 modelyeargroups includes all groups from 1980 & earlier through 2021-2050. The count of 31 opModes includes all modes for both the start and running processes. The counts for dataSourceID 4910 is similar, except that the 12 modelyeargroups include only 2010-20212050, as mentioned.

Table 1 - 23. Accounting for the Segment of the Draft emissionRateByAge Table contributed by Rates for Criteria-Pollutant Emissions, Light-Duty Vehicles.

Process(es)	fuelTypeID	engTechID	dataSourceID	Accounting (No. classes or groups)							No. records
				fueltypes	engTechs	regClasses	MYG	ageGroups	opModes	polProcesses	
Start	01	01	101	1	1	2	10	1	1	3	60
Running	01	01	4400	1	1	2	15	7	23	3	14,490
Running	01	01	4427								
Running	01	01	4437								
Running	01	01	4500								
Running	01	01	4527								
Running	01	01	4537								
Running	01	01	4601								
Running	01	01	4602								
Running	01	01	4800	1	1	2	21	1	23	3	2,898
Running	01	01	4801	1	1	2	21	4	23	3	11,952
Running	01	01	4802	1	1	2	21	1	23	3	2,898
Running	01	01	4803	1	1	2	21	1	23	3	2,898
Start	01	01	4805	1	1	2	26	1	1	3	156
Start	01	01	4806	1	1	2	36	6	1	3	1,296
Start	01	01	4807	1	1	2	36	7	7	3	10,584
SUBTOTAL											46,872
Running & start	05	01	4900	1	1	2	36	7	31	3	46,872
Running & start	01	02	4901	1	1	2	36	7	31	3	46,872
Running & start	01	11	4901	1	1	2	36	7	31	3	46,872
Running & start	01	12	4901	1	1	2	36	7	31	3	46,872
Running & start	01	20	4901	1	1	2	36	7	31	3	46,872
Running & start	01	21	4901	1	1	2	36	7	31	3	46,872
Running & start	01	22	4901	1	1	2	36	7	31	3	46,872
Running & start	05	02	4901	1	1	2	36	7	31	3	46,872
Running & start	05	11	4901	1	1	2	36	7	31	3	46,872
Running & start	05	12	4901	1	1	2	36	7	31	3	46,872
Running & start	05	20	4901	1	1	2	36	7	31	3	46,872
Running & start	05	21	4901	1	1	2	36	7	31	3	46,872
Running & start	05	22	4901	1	1	2	36	7	31	3	46,872
Running & start	02	01	4910	1	1	2	12	7	31	3	15,624
Running & start	02	02	4910	1	1	2	12	7	31	3	15,624
Running & start	02	11	4910	1	1	2	12	7	31	3	15,624
Running & start	02	12	4910	1	1	2	12	7	31	3	15,624
Running & start	02	20	4910	1	1	2	12	7	31	3	15,624
Running & start	02	21	4910	1	1	2	12	7	31	3	15,624
Running & start	02	22	4910	1	1	2	12	7	31	3	15,624
TOTAL											765,576

2. Particulate-Matter Emissions from Light-Duty Vehicles

2.1 Introduction and Background

There is currently a large body of research on the formation and measurement of Particulate Matter (PM) emissions from combustion engines. This chapter describes the process by which emissions measured in a subset of the past PM research programs from light-duty gasoline vehicles was employed to generate emission rates for MOVES. The emission rates determined by this approach embody strictly “bottom-up” methodology whereby emission rates are developed from actual vehicle measurements following intensive data analysis, and are then input into an emissions inventory model. This is in contrast to a “top-down” approach which uses measurements of ambient PM concentrations from local regions and may apportion these emissions to vehicles (and other sources), which are then input into inventory models.

The primary study that this chapter relies on is the “Kansas City Characterization Study” conducted in 2004-2005[14]. The Environmental Protection Agency and several research partners conducted this study to quantify tailpipe particulate-matter emissions from gasoline-fueled light duty vehicles in the Kansas City Metropolitan Area. This study is the most comprehensive and representative study of its kind. In the context of a rigorous recruitment plan, strenuous efforts were made to procure a representative sampling of the fleet. During the summer phase, 261 vehicles were tested, while 278 vehicles were tested in the winter. The testing was conducted on a portable dynamometer using the LA92 driving cycle in ambient temperature conditions.

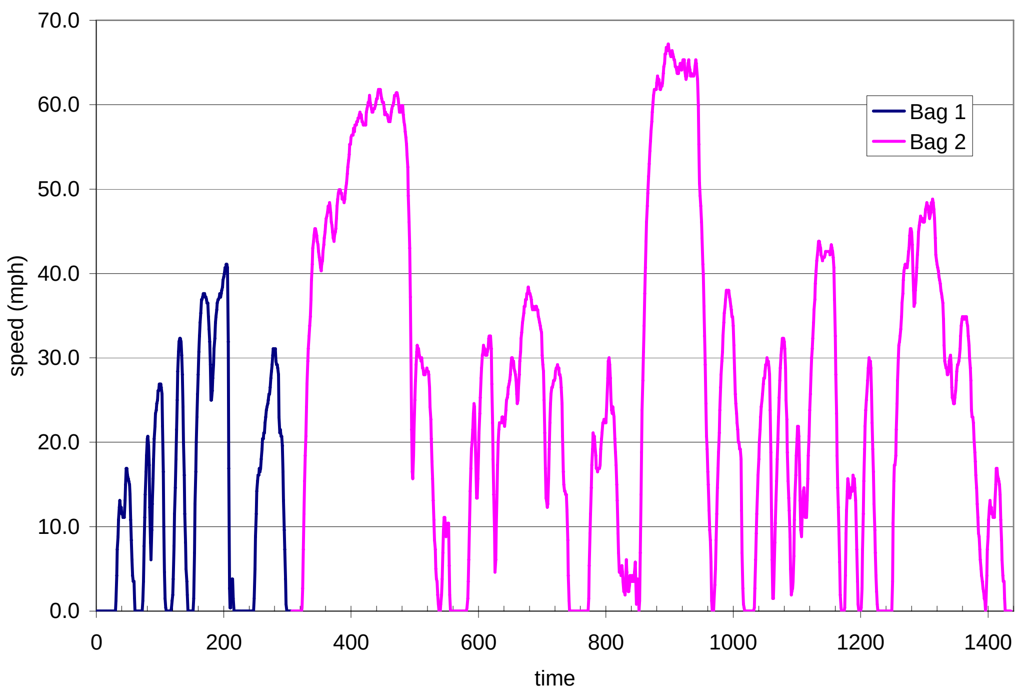
Much of the data from this program was analyzed in the report: “Analysis of Particulate Matter Emissions from Light-Duty Gasoline Vehicles in Kansas City”[15] This “analysis report” (which is the partner to this “modeling” chapter) presented preliminary emission rates for PM, elemental carbon fraction (EC), organic carbon fraction (OC), as well as temperature adjustment factors for start as well as hot running emissions processes for MOVES. These emission rates form the basis for the emission rates developed in this chapter. The rates from the analysis report were based on the aggregate or “bag” emissions measured on the PM filters in the program, thus they are in units of grams per start for starts and grams/mile for hot running operations. The rates were inclusive of gasoline powered light-duty vehicles of all ages, but only modeled calendar year 2005, when the emission rates were actually measured. The measurement program could say very little of what the emission rates of vehicles were in, say 1990, or will be in 2020 because the vehicle fleet looks very different in these other calendar year scenarios. This chapter describes the development of a deterioration model based on a comparison of past PM studies with the 2005 Kansas City study. The rates from this deterioration model include calendar years in the past, present and future as required by MOVES.

The analysis in the 2008 paper also did not describe how the emissions would change if the driving differed from the LA92 (unified) drive cycle used in the program as it would in the real-world. MOVES has the capability to capture hot running “modal” emission rates so that emissions vary by the Vehicle Specific Power (VSP) of the vehicle. This chapter describes how the real-time PM measurements collected in the study were used to populate the VSP modal rates for MOVES. Because of the reliance on real-time PM measurement, it is worth describing the measurement procedures used in Kansas City.

2.1.1 Particulate Measurement in the Kansas City Study

For measurements conducted on the dynamometer, vehicles were operated over the LA92 Unified Driving Cycle (see Figure 2 - 1). The LA92 cycle consists of three phases or "bags". Phase 1 ("bag 1") is a "cold start" that lasts the first 310 seconds (1.18 miles). "Cold start" is technically defined as an engine start after the vehicle has been "soaking" in a temperature controlled facility (typically ~72°F) with the engine off. In the Kansas City study, the vehicles were soaked over night in ambient conditions. Phase 1 is followed by a stabilized Phase 2 or "hot running" (311 – 1427 seconds or 8.63 miles). At the end of Phase 2, the engine is turned off and the vehicle is allowed to "soak" in the test facility for ten minutes. At the end of the soak period, the vehicle is started again, and is driven on the same driving schedule as Phase 1. This Phase 3 is called a "hot start" because the vehicle is started when the engine and after-treatment systems are still hot. Criteria pollutants were measured both in continuous and aggregate modes. Particulate was collected during each of the three phases on 47 mm Teflon filters at 47°C ± 2°C.

Figure 2 - 1. Phases 1 and 2 of the LA92 Cycle, "cold-start" and "hot-running," respectively.



In addition to the regulated gas pollutants measured via the constant-volume sampler (CVS), continuous measurements of PM mass were taken using an EPA-supplied Booker Systems Model RPM-101 QCM manufactured by Sensors, Inc. and a Thermo-MIE Inc. DataRam 4000 Nephelometer. An estimate of black carbon was measured continuously with a DRI photoacoustic instrument and integrated samples were collected and analyzed by DRI for PM gravimetric mass, elements, elemental and organic carbon, ions, particulate and semi-volatile organic compounds, and volatile organic air toxics. All sampling lines were heated and maintained at 47°C ± 2°C. The samples were extracted from the dilution tunnel through a low particulate loss 2.5 μm cutpoint pre-classifier. Further details and a schematic of the sampling instrumentation are shown in Figure 2 - 2 and Figure 2 - 3.

Figure 2 - 2. Schematic of the constant-volume sampling system used in the Kansas-City Study.

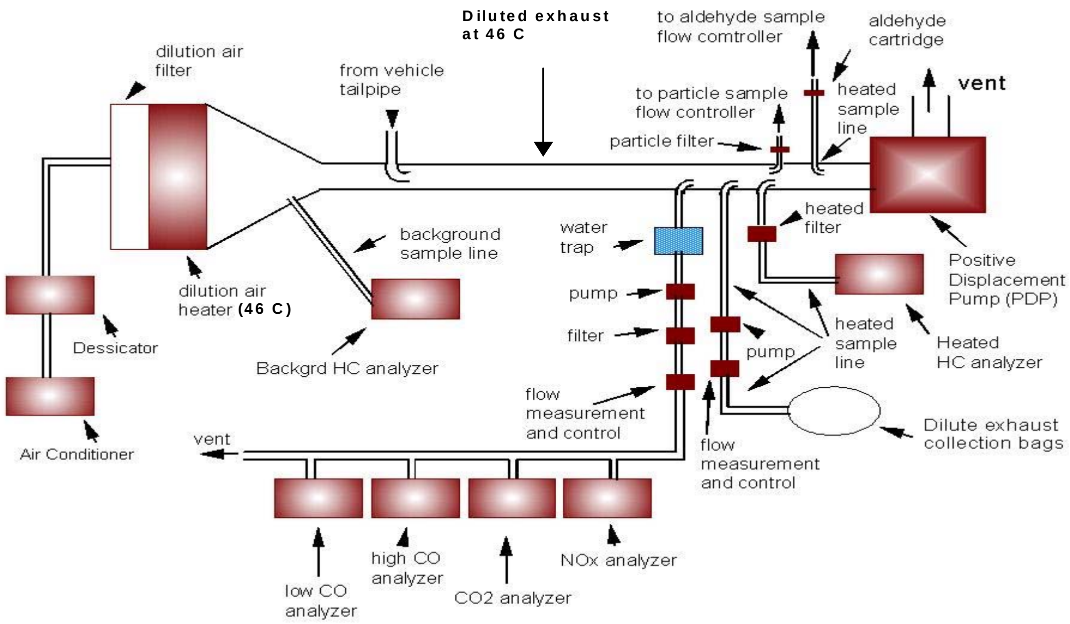
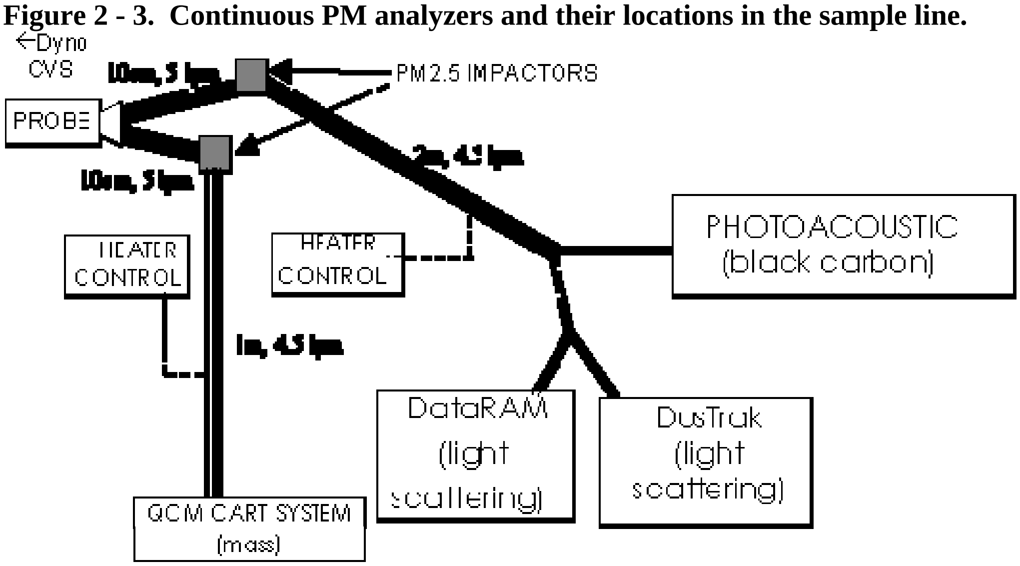


Figure 2 - 3. Continuous PM analyzers and their locations in the sample line.



It is worth briefly describing the real-time PM measurement apparatus used in the study. A more thorough description maybe found in the contractor's report[14]. As of the date of this measurement program, there existed no perfect means of measuring real-time PM. Each of the devices has specific advantages and disadvantages. For this study, it has been assumed that the cumulative mass as measured (weighed) on the Teflon filters is the benchmark. Thus all real-time measurements have been normalized to the filters in order to minimize systematic instrument errors.

The Quartz Crystal Microbalance measures the cumulative mass of the PM deposited on a crystal face by measuring the change in oscillating frequency. It is highly sensitive to many artifacts such as water vapor and desorption of lighter organic constituents. Due to the high degree of noise in the continuous time series, the measurements were averaged over 10 seconds, thus diluting the temporal effects of transients. The QCM can accurately capture cumulative PM that collects over time, however the measurement uncertainties increase at any given moment in time because they are dependent on a calculation difference between two sequential, and similar, measurements. Due to the resulting high variability, including large and rapid fluctuations from positive to negative emissions at any given instant, and vice versa, QCM measurements were not viewed as practical for use with MOVES at this time, except as a check for the other instruments.

The Dustrak and Dataram both work on light-scattering principles. As such, they have very rapid response times and can measure larger PM volumes with reasonable accuracy. However, their accuracy degrades when measuring low PM volumes. Since most PM mass lies within the larger particles, the instruments should be able to capture most of the real-time mass concentrations though it may miss a substantial portion of the smaller (nano) particles. To provide a qualitative check on supposition, the time-series for the QCM and optical instruments aligned and checked to ensure that significant mass was not missed. Based on this analysis, the Dustrak instrument was observed to be the more reliable of the 3 instruments, and mass correction at low loads was not judged to be worth the effort given the uncertainties involved. This time-consuming analysis was done by eye for each test and the results are not presented in this chapter.

The photoacoustic analyzer (PA) is unique among the real-time instruments in its ability to capture only the soot or elemental carbon components of PM. The fast analyzer detects the resonances coming off the carbon-carbon bonds in soot. It has been validated in a number of studies[references]. Unfortunately, there were insufficient Thermal Optical Reflectance (TOR) elemental carbon (EC) measurements from quartz filters to normalize the PA data, but some comparisons are shown in the contractor's report[14]. In this study, the PA data were compared qualitatively with the Dustrak and Dataram and found to be consistent with expected ratios of elemental to total carbon during transient events, leading to the conclusion that these instruments were largely consistent with one another. These results are also not presented in this chapter as every single trace was compared by eye. The data is used to determine the modal relationship of elemental to total PM.

Due to the uncertainty of experimental measurement techniques for real-time PM at the time of the Kansas City study, these instruments are employed only as a semi-qualitative/quantitative means of determining modal emission rates, and the use of such data do not qualify them as EPA recommended or approved devices or processes.

2.1.2 Causes of Gasoline PM Emissions

Particulate matter is formed from gasoline-fueled engines originating from incomplete fuel and oil combustion (although the amount of oil consumed in combustion and its contribution to PM varies greatly from vehicle to vehicle). During operation, numerous distinct technologies used in vehicles are in various states of repair or disrepair which also affect PM emissions. Even brand new vehicles emit PM from combustion but at very low levels. A complete description of the causes of PM emissions and associated mechanisms is beyond the scope of this report, as many aspects of the science that are still not well understood. We will briefly summarize factors that contribute to gasoline PM in the vehicle fleet in this section. Where appropriate, we will also compare to the mechanisms of hydrocarbon (HC) formation, since parallels are often drawn in the literature.

Simply put, particulate matter forms primarily during combustion when carbon-containing molecules condense or otherwise form particulate. This PM is generally ccomposed of higher molecular weight hydrocarbon compounds, some of which originate in the fuel/oil and some of which are formed during combustion. Unlike diesel engines, elemental (molecular) carbon or soot is not very prevalent with gasoline engines as compared to diesels but does form in larger quantities under relatively rich air:fuel ratios. The amount of elemental carbon in PM varies from vehicle to vehicle (and, even for a given vehicle, varies depending on operating conditions and state of repair). For gasoline-fueled vehicles, a typical number is about 20% of PM mass compared to about 70% for a diesel engine. There are also other compounds in the fuel or engine oil such as trace levels of sulfur and phosphorus which, in combustion, form sulfates and phosphates, both of which form particulate. The sulfur level in gasoline is now very low, almost eliminating sulfate formation from gasoline sulfur content but motor oil contains significant sulfur (and phosphorus) compounds. Also, trace metal constituents in gasoline and oil form PM in the combustion process as metallic oxides, sulfates, nitrates, or other compounds. Catalyst attrition products from the substrate and trace amounts of noble metals also form PM but not in the combustion process. The catalyst attrition products are mechanically generated and are usually in larger size ranges compared to exhaust PM. Exhaust PM as formed in the engine is generally very small in size (possibly much of it is nuclei mode PM in the range of 0.05 microns or smaller). In the exhaust system, including the muffler, some of the PM agglomerates and increases in size.

The wide assortment of technologies used in vehicles can affect PM formation. These technologies were mainly developed to control HC, CO and NOx emissions, but most have the side benefit of reducing PM, since reducing exhaust HC generally also reduces exhaust PM although not to the same extent. Older engines from the 1980s and earlier that deliver fuel through a carburetor typically have poorer fuel droplet quality, as well as looser control of fuel air stoichiometry. These older vehicles are expected to produce more PM (on average) then their fuel injected counterparts that followed generally in the late 1980s and early 1990s.

Among fuel injected engines, throttle body fuel injection (TBI) used in earlier engines with fuel injection typically has poorer fuel atomization quality and air:fuel ratio control than the port fuel injection (PFI) technology that supplanted it; thus, one might expect older model-year fuel-injected vehicles to have higher PM emissions (on average) than newer ones. Somewhat before the widespread adoption of fuel injection, closed-loop control systems were developed in tandem with oxygen sensors to improve the stoichiometric chemistry of combustion These closed loop controls improved combustion as well as the effectiveness of the after-treatment system.

The after-treatment system on most vehicles consists of a 3-way catalyst. The 3-way catalyst was designed for simultaneous control of hydrocarbons, carbon monoxide, and nitrogen oxides. Vehicles with 3-way catalysts would meet more stringent hydrocarbon and carbon monoxide emission standards while also meeting the first stringent nitrogen oxide standard. In oxidizing hydrocarbons, these systems are resulted in additional PM control. These systems were utilized on almost all gasoline-fueled vehicles beginning in the 1981 model year. On some model-year vehicles in the 1980s and a few more recently, a secondary air injection system was added between the engine and oxidation portion of the catalyst in order to add supplementary air to the oxidation reactions on the catalyst. These systems also helped oxidize PM (though probably not to the extent that it oxidizes CO or HC). The deterioration of these technologies may affect PM and HC quite differently. Throughout this chapter, there are parallels drawn between HC and PM formation as well as controls, however it should be noted that the correlation between these emissions is far from perfect. Many examples of this are shown in the 2008 analysis report.

Amounts of of PM emitted are very sensitive to the amount of fuel in combustion as well as the air:fuel ratio. As mentioned above, over-fueled mixtures result in higher PM formation and in some cases, also excess soot formation. Over-fueling can occur under several different conditions. During cold start, engines are often run rich in order to provide sufficient burnable fuel (i.e. light ends that vaporize at colder temperatures) to start combustion when the cylinder walls are still cold (which results in increased flame quench). When high acceleration rates or loads are encountered (such as in a wide-open throttle event), an extra amount of fuel is often injected for greater power or for catalyst and component temperature protection. Emission control systems in the late 1990s are better designed to control this enrichment. Finally, engines can run rich when a control sensor (e.g. oxygen, MAF, MAP, or coolant sensors) or the fuel system fails.

In addition to fuel, lubricating oil can also get into the combustion chamber via several pathways. Some manufacturers may have poor tolerances for pistons and piston rings, thus the negative pressures (engine intake vacuum) can pull oil through these larger gaps during the intake stroke. Furthermore, engine components, such as valves, valve seals, piston rings, and turbochargers can wear and deteriorate resulting in increasing emissions over time. In all gasoline automotive engines, the crankcase (where the oil bathes the engine components) is vented back into the combustion chamber through the intake manifold. This is known as Positive Crankcase Ventilation (PCV), and is required in order to remove and burn the excess hydrocarbons in the hot crankcase. Unfortunately, it can also introduce PM precursors and oil into the engine combustion chamber. Because of the relatively small amount of oil consumption compared to the volume of gasoline burned in a vehicle, HC from oil is also small. However, organic PM from oil consumption can be quite significant because oil is a high molecular weight hydrocarbon, and more likely to remain as uncombusted droplets. Therefore as vehicles age, those that consume more oil will probably have very different emissions behavior for HC than for PM, compared to when they were new. However, oil consumption can "poison" the catalyst substrate, reducing the effectiveness of the catalyst at oxidizing HC.

The fuel itself may have properties that exacerbate PM formation, which may be affected by concentrations of sulfur, lead, aromatics, and impurities. With the lower levels of lead and sulfur in fuels recently, the first two are less of a factor in the Kansas City program than aromatics would be. In the calendar years that MOVES models, lead is not a significant portion of the inventory, thus is largely ignored. Sulfur (as a fuel rather than a tailpipe phenomenon) is modeled separately and is described in another MOVES document. Impurities may be captured in a future version of MOVES.

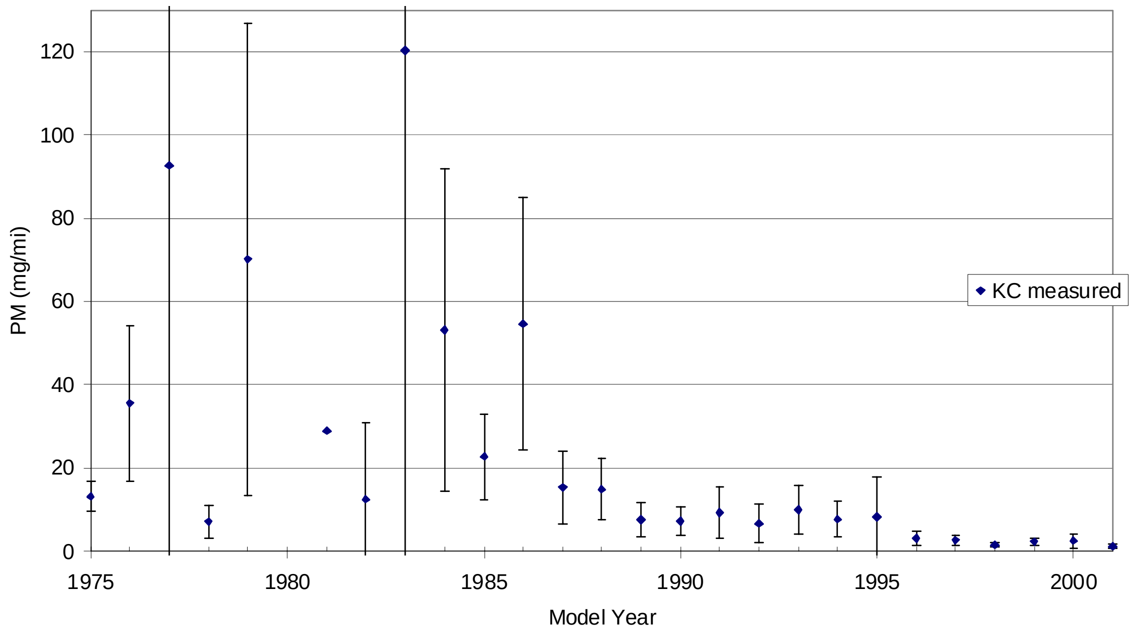
Some of these PM forming mechanisms clearly affect HC emissions. So a control technology or a deterioration path for HC may or may not similarly affect PM depending on the source. It is also likely that the processes that cause high PM may not be the same processes that cause organic PM. Some of the mechanisms also form visible smoke. Smoke takes on a variety of characteristics depending on the source, and can be due to oil consumption or overfueling. The smoke is visible because of the relative size of the particles compared to the light wavelengths that are scattered. However, visible smoke is not necessarily a reliable indicator of high PM emissions.

2.2 New Vehicle or Zero Mile Level (ZML) Emission Rates

In this section, we develop a modeling approach to extend the PM emission trends from Kansas City presented in the analysis report to average emissions across the fleet. The section also compares the new vehicle results from many different studies in order to determine the zero mile level (ZML) emission rates for all model years in MOVES. Before modeling deterioration, it is first necessary to capture ZML emission rates.

In constructing a model of emissions from the Kansas City data (**Error! Reference source not found.**), the most significant challenge is distinguishing between model year and age effects. This problem arises because program was conducted over a two-year period, thus ensuring a 1 to 1 correspondence between model year and age. As a result, one cannot know for sure whether emissions are decreasing with model year, or increasing with age, or both. Emissions tend to decrease as technologies are introduced on vehicles (with later model years) in order to comply with more stringent emissions standards. However, these technologies and vehicles tend to deteriorate over time, thus for the same model year vehicle, older vehicles (greater age) will have higher emissions (on average) than newer vehicles.

Figure 2 - 4. Average PM Emissions from the Kansas City by Model Year.



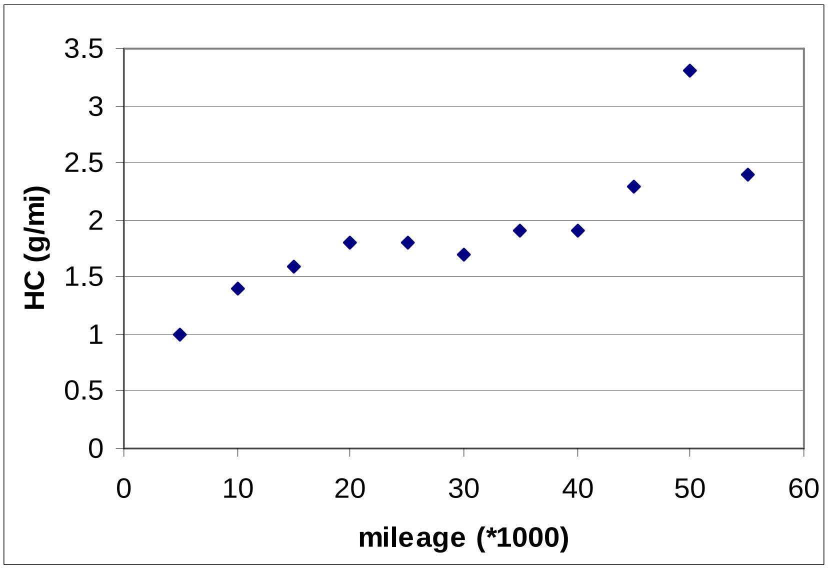
In concept, the most accurate means of quantifying emissions from vehicles over time is to conduct a longitudinal study, where emissions are measured for the same vehicles over several (or many) years. However, implementing such a study would be costly. Moreover, it is impossible to obtain recent model year vehicles that have been significantly aged. In the following sections, we will describe some limited longitudinal studies conducted in the past. Then we will present our modeling methodology to isolate model year (technology) in this chapter from age (deterioration) in the next.

2.2.1 Longitudinal Studies

There have been a few longitudinal studies conducted in the past that are relevant for PM emissions. Unfortunately, they are all limited in their ability to conclusively discern model from age effects.

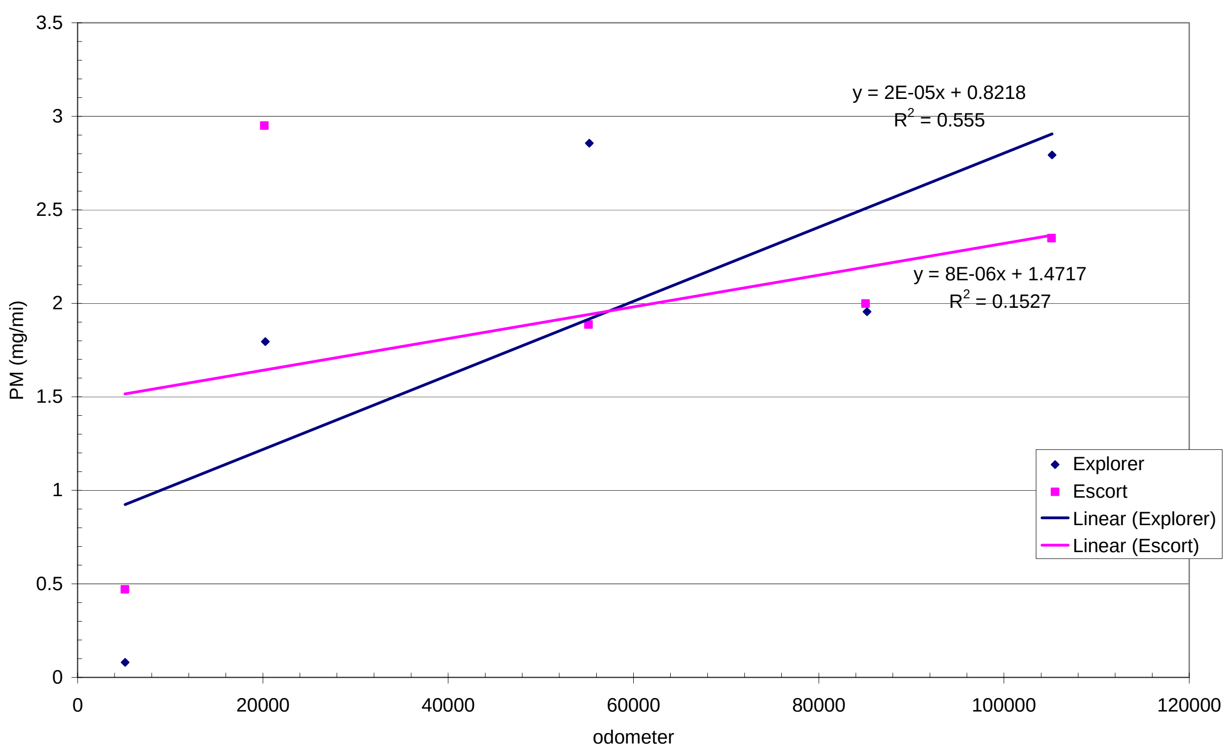
Gibbs et al. (1979) measured emissions from 56 vehicles from 0 to 55,000 miles (odometer) on 3 different cycles for the EPA[16]. Hydrocarbon emissions were analyzed, but unfortunately, PM results were not reported as a function of mileage. The authors state that “emission rates of measured pollutants were not found to be a consistent function of vehicle mileage,” however, the following figure shows that some increasing trend seems to exist for HC.

Figure 2 - 5. Hydrocarbon emission as a function of mileage (Gibbs et al., 1979)



Hammerle et al. (1992) measured PM from two Ford model vehicles over 100,000 miles.[17] However, their results for PM deterioration are somewhat inconclusive, as the following figure shows, since the deterioration seems to occur mainly in the beginning of life, with very little occurring after 20,000 miles. Also, the study is limited to only 2 vehicle models.

Figure 2 - 6. Particulate emissions as a function of odometer for two Ford vehicles (Hammerle et al., 1992)



Both of these studies assume that odometer is a surrogate for age. While there are some deterioration mechanisms that worsen with mileage accumulation, there are others that deteriorate with effects that occur over time, such as number of starts, corrosion due to the elements, deposits and impurities collecting in the gas tank and fuel system, etc. Therefore, we believe that any study that describes deterioration as a function of odometer (alone) is not capable of accounting for all causes of deterioration.

Whitney (2000) re-recruited 5 vehicles that had been tested from a previous large study 2 years prior (CRC-E24)[18]. There are two significant limitations of this follow-up study: (1) the interval between studies was only 2 years, though the odometers had increased 22,200 miles (on average) and (2) these vehicles were tested on a different drive cycle, the LA92 compared to the previous study, which used the FTP. We will explore the potential cycle differences on PM later, but assuming the cycles give similar PM results, the PM emissions were only 8% higher (on average). This increase is due to a single vehicle, which had significantly increased PM emissions (the rest were the same or slightly lower). Unfortunately, this is not a large enough sample and time period on which to resolve age effects, but it may be sufficient to conclude that the differences between PM from the FTP and LA92 drive cycles are minimal for PM.

The three longitudinal studies described above are inconclusive, though they do hint that deterioration does occur.

2.2.2 New Vehicle, or ZML Emission Rates and Cycle Effects

In order to isolate the effect of model year (technology) from age (deterioration), it is useful to look at the model-year effect independently. This can be done by analyzing emissions from new vehicles from historical PM studies. This entails capturing the near ZML or Zero Mile Level emission rates. New vehicle emission rates tend to have a much smaller variability than older vehicles (in absolute terms) since they have lower emissions that comply with more stringent standards. These standards, which decrease over time, are for hydrocarbons, but tend to have an effect on PM emissions as well since many of the same mechanisms for HC formation also form PM.

Several independent studies have been conducted , which have measured PM emissions from nearly new vehicles. For our purposes, we will define "new" as a vehicle less than 3 years old, i.e., vehicles within the 0-3 year age Group. The following table lists the 15 studies employed for this analysis.
Historical gasoline PM studies including new vehicles

Table 2 - 1. Historical gasoline PM studies, including new vehicles.

Program	Yr of Study	# new vehicles	Drive cycle
Gibbs et al.[19]	1979	27	FTP
Cadle et al.[20]	1979	3	FTP
Urban & Garbe[21],[22]	1979, 1980	8	FTP
Lang et al.[23]	1981	8	FTP
Volkswagen[24]	1991	7	FTP

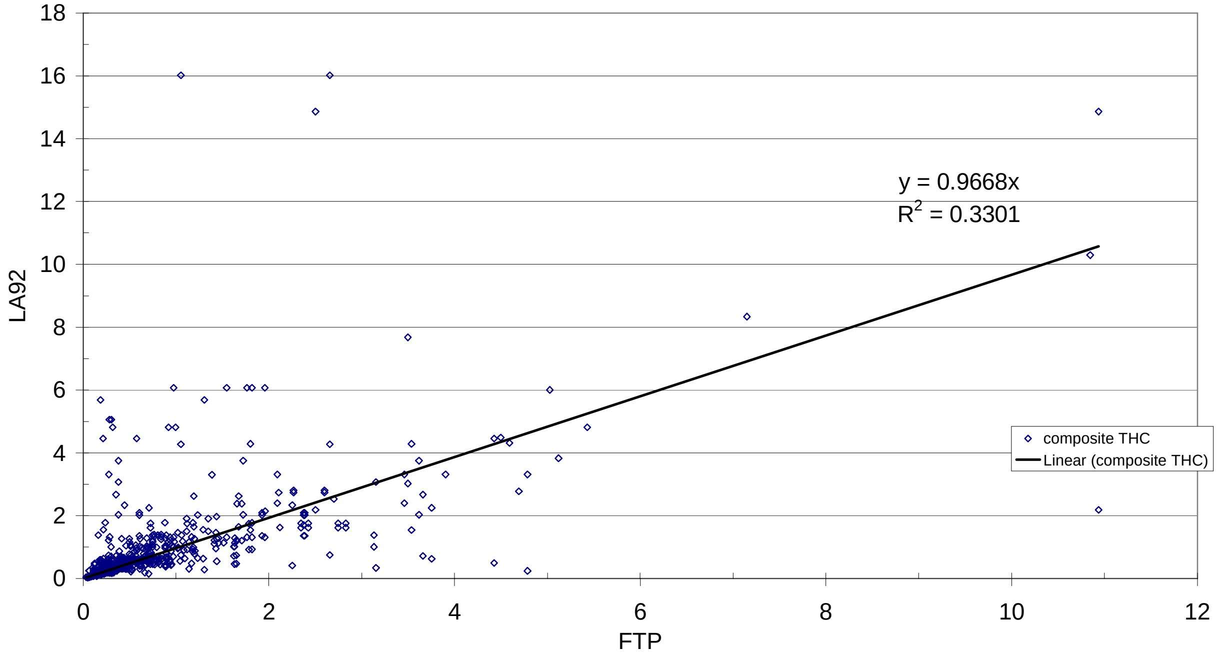
CARB[25]	1986	5	FTP
Ford[26]	1992	2	FTP
CRC E24-1 (Denver)[27]	1996	11	FTP
CRC E24-3 (San Antonio)[28]	1996	12	FTP
CRC E24-2 (Riverside)[29]	1997	20	FTP
Ford,Chrysler,GM Chase et al.[30]	2000	19	FTP
Whitney (SwRI)[31]	1999	2	LA92
KC (summer)[32]	2004	13	LA92
EPA (MSAT)[33]	2006	4	FTP

Before, we plot these emissions, we should convince ourselves that the LA92 driving cycle will not give significantly different PM emissions than the FTP so that we can compare these test programs directly. As described above, the results from Whitney (2000) seem to indicate little difference between the two cycles. Even though the tests were conducted 2 years apart, one would expect that the aging effects in combination with the slightly more aggressive LA92 cycle (used later) would have given higher PM emissions. However, this was not the case, and only one of the 5 vehicles showed significantly increased emissions.

Li et al., (2006) measured three vehicles on both cycles at the University of California, Riverside[34]. The PM emissions from the LA92 were 3.5 time larger (on average) than the FTP results. However, the HC emissions were only 1.2 times higher. These results seem rather contradictory and inconclusive. The 3.5 factor also seems excessive.

Finally, the California Air Resources Board conducted an extensive measurement program over several years comparing many different drive cycles. Unfortunately, PM was not measured in this program. However the Figure 2 - 7 shows the HC emissions compared for the two cycles. The trends indicate that there is little cycle effect for HC.

Figure 2 - 7. Hydrocarbon emissions on the LA92 versus corresponding results on the FTP.

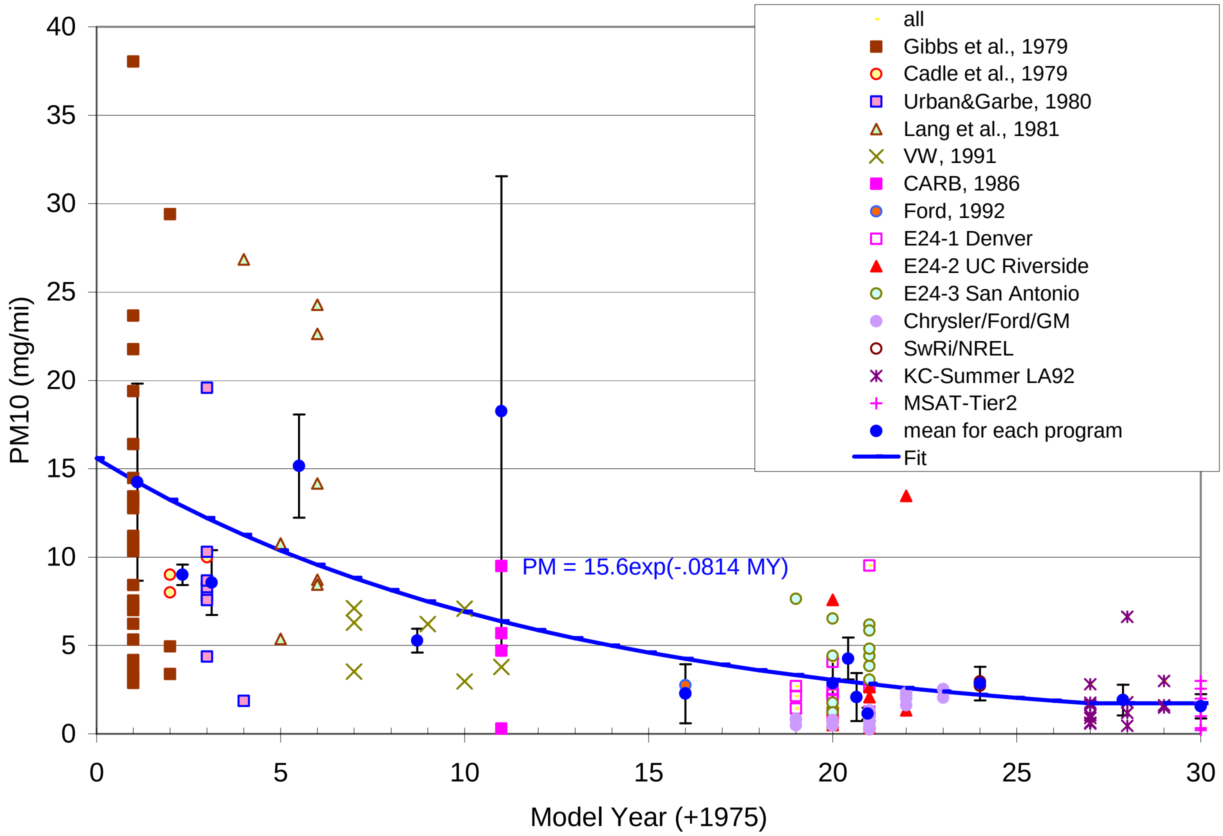


Based on these studies, we conclude that there is little difference in PM emissions between the LA92 and FTP cycles on an aggregate basis (though their bag by bag emissions may differ). We shall demonstrate that, for the purposes of ZML analysis, the results will be nearly identical even if we omit the LA92 data, thus minimizing the significance of this issue.

Figure 2 - 8 shows the new-vehicle emission rates from the 11 studies listed in Table 2 - 1 The data points represent each individual test, and the points with error bars represent the average for each test program. The plot presents evidence of an exponential trend (fit included) of decreasing emissions with increasing model year. The fit is also nearly identical if we omit the 2 programs that employed the LA92 cycle. We will use this exponential ZML relationship as the baseline on which to build a deterioration model. However, the measurements from the older programs primarily measured total particulate matter. These have been converted to PM10 (for the plot), which is nearly identical (about 97% of total PM is PM10. We also assume that 90% of PM10 is PM2.5 (EPA,1981). For the older studies, we accounted for sulfur and lead directly if they were reported in the documentation. In those cases where sulfur was not reported, the levels were approximated using MOBILE6 sulfur emission factors and subtracted as an adjustment.

Unfortunately, many of the older studies used a variety of methods for measuring particulate matter. There were many differences in filter media, sampling temperature, sample length, dilution, dynamometer load/settings etc. It is beyond the scope of this project to normalize all of the studies to a common PM metric. It is likely that there is insufficient documentation to even attempt it. Therefore no attempts at adjustment or normalization were made except for size fraction, lead and sulfur, as described above.

Figure 2 - 8. Particulate emission rates for new vehicles compiled from 11 independent studies.



To determine the ZML emission rates from these data, the next step was to separate results for cars and trucks, and to separate cold-start from hot-running emissions. Unfortunately, the historical data does not present PM results by bag. Therefore, the 2005 hot-running ZMLs for cars vs trucks were determined from the KC dataset, and the model year exponential trend from the aggregate trendline (-0.08136) is used to extend the ZMLs back to model year 1975. The base hot running ZML emission rate for LDV ($E_{HR,y}$) is:

$$E_{\mathrm{HR},y} = E_{\mathrm{HR},2005}\mathrm{e}^{-0.814y} \qquad \textbf{2 - 1}$$

Where

y = model year – 1975, and

$E_{\mathrm{HR},2005}$ = hot running zml rate for MY 2005.

To estimate equivalent rates for trucks, we multiplied this expression by 1.43. This value is based on an average of all the studies with new vehicles from 1992 onward (before this model year, there were no trucks measured). It is also multiplied by 0.898 to give hot running bag 2 rates and 1.972 to give the cold start emission rate (here defined as bag 1-bag 3 in units g/mi). These values were estimated by running a general linear model of bag 2 and bag1-3 with respect to composite PM respectively in the SPSS statistical software tool. The averages of these ratios by model year are shown in Figure 2 - 9, in which no clear trend is discernable. The parameters of the model are summarized in Table 2 - 2.

Figure 2 - 9. Ratios of hot-running/composite and cold-start/composite, Bag2 and Bag1-Bag3, respectively, averaged by model year.

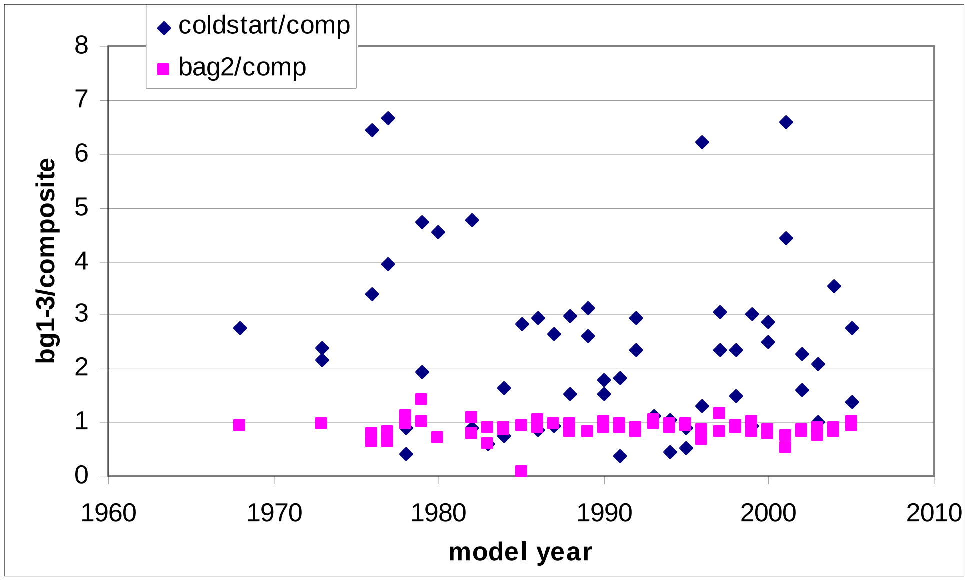
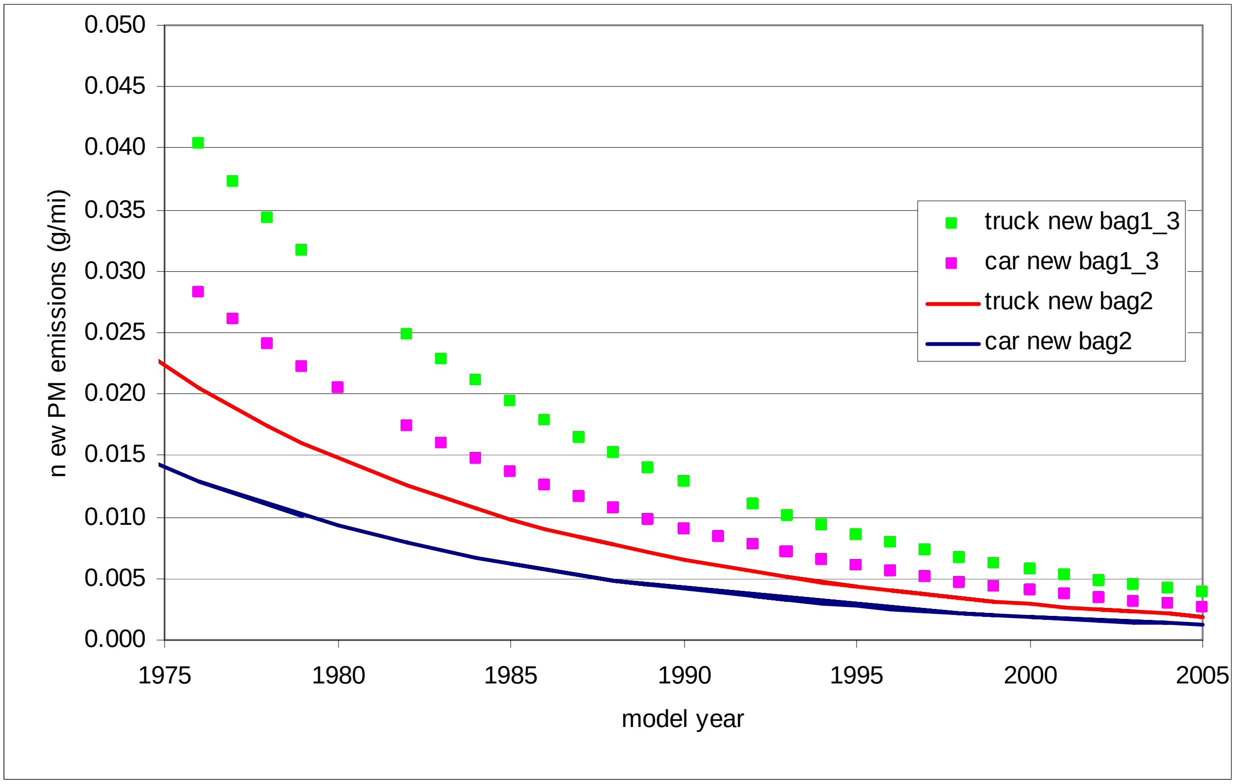


Table 2 - 2. Best-fit parameters for cold-start and hot-running ZML emission rates.

Parameter	Value
LDV hot-running ZML (g/mi)	0.01558
Exponential slope	0.08136
Truck/car ratio	1.42600
Bag-2 coefficient	0.89761
Cold-start coefficient	1.97218

Figure 2 - 10 shows the ZML emission rates. The rates are assumed to level off for model years before 1975 and again after 2005. Elemental and organic carbon fractions are another modification to the ZML rates. These fractions are already reported in the analysis report.

Figure 2 - 10. Particulate ZML emission rates (g/mi) for cold-start and hot-running emissions, for LDV and LDT.



2.2.3 Aging or Deterioration in Emission Rates

In this section, a deterioration model is introduced that captures how new vehicles in all model years deteriorate over time so that gasoline PM from any given calendar year can be modeled in MOVES. The purpose of this model is to characterize the PM emissions from the fleet and to hindcast the past as well as forecast the future, as must be done in inventory models.

2.2.2.4 Age Effects or Deterioration Rates

The ZMLs determined in the previous section represent baseline emissions for new vehicles in each model–year group. By comparing the emissions from the "aged" Kansas City vehicles in calendar year 2005, to the new rates determined earlier, we can deduce the "age effect" for each corresponding age. However, simple an approach as this seems, there are many ways to connect two points (even if there are a family of 2 points). This section describes the procedure and the assumptions made to determine the rate at which vehicle PM emissions age.

We first break the data into age Groups. We use the MOVES age bins which correspond to the following age intervals: 0-3 (new), 4-5, 6-7, 8-9, 10-14, 15-19, 20+. Having a single age category for 20 years and older implies that emission rates have stabilized by 20 years of age.

The bag measurements from all of the vehicles measured in Kansas City were first adjusted for temperature using the equation derived in the analysis paper. The equation used is:

$$E_{\mathrm{PM},72} = E_{\mathrm{PM},T}\mathrm{e}^{-0.03344(72-T)} \qquad \mathbf{2\text{ - }2}$$

where $E_{\mathrm{PM},72}$, is the adjusted rate at 72ºF for cold-start or hot-running emissions, $E_{\mathrm{PM},T}$ is the corresponding measured emissions for cold-start or hot-running, respectively, at temperature T, respectively.

The temperature-adjusted measurements are the “aged” rates.

There are two simple methods for determing the deterioration rates. The first method is to subtract the aged rates from the new rates, the difference by age would define the deterioration rate. This model will be referred to as the “additive deterioration model”. The second method is to ratio the aged rates with the new rates so that the deterioration rates are all proportional. This will be referred to as the “multiplicative deterioration model”.

In the additive mode, the Kansas City data were grouped or averaged within the age Groupss, with the prerequisite that no newer age Group can have greater rates than an older age Group. This constraint is maintained by averaging 2 (or more) age bins together as necessary. Figure 2 - 11 shows the additive age deterioration rates in units of g/mi for running and g/start for starts. Figure 2 - 12.shows the aggregate deterioration rates compared to the data. The cold start and hot running were combined with the proper weighting to calculate an LA92-equivalent rate. Finally, **Error! Reference source not found.** shows the LA92 equivalent emission rates, combining ZMLs and deterioration, compared to the data.

Figure 2 - 11. The incremental effect of age on Particulate emissions based on the Kansas-City Results.

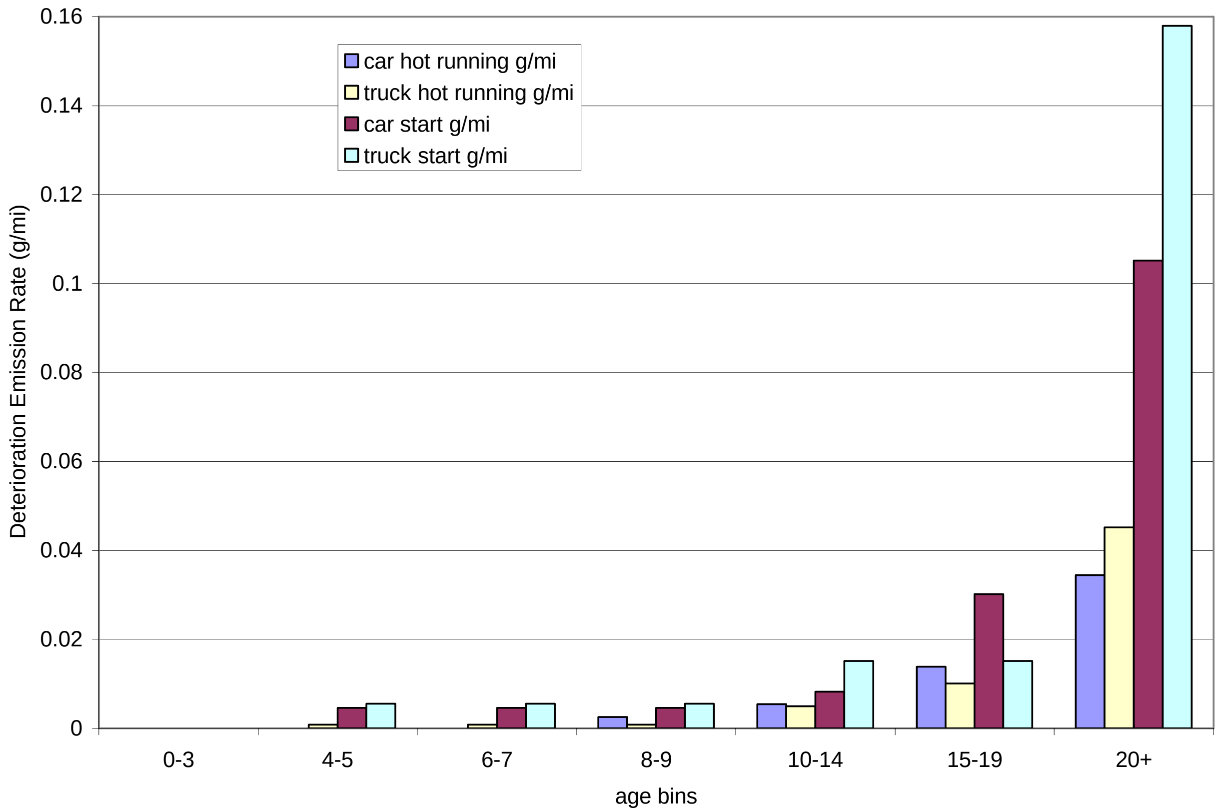


Figure 2 - 12. Cold-start and hot-running Particulate emissions by age, with estimated rates compared to data (as LA92 composite, mg/mi).

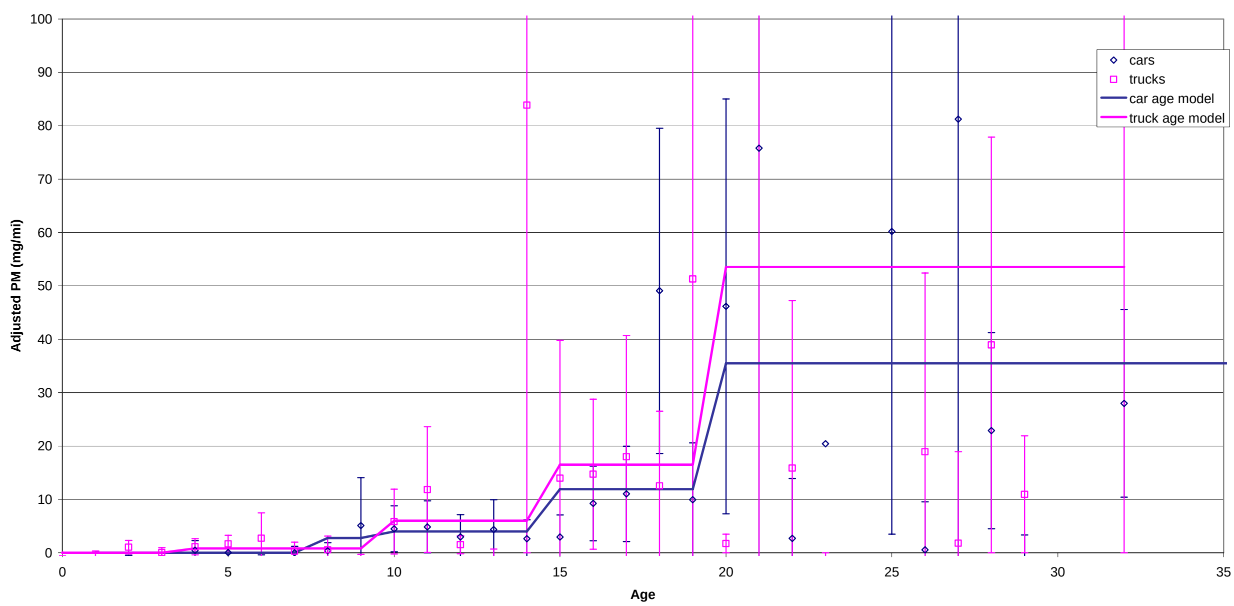
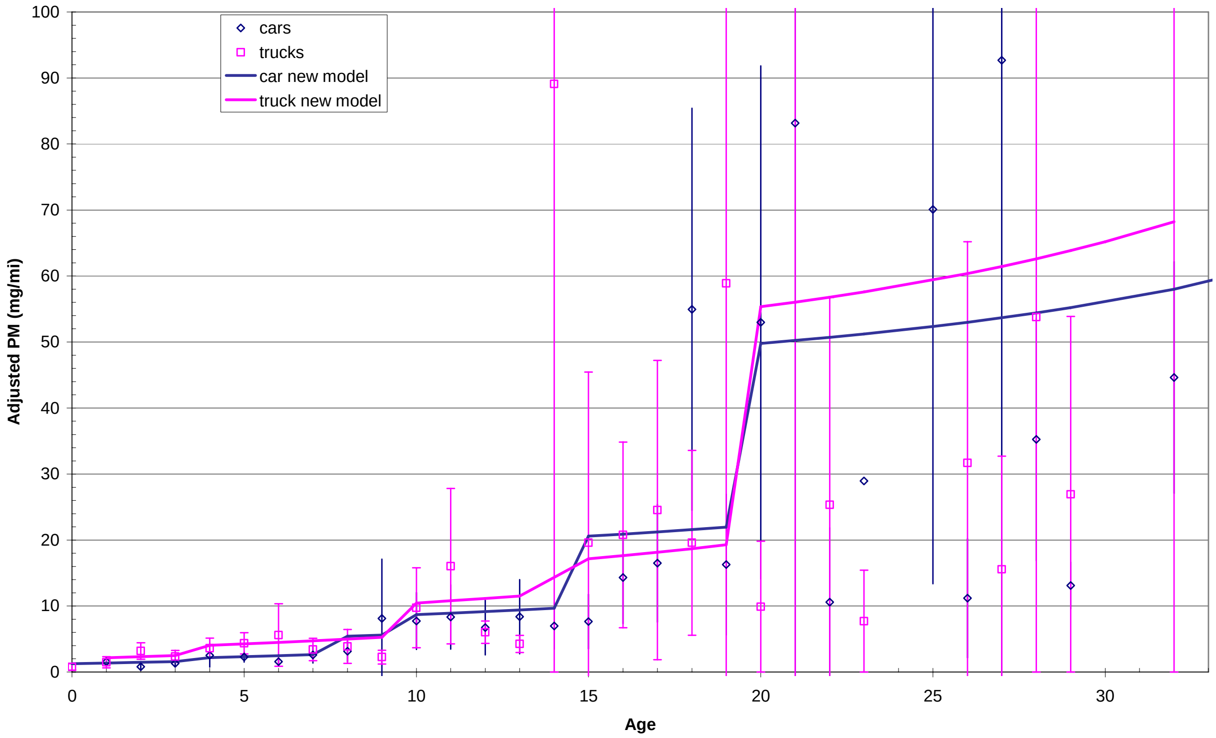


Figure 2 - 13. Emission rates by age, as LA92 composites (mg/mi), compared to data.



The model appears to fit the data well. One can also observe from **Error! Reference source not found.** the relative effect of ZML vs deterioration. The slight curvature after each large step represents the ZML exponential curves, whereas the larger steps represent the deterioration.

This additive deterioration model suffers from some significant limitations. The first is that it assumes that all model years will deteriorate the same way independent of the emission control technologies employed. As an example, the model assumes that a 1975 carbureted engine will deteriorate in the same fashion as a 2005 fuel injected engine car; i.e. they will have different ZML rates to start, but both will add about 50 g/mi after 20 years. So, while new 1975 and 2005 cars have emission rates of approximately 15 and 2 g/mi, respectively, (a factor of 7.5), after 20 years the rates would be 65 and 52 g/mi, respectively, thus the cars will have nearly the same emission rates. While it is possible that a small number of fuel-injected vehicles would experience complete failures of their aftertreatment and fuel control systems, it is unlikely that they would (on average) have similar emission rates as a carbureted vehicle. The second problem with the model is that it is not consistent with the way hydrocarbon deterioration rates are represented in MOVES.

It is likely that some of the same mechanisms that cause HC to increase over time would also result in PM increases. These factors include deterioration in the catalyst, fuel control, air:fuel-ratio control, failed oxygen sensors, worn engine parts, oil leaks, etc. **Error! Reference source not found.** shows trends in the natural logarithm of THC rates over approximately 10 years, based on random-evaluation samples in the Phoenix I/M program. On a ln-linear scale, the deterioration rates appear approximately linear over this time period. This pattern means that the deterioration rate is exponential over this time interval. This observation, combined with the approximate parallelism of the trends for successive model years, imply that emissions follow a multiplicative pattern across model-year or technology groups. thus calling for a multiplicative deterioration model. In such a model, the aged rates and the new rates are converted to a logarithmic scale, then the slopes are estimated by fitting a general linear model. The average slope is determined, and the ZMLs determined earlier define the y-axis offsets. This results in a series of ladder-like linear lines in on a log scale as show in **Error! Reference source not found.**. The lines fan out exponentially on a linear scale as shown in **Error! Reference source not found.**. The dotted lines and the points with uncertainty bars represent the Kansas City data overlaid onto the model and indicate that the model is consistent with the data.

Figure 2 - 14. The natural log of THC emissions vs. Age for LDV in the Phoenix (AZ) Inspection and Maintenance program over a ten-year period (1995-2005).

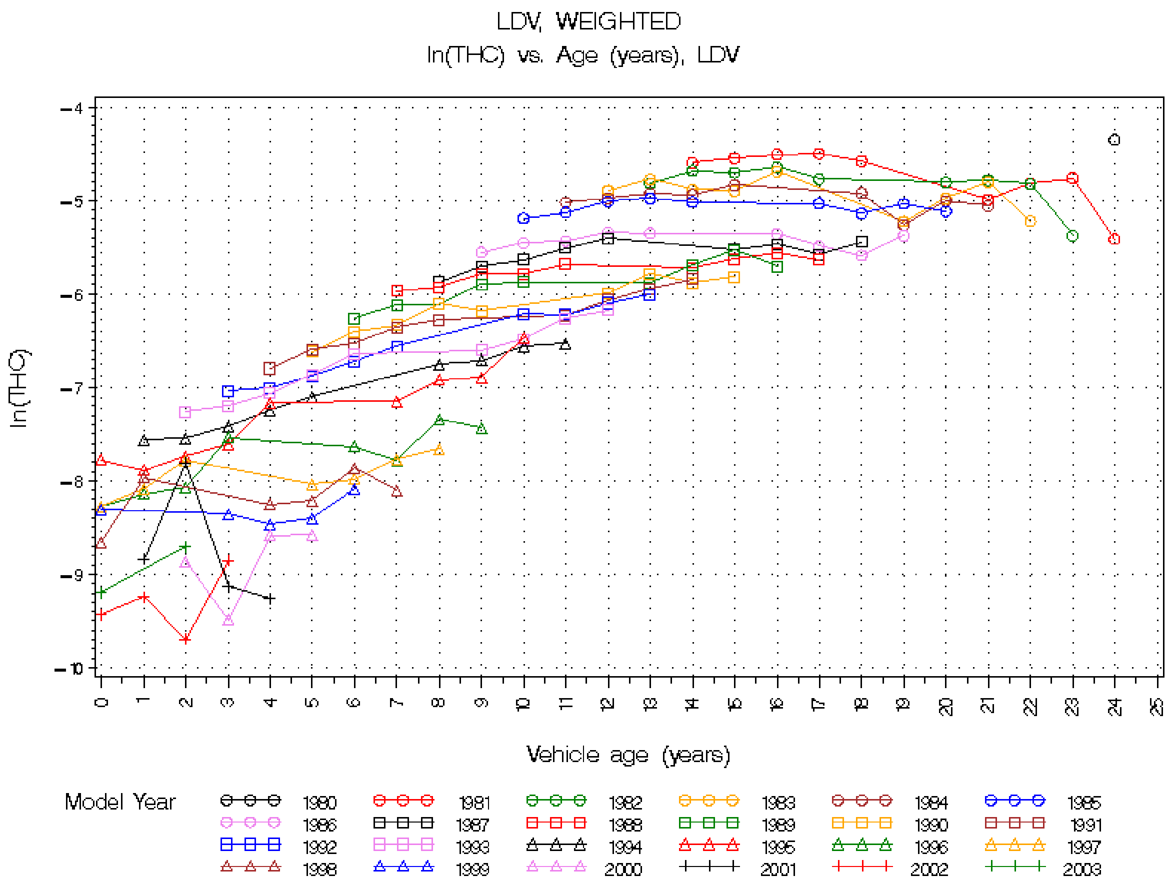


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Figure 2 - 15. The Multiplicative deterioration model applied to PM results from Kansas City. The y-axis offsets represent ZML rates. The dotted line represents the Kansas-City Data.

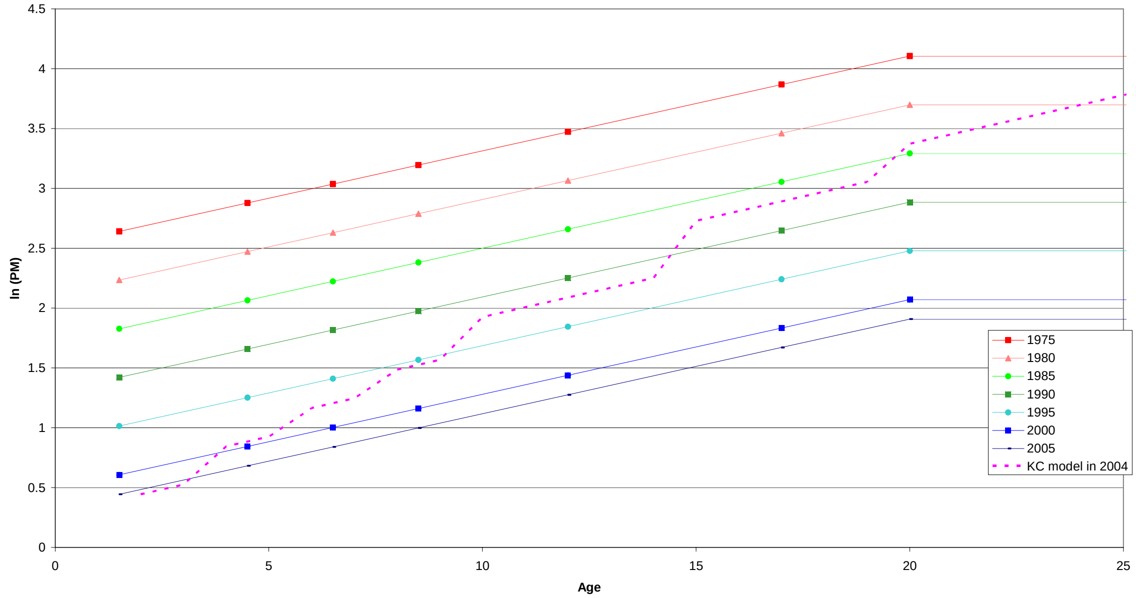
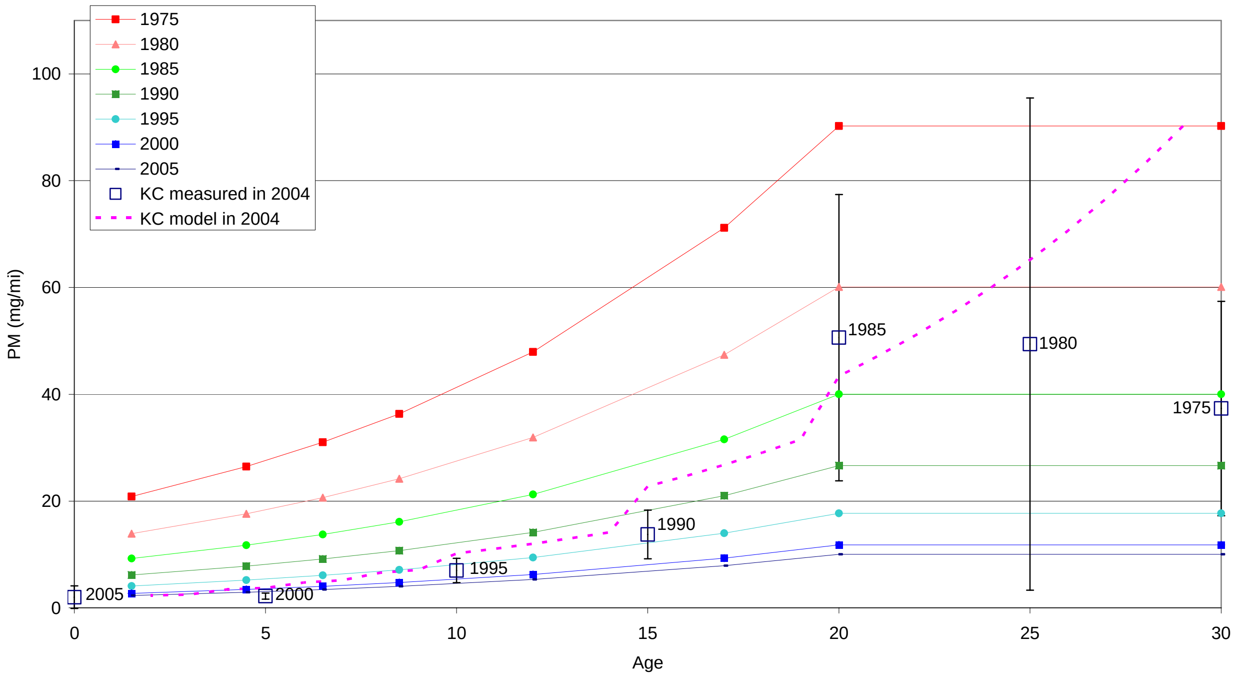


Figure 2 - 16. The multiplicative deterioration model shown on a linear scale. The y-axis offsets capture the new-vehicle ZML rates. The dotted lines and points with error bars represente the Kansas-City results (with 95% confidence intervals).



Because the model is multiplicative, the deterioration factors can be applied directly to trucks, cold start, hot-running, EC, and OC, since the order of operations does not matter. The start process requires only a soak time model to fill the remainder of the rates. Because no data is available describing how particulate start emissions vary by soak time, we have used the HC soak curves shown previously (see Figure 1 - 35).

Substantial analysis is yet required to fill modal particulate emission rates for emissionRateByAge table in the MOVES input database. Because the simple multiplicative model can be applied across the range of VSP, deteriorated rates by operating mode can be directly generated, as described in the next section.

2.3 Modal PM Emission Rates

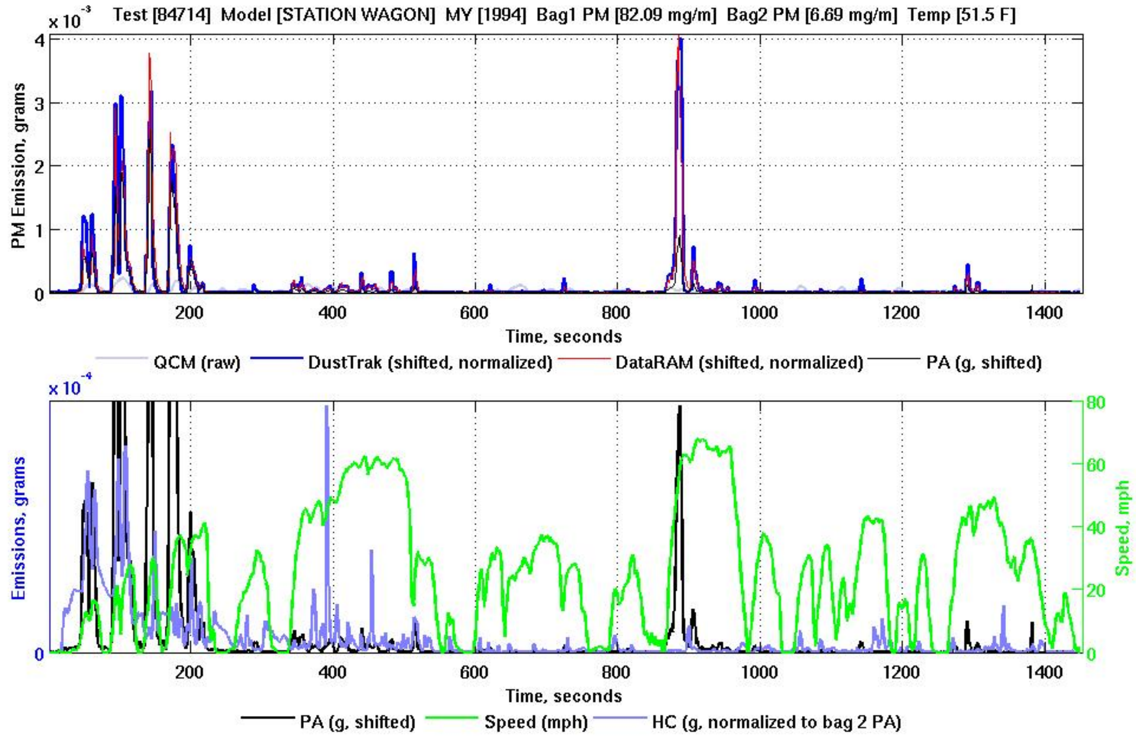
As mentioned earlier, the continuous emissions measurements from the Kansas City study were examined at great length, after which we determined that the Dustrak gave the most reliable second-by-second PM time-series data when compared to the quartz-crystal microbalance (QCM) and the Nephelometer. In the following sections, we describe some of the trends in continuous PM for "typical" normal and higher emitters. We conclude by describing the procedure by which results from the Dustrak were used to develop emission rates by operating mode.

2.2.1 Typical behavior in particulate emissions as measured by the Dustrak and Photoacoustic Analyzer

After looking at over 500 second-by-second traces, it became apparent that most of the vehicles fell into certain general patterns. The most common behavior involved a highly non-linear PM emissions release as engine load increased. This pattern led to a monolithic "spike" in emissions during the most aggressive acceleration event in the LA92 drive cycle during the 2nd (hot running) bag at around 850 seconds. This peak is captured in Figure 2 - 17, which includes 2 plots. The higher emissions prior to 300 seconds can be attributed to cold start, during which the engine is still cold and the fuel:air mixture tends to be on the rich side. The plot on the bottom confirms this supposition since it indicates that elemental carbon is relatively high during the start. The hydrocarbons are overlaid on the bottom plot merely for comparison, and provide a loose and qualitative comparison to organic PM emissions. Some vehicles had variations on this spike where it was much larger than even the cold start emissions, but this pattern is more typical of the newer vehicles tested on the warmer days.

On the following series of plots the dustrak (most prominent), nephelometer and QCM are overlaid on the top chart, while the photoacoustic analyzer, hydrocarbon and speed are overlaid on the bottom chart. Ordinate values are all relative and not absolute. "Shifted" means time-aligned, "Temp" means ambient temperature and the filter measurements as well as vehicle type and model year are written above the figures.

Figure 2 - 17. A typical time-series plot of continous particulate emissions as measured by several instruments.



The next series of two figures shows how in some cases, the cold-start emissions appear to be persist into the "hot-running" phase of the cycle (bag 2). Figure 2 - 18 shows an older 1976 vehicle tested at 54°F, for which one might expect the cold start emissions to have a longer duration than a newer vehicle. In this case, the cold start emissions seem to end at around 550 seconds (based on the HC trace). However, such cases where large portions of the cold start emissions leak occur during bag 2 were rare in the dataset, and thus they were not "corrected". This step can be considered for future study.

Figure 2 - 18. Continous particulate emissions from a 1976 Nova measured at 54°F.

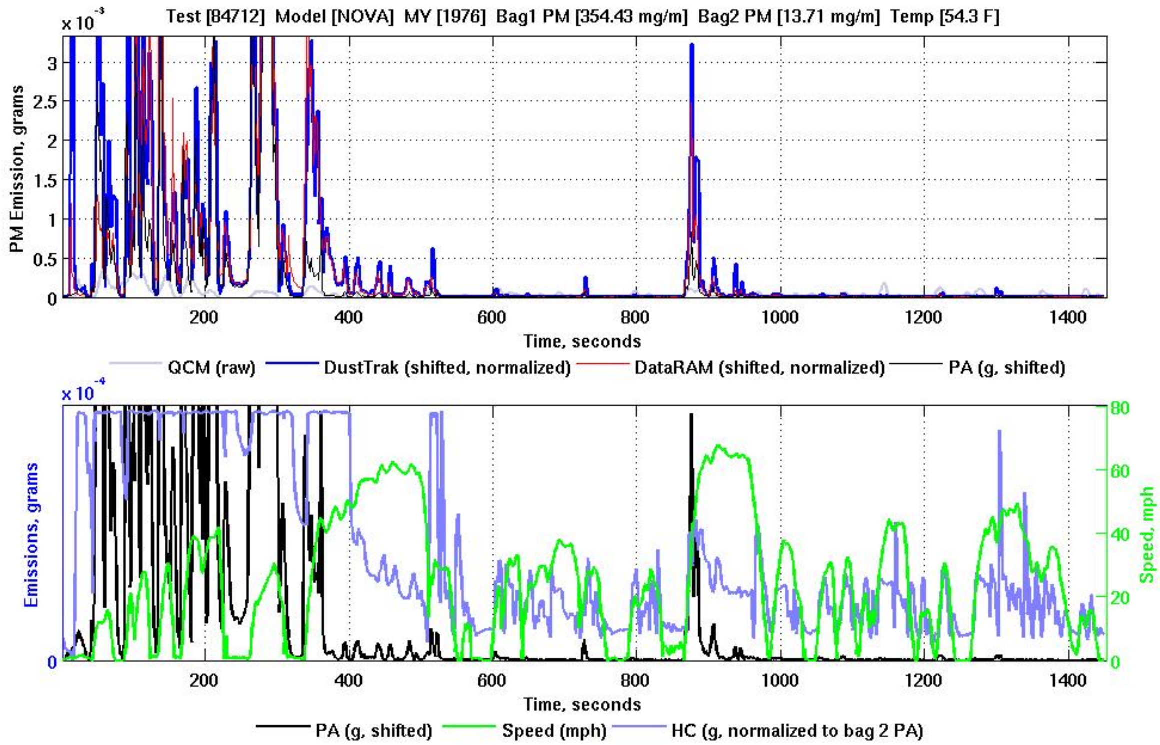
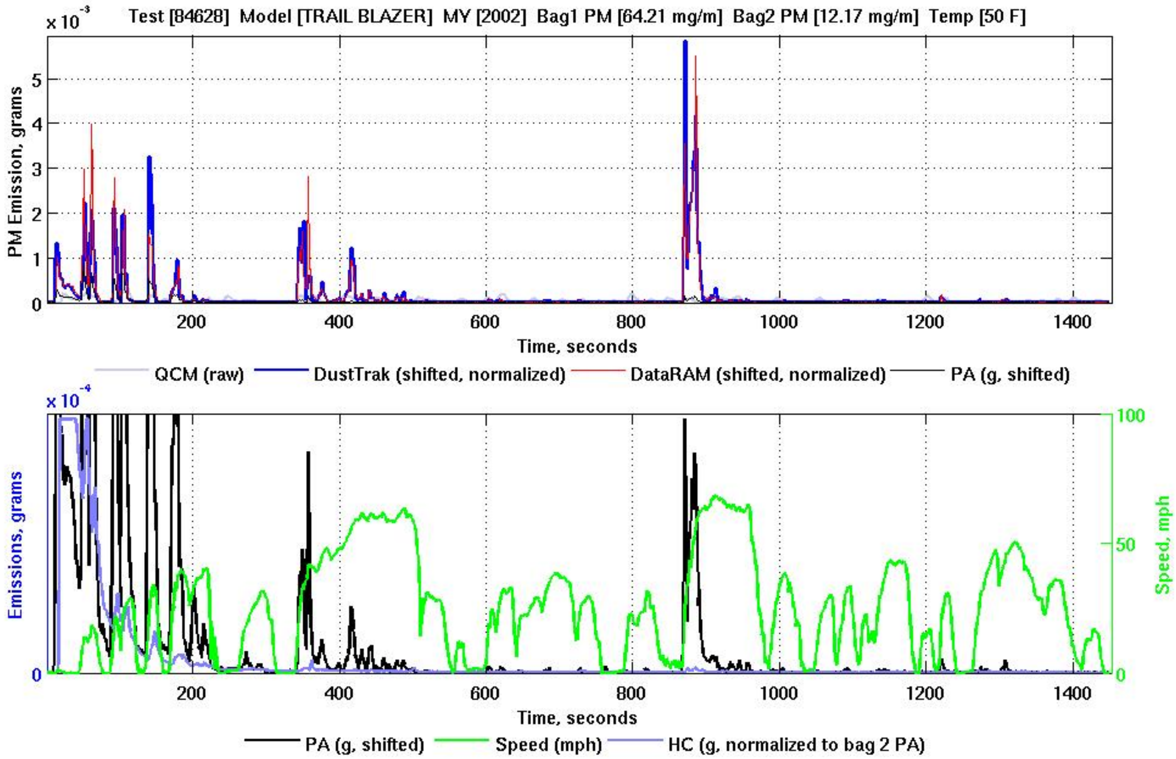


Figure 2 - 19 shows a similar but slightly more commonly seen effect for a newer vehicle. The difference is that the cold start seems to end at around 250 seconds in bag 1, but then is high again when bag 2 starts at around 350 seconds. Here the HC is low, but the EC (as indicated by the PA) is relatively high hinting at a slightly fuel rich mixture. It is uncertain at this time, why

Figure 2 - 19. Particulate time-series for a 2002 Trailblazer at 50°F.

these vehicles need to go into enrichment during this relatively mild acceleration.



The traces shown so far have been "normal emitters" during hot running operation, i.e. they did not have unusually high emissions during bag 2. These vehicles represent the bulk of the data. However, some vehicles do exhibit higher or otherwise unusual hot-running PM emissions. Examples are shown in the following series of figures.

Figure 2 - 20 shows a large "hump" of PM emissions starting at the beginning of bag 2 that lasts for nearly 600 seconds. The dustrak, nephelometer and the QCM all register this hump to varying degrees, so it's unlikely that it is a mere instrument artifact. The bulk of the bag 2 PM emissions lies in this "hump,", which does not coincide with a high load event. It is interesting that the PA is not detecting a broad EC portion, so this hump is most likely organic carbon (OC), which leads us to deduce that this hump probably represents OC particulate due to oil consumption. Because these humps are not load based events, they don't suit themselves well to characterization by VSP as correlation to power should not be high during the event. Moreover, it is interesting to note that the broad hump does not repeat. Some vehicles have the hump at different locations in the cycle (or throughout the whole cycle in rare cases), thus making this effect impossible to model physically using only a VSP methodology. Therefore, the effect can only be captured on an aggregate level by simply averaging with the normal emitters described earlier. It follows logically that if the recruitment of these "high emitters" was representative in Kansas City, and these high emissions humps are not load dependent, then this effect on the inventory should be captured by normalizing the modal rates to the filter measurements; i.e. they are captured in the base emission rates.

Figure 2 - 20. Particulate time-series for a 1988 Dynasty.

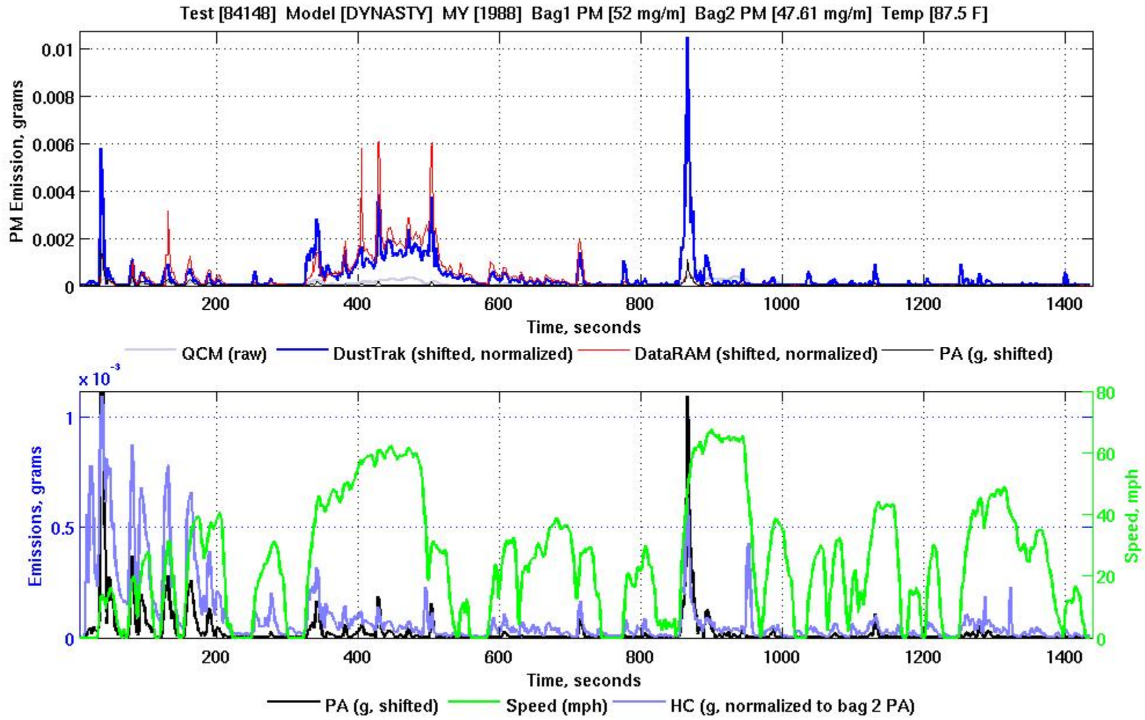
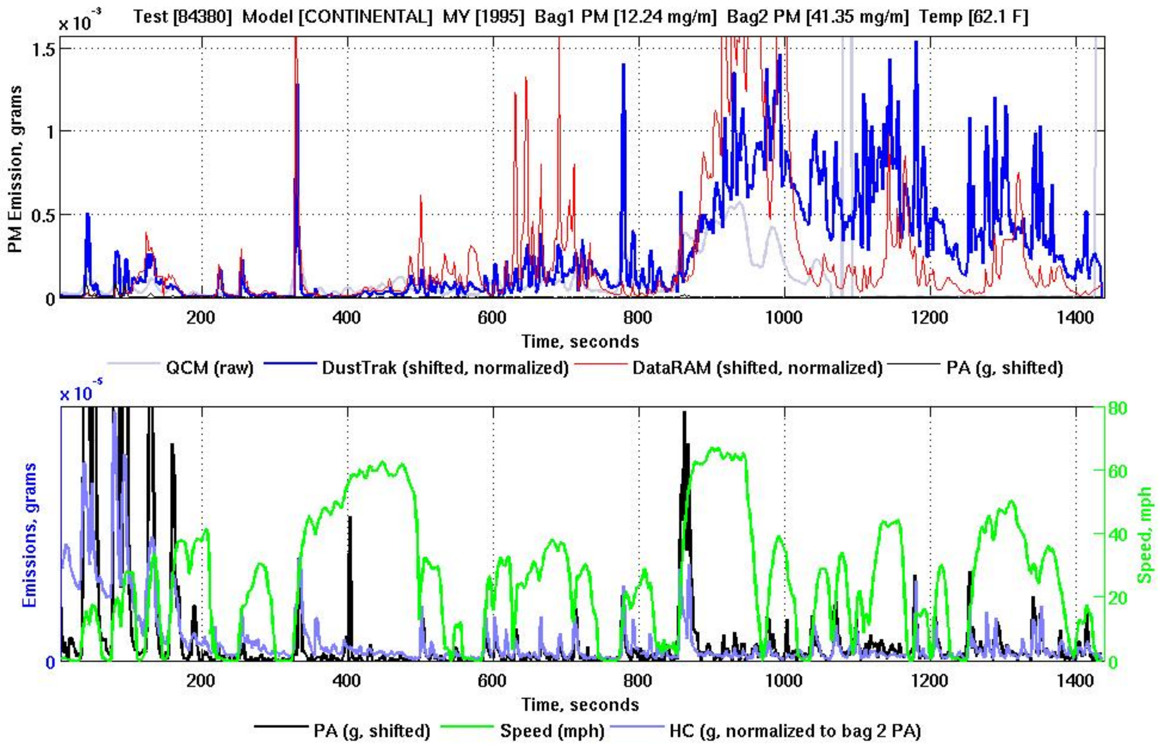


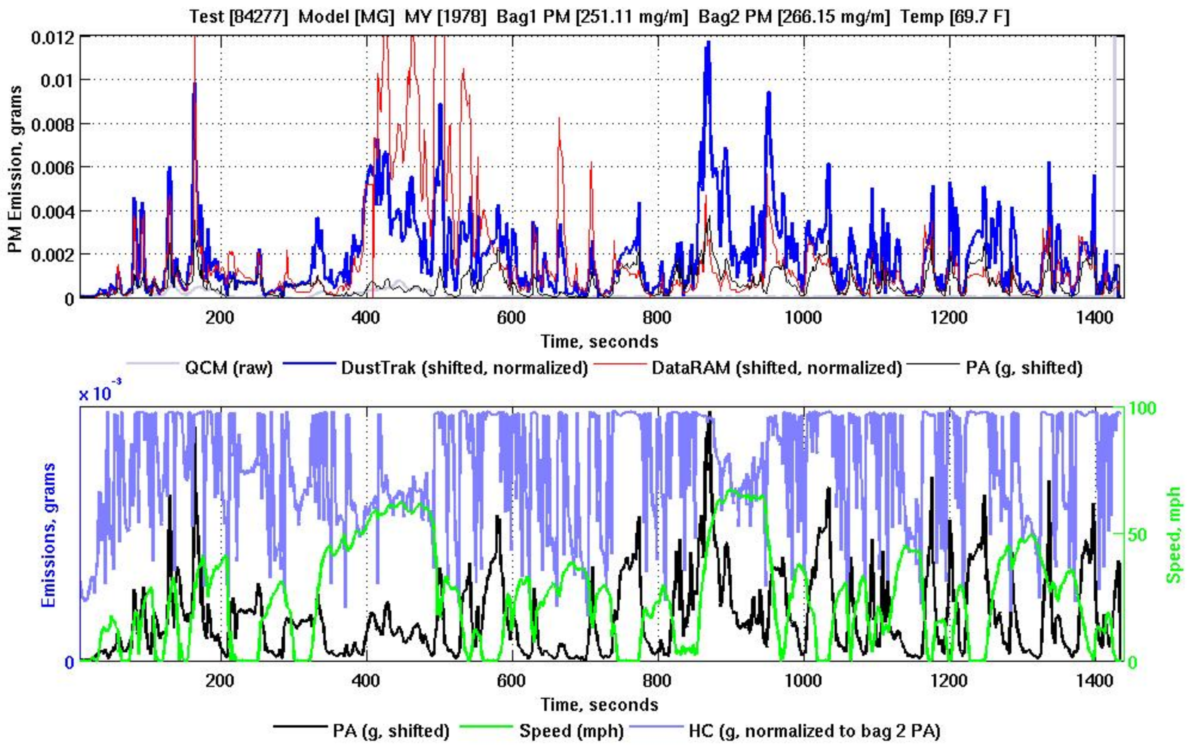
Figure 2 - 21 shows another likely candidate for designation as an oil burner. The emissions humps are much broader, though the absolute emissions are similar to the Dynasty. Note again that the dustrack, nepholometer, and the QCM all register the hump, while the PA shows very little EC, one of the "fingerprints" of oil-based particulate. In one of the repeat test vehicles in the study, one test exhibited a hump in emissions and the repeat test did not. The inconsistency and non-repeatability of some of these humps arising from oil consumption explains how some vehicles can flip from high to normal emitter or vice-versa in back-to-back tests. These observations have ramifications for future PM test programs, in that sample sizes should be large and fleets properly representative.

Figure 2 - 21. Continuous particulate time series for a 1995 Lincoln Continental.



The next figure (Figure 2 - 22) shows a more typical high PM emitter, where the bag 2 emission rate is 266g/mi. Here the EC does mirror the high emissions seen in the other instruments. Even the HC measurements are saturated. This trace, representing a 1978 MG, is an indicator of poor fuel control, as might be expected with an older (1978) carbureted engine.

Figure 2 - 22. Continuous particulate (and HC) time series for a 1978 MG.



We are now ready to bin the emission rates into operating modes based on vehicle-specific power (VSP) The following two figures show Dustrak PM emissions binned by VSP and classified by model year Groups. **Error! Reference source not found.** shows this relationship on a linear scale and **Error! Reference source not found.** shows the relationship on a logarithmic scale. It is clear from the latter plot that VSP trends for PM tend to be exponential with VSP load, i.e. they are linear on a log scale. [*This is consistent with some of the other criteria pollutant trends – is this true Jim?*]. Thus we assume smooth log-linear relations when calibrating our VSP based emission rates.

Figure 2 - 23. Particulate emissions, as measured by the Dustrak, averaged by VSP and model-year Group (LINEAR SCALE).

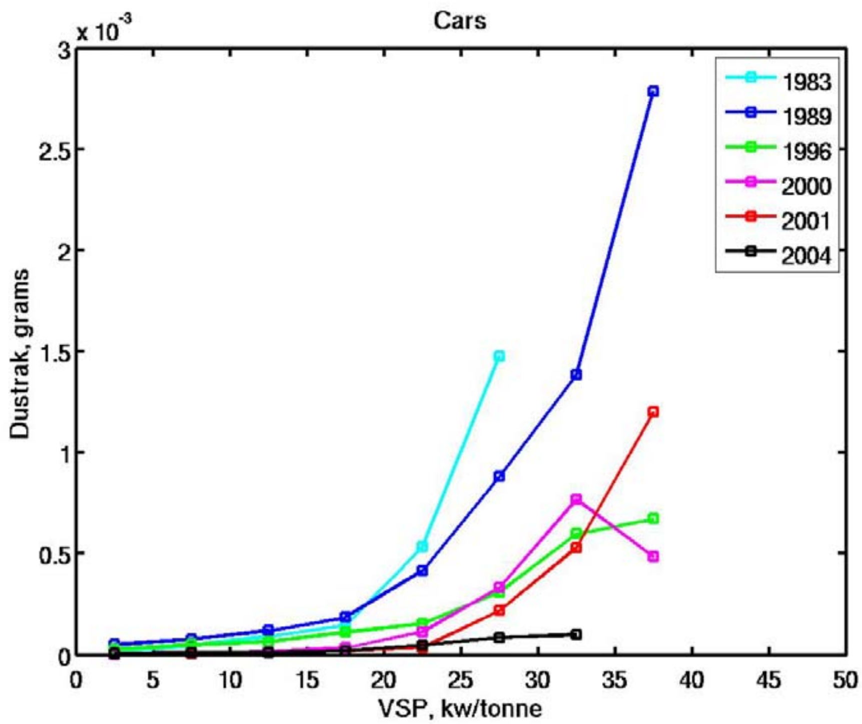
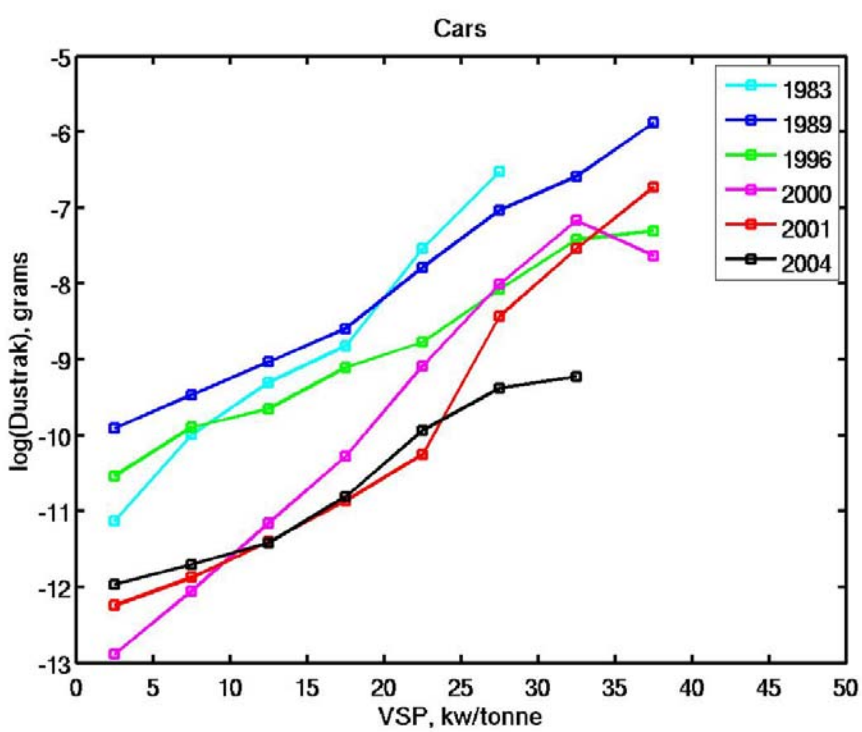


Figure 2 - 24. Particulate emissions, as measured by the Dustrak, averaged by VSP and model-year Group (LOGARITHMIC SCALE).

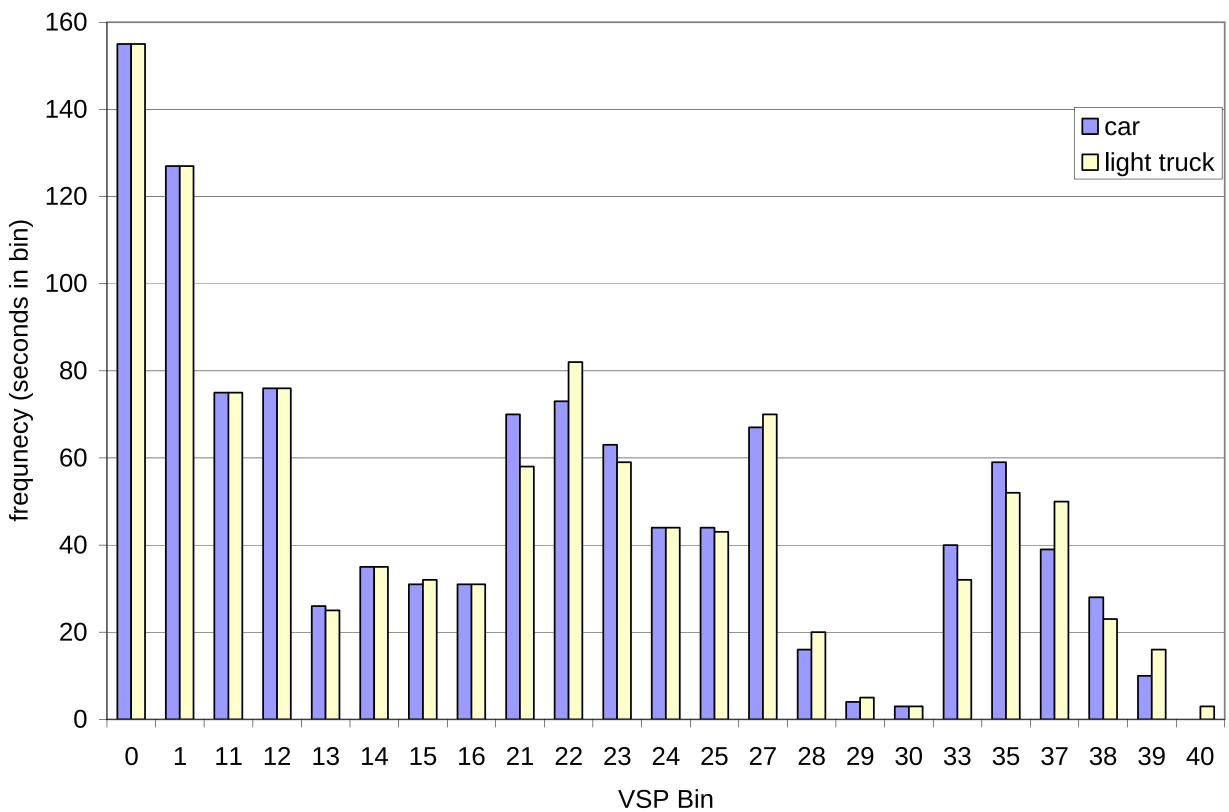


In order to determine the actual MOVES VSP based rates, followed seven steps:

1. The LA92 equivalent hot running emission rate in g/mi is determined for every model year and age group from the model described in section 2.2.
2. The gram per second (g/s) emission rate is determined from the dustrak for cars and trucks based on the KC data. These trends are then extrapolated to the higher VSP bin levels where data is missing.
3. The VSP activity distribution is calculated for bag 2 of the LA92 drive cycle for cars and trucks separately – this step is equivalent to determining the number of seconds in each VSP bin.
4. The VSP bin rates are then combined with the VSP activity rates by operating mode and then summed to give a total bag 2 emission factor that must match the aggregate LA92 emission rates in step 2 (as calculated from the filter measurements).
5. The emission rates are constrained to match the filter values through a normalization factor that is applied to every model year age group (test by test??).
6. The rates from step 5 are then multiplied by the corresponding EC and OC factors from the analysis report(cite) to give all of the hot running rates.
7. Steps above are repeated for all ages and model years.

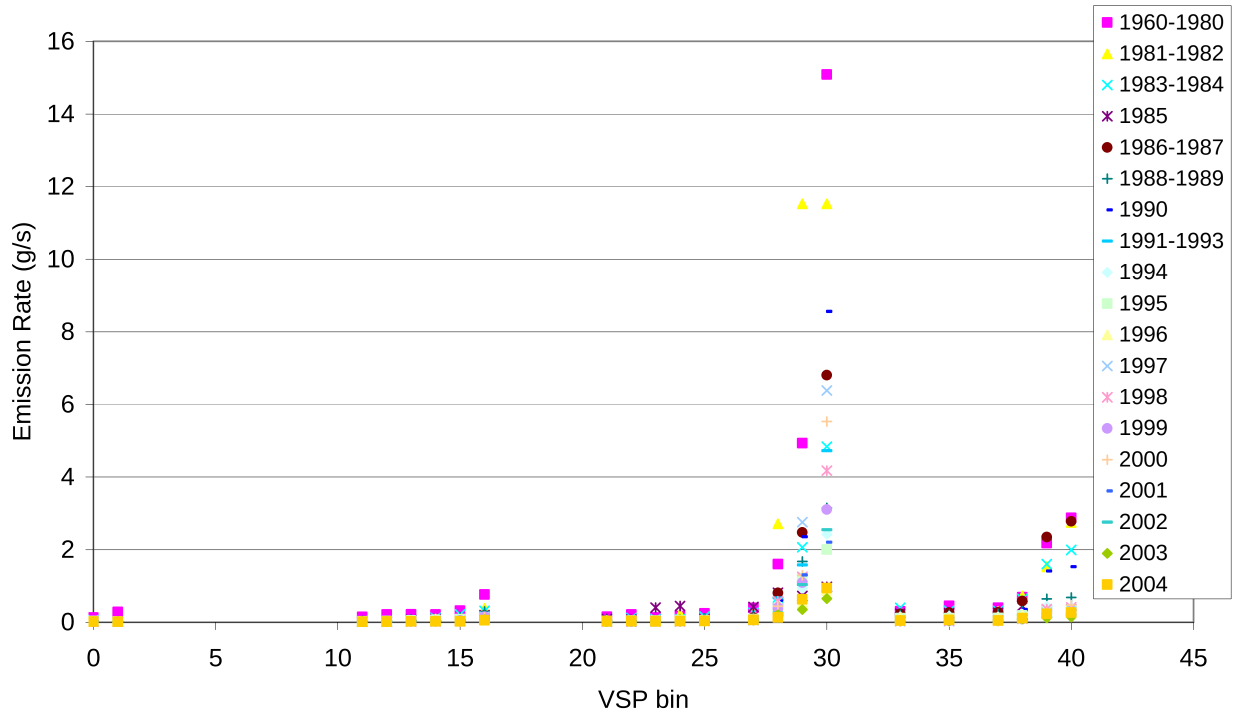
The output from step 3 (operating-mode distribution) for cars and light trucks is shown in Figure 2 - 25. For operating-mode definitions, see Table 1 - 2.

Figure 2 - 25. Operating-Mode distribution for cars and light trucks representing the hot-running phase (Bag 2) of the LA92 cycle.



The output of step 5 for each model year ZML (0-3 year age Group) is shown in **Error! Reference source not found.**.

Figure 2 - 26. Particulate emissions for passenger cars (LDV) from Kansas City results, by model year Group, normalized to filter mass measurements.



After the rates were calculated, a quality check was performed to ensure that the aged rates in any particular bin were not too high. A multiplicative model that has exponential factors risks having excessively high emission rates under extreme conditions. For example any rate over 100 g/s was considered too high, this would be an extremely high-smoking vehicle. This behavior was corrected in only two cases bins in operating mode 30, representing values for cars and trucks in the 1975 model-year Group. In these cases, the value from operating mode 29 was copied into mode 30.

2.4 Conclusions

The previous discussion describes analyses of particulate-matter emissions designed to develop operating-mode based emission rates for use in the MOVES emissionRateByAge table, incorporating the effects of temperature, model year and age. These rates include organic and elemental carbon for cold-start and hot-running emissions from cars and light trucks (e.g., LDV and LDT).. This analysis is crucial for understanding how PM emissions have changed over the years and how new vehicle PM rates are projected to deteriorate over time. The new vehicle (zero mile level) PM emissions are estimated by analyzing the new-vehicle emissions rates from historical PM studies. The trends indicate that emissions have been decreasing exponentially with model year as the engine and fuel controls have improved and after-treatment devices have been installed. The new truck rates are found to be larger than the car rates. The deterioration effect of age is determined by comparing the new vehicle rates to the Kansas City data. It is determined that emissions deteriorate exponentially with the age of the vehicle, but remain constant after about 20 years. It is also found that PM emission increase exponentially with VSP (or road or engine load).

There is still much analysis that can be conducted with these data. In the future, it would be important to examine trends in the speciated hydrocarbons and organic PM from the standpoint of toxic emission and also quantifying the PM emissions due to oil consumption. This analyses are likely to expand the scientific understanding of PM formation and why certain gasoline fueled vehicles emit more PM than others under certain conditions. It would also be useful to explicitly capture the non carbon portion of the PM. It is important to continue to collect PM emissions data in the field, since it validates the deterioration model.

3. Criteria Pollutant Emissions from Light-Duty Diesel Vehicles (THC, CO, NOx)

In MOVES, emission rates for running emissions are calculated for each operating mode. However, for the diesel-fueled passenger cars (LDV) and light-duty trucks (LDT), we lack the necessary continuous or "second-by-second" measurements to directly calculate the emission rates by VSP. Therefore, we used results (in grams per mile) from the Federal Test Procedure (FTP) to estimate corresponding modal rates (in grams per hour).

3.1. Estimating Zero-Mile FTP Emissions:

We identified FTP results on the Annual Certification Test Results & Data website (http://www.epa.gov/otaq/crttst.htm) and on the Test Car List Report Files Website (http://www.epa.gov/otaq/tclrep.htm) for 513 diesel-powered LDV and 187 LDT from the 1978 through 2008 model years. These vehicles had been tested either to certify or to generate fuel economy estimates (labels or CAFE). These test vehicles were all new (age = zero years), each vehicle having accumulated about 4,000 miles. These individual tests were used to calculate mean (composite) FTP emissions (grams per mile of HC, CO, NOx, and PM10) for each model year group. (We examined, but did not include data on European diesels since those vehicles might not be representative of those sold in the USA.) The sample sizes (by model year group) and the mean composite FTP emissions are given in Table 3 - 1 for LDV and Table 3 - 2 for LDT:

Table 3 - 1. Mean Composite FTP Emissions (g/mile) for diesel-fueled LDV.

Model Year Group	Sample Size	HC	CO	NOx	PM[1]
Pre-1981	104	0.4883	1.3425	1.4126	---*
1981-82	114	0.2508	1.0861	1.1859	0.2999
1983-84	116	0.2006	0.9809	1.0517	0.2881
1985	73	0.2178	1.1386	0.8436	0.2751
1986-90	79	0.2075	1.3581	0.5952	0.5668
1991-93	13	0.2123	1.6854	0.5685	0.4990
1994	3	0.2273	1.2233	0.8567	0.1747
1995-2005	5	0.1364	0.4140	0.8180	0.0848
2006-2008	6	0.0196	0.5367	0.3925	0.0312
2011+[2]	---	0.0196	0.5367	0.0500	0.0100

[1] Measurements of PM emissions were not performed for the Pre-1981 model year cars (or trucks). For this analysis, we applied the (later) 1982 standard of 0.6 grams per mile to those earlier model years.
[2] For 2011 and newer model years, we used the smaller of the Tier-2 Bin-5 standard or actual test results from the 2006-2008 LDV.

Table 3 - 2. Mean Composite FTP Emissions (g/mile) for diesel-fueled light-duty trucks.

Model Year Group	Sample size	HC	CO	NOx	PM
Pre-1981	13	0.6900	1.7923	1.6577	---*
1981-82	45	0.3478	1.3277	1.3748	0.3296
1983-84	56	0.2578	1.0302	1.3052	0.2700
1985	11	0.2297	1.1200	0.9473	0.2673
1986-90	20	0.2364	0.9985	1.4435	0.2790
1991-93	5	0.3020	1.7000	1.2600	0.1280
1994	17	0.2213	1.6256	1.3814	0.1114
1995-2005	14	0.1526	1.6179	1.4629	0.0960
2006-2008	6	0.0181	0.2767	0.4583	---*
2011+**	---	0.0181	0.2767	0.0500	0.0100

[1] Because measurements of PM emissions were not performed for the Pre-1981 model year cars (or trucks), we applied the (later) 1982 standard of 0.6 grams per mile to those earlier model years. Due to questionable PM results for the 2006-2008 LDT, we used the LDV average PM value (0.0312 grams/mile).

[2]. For 2011 and newer model years, we used the smaller of the Tier-2 Bin-5 standard or actual test results from the 2006-2008 LDT.

To achieve the substantially lower FTP PM emissions, manufacturers are now equipping their diesel-fueled vehicles (cars and trucks) with particulate traps.

All of the Tier-2 diesel test vehicles in this sample were certified to the Bin-10 standards, implying that all 2006-2008 light-duty Tier-2 diesels sold in the USA were certified to the same standards. Since manufacturers are likely to target the Bin-5 standards in future model years, those average FTP emissions are probably not appropriate to represent model years beyond 2010. Therefore, we used the mean Bin 10 emissions of those Tier-2 vehicles to estimate the typical fleet emissions for only model years 2006 through 2010. For model years 2011 and later, we used the Bin-5 standards for NOx and PM. However, since the actual HC and CO (average) emissions (of those 2006 to 2008 test vehicles) were lower than the Bin-5 standards, EPA proposes using those lower test results for HC and CO (assuming that moving from Bin-10 to Bin-5 will not lead to an increase in HC or CO).

3.1.2 Estimating Bag Emissions:

The 700 certification (car and truck) test results were composite FTP results (HC, CO, NOx, and PM), not differentiated by test phase (bag). Therefore, the first task was to estimate the individual bag results based on the composite results.

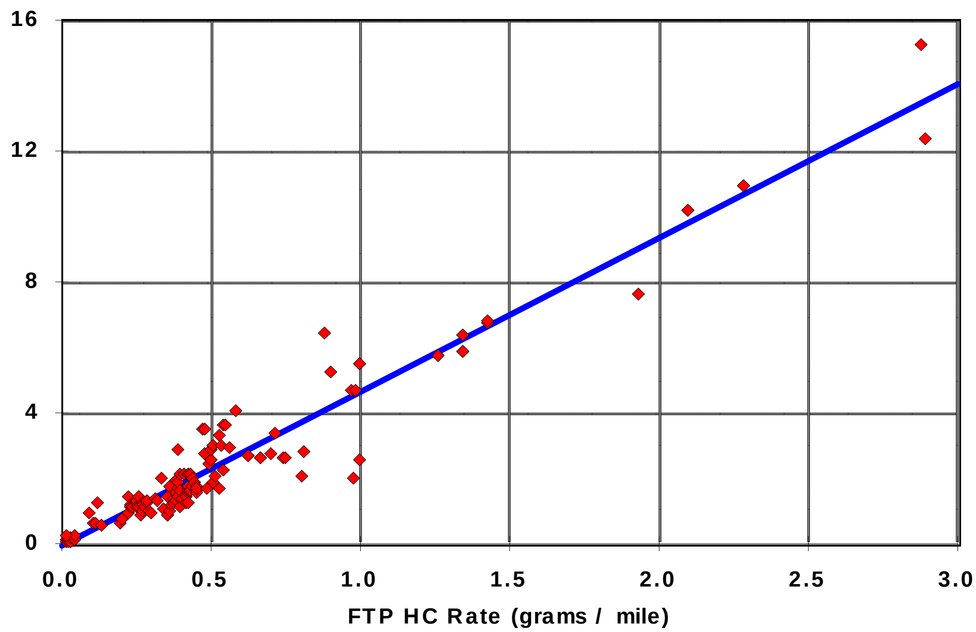
A smaller sample (151 tests) of FTPs from other data sets had emission results by bag. These FTPs of in-use vehicles (of various ages from various model years) were used only to develop correlations between the composite FTP emissions and the corresponding emissions of each of the three bags/modes.

We regressed the Bag-2 emissions (in grams per hour) against the corresponding composite FTP emissions (in grams per mile) to obtain an estimate of running emissions. For these regressions, we used a piecewise linear approach rather than a polynomial regression to account for slight curvature in the relationships. Similar analyses were performed regressing Bag-1 emissions and Bag-3 emissions (in total grams) each against the corresponding composite FTP emissions (in grams per mile). Each of the 14 regressions produces an equation, such as the following example, which correlates the Bag-1 "cold-start" HC emissions ($E_{HC,Bag1}$, g) to the corresponding composite FTP HC emission rate ($E_{HC,composite}$,g/mile):

$$E_{\text{HC,Bag1}} = -0.6433 + 4.702885\, E_{\text{HC,composite}} \qquad \textbf{3 - 1}$$

Graphing this equation along with the 146 FTP test results, as shown in Figure 3 - 1 below, illustrates the relationship between the individual bag HC emission and the composite HC emission.

Figure 3 - 1. Example: Bag-**1 HC (g) versus Composite FTP HC (g/mile)**



We then applied those 14 equations (derived from the regression analyses) to the corresponding composite FTP emissions from Tables 1 and 2. This step yielded (for each model year group in Tables 1 and 2) estimates of the emissions rate (in grams per hour) for Bag-2 as well as the total emissions (in grams) for each of Bag-1 and Bag-3.

We then assumed that the running emission rates (in grams per hour) on Bag-2 were comparable to the rates on the running portion of the Bag-1 (and Bag-3). Subtracting the total emissions associated with those running rates from the estimated total emissions of Bag-1 (based on the regressions of Bag-1 versus composite FTP) yielded estimates of the cold-start emissions (by model year). Similarly, subtracting the estimated running emissions from the estimated total

Bag-3 emissions produced estimates of hot-start emissions. Those estimated emission rates (running, cold-start, and hot-start) are summarized in the four following tables (Table 3 - 3 to Table 3 - 6), one table for each of the four pollutants.

Table 3 - 3. Estimated Aggregate HC Emission Rates.

Model Year Group	Diesel-Fueled Passenger Cars				Diesel-Fueled Light-Trucks		
	Running (g/hr)	Cold-Start (g)rt	Hot-Start (g)		Running (g/hr)	Cold-Start (g)rt	Hot-Start (g)
Pre-1981	8.0991	1.0961	0.1688		11.2131	1.6077	0.3280
1981-82	4.0262	0.5505	0.1626		5.6533	0.7784	0.2161
1983-84	3.1838	0.4325	0.1349		4.1427	0.5668	0.1664
1985	3.4727	0.4729	0.1444		3.6724	0.5009	0.1510
1986-90	3.2992	0.4486	0.1387		3.7835	0.5165	0.1546
1991-93	3.3802	0.4600	0.1414		4.8847	0.6707	0.1908
1994	3.6322	0.4953	0.1496		3.5308	0.4811	0.1463
1995-2005	2.1069	0.2816	0.0995		2.3782	0.3196	0.1084
2006-2008	0.1477	0.0071	0.0351		0.1226	0.0036	0.0342
2011+	0.1477	0.0071	0.0351		0.1226	0.0036	0.0342

Table 3 - 4. Estimated Aggregate CO Emission Rates.

Model Year Group	Diesel-Fueled Passenger Cars				Diesel-Fueled Light-Trucks		
	Running (g/hr)	Cold-Start (g)rt	Hot-Start (g)		Running (g/hr)	Cold-Start (g)rt	Hot-Start (g)
Pre-1981	21.3626	3.0900	1.0957		28.8186	4.0993	1.5010
1981-82	17.1121	2.5146	0.8647		21.1168	3.0567	1.0824
1983-84	15.3696	2.2787	0.7700		16.1856	2.3892	0.8144
1985	17.9833	2.6326	0.9121		17.6745	2.5908	0.8953
1986-90	21.6212	3.1250	1.1098		15.6605	2.3181	0.7858
1991-93	27.0463	3.8594	1.4046		27.2886	3.8922	1.4178
1994	19.3873	2.8226	0.9884		26.0552	3.7252	1.3508
1995-2005	5.9718	1.0066	0.2592		25.9270	3.7079	1.3438
2006-2008	8.0052	1.2818	0.3698		3.6954	0.6984	0.1355
2011+	8.0052	1.2818	0.3698		3.6954	0.6984	0.1355

Table 3 - 5. Estimated Aggregate NOx Emission Rates.

Model Year Group	Diesel-Fueled Passenger Cars			Diesel-Fueled Light-Trucks		
	Running (g/hr)	Cold-Start (g)rt	Hot-Start (g)	Running (g/hr)	Cold-Start (g)rt	Hot-Start (g)
Pre-1981	23.4257	1.6481	1.5561	27.6186	1.8543	1.7824
1981-82	19.5462	1.4573	1.3466	22.7786	1.6162	1.5211
1983-84	17.2503	1.3444	1.2227	21.5870	1.5576	1.4568
1985	13.6886	1.1692	1.0304	15.4631	1.2565	1.1262
1986-90	9.4389	0.9602	0.8009	23.9537	1.6740	1.5846
1991-93	8.9815	0.9377	0.7762	20.8139	1.5196	1.4151
1994	13.9128	1.1802	1.0425	22.8916	1.6218	1.5272
1995-2005	13.2512	1.1477	1.0067	24.2849	1.6903	1.6025
2006-2008	5.5883	0.8433	0.6673	6.4738	0.9325	0.7619
2011+	0.9813	0.3793	0.1751	0.9813	0.3793	0.1751

Table 3 - 6. Estimated Aggregate PM Emission Rates.

Model Year Group	Diesel-Fueled Passenger Cars			Diesel-Fueled Light-Trucks		
	Running (g/hr)	Cold-Start (g)rt	Hot-Start (g)	Running (g/hr)	Cold-Start (g)rt	Hot-Start (g)
Pre-1981	7.0131	2.4362	1.2789	7.0131	2.4362	1.2789
1981-82	3.3778	1.2427	0.6436	3.7378	1.3609	0.7065
1983-84	3.2356	1.1960	0.6188	3.0160	1.1239	0.5804
1985	3.0774	1.1441	0.5911	2.9830	1.1131	0.5746
1986-90	6.6108	2.3041	1.2086	3.1250	1.1597	0.5995
1991-93	5.7897	2.0346	1.0651	1.7460	0.4961	0.2167
1994	2.4073	0.6682	0.3020	1.5101	0.4347	0.1863
1995-2005	1.1338	0.3368	0.1378	1.2931	0.3782	0.1583
2006-2008	0.3738	0.1390	0.0397	0.3738	0.1390	0.0397
2011+	0.0739	0.0609	0.0010	0.0739	0.0609	0.0010

The PM rates in the preceding table represent the PM10 rates for all particulate matter on the collection filter (i.e., elemental carbon (EC), organic carbon (OC), sulfates, etc.). Disaggregating the PM estimates to obtain rates separately for EC and for OC, will be described in another report.

NOTE: start and running rates for light-duty diesels in model years 2010 and later were assumed to equal those for light-duty gasoline vehicles, as vehicles running on both fuels would be certified to the same standards. See Table 1 - 23 (dataSourceID 4910)

3.1.2 Assigning Operating Modes for Starts (Adjustment for Soak Time)

MOVES has start emission rates for eight different operating modes (opModes), each based on the length of the soak time prior to engine start. One mode corresponds to the 12 hour cold-soak (opmodeID = 108). The remaining seven modes have soak times ranging from three minutes up to nine hours (opModeID = 101-107).

Assuming that the start emissions change as functions of the temperature of the engine, and assuming that the engine temperature decreases (cools) exponentially with the soak period (i.e., length of time the engine is shut off), then we should be able to approximate the start emissions (following a soak E_{opModeID}) by exponential functions of the form:

$$E_{\text{opModeID}} = E_{108}\left(1.001 - \alpha\, \mathrm{e}^{-\beta t}\right) \qquad \textbf{3 - 2}$$

where E_{108} = cold-start emissions (g) and t = soak time (min), in minutes.

(Note that the factor of 1.001 (rather than 1.0) in the preceding equation allows the exponential curve to pass through the cold-start value at 720 minutes rather than simply approaching it.)

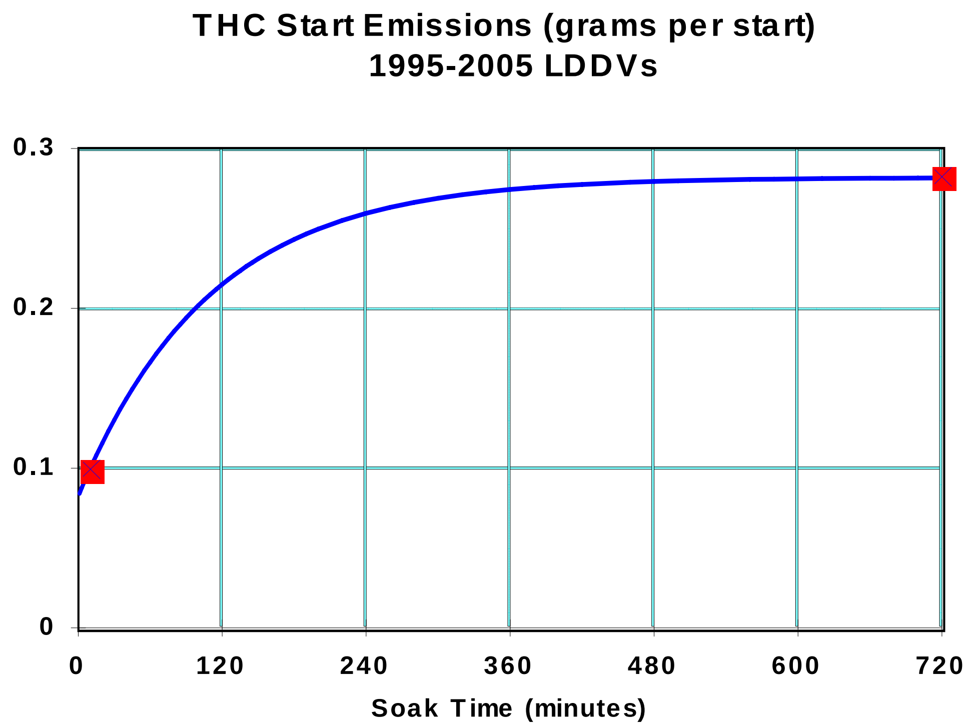
Using the estimated cold-start (CS) emissions i.e., emissions following a soak of at least 720 minutes (E_{108}) and the hot-start emissions i.e., the emissions following a soak of only 10 minutes (E_{101}) from the preceding four tables, we solved algebraically for both the α and β coefficients, specifically:

$$\begin{aligned} \alpha &= \mathrm{e}^{720\beta + \ln 0.001} \\ \beta &= \frac{\ln\left(1 - \dfrac{E_{101}}{E_{108}}\right) - \ln 0.001}{710} \end{aligned} \qquad \textbf{3 - 3}$$

This approach yielded a unique start emission curve (as a function of soak time) for each pollutant and for each model year group.

The effect of this exponential approach is illustrated in the following example plot which was created using the estimated cold-start THC emissions of 0.281593 grams for the 1995-2005 model year diesel-fueled passenger cars and the estimated hot-start THC emissions of 0.099486 grams from the preceding table.

Figure 3 - 2. Estimated THC Start Emissions (g) in terms of Soak Time (1995-2005 LDV).



This continuous concave curve is broadly comparable to the piecewise approach that the California Air Resources Board used in its analysis of the effect of soak time on the start emissions of gasoline-fueled vehicles and that EPA used in MOBILE6[35,36].

3.2 Running Emissions by Operating Mode

In MOVES, running emission rates are estimated for a set of operating modes defined in terms of vehicle-specific power, speed and acceleration (see Table 1 - 2). However, we lacked the requisite second-by-second data for the diesel-fueled passenger cars and light-trucks to perform those calculations. Therefore, we developed modal rates for LDT from corresponding rates for light heavy-duty diesel-fueled trucks (LHD<=14K) (i.e., from trucks with gross vehicle weight ratings between 8,500 and 14,000 pounds).

To adapt the LHDDT operating modes for application to LDDs, we developed operating mode frequencies in each mode for the 1,372-second LA-4 drive cycle (the first two phases of the FTP run sequentially). Due to differences in vehicle weight, we obtained separate (slightly different) distributions for passenger cars and and light-trucks, as shown in Table 3 - 7.

Table 3 - 7. Operating-Mode Distribution for the LA-4 Drive Cycle.

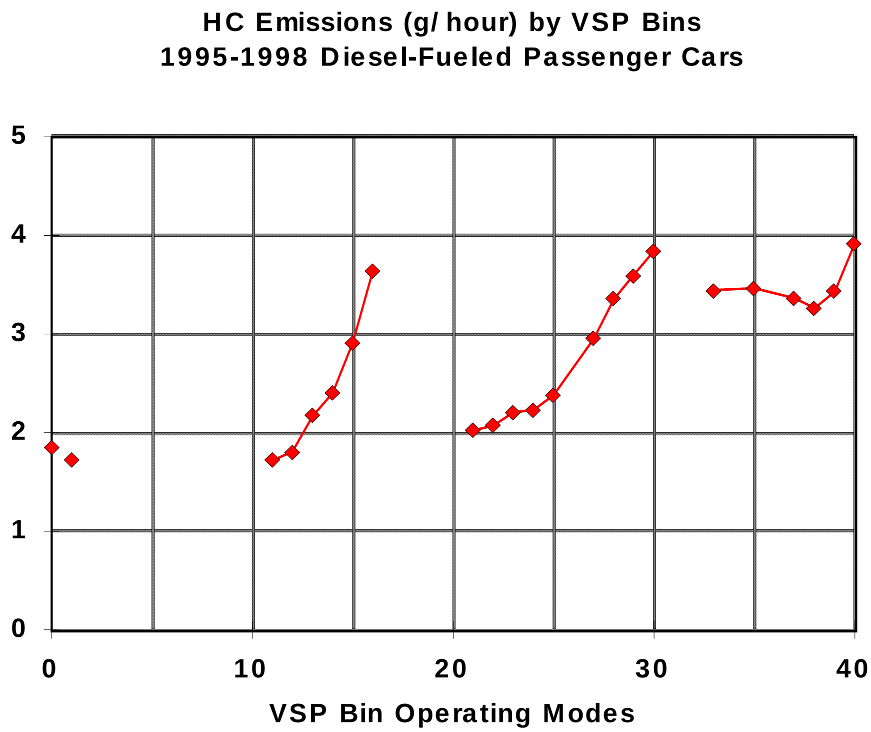
opModeID	LDV	LDT
0	164	164
1	255	255
11	93	96
12	142	139
13	99	103
14	69	66
15	34	33
16	20	20
21	68	70
22	149	164
23	123	110
24	35	33
25	21	19
27	14	15
28	8	7
29	2	2
30	0	0
33	25	29
35	35	33
37	13	11
38	3	3
39	0	0
40	0	0

Applying the appropriate distribution to the modal emission rates for the LHDDVs, we obtained estimates of the emission rates (in grams per hour) over a simulated LA-4 driving cycle. Dividing those rates into the hour running rates for LDD (Table 3 - 3 through Table 3 - 6), by model-year group, yielded ratios of the light-duty emission rates to the light heavy-duty rates. The resulting ratios are then used as adjustment factors to scale the modal LHD rates to give estimated .modal LDD rates For example, applying the LA-4 operating-mode distribution to the NOx modal rates for the 1999-2002 model year LHDDVs produces an estimated NOx rate of 143.66993 grams per hour compared to the actual passenger car average rate of 13.2512 grams per hour. Dividing yields a ratio of 0.092234. Therefore, we used that ratio (0.092234) as an adjustment factor to multiply all of the modal LHDDV rates for that model-year group to produce the corresponding VSP bins for the 1999-2002 model year diesel-fueled passenger cars. Thus, summing all of the LA-4 modal rates will exactly match the total estimate LA-4 (running) emissions.

Not all of the operating modes are represented by the LA-4 driving cycle. Specifically, modes 30, 39, and 40 do not occur during the LA-4. For this analysis, we applied the same adjustment factor to all operating modes.

This approach is illustrated by the following plot of the estimated zero-mile HC emission rates by VSP bins for 1995-1998 model year diesel-fueled passenger cars.

Figure 3 - 3. Modal Emission Rates for HC (g/hour) for 1995-98 diesel-fueled LDV.



4. Crankcase Emissions

In an internal combustion engine, the crankcase is the housing for the crankshaft. The enclosure forms the largest cavity in the engine and is located below the cylinder block. During normal operation, a small amount of unburned fuel and exhaust gases escape around the piston rings and enter the crankcase, and are referred to as "blow-by." This blow-by is potentially a source of vehicle emissions.

To alleviate this source of emissions, the Positive Crankcase Ventilation (PCV) system was designed as a calibrated air leak, whereby the engine contains its crankcase combustion gases. Instead of the gases venting to the atmosphere, they are fed back into the intake manifold where they reenter the combustion chamber as part of a fresh charge of air and fuel. A working PCV valve should prevent virtually all crankcase emissions from escaping to the atmosphere.

PCV valve systems have been mandated in gasoline vehicles since model year 1969. Turbocharged diesel engines have only required PCV valves since model year 2008. MOVES emission rates assume that all 1968 and earlier gasoline vehicles, and 2007 and earlier diesel vehicles do not have PCV valves.

The MOBILE series of models included crankcase emission factors solely for gasoline hydrocarbons. For purposes of MOVES, we have developed additional emission factors, as explained below.

Crankcase emissions are calculated in MOVES by chaining the emission calculators which calculate start, running, or extended idling emissions to a crankcase emission ratio. Crankcase emissions are calculated as a percentage of tailpipe emissions. These emissions are calculated selected pollutant processes, including THC, CO, and $NO_{x,}$ and the particulate fractions Organic Carbon PM 2.5, Elemental Carbon PM 2.5, Sulfate PM 2.5, Sulfate PM 10,. For each of these pollutants, the crankcase emissions are calculated from the start, running exhaust, or extended idling emissions of the same pollutant and then multiplying by the appropriate ratio in the CrankcaseEmissionRatio table.

With a working PCV valve, emissions are considered zero. Based on EPA tampering surveys, MOVES assumes a 4% PCV valve failure rate.[37] Consequently, the emission rates in fuel type/model year combinations that have PCV valves are estimated as 4% of the emission rates of those years with uncontrolled emissions.

Very little information is available on crankcase emissions, especially those for gasoline vehicles. A literature review was conducted in order to determine the best data sources for emission factors (Table 4 - 1).

Table 4 - 1. Selected Results for Crankcase Emissions Data

Authors	Year	Type	# Vehicles	HC	PM(all species)	CO	NOX	Units
Hare and Baines[38]	1973	Diesel	1	0.2-4.1	0.9-2.9	0.005-0.43	0.005-0.43	% of exhaust
Heinen and Bennett[39]	1960	Gasoline	5	33	X	x	x	% of exhaust
Bowditch[40]	1968	Gasoline	x	70	X	x	x	% of exhaust
Montalvo and Hare[41]	1985	Gasoline	9	*1.21-1.92*	*X*	*x*	*x*	g/mi
Williamson[42]	1995	Diesel	1	50	35	x	x	% of exhaust
Kittelson[43]	1998	Diesel	1	*x*	*0.038*	*x*	*0.005*	g/hp-hr
Hill[44]	2005	Diesel	9	x	100	x	x	% of exhaust
Ireson[45]	2005	Diesel	12	x	25-28	*x*	*x*	% of exhaust
Zielinka[46]	2008	Diesel	2	x	20-70	*x*	*x*	% of exhaust
	x = no data							

Based on this literature review, emission factors were estimated for years without mandated PCV valves (Table 4 - 2). In absence of better information, gasoline emission factors are largely a reflection of diesel research. As noted previously, model years with PCV valves were assigned emission factors which were 4% of the emission factors belows

Table 4 - 2. Emission Rates for Vehicles without PCV systems (percent of exhaust emissions)

Emission Type	Gasoline	Diesel
HC	33%	2%
NO_x	0.03%	0.03%
CO	0.005%	0.005%
PM (all speciations)	20%	20%

The crankcase emission factors for HC, CO and NO_x may underestimate emissions. These percentages of exhaust emissions are generally based on uncontrolled exhaust, which is not calculated by MOVES. MOVES produces exhaust estimates based on a number of control technologies (such as catalytic converters). Uncontrolled exhaust in the 1970s was significantly higher than current tailpipe exhaust.

A 1995 study by Williamson estimated a significantly higher proportion of HC, CO, and NO_x exhaust due to crankcase than earlier works. However, Williamson tested only a single engine. In absence of more consistent or compelling evidence, the emission factors in MOVES maintain consistency with those emission factors used in the NONROAD model.

Emission factors for other fuels (LPG, methanol, etc) were set equivalent to diesel emission factors. Emission factors for electric vehicles were set to zero.

Generally, the contributions of crankcase emissions to the overall emission inventory are expected to decrease as additional diesel vehicles acquire PCV systems.

. References

[1] Nam, E.K.; Giannelli, R. *Fuel Consumption Modeling of Conventional and Advanced Technology Vehicles in the Physical Emission Rate Estimator* (PERE). USEPA Office of Transportation and Air Quality, Ann Arbor, MI. EPA420-P-05-001, February, 2005.

[2] Heirigs, Philip, Robert Dulla, Robert W. Crawford. *Processing of IM240 Data for Use in MOVES*. SR007-05-02. Sierra Research, Inc., Sacramento, CA. May, 2007.

[3] Singer, B.C., R.A. Harley, D. Littlejohn, Jerry Ho and Thu Vo. 1998. Scaling of infrared remote sensor hydrocarbon measurements for motor vehicle emission inventory calculations. *Environ. Sci. Technol*. 32:21 3241-3248.

[4] Bishop, G.A. and D.H. Stedman. *On-road Remote Sensing of Automobile Emissions in the Denver Area: Year 6, January 2007.* Department of Chemistry and Biochemistry, University of Denver, Denver, CO. June 2007.

[5] Kish, L. 1965. *Survey Sampling*. John Wiley & Sons, New York.

[6] Neter, J.; Kutner, M.H.; Nachtsheim, C.J.; Wasserman, W. *Applied Linear Statistical Models*. Irwin, Chicago. Fourth Edition. 1996.

[7] McClement, D., Dulla, R.G. *Identification of Non-I/M Vehicles in I/M Program Vehicle Emission Datasets*. Draft Report, Sierra Research, Sacramento, CA., October, 2007.

[8] Air Quality Group, Aerospace, Transportation and Advanced Systems Laboratory, Georgia Technical Research Institute. *Biennial Evaluation of the Emissions Reduction Effectiveness of the Atlanta Vehicle Inspection and Maintenance Program for 2003-2004*. Prepared for the Air Protection Branch, Environmental Protection Division, Georgia Department of Natural Resources, Atlanta. September, 2007.

[9] DeHart-Davis, L., Corley, Elizabeth and Micheal O. Rodgers. 2002. Evaluating vehicle inspection/maintenance programs using on-road emissions data. *Evaluation Review*. 26(2) 111-146.

[10] Bishop, G.A., and D. H. Stedman. 2008. A decade of on-road emissions measurements. *Environ. Sci. Technol*. Xxs, sss-sss.

[11] M6. EXH.??? Report for NOx
[12] M6.EXH.012. Report for HC, CO
[13] M6.STE.003.

[14] Kishan, S.; Burnett, A.; Fincher, S.; Sabisch, M.; Crews, B.; Snow, R.; Zmud, M.; Santos, R.; Bricka, S.; Fujita, E.; Campbell, D.; Arnott, P. (2007), *Kansas City PM Characterization Study – Final Report.* EPA Contract Report (ERG No. 0133.18.007.001), October, 2006.
[15] Nam et al., 2008.

[16] Gibbs, R.E., Wotzak, G.P., Byer, S.M., Kolak, N.P., *Sulfates and Particulate Emissions from In-Use Catalyst Vehicles. Regulated/Unregulated Emissions and Fuel Economy*. Report: EPA-600/9-79-047, 1979.

[17] Hammerle, R.H., Korniski, T.J., Weir, J.E., Cladek, E., Gierczak, C.A., Chase, R.E., Hurley, R.G., *Effect of Mileage Accumulation on Particulate Emissions from Vehicles Using Gasoline with Mehthylcyclopentadienyl Manganese Tricarbonyl.* SAE920731, 1992.

[18] Whitney, K.A., *Collection of In-Use Mobile Source Emission Samples for Toxicity Testing,* Southwest Research Institute. 08.02602, October, 2000..

[19] Gibbs et al., 1979
[20] Cadle et al., 1979

[21] Urban, C.M., Garbe, R.J., *Regulated and Unregulated Exhaust Emissions from Malfunctioning Automobiles*. SAE790696, 1979.

[22] Urban, C.M., Garbe, R.J., *Exhaust Emissions from Malfunctioning Three-Way Catalyst Equipped Automobiles*. SAE800511, 1980.

[23] Lang et al., 1981
[24] Volkswagen, 1991

[25] California Air Resources Board. *Particulate Exhaust Emissions: Gasoline Powered Vehicles*. Emissions Studies Section, El Monte, CA. Project No. 2R8618, 1986.

[26] Ford, 1992
[27] CRC E-24-1 (Denver)
[28] CRC E-24-3 (San Antonio) <same cite as #31?>
[29] CRC E-24-2 (Riverside)
[30] Ford, Chrysler, and Chase, 2000.

[31] Whitney, K., *Measurement of primary exhaust particulate matter emissions from light-duty motor vehicles*. Southwest Research Institute Report E-24-3, 1998. <is this citation correct?>

[32] Kansas City study (summer)
[33] EPA (MSAT)

[34] Li, W., Collins, J.F., Norbeck, J.M., Cocker, D.R., Sawant, A., *Assessment of Particulate Matter Emissions from a Sample of In-Use ULEV and SULEV Vehicles*. SAE 2006-01-1076, 2006.

[35] Sabate, S. *Methodology for Calculating and Redefining Cold and Hot Start Emissions.* March 1996. (Available at: www. epa.gov/otaq/models/mobile6/m6tech.htm)

[36] Glover, E. and P. Carey, *Determination of Start Emissions as a Function of Mileage and Soak Time for 1981-1993 Model Year Light-Duty Vehicles*. USEPA Office of Transportation and Air Quality. EPA420 R 01 058 (M6.STE.003), November 2001. (http://www.arb.ca.gov/msei/onroad/downloads/pubs/starts.pdf)

[37] US Environmental Protection Agency. EPA Motor Vehicle Tampering Survey. EPA # 420-A-90-001. 1990.
[38] Hare, Charles T.; Baines, Thomas M.; Characterization of Diesel Crankcase Emissions. Society of Automotive Engineers Off-Highway Vehicle Meeting and Exhibition. MECCA, Milwaukee. 1977.

[39] Bennet, PA et al. Reduction of Air Pollution by Control of Emissions from Automotive Crankscase, SAE Paper 142A, January 1960.

[40] Bowditch, Fred W. The Automobile and Air Pollution, SAW Paper No. 680242 presented at SAE Mid-Year Meeting Milwaukee, WI, May 1968.

[41] Montalvo and Hare, 1985 EPA 460/3-84-011 *Crankcase Emissions with disabled PCV systems*

[42] Williamson, James. 1995. Exhaust emissions and fuel economy evaluation of SV technologies power valve crankcase ventilation and oil recovery system. SV Technologies report

[43] Kittelson, David B. Matthew Spears, Todd Taubert, and Bob Waytulonis, "Measurement and Characterization of Crankcase Blow-by Emissions Phase I Report to Caterpillar, Inc.", University of Minnesota, Center of Diesel Research, February 25, 1998

[44] Hill, Zimmerman, Gooch A Multi-City Investigation of the Effectiveness of Retrofit Emissions Controls in Reducing Exposures to Particulate Matter in School Buses. January 2005. Clean Air Task Force.

[45] Ireson, Robert. Intentional Tracer Measurements of Tailpipe and Crankcase Particulate Matter Concentrations Inside School Buses: Results, Refinements, and Effects of Bus Operating Conditions . 18th CRC.

[46] Detailed Characterization and Profiles of Crankcase and Diesel Particulate Matter Exhaust Emissions Using Speciated Organics .Zielinska, B.; Campbell, D.; Lawson, D. R.; Ireson, R. G.; Weaver, C. S.; Hesterberg, T. W.; Larson, T.; Davey, M.; Liu, L.-J. S. Environ. Sci. Technol.; (Article); 2008; 42(15); 5661-5666.