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REVIEW OF THE 2009 DRAFT MOTOR VEHICLE EMISSIONS SIMULATOR (MOVES) MODEL

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COORDINATING RESEARCH COUNCIL, INC.
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Final Report

Review of the 2009 Draft Motor Vehicle Emissions Simulator (MOVES) Model

CRC Project: E-68a

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List of Acronyms

AAM**:** Alliance of Automobile Manufacturers
AIAM: Association of International Automobile Manufacturers
API: American Petroleum Institute
AZ: Arizona
BAR: Bureau of Automotive Repair (of California)
CAAA: Clean Air Act Amendments
CARB: California Air Resources Board
CAVTC: Clean Air Vehicle Technology Center
CO: Carbon monoxide
CRC: Coordinating Research Council
CY or CYR: Calendar year
DPF: Diesel particulate filter
DR: Deterioration rate
E85: Mixture of 85% ethanol, 15% gasoline
EC: Elemental carbon particulate matter
EMA: Engine Manufacturers Association
EPACT: Energy Policy Act of 2005
ERG: Eastern Research Group
FACA: Federal Advisory Committee
FID: Flame ionization detector (emissions measurement instrument)
FTP: Federal Test Procedure
FVV: Fuel vapor venting
GVWR: Gross vehicle weight range
HC: hydrocarbons
HD: Heavy-duty
HHD: Heavy-heavy-duty vehicle
HDDV: Heavy-duty diesel vehicle
I/M: Inspection and maintenance
I/M147: I/M 147 test (147 seconds long)
I/M240: I/M 240 test (240 seconds long)
IUVP: In-use vehicle program
KC: Kansas City
LA: Los Angeles
LA92: Higher average speed driving cycle
LDGV: Light-duty gasoline vehicle (passenger car)
LDGT: Light-duty gasoline truck
LDT1: Light-duty gasoline truck in 0-3750 lb gross vehicle weight range
LDT2: Light-duty gasoline truck in 0-5750 lb gross vehicle weight range
LDT3: Light-duty gasoline truck in 5750-8500 lb gross vehicle weight range
LDT4: Light-duty gasoline truck in 5750-8500 lb gross vehicle weight range
LDV: Light-duty vehicle
LEV: Low emissions vehicle
LEV1: Low emissions vehicle meeting the LEV1 standards (prior to model year 2004)
LEV2: Low emissions vehicle meeting the LEV2 standards (2004 and later model years)

LHDT: Light heavy-duty truck (8500-10000 lbs)
MEC: Modal emissions cycle
MEMS: Mobile emissions measurement system
MDPV: Medium-duty passenger vehicle (passenger vehicle above 8500 lbs)
MHD: Medium heavy-duty vehicle
MIL: Malfunction indicator light
MOVES: Motor Vehicle Emissions Simulator
MSAT: Mobile source air toxics
MSOD: Mobile Source Operations Division of EPA
MSTRS: Mobile Source Technical Review Subcommittee
NAS: National Academy of Sciences
NDIR: Non-dispersive infra-red emissions measurement instrument
NFRAQS: Northern Front Range Air Quality Study
NGM: New Generation Model
NH_3: Ammonia
NLEV: National Low Emissions Vehicle
NMHC: Non-methane hydrocarbons
NMIM: National Mobile Inventory Model
NMOG: Non-methane organic gas
NO_2: Nitrous oxide
NOx: Oxides of nitrogen
NPRM: Notice of proposed rulemaking
NREL: National Renewable Energy Laboratory
NYCC: New York City Cycle
OBD: Onboard diagnostics
OC: Organic carbon particulate matter
OpMode: Operating mode (VSP bin)
PCV: Positive crankcase ventilation
PFI: Port fuel injection
PM: Particulate matter
PM_{10}: Particulate matter less than or equal to 10 microns in size
$PM_{2.5}$: Particulate matter less than or equal to 2.5 microns in size
PZEV: Partial zero emissions vehicle
QCM: Quartz crystal microbalance
RFG: Reformulated gasoline
RFS: Renewable fuel standard
RVP: Reid vapor pressure (measure of gasoline volatility)
ROVER: Real-time on-road vehicle emissions reporter
SHO: Source hours operating (measure of activity)
SFTP: Supplemental federal test procedure
SO_2: Sulfur dioxide
SwRI: Southwest Research Institute
T&M: Tampering and malmaintenance
TBI: Throttle body injection
THC: Total hydrocarbons
TOR: Thermal optic reflectance system

UDDS: Urban dynamometer driving schedule
ULEV: Ultra low emissions vehicle
US06: US06 testing procedure at higher speeds and loads
VOC: Volatile organic carbon
VMT: Vehicle miles traveled
VSP: Vehicle Specific Power
WVU: West Virginia University
ZML: Zero mile level

Follow-Up MOVES Review
CRC Project: E-68a

1.0 Summary

EPA's new MOVES model represents a significant technical breakthrough in modeling on-road emissions. The model replaces the "MOBILE" series of emissions models.

The technical breakthrough in emission modeling is due to two primary factors. First, the MOVES model is based on a new understanding and new methods of evaluating emissions processes, for both exhaust and evaporative emissions. The MOBILE models were based on emission results from a set of test cycles. For example, the Federal Test Procedure driving cycle, or FTP, is commonly used to estimate exhaust emissions from on-road vehicles. Similarly, the diurnal and hot soak tests were used to evaluate evaporative emissions. As a result, the output from the MOBILE model for exhaust emissions is in grams of emissions per mile, which is considered a "cycle-average" output. While this could be corrected for average speed, it is difficult to use this cycle-average output to evaluate emissions in micro-environments such as intersections or vehicle tunnels. The MOVES model estimates running exhaust emissions as a function of vehicle specific power, or VSP. This methodology allows the user to input any vehicle operation cycle and evaluate emissions, giving MOVES much greater flexibility than MOBILE. The same model can be used to produce emissions on a micro-scale, meso-scale, and macro-scale.

MOBILE6 model evaporative emissions are based on testing procedures that include multiple evaporative processes. For example, the diurnal test, which measures emissions over the 1-day or 3-day period while the ambient temperature is cycling, actually measures emissions from three different evaporative emission processes – fuel permeation, fuel vapor venting, and leaks (if present). Instead of basing the evaporative emissions on the summed test results, the MOVES model bases the evaporative emissions on the underlying evaporative processes. The result should be a more accurate method of evaluating evaporative emissions.

The second major factor is that the MOVES exhaust emissions for light-duty vehicles are based on inspection and maintenance tests. The MOBILE model is based on in-use surveillance tests, where vehicles were recruited from the general public for testing. Response rates for these tests were usually significantly below 50%, which raised concerns that the test sample is biased. In states with I/M programs, vehicles must pass an I/M test in order to be re-registered. This approach captures a much greater fraction of the driving population than in-use surveillance testing. It does not include vehicles whose owners refuse to re-register their vehicles but drive them anyway, vehicles excluded from participation in the program, or vehicles that are re-registered outside of the I/M area but are driven in the I/M area. But the biases for I/M data are thought to be less than biases for in-use surveillance testing. A correction factor is used to estimate emissions in non-I/M areas.

There are no I/M programs that measure all criteria pollutants from heavy-duty diesel vehicles, so I/M data can not be used to model this category in MOVES. Instead, the MOVES model is based on in-use surveillance testing of a number of trucks in several different testing programs. In addition, the exhaust emissions are calculated using the same VSP methodology as applied to the light-duty vehicles.

In addition to the above factors, there are many other ways in which MOVES differs from MOBILE and where new methods and new data were used to develop the new emissions model.

Given the groundbreaking nature of the MOVES model, the Coordinating Research Council (CRC) contracted with a study team consisting of Air Improvement Resource, E. H. Pechan and Associates, and Dr. Albert Hochhauser to critically review the draft MOVES model. Work was started in February 2009 and concentrated in two areas: (1) reviewing the data and methods used to estimate emission rates and emission correction factors and summarizing the major assumptions EPA had made, and (2) performing many sensitivity runs of the MOVES model. The vehicle activity inputs, which can have a significant impact on overall emissions, were not evaluated.

This evaluation is based primarily on the draft MOVES model, which was released in April 2009. Some evaluations were performed with an August version of the model that was essentially the same as the April version, except that it contained features making it easier to use. EPA has continued to modify the model during the evaluation period, including addressing issues raised in this report, since they were included on the technical panel that reviewed the project report.

EPA has included error analysis in the model. However, this feature makes the model run too slowly to be practical. Error analysis could be a valuable tool to point towards the most critical data needs. Evaluation of the error analysis methodology and the error inputs was beyond the scope of this project.

A number of comments on the methods and data used in the MOVES model were developed from this review. The most important issues are summarized below. The discussion is divided into the following sections:

- Exhaust Emissions
- Evaporative Emissions
- Heavy-Duty Vehicle Emissions
- Emission Correction Factors
- Particulate Matter Emissions
- MOVES versus MOBILE6

Exhaust Emissions

One of the critical issues is whether exhaust emissions are correctly apportioned between the start and running modes of vehicle operation. MOVES shows that HC exhaust emissions in the summer occur mostly during vehicle starts, while CO and NOx are generated primarily from the running operating mode. In the winter, HC and CO emissions are dominated by starts, while NOx is about ½ starts and ½ running. All three pollutants, however, decline dramatically between 2008 and 2020 due to the light-duty Tier 2 vehicle emission standards.

The start emissions consist of cold starts, hot starts, and intermediate starts. EPA developed the emission rates for cold and hot starts from very recent data. However, the emission rates for intermediate starts were developed by relying, in part, on ratios of intermediate soak emissions to cold start emissions determined from tests limited to Tier 1 and earlier vehicles. It is not known whether these ratios are still valid for newer technology vehicles. *It is recommended that intermediate start testing be performed on Tier 2 and LEV2 (LEV2, ULEV2 and SULEV) vehicles to determine whether the factors currently being used are still appropriate for these newer technology vehicles. A pilot program testing a few vehicles could be conducted to determine if the current factors are still appropriate.*

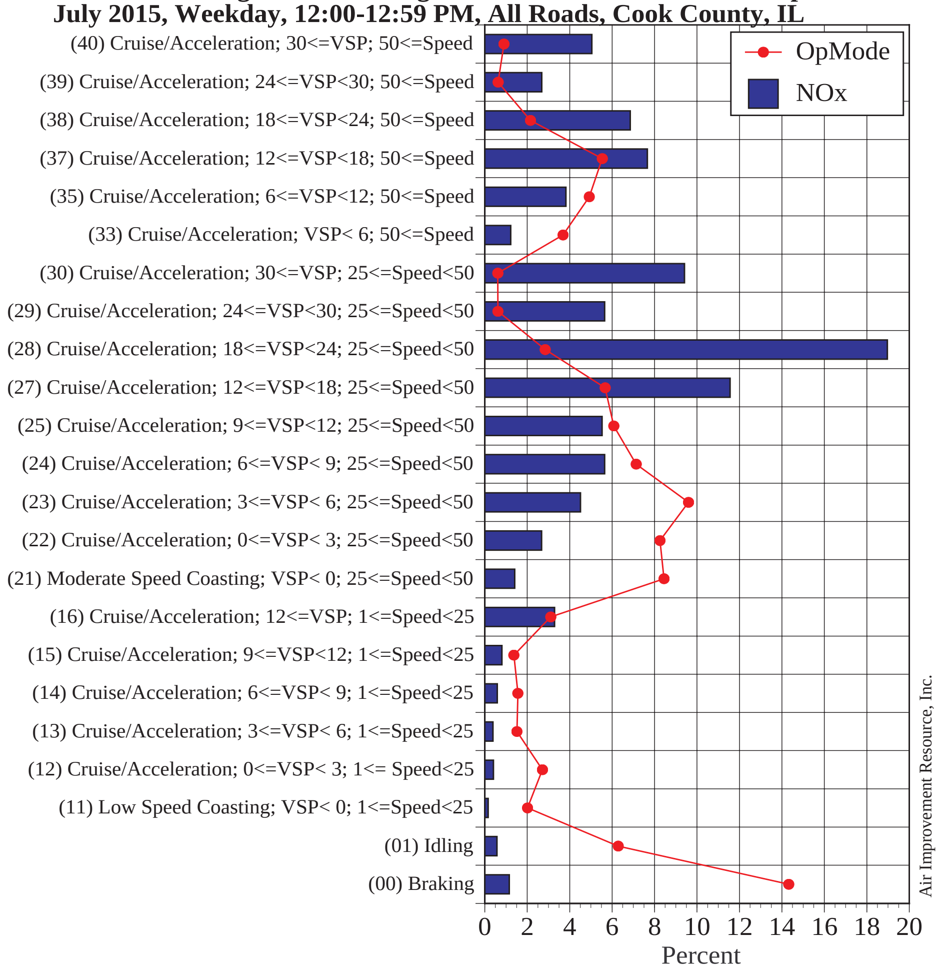
A second concern is exhaust emissions deterioration. Where the MOBILE model evaluated emissions versus mileage, MOVES estimates emissions versus vehicle age (or age bin). EPA's analysis of the Arizona I/M data concluded that the log of emissions increased linearly versus vehicle age until about 15-17 years and then leveled off. EPA developed an S-curve to fit this type of deterioration model, where the deterioration starts out gradually, then increases faster, and finally flattens.

EPA did not divide the vehicles into low and high emitters for this analysis, as has been done with the MOBILE models. This project's analysis of the Arizona data for Tier 1 vehicles (1996 and later), however, did not find evidence that the log of emissions increased with age. Not only is the deterioration for these vehicles much lower than previous model years and technologies, but it appeared that emissions, instead of the log of emissions, increased linearly with age.

The authors concur with EPA that these emissions flatten at the moderate to higher ages. This flattening in emissions is thought to be due to the higher emitting vehicles being retired from use, leaving the older vehicles that are generally well maintained and with lower emissions. It is recommended that EPA revise the emissions deterioration for Tier 1 and later vehicles to be linear rather than log-linear. If emissions deteriorate as the analysis suggests instead of an "S-curve", then overall future light-duty emission inventories would be lower, and cold start emissions would probably be a greater fraction of total emissions than currently estimated by MOVES. *The authors recommend obtaining more up-to-date data from the Arizona I/M program, especially for 2001 and later vehicles, and performing further analysis of vehicle deterioration using these data.*

A third concern is that emissions at higher VSP bins, which represent emissions at higher speeds and acceleration rates, are driving the overall emission inventories. EPA includes "high-VSP" correction factors to adjust the emissions in some of the higher VSP bins. These correction factors are based on a different data set than the Arizona I/M data used to develop the lower VSP bins. An analysis of the distribution of emissions by VSP bin, for the calendar year 2015 passenger car fleet, and for Tier 2 vehicles in calendar year 2015, showed that about 50% of the HC, CO, and NOx emissions were in the four highest VSP bins for the 2015 fleet. This is shown for NOx from the passenger car fleet in calendar year 2015 in the figure below.

Percent of Gas Passenger Car Running NOx Emissions and MOVES OpMode
July 2015, Weekday, 12:00-12:59 PM, All Roads, Cook County, IL



These four highest bins represented only about 5% of the total driving (by time). When this analysis is extended to just Tier 2 vehicles, 70% of the emissions were in the highest VSP bins with only 5% of the operation. However, the draft MOVES model did not include the estimated effect of supplemental FTP (SFTP) standards, which reduce emissions at high speeds and loads, so these percentages (50% and 70%) may be different in the final MOVES model, because EPA did include the effect of these regulations in the final model. This analysis concludes that for estimating either current or future emissions, attention should focus on accurately portraying emissions in the higher VSP bins, which have little actual operation and are almost entirely outside of the I/M tests. *It is recommended that the distribution of emissions analysis by VSP bin be re-done with the final MOVES model. If the majority of emissions are still in the top four bins, then much more data in these VSP bins should be collected, with random tests of I/M vehicles on cycles other than the IM147.*

A fourth concern is the emission rates of recent and future light-duty trucks. EPA started with estimates of Tier 1 emission rates for cars and light trucks based on analysis of Arizona I/M data and then reduced these estimates by the ratio of Tier 2 to Tier 1 emissions standards in order to develop emission rates for Tier 2 and later vehicles. In model year 2010, the Tier 2 HC and NOx standards of cars and LDTs are almost equal. However, since the Arizona I/M based Tier 1 emission rates for cars versus light trucks are very different, the resulting emissions estimates by VSP bin in 2010 are not nearly as close as the emission standards. *It is recommended that EPA re-estimate light-duty truck emission rates meeting the Tier 2 standards so that the LDT emissions by VSP bin are nearly the same as for passenger cars by 2010 (that is, LDT emission rates reflect the very small difference in Tier 2 emission standards). The activity by VSP bin can still be different, resulting in different overall emissions, but the emissions by VSP bin should be approximately the same.*

EPA developed base emission rates for I/M areas, and therefore it is necessary to develop separate emission rates for non-I/M areas as well. EPA examined emissions of vehicles migrating into the Arizona I/M program from non-I/M areas. In general, these vehicles averaged higher emissions than the vehicles of the same age and vintage in Arizona. Ratios of the non-I/M migrating vehicle emissions to I/M vehicle emissions were developed from the Arizona data, and were subsequently used to estimate emissions in non-I/M areas. This method and the underlying data were not examined in great detail, partially because the I/M adjustment factors were deemed to be reasonable. *With dramatically declining vehicle emissions, the value of I/M could also be declining rapidly. It is recommended that a more detailed vehicle migration study of exhaust I/M benefits be performed for additional I/M programs other than Phoenix.*

Evaporative Emissions

EPA developed a completely new approach to modeling evaporative emissions for MOVES. In the new approach, evaporative emissions are estimated for the major evaporative processes – leaks, vapor venting emissions, and permeation through fuel system components.

MOVES predicts that evaporative emissions from light-duty gasoline vehicles are about half of total VOC emissions (annual or summer), and that this ratio will remain approximately constant as VOC emissions are reduced in the future. Evaporative VOC emissions in the summer of 2008 from the passenger car fleet in Cook County, IL are about 0.195 g/mi, and are dominated by vapor venting emissions. These are reduced to 0.075 g/mi in 2020, and arise equally from leaks, permeation, venting, and refueling emissions. MOVES estimates that I/M programs inspecting for missing gas caps have little effect on fleet average evaporative emissions.

The first concern is that for new technology vehicles subject to the Tier 2 evaporative standards and the Mobile Source Air Toxics evaporative standard, EPA assumes that only vapor venting evaporative emissions are reduced. The authors assert that since the evaporative standards measure permeation as well as vapor venting, there will be reductions in permeation emissions as well as vapor venting emissions. *Data are or will be available from the CRC E-77 program to evaluate EPA's assumption. The CRC E-77 data should be examined by EPA as soon as possible to determine if the assumption of no improvement in permeation emissions is valid.*

A second concern is that there are no evaporative emission estimates for Partial Zero Emission Vehicles (PZEVs). These vehicles are sold throughout the nation, and in particular in states that have adopted the California Low Emission Vehicle Program. *This technology category and its associated evaporative emissions should be incorporated into the model.*

Finally, the evaporative emissions of new technology vehicles are very low, as expected. The increase in evaporative emissions is driven by two factors: (1) the estimated frequencies of leaking vehicles and their associated emissions, and (2) the estimated frequencies of vapor leaks and their associated emissions. EPA has a testing program underway to evaluate the frequency of liquid leaking vehicles and high evaporative emissions vehicles. This will provide valuable information which should impact MOVES. In addition, the CRC E-77 program is investigating the emissions associated with vapor leaks in new technology vehicles. *However, a more complete review is needed of the frequency of vapor leaks in-use in I/M and non-I/M areas by vehicle technology and age. Issues to be examined are (1)if the current EPA method double-counts vapor leaks for different tests (gas cap check, OBD, and fueling inlet leak check) that are being added together to estimate the failure rates, and (2) if newer data from I/M programs exists for estimating evaporative system OBD failures.*

Heavy-Duty Vehicle Emissions

In MOBILE6, heavy-duty emissions are based on the emission standards for engines that are in g/bhp. Conversion factors are required to translate the engine emission rates to units of g/mi for vehicles. In MOVES, emissions are based on chassis emission tests and on-road truck testing that provide g/mi emission rates, so there is no need for the conversion factors. In addition, similar to the light-duty vehicle emissions modeling approach, the heavy-duty emission rates are divided into VSP bins. Extended idle emissions have been added to the inventory, which is a category of emissions not accounted for in MOBILE6. Lastly, EPA includes tampering and malmaintenance factors, which increase heavy-duty vehicle emissions with age.

The first concern is that EPA has not included the effects of heavy-duty reflash programs on vehicle emissions. The Heavy-Duty Consent Decree signed in 1998 between EPA and seven heavy-duty engine manufacturers required, among other things, that the onboard computers of on-road heavy-duty truck engines built in the mid-1990s receive a "reflash" calibration to reduce NOx emissions at the time of the first engine rebuild. This should reduce NOx emissions from in-use trucks. *It is recommended that this factor be included in the final MOVES model.*

The second concern is that EPA did not include the effects of heavy-duty onboard diagnostic (OBD) regulations on reducing tampering and malmaintenance. EPA projects that tampering and malmaintenance will increase the emissions from trucks equipped with particulate traps and NOx reduction strategies. OBD requirements should significantly reduce any tampering and malmaintenance on these vehicles. At the November 10, 2009 FACA meeting, EPA proposed to incorporate OBD requirements by assuming that tampering and malmaintenance would be reduced by 33% as compared to vehicles without OBD. *Any assumption made is arbitrary, since there are not yet any heavy-duty vehicles on the road with OBD. However, 33% is a very low effectiveness rate. There will be a need to determine how heavy-duty diesel vehicle owners comply and respond to the OBD requirements.*

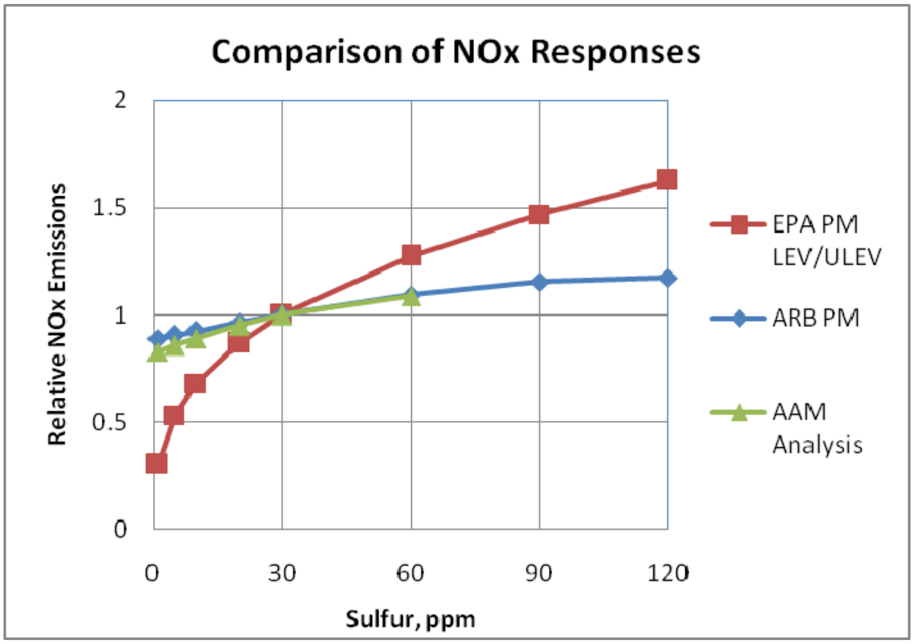
The third concern is that idle emissions from heavy-duty trucks are overestimated by assuming (1) that all extended idle events occur at high idle conditions, (2) there is no compliance with any state anti-idling regulations, and (3) there is no use of "hoteling" facilities at truck stops that eliminate the need for extended idling. There is survey data on truck operators who were asked what idle speed they set their trucks on. Responses indicate that a significant fraction of the time they use lower idle settings. Finally, many trucks are using hoteling facilities that eliminate the need for idling. *It is recommended that the idling assumption be revised to include the surveyed fraction of idling at lower speeds, and the estimated impacts of state anti-idling requirements and hoteling frequencies on average idle times for long-haul truck be recalculated. Additional truck survey data may be needed to quantify these effects.*

The fourth concern is that the chassis and on-road emissions test data for pre-2007 heavy-duty vehicles used to develop emission rates by VSP bin includes some possible tampering and malmaintenance. Yet the MOVES model adds additional tampering and malmaintenance effects for HC and CO on top of these emissions, thereby double-counting tampering and malmaintenance effects on these vehicles. *It is recommended that one of two options be applied to the model: either removed the vehicles that are tampered or malmaintained from the chassis and on-road tests data used to develop the emission rates by VSP bin, or refrain from the separate application of tampering and malmaintenance effects to these vehicles.*

Correction Factors (Fuels and Temperature)

The fuel correction factors for MOVES are based on MOBILE6 and the EPA COMPLEX Model fuel parameter relationships. New test data are available to estimate new fuel correction factors, but EPA has decided to wait until an extensive project (EPAct) testing fuel effects is completed to update the fuel correction factors.

The most serious fuel-related concern is that the sulfur correction factors are based on a log-log relationship of emissions versus sulfur that was developed on fuel sulfur levels that are much higher than today's levels. The log-log relationship predicts a very steep decline in emissions below 30 ppm that is not represented by more recent tests on vehicles at lower sulfur levels. This steep decline in emissions is shown in the following figure.



The log-log relationship is represented by the line labeled "EPA PM (Predictive Model) for LEV/ULEV. The other two lines represent more recent tests collected by the AAM (Alliance of Automobile Manufacturers) and newer data embodied in the ARB PM (Air Resources Board Predictive Model). *The emissions versus sulfur relationship should be revised for the final MOVES model. Data from EPAct and MSAT should be helpful.*

MOVES does not provide a means for assessing the impact of certain metal- or non-metallic additives used in fuel. These additives are not widely used in the US, and suitable emissions data would be needed before they could be included.

Regarding temperature correction factors, EPA's examination of recent data found that cold start HC, CO, and NOx emissions should be adjusted for temperature, but found no ambient temperature effect on running exhaust emissions. EPA developed additive cold start increments for HC, CO, and NOx that increase with lower temperatures.

One concern with the temperature increments is that there is no analysis of how these may change as vehicles age. Additionally, the available data seem to omit the CRC E-74b testing program, which was completed in May 2009. *EPA could utilize the Kansas City temperature data to determine if the temperature relationships change with vehicle age. Also, the CRC E-74b testing program data could be used to further check the MOVES cold start correction factors.*

A second concern is that the method used to develop HC temperature increments for the MSAT rule (which requires lower HC standards at cold temperatures) assumes a compliance margin with respect to the HC standards at 75° F, but no compliance margin with respect to the HC standards at 20° F. As a result, the HC increments for vehicles meeting the MSAT requirements are over-estimated. *The method should be revised to include a compliance margin at 20°F to be consistent with the margin currently being utilized at 75°F.*

A third concern is that vehicles subject to the lower MSAT HC standards will very likely have much lower CO emissions as well. *Once vehicles are certified to the MSAT cold HC standards, an analysis should be conducted of certification or other data to determine how much the CO increments change for these vehicles as well.*

Particulate Matter Emissions for Gasoline Vehicles

EPA combined results from the recent CRC Kansas City Program with many other recent PM programs in determining average PM emissions from gasoline vehicles. However, the Kansas City testing program was the first large scale PM testing program to collect second-by-second PM data using several different instrumentation methods in addition to filter data. The second-by-second data are needed to estimate emissions by VSP bin similar to HC, CO, and NOx. In addition, the Kansas City data included tests at summer and winter temperatures on the same vehicles (i.e., "matched pairs"). The test cycle used in Kansas City to collect PM data is the LA-92 cycle.

PM emissions were developed for newer vehicles in the earliest age category by combining the newer vehicles in all of the different studies, and developing a curve of emissions by model year. The newer vehicle emissions were then grouped into similar model year groups. Deterioration rates were estimated by regressing the log of PM emissions versus age, and capping the deterioration at 20 years of age, substantially longer than HC, CO, and NOx emissions. Newer model year vehicles were assumed to have the same percent deterioration in age as older vehicles. This did result in much lower emissions versus age than the older vehicles, since the newer vehicles had much lower starting levels than the older vehicles.

Temperature correction factors were estimated from the matched vehicle pairs in the Kansas City study, and from two EPA test programs, one of which supported the development of MSAT regulations. Unlike HC, CO, and NOx emissions, where the temperature correction factors were only for cold start emissions, EPA found an increase in running PM emissions with decreasing ambient temperatures, albeit lower than for the cold start.

The first concern is that the combined MSAT and Kansas City data on matched pairs do not appear to support a cold temperature adjustment for running emissions. *Results from other studies such as NFRAQS should be included in the analysis, with special regard to high PM emitters.*

A second concern is that EPA should check the low mileage emissions of the older model year vehicles to determine the fuels and measurement methods that were used. The use of higher sulfur fuels and older measurement methods could be biasing the PM emissions for older vehicles on the high side.

A third concern is that in the draft model, vehicles meeting lower HC standards in response to the MSAT rule are not assumed to have lower PM emissions. Since HC and PM emissions seem to correlate well, it is thought that there will be lower PM emissions with a lower HC standard at cold temperature. *The section on correction factors recommends evaluating certification data or other data to examine the effect of cold HC standards on HC and CO emissions. This should be extended to PM as well if possible.*

In addition to the above, further testing of PM emissions from gasoline vehicles should be conducted soon, since the Kansas City data are now over five years old. This testing could include the following:

- Improved real-time PM emissions data base with more data and improved instrumentation
- Data base broader than one location – e.g., Kansas City
- Effect of fuels, especially oxygenated fuels, should be monitored
- Effect of soak time on PM emissions (rather than using a HC surrogate)
- Temperature effects are very important. More data are needed, especially for hot running emissions.
- Improved brake and tire wear emissions data and apportionment of tire wear emissions based on driving condition.

MOVES Compared to MOBILE6

MOBILE6 is not a "gold standard" that can be used to benchmark MOVES, but it is instructive to compare the two models and determine possible reasons for different results. MOVES emission rates from the draft emissions model are significantly higher than MOBILE6 in most geographical areas and in most years. The reasons for higher exhaust HC, CO, and NOx emissions from light-duty vehicles are the log-linear emissions versus age model for deterioration, coupled with high VSP correction factors. For heavy-duty vehicle HC, CO, NOx, and PM, the increases in emissions are due mainly to the addition of the extended idle operation and the addition of tampering and malmaintenance factors that were not included in MOBILE6. PM emission rates for light-duty gasoline vehicles are higher in MOVES because of the inclusion of deterioration and cold temperature effects, neither of which was included in MOBILE6.

For states trying to attain the ambient air quality standards for ozone and PM who are using the MOVES model in conjunction with an ambient air quality model, the percent reductions in emissions from their base year to a projection year are more important than the absolute emissions. Based on an analysis of the comparison of the percent reductions in emissions from 2008 to 2015 in MOVES and MOBILE6, the draft MOVES model generally estimates less HC, CO, and NOx percent reductions over time than MOBILE6. However, this varies significantly by geographical area. If the final MOVES model is similar to the draft model, the use of MOVES by states to project future air quality could make the development of State Implementation Plans that predict attainment more difficult.

2.0 Introduction

EPA's website for modeling describes MOVES as the **MO**tor **V**ehicle **E**mission **S**imulator. MOVES is a model developed by the EPA that estimates emissions produced from on-road vehicles; it will eventually include nonroad vehicles as well. Also previously known as the "New Generation Model," MOVES encompasses all pollutants including hydrocarbons (HC), Carbon Monoxide (CO), oxides of nitrogen (NOx), particulate matter (PM), air toxics, and greenhouse gases and all mobile sources at the levels of resolution needed for the diverse applications of the system.

MOVES is the follow-on model to the MOBILE series of models, which were used from 1976 through today. Until MOVES is fully implemented in the 2012-2013 timeframe, MOVES will be used by (1) EPA to estimate the benefits of future criteria pollutant and fuel economy standards, (2) states to estimate the benefits of their state implementation plans (SIPs), and (3) by various entities to project air quality (mainly ozone and PM) in certain regions in the future.

MOVES has been under development for approximately seven years. EPA released an initial "proof of concept" version of the model (known as MOVES2004) in January 2005 that incorporated on-road vehicle fleet and activity factors, energy consumption, and emissions of methane and nitrous oxides for all on-road sources. The draft model, released in April 2009, built substantially upon MOVES2004 by including criteria pollutants, mobile source air toxics, and other factors for all highway vehicles. EPA released the final on-road model in December 2009.

The Coordinating Research Council commissioned a project with Air Improvement Resource, E.H. Pechan and Associates, and Dr. Albert Hochhauser to perform a critical review of the MOVES model. There are six tasks in the review:

Task 1 – Describe MOVES Methods
Task 2 - Perform MOVES Sensitivity Analysis
Task 3 – MOVES Validation
Task 4 – Status Reports
Task 5 – Provide Recommendations
Task 6 – Complete Final Report

This draft report fulfills Tasks 1, 2, 3, 5 and 6. The report is organized in the following sections:

- 3: Background
- 4: Review of EPA's MOVES and MOBILE6 Comparisons
- 5: Exhaust HC, CO, NOx Emissions - Light-Duty
- 6: Evaporative Emissions – All Gasoline Vehicles
- 7: Heavy-Duty Vehicles – All Pollutants
- 8: HC, CO, NOx Correction Factors
- 9: PM Emissions – Light-Duty

- 10: MOVES Sensitivity Runs
- 11: Recommendations

The activity data (e.g., miles traveled per day by vehicle class and age) used for MOVES were not reviewed in this report. These activity data have a significant effect on overall emission inventories. Readers desiring to know more abut MOVES activity data should consult several reports on this subject that are available on the EPA MOVES website.[1]

Also, all EPA FACA references used are listed at the start of each section of the report. Other references are included as footnotes.

The report also contains a number of appendices, as shown below. The Task 3 report, which includes recommendations for validating MOVES, is presented in Appendix G. No real validation of the model was conducted in this task; rather it was a literature search of previous, recent validation methods that have been used for MOVES (by EPA) and MOBILE6.

Appendix A: Method Used to Compare NMIM and MOVES Emissions Based on NMIM VMT
Appendix B: Sample Sizes for Cold Start Emissions
Appendix C: Additional Information from EPA on Log-Linear Deterioration in Exhaust Emissions
Appendix D: Distribution of Emissions by VSP Bin for LDTs
Appendix E: Additional Information from EPA on Cold Temperature Correction Factors
Appendix F: Temperatures Modeled in MOVES for Chicago (Cook County)
Appendix G: Investigation of Validation Methods (Task 3 report)

[1] See http://www.epa.gov/otaq/models/moves/index.htm.

3.0 Background

The MOBILE model for estimating on-road vehicle emission factors (in grams per mile) was first developed by the EPA in the late 1970s. Prior to that time, the Agency published look-up tables for estimating mobile source emissions. The model, originally and still written using the FORTRAN scientific programming language, has had significant updates and new releases every few years as new data became available, new regulations were promulgated, emission standards were established, and sources and processes of vehicle emissions were better understood. Each new version of the model has become more complex in the approach to modeling average in-use vehicle emissions, and has provided the user with additional options for tailoring emission-factor estimates to local conditions. The current version of the model, released in 2002, is MOBILE6.

In response to a request from Congress in 2000, the National Academy of Sciences (NAS) released a review of mobile source modeling in general, and MOBILE in particular.[2] One of the key findings and recommendations of the NAS is that it

> …recommended the development of a toolkit of models that would include…an aggregated regions emission factor modeling component…a mesoscale emissions modeling component…and a microscale instantaneous emissions modeling component…

The EPA has already adopted many other recommendations in the NAS report.

In response to the NAS report, EPA conceived a New Generation Model, or NGM, that would become this toolkit, and started work on designing the model. In 2001, EPA sponsored a "shootout" competition between different contractors and universities to develop a modeling framework. These organizations were given substantial second-by-second emissions data collected by EPA and asked to develop models that would use the data to predict emissions over different cycles and conditions. EPA prepared a summary report of these concepts.[3]

The basic concept that emerged for the exhaust emissions portion of the model is to place second-by-second emissions data from I/M and other programs into "bins" defined by vehicle specific power, or VSP. EPA also contracted with Harold Haskew and Associates to design concepts for changing its methods of modeling evaporative emissions. Along the way, the NGM name was dropped, and the model was renamed MOVES. A new program language was selected – JAVA, as it is much more flexible for using large databases, among other reasons. MySQL is also used. Another touted feature of the model is that it could be easily updated as additional I/M, remote sensing, or onboard emissions data became available.

Prior to 2006, EPA held periodic meetings with the Modeling Work Group under the umbrella of the Mobile Source Technical Review Subcommittee (MSTRS) of the Clean

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[2] "Modeling Mobile Source Emissions", National Research Council, 2000.
[3] "EPA's Onboard Analysis Shootout: Overview and Results", EPA420-R-02-026, October 2002.

Air Act Advisory Committee established per the Federal Advisory Committee Act (FACA). In 2006, the Modeling Work Group was dissolved and replaced by the MOVES Review Work Group under FACA, which was composed of selected states, stakeholders such as autos and oil companies and their associations, and environmental groups, to serve as a sounding board for the voluminous technical information that would be developed. Mr. John Koupal of the EPA and Prof. Matthew J. Barth of the University of California Riverside are co-chairs of this group. The first FACA meeting was held on August 8, 2006. Thirteen subsequent FACA meetings were held on the dates shown below, where EPA briefed the group on its research:

- May 18, 2007
- June 26, 2007
- August 16, 2007
- September 18, 2007
- November 8, 2007
- December 13, 2007
- January 29, 2008
- April 24, 2008
- October 27, 2008
- December 15, 2008
- April 28, 2009
- September 14, 2009
- November 10, 2009

The draft MOVES model was released in April 2009. EPA released its final model in December 2009. This timing was considered important to allow states to use the model in the next round of State Implementation Plans (SIPs) due in 2012 and 2013. In addition, EPA prepared reports on six areas of MOVES development, and released these for comment and peer review on September 3, 2009:

- Development of Emission Rates for Light-Duty Vehicles
- Development of Emission Rates for Heavy-Duty Vehicles
- Development of Evaporative Emissions Calculations
- Development of Gasoline Fuel Effects
- Draft MOVES2009 Highway Vehicle Temperature, Humidity, Air Conditioning, and Inspection and Maintenance Adjustments
- Draft MOVES2009 Highway Vehicle Population and Activity Data

The work undertaken in this CRC effort was started well before these technical reports were released. Therefore, much of this review was based on the materials presented in the previous stakeholder meetings. A preliminary draft report was released to CRC members and EPA at the end of June 2009. While the report's authors did not review all of the above reports in detail for this effort, they believe their comments on the methods presented by EPA in the stakeholder meetings are not outdated by these latest EPA reports.

Based on peer review, a review of the draft CRC E-68a report, and other comments raised by stakeholders and others, EPA made changes to some key methodologies between the September 14, 2009 and November 10, 2009 workshops. The authors of this report have not reviewed EPA's latest methods as reflected by any changes made between September 14 and November 10, 2009.

Clearly, the release of the final MOVES2009 model to replace MOBILE marks an important milestone in on-road mobile source emissions modeling in the U.S. Careful review of the model is critical to its success, and also critical to the adoption of correct policies to reduce emissions in the future.

4.0 Review of EPA's MOVES and MOBILE6 Comparisons

EPA performed an analysis to compare emissions calculated by MOVES with those calculated using MOBILE6 emission factors for three cities.[4] This analysis provides valuable information for CRC and others regarding what can be expected with MOVES in relation to MOBILE6 results. The analysis was performed before the new fuel economy standards were finalized and hence does not reflect the impact of those standards.

The EPA analysis was designed to compare the emission rates of MOVES to MOBILE6, rather than activity differences such as vehicle miles traveled. In practice, however, achieving a pure emission rate comparison is very difficult. Both MOVES and MOBILE6 were run with the same local temperature and humidity levels, for both January and July. MOVES emissions in tons were then divided by MOVES vehicle miles traveled to produce emission rates in g/mi. These emission rates were multiplied by National Mobile Inventory Model (NMIM) vehicle miles traveled to produce emissions in tons per month for both January and July. To produce annual emissions, the January and July emissions were added, and then multiplied by a factor of six. MOBILE6 outputs emissions in g/mi, and these were also multiplied by the same NMIM vehicle miles traveled as used for MOVES.

Differences remain between the two models that are not strictly related to emission rate differences. For example, the two models have different vehicle class age distributions, and the MOVES model has a different internal VMT distribution by roadway type (and speed by roadway type) than MOBILE6. As a result, EPA prepared a set of auxiliary inputs for MOVES. These inputs, which include the county-specific age, class, speed, and VMT distributions, are based on NMIM. They are designed to remove as many basic differences between the two models as possible.

The cities analyzed by EPA were: Atlanta, represented by Fulton County; Chicago, represented by Cook County; and Salt Lake City, represented by Salt Lake County. All three cities have inspection and maintenance (I/M) programs, and Chicago is a reformulated gasoline area. Chicago was noted to have a young fleet, which would lead to a faster turnover to the newer emission standards. Four pollutants were analyzed: HC, CO, NOx, and $PM_{2.5}$. The analysis was performed for the years 2008, 2015, and 2020. Results were analyzed separately for light-duty and heavy-duty vehicles.

Results from this analysis showed significantly higher NOx emissions with MOVES than MOBILE6. HC emissions were somewhat lower with MOVES than with MOBILE6. The $PM_{2.5}$ emissions were higher with MOVES than with MOBILE6, primarily due to the PM temperature effect now included in MOVES, while MOBILE6 included no

[4] EPA, 2009: US EPA, Office of Transportation and Air Quality, "Air Pollution from Highway Vehicles: What MOVES Tells Us," presentation for the International Emission Inventory Conference, April 15, 2009.

temperature effect for PM emissions. CO emissions were not analyzed by EPA in their comparison of MOVES and MOBILE6, but that comparison was conducted by AIR for this report.

For this comparison, AIR obtained and used all of EPA's auxiliary MOVES inputs for the three cities, and repeated the comparison for two cities – Chicago and Atlanta. The MOVES model was run in a manner very similar to the method used for the MOBILE6 model. In NMIM, MOBILE6 is executed using a set of codes based on the SCC (Standard Classification Code). The NMIM SCC codes contain 12 vehicle classes and 12 roadway types, for a total of 144 class/roadway combinations. To obtain similar detail, MOVES was run using its SCC output mode. Although different in design, the MOVES SCC codes still contain the same 144 class and road combinations as NMIM. Therefore, for both models, the total emissions were based on the summation of all of these class/road emissions. Appendix A shows the detailed procedure that was used.

AIR's results for THC for the two cities are shown in Figures 4-1 through 4-2 below. Results for CO are shown in Figures 4-3 and 4-4, for NOx are shown in Figures 4-5 and 4-6, and for exhaust $PM_{2.5}$ are shown in Figures 4-7 and 4-8.

THC – Figures 4-1 and 4-2 show that MOVES THC emissions are higher than for MOBILE6 in most cases. Most of the increase is due to higher light-duty vehicle emissions, although heavy-duty emissions are slightly higher for the two cities as well. The Atlanta THC emissions for MOVES in 2020 are lower than MOBILE6. This is the one exception to higher THC emissions for MOVES. The difference in MOVES and MOBILE6 narrows in the future. This would indicate that the percent reductions in VOC emissions are higher for MOVES than for MOBILE6. Percent reductions for the various pollutants from 2008 to 2020 are shown in Table 4-1 at the end of this section. AIR's results are somewhat different than the results that EPA obtained with the April version of the draft MOVES model.

CO - CO emissions between the two models are very similar for Atlanta and Chicago.

NOx – NOx emissions are higher for MOVES than MOBILE6 for both cities and all three years. Both heavy-duty NOx and light-duty NOx is higher for MOVES than for MOBILE6. A major reason that heavy-duty NOx is higher in MOVES than in MOBILE6 is because idle NOx emissions have been added to the heavy-duty inventory, whereas they are not included in MOBILE6.

$PM_{2.5}$ –While MOVES shows large reductions in $PM_{2.5}$ from 2008 to 2020 (similar to MOBILE6), the MOVES $PM_{2.5}$ emissions are substantially higher than MOBILE6. The higher emissions for MOVES are due to both heavy-duty vehicles and light-duty vehicles. The higher emissions for both heavy and light-duty vehicles are due to the inclusion of deterioration on PM for both major vehicle classes (MOBILE6 assumed zero PM deterioration), the inclusion of temperature correction factors for PM, and the inclusion of PM emissions during heavy-duty truck idling. EPA is repeating these comparisons, but with its final model.

Figure 4-1. MOVES versus MOBILE6, HC, Atlanta

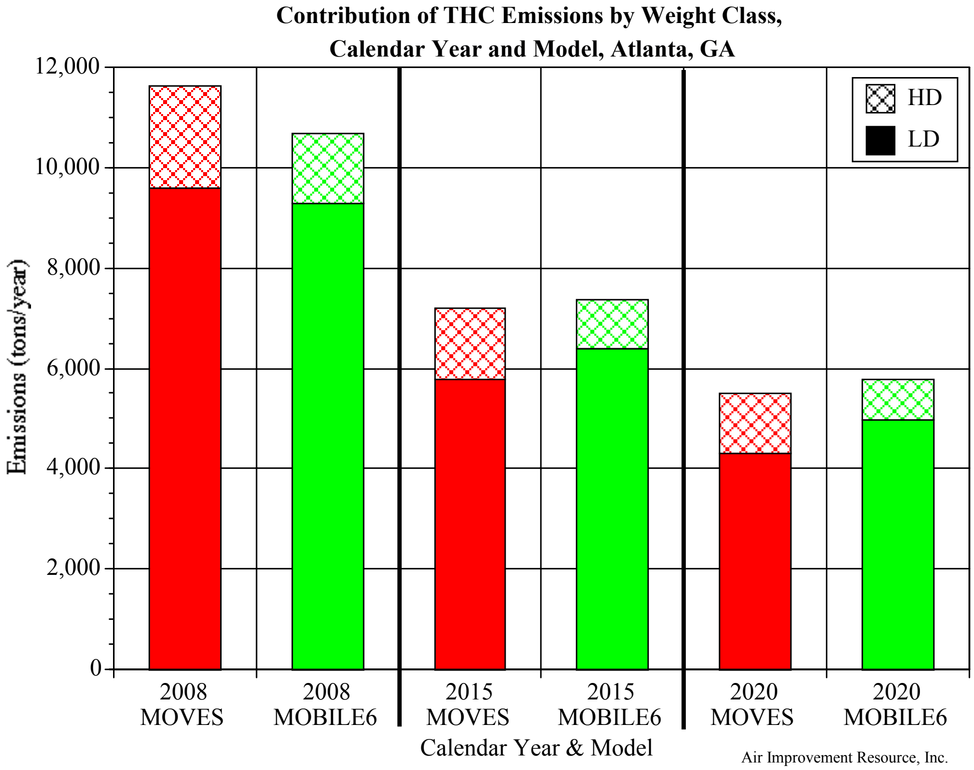


Figure 4-2. MOVES versus MOBILE6, Chicago, HC

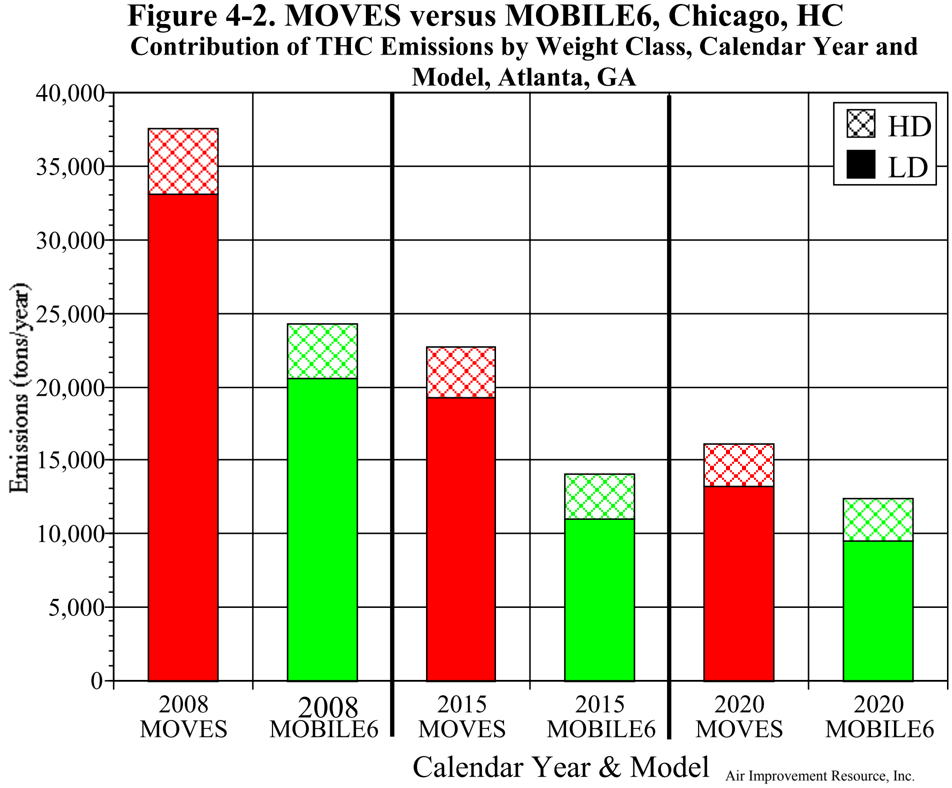


Figure 4-3. MOVES versus MOBILE6, Atlanta, CO

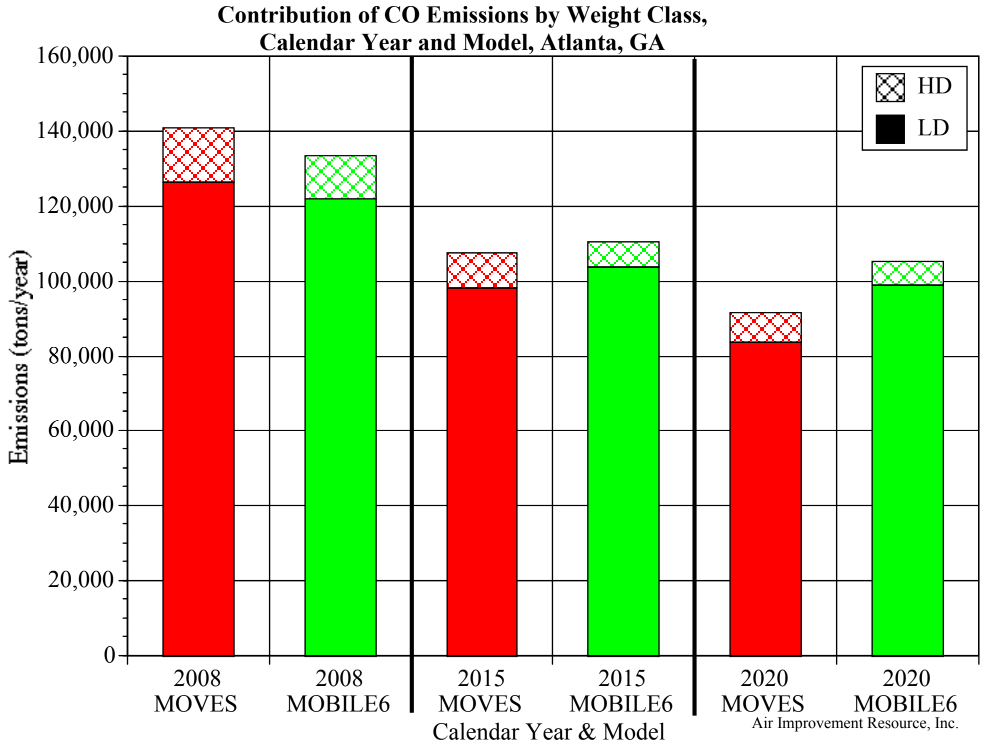


Figure 4-4. MOVES versus MOBILE6, Chicago, CO

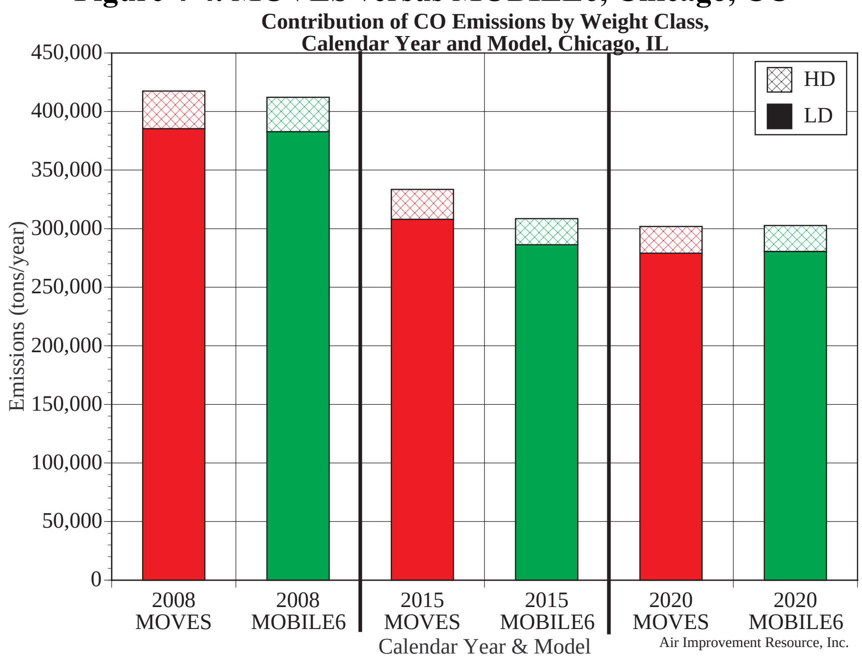


Figure 4-5. MOVES versus MOBILE6, Atlanta, NOx

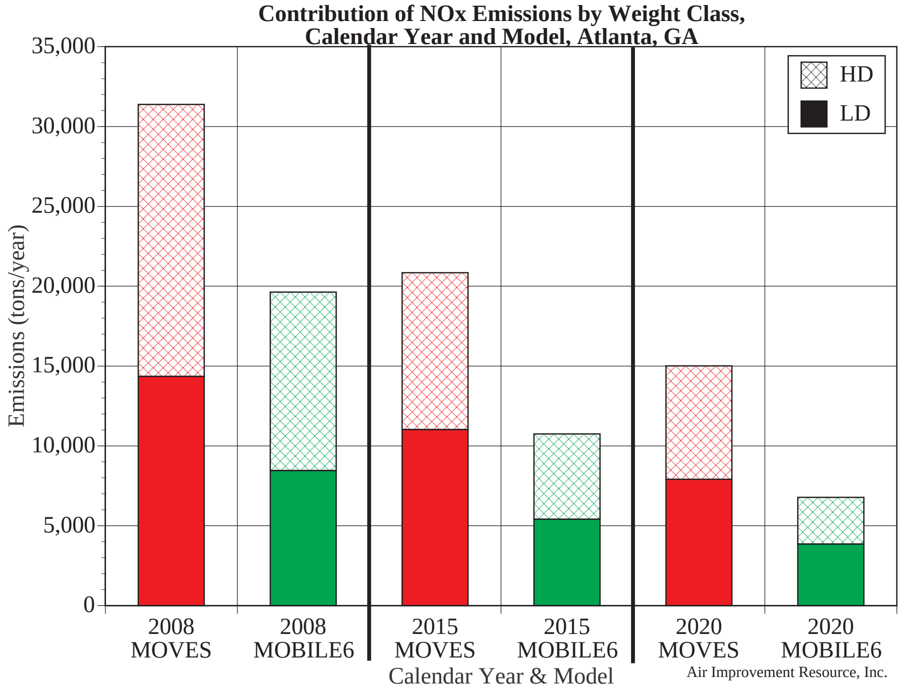


Figure 4-6. MOVES versus MOBILE6, Chicago, NOx

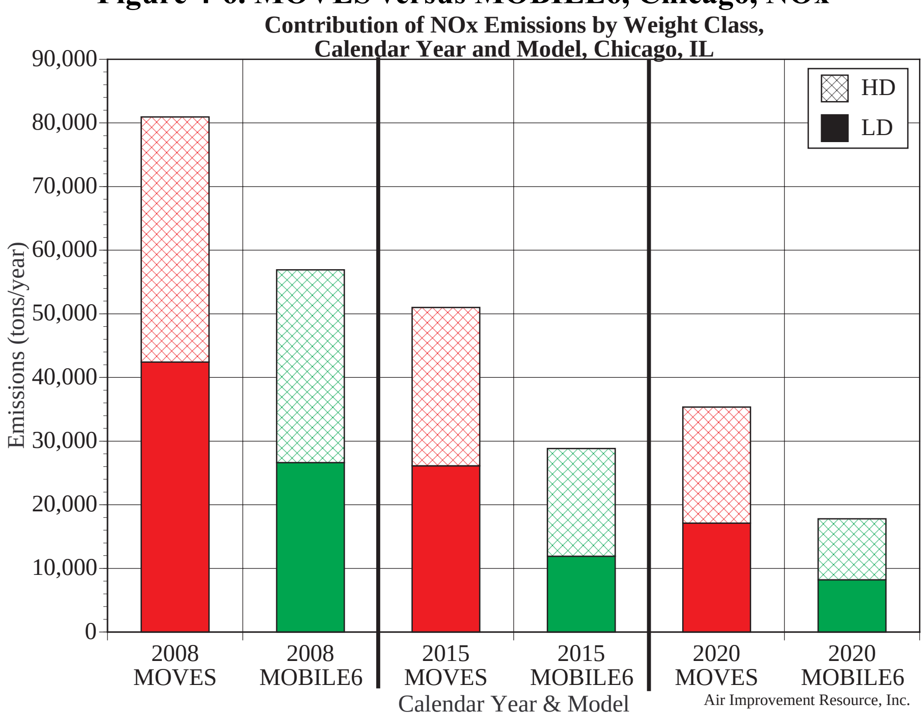


Figure 4-7. MOVES versus MOBILE6, Atlanta, $PM_{2.5}$

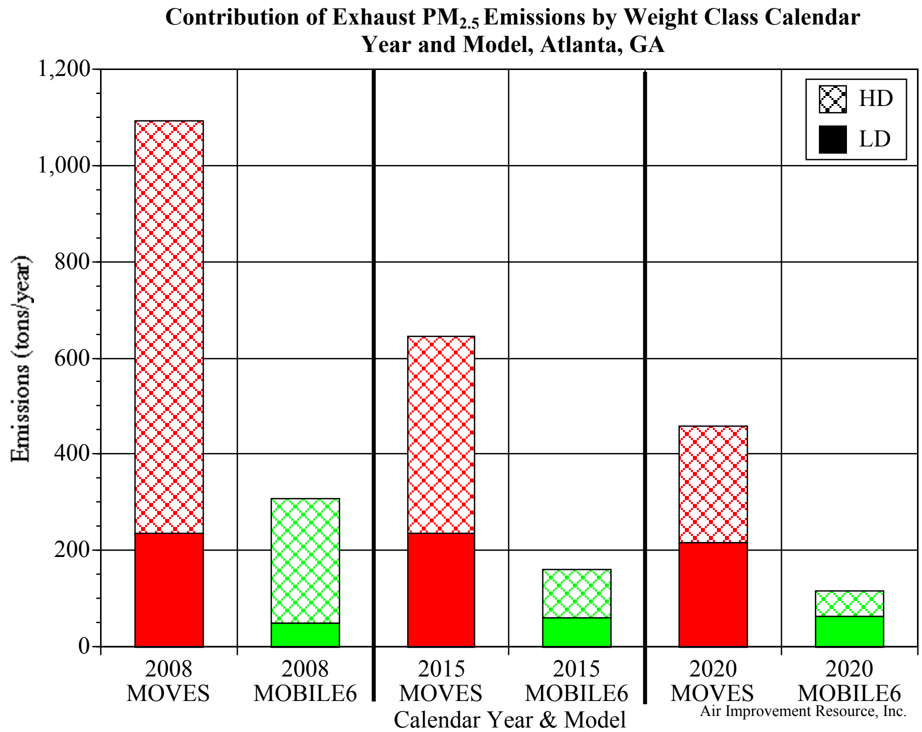


Figure 4-8. MOVES versus MOBILE6, Chicago, $PM_{2.5}$

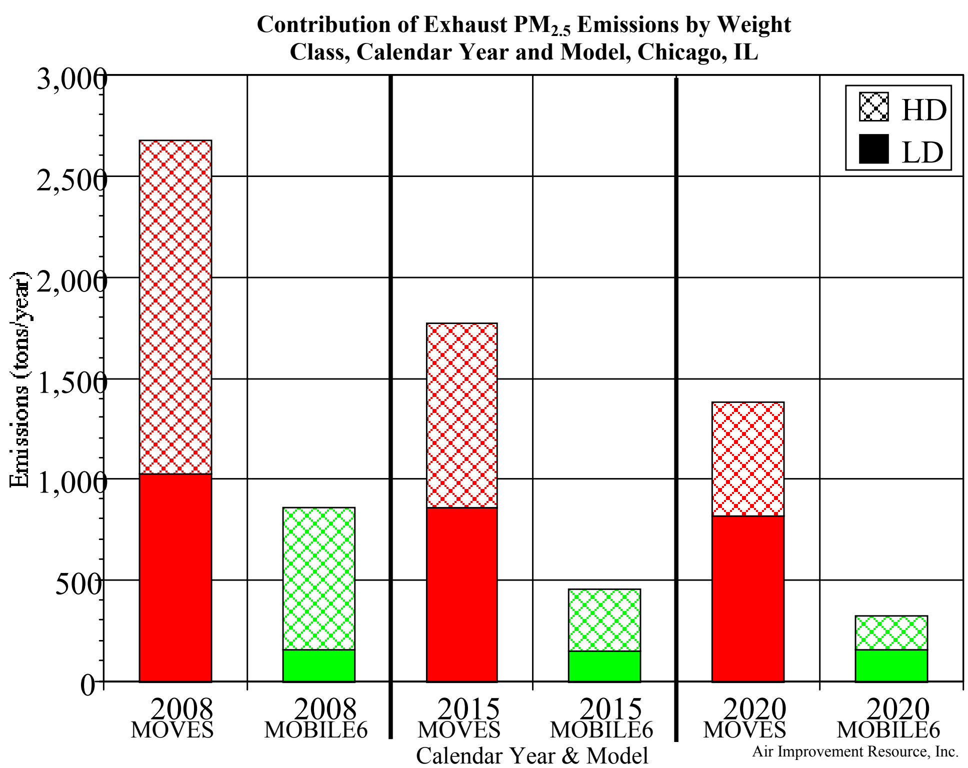


Table 4-1 compares the reduction in emissions from 2008 to 2015 when calculated with MOVES and with MOBILE6 for the two cities. For HC, the percent reductions between the two models for the two cities are similar. For CO, MOVES shows similar reductions to MOBILE6 for Chicago and Atlanta. For both NOx and $PM_{2.5}$, MOVES shows less reduction for the two cities than MOBILE6.

These comparisons indicate that when states make the transition from MOBILE6 to MOVES (assuming the final MOVES model is similar to the draft model), they could have a more difficult time demonstrating attainment with the ambient air quality standards for ozone and PM.

Table 4-1. Percent Reduction in On-Road Emissions, 2008 to 2015

Pollutant	Chicago		Atlanta	
	MOVES	MOBILE6	MOVES	MOBILE6
HC	39.6	42.1	38.0	31.0
CO	20.1	25.1	23.5	17.2
NOx	37.0	49.4	33.6	45.2
$PM_{2.5}$	33.6	46.7	40.8	47.5

5.0 Exhaust HC, CO, and NOx Emissions – Light-Duty

This section discusses EPA's methods for estimating HC, CO, and NOx exhaust emissions. EPA divides the exhaust emissions into start emissions and running emissions. Start emissions include all starts, whether cold, warm, or hot. The soak time preceding the start determines where in this spectrum the start actually is.

This section is organized in the following subsections:

- Start Emissions
- Running Emissions
- Distribution of Emissions by VSP Bin
- Summary of Recommendations

The primary FACA workgroup materials used in this section are the following:

"Developing Draft Start Rates for MOVES: Criteria Pollutant Emissions from Light-Duty Gasoline Fueled Vehicles", Landman and Brzezinski, May 18, 2007

"Developing Draft Emission Rates for MOVES: Criteria Pollutant Emissions from Light-Duty Vehicles", Warila and Glover, May 18, 2007

"Emission Rates for MOVES: HC & NOx Emissions from Light-Duty Vehicles", Warila, Brzezinski, Beardsley, and Koupal, October 27, 2008

"Verifying and Correcting Rates at High Speed and VSP: Criteria Pollutant Emissions from Light-Duty Vehicles", Warila, April 24, 2008

"Inspection/Maintenance for Exhaust Emissions", (no author listed), November 8, 2007

"Release of Draft MOVES2009", Beardsley, April 28, 2009

5.1 Start Emissions

Start emissions are included in MOVES, and are estimated for all types of starts, which include cold starts, hot starts, and intermediate starts. Cold starts generally involve a soak period of 12 hours or greater, while hot starts have a soak period of 10 minutes or less. Intermediate starts are starts with a soak period of anywhere between 10 minutes and 12 hours. The start emissions are estimated from two data sources, (1) cold start and hot start data on both the Federal Test Procedure and the LA-92, and (2) California data on the emissions of various vehicles under warm start conditions with soaks between 0 minutes and 12 hours (these are discussed in more detail and referenced later).

There are two steps in estimating start emissions: (1) the creation of a cold start emission estimate from Bag 1 and Bag 3 of either the Federal test procedure (FTP) or the LA-92 for use as a cold start emission level, and (2) adjustment of this cold start level to any non-cold start level (i.e., intermediate start level or hot start level), as needed by the MOVES program for all intermediate starts or hot starts.

The Bag 1 of the FTP is immediately preceded by a minimum 12-hour vehicle soak. The bag 1 driving cycle consists of a cold start followed by a relatively low speed urban driving simulation. Bag 3 uses the same driving cycle as bag 1, but is preceded by a 10-minute soak. Thus, the vehicle must be restarted at the beginning of bag 3. This restart is referred to as a hot start. Were it not for the hot start in Bag 3, the cold start emissions could be estimated simply by subtracting Bag 3 emissions from Bag 1 emissions. To determine cold start emissions, an adjustment is necessary to remove the hot start contribution. Cold start emissions are estimated from the following equation:

$$\text{Cold start} = (\text{Bag 1} - \text{Bag 3})/(1-A)$$

Where:

Bag 1 = Bag 1 emissions in grams
Bag 3 = Bag 3 emissions in grams
A = ratio of hot start emissions at 10 minutes/cold start emissions

Bag 1 and Bag 3 emissions are from emissions test data. The value “A” is determined from CARB’s testing of catalyst vehicles over a number of different soak periods (more on this presently). The “A” value, however, is determined from the ratio of 10-minute soak emissions to the cold start emissions for catalyst vehicles. Table 5-1 below shows the ratio of hot start to cold start emissions for these catalyst vehicles.

Table 5-1. Ratio of Hot Start Emissions to Cold Start Emissions

Pollutant	Ratio of 10-minute soak emissions to cold start emissions for catalyst vehicles (A)	Percent Increase in (Bag 1 minus Bag 3) emissions to produce cold start emissions
HC	0.160	19%
CO	0.112	13%
NOx	0.204	26%

An example of this adjustment is as follows. The Bag 1 minus Bag 3 emissions for HC for 1990-1993 passenger cars are 1.9 grams, so adjusted for the hot start, cold start emissions are 1.9/(1-0.160) = 2.26 grams. The A values do not change from those listed in Table 5-1, so the Bag 1 minus Bag 3 values for HC are always adjusted upward by 19%, CO by 13% and NOx by 26% to predict the “true” cold start levels. The adjusted cold start levels (which have running emissions removed) are shown in Figures 5-1 through 5-3 below. These are cold start increments in grams/start. The Tier 2 cold start levels are much lower than all other cold start levels.

Figure 5-1. Cold Start HC Emissions (grams/start)

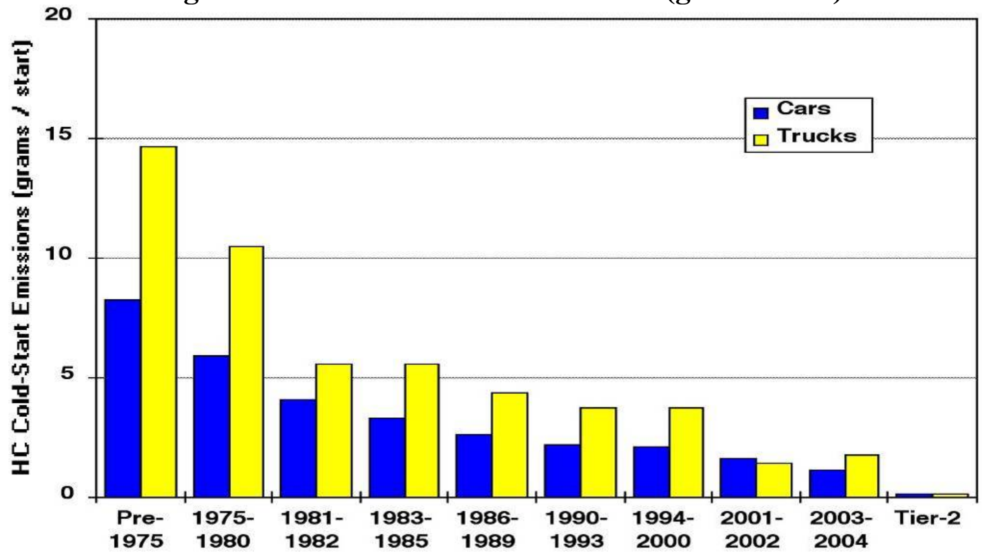


Figure 5-2. Cold Start CO emissions (grams/start)

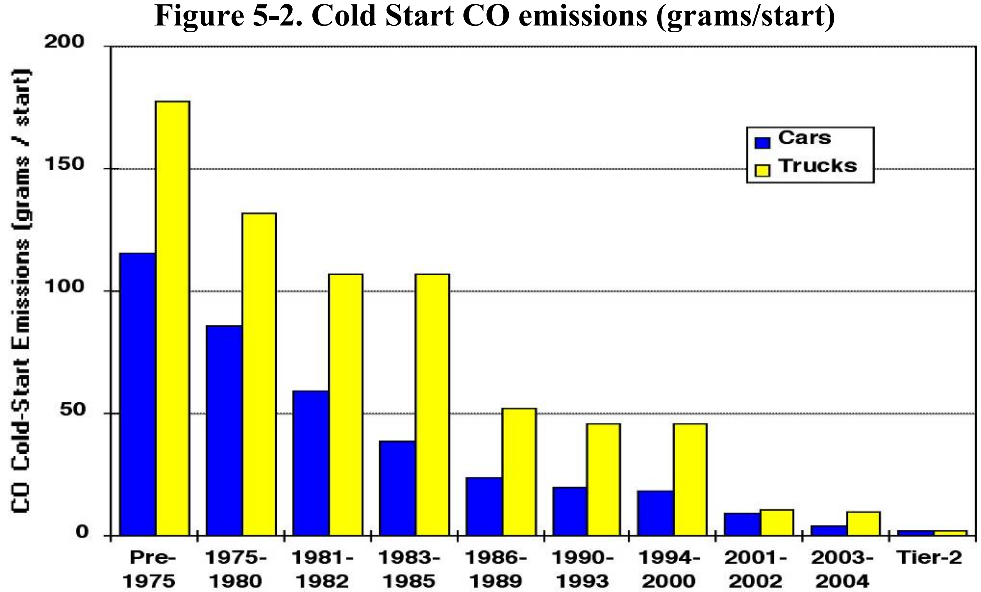
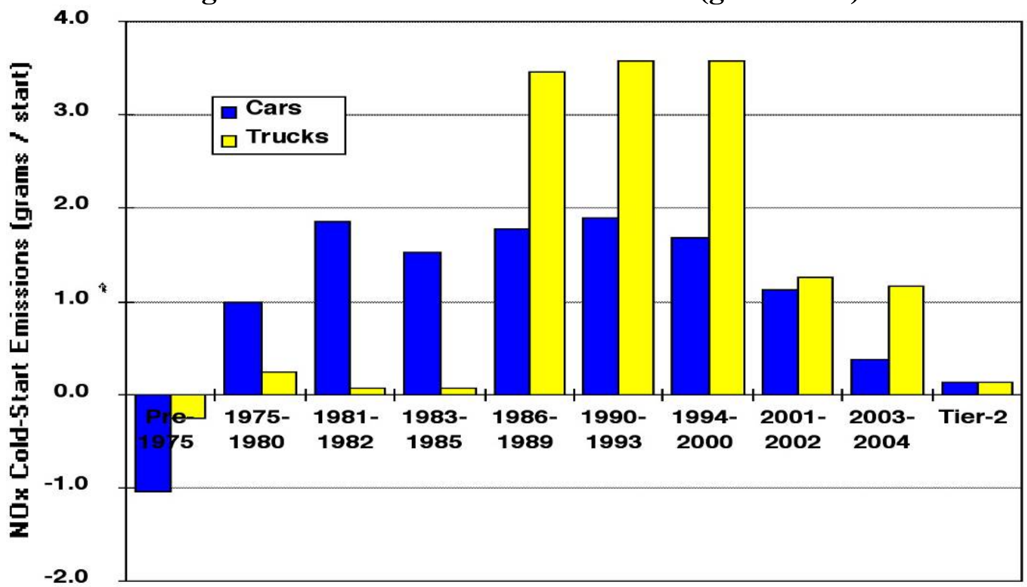


Figure 5-3. Cold Start NOx Emissions (grams/test)



The pre-1975 cold start NOx values for both cars and light-duty trucks (LDTs) are negative. This does not mean that the Bag 1 NOx emissions are negative, but only that the Bag 1 emissions were less than the Bag 3 emissions. These were non-catalyst equipped vehicles, and it is reasonable to expect that the NOx emissions during cold start would be less than during a hot start, since NOx emissions are higher when the engine is warmer, and lower when the engine is colder. The assumption made by EPA is that MOVES trips are at least the distance of Bag 1 of the FTP, or 3.59 miles. If a significant fraction of trips are significantly shorter than this, then pre-1975 NOx emissions could be negative, which would be impossible. Given that most MOVES runs will go no further back than 1999, and there is a 25 model year window for light-duty vehicles (LDVs) and LDTs in MOVES, and there are probably few trips less than 3.59 miles, this anomaly is probably not worth being concerned about.

After adjusting the Bag 1-Bag 3 values to give the cold start emissions, the cold start levels must be adjusted to account for shorter soak periods. This adjustment again uses the California test data, which is discussed in more detail in Section 5.1.1.

MOBILE6 used a somewhat different approach.[5] Instead of estimating cold start emissions as the difference in Bag 1 and Bag 3 (adjusted for the start), EPA had test data on some vehicles on the Bag 3 test procedure without a hot start (called a "Hot 505"). This allowed EPA to develop a relationship between FTP Bag 1 and Bag 3 emissions and the Hot 505 emissions. EPA used this relationship to predict Hot 505 emissions for all of its in-use data, and developed hot and cold starts by subtracting the predicted Hot 505 emissions from both Bag 1 and Bag 3. EPA did not carry-over this approach to MOVES

[5] "Determination of Start Emissions as a Function of Mileage and Soak Time for 1981-1993 Model Year Light-duty Vehicles", EPA420-R-01-058, November 2001.

because of the need to run additional testing programs to develop Hot 505 emissions for newer technology vehicles such as National Low Emission Vehicles (NLEVs) and Tier 2 vehicles.

5.1.1 California Test Data

The intermediate soak analysis is based on test data and analysis of the data conducted by CARB. There were 29 vehicles tested in two different testing programs, with three carbureted vehicles excluded for various reasons.[6] The vehicle model years ranged from 1970 to 1993, with none of the vehicles being Tier 1 vehicles or later. Most of the vehicles were equipped with three-way catalyst systems and had ported fuel injection (PFI), but two of the vehicles were throttle body injected (TBI), a technology that was phased out in favor of port fuel injection in the late 1980s and early 1990s. Interestingly, the analysis on which EPA based its correction factors did not include testing that was later conducted by California Air Resources Board (CARB) on ten vehicles with a model year range of 1972 to 1996, which included two Tier 1 vehicles.[7]

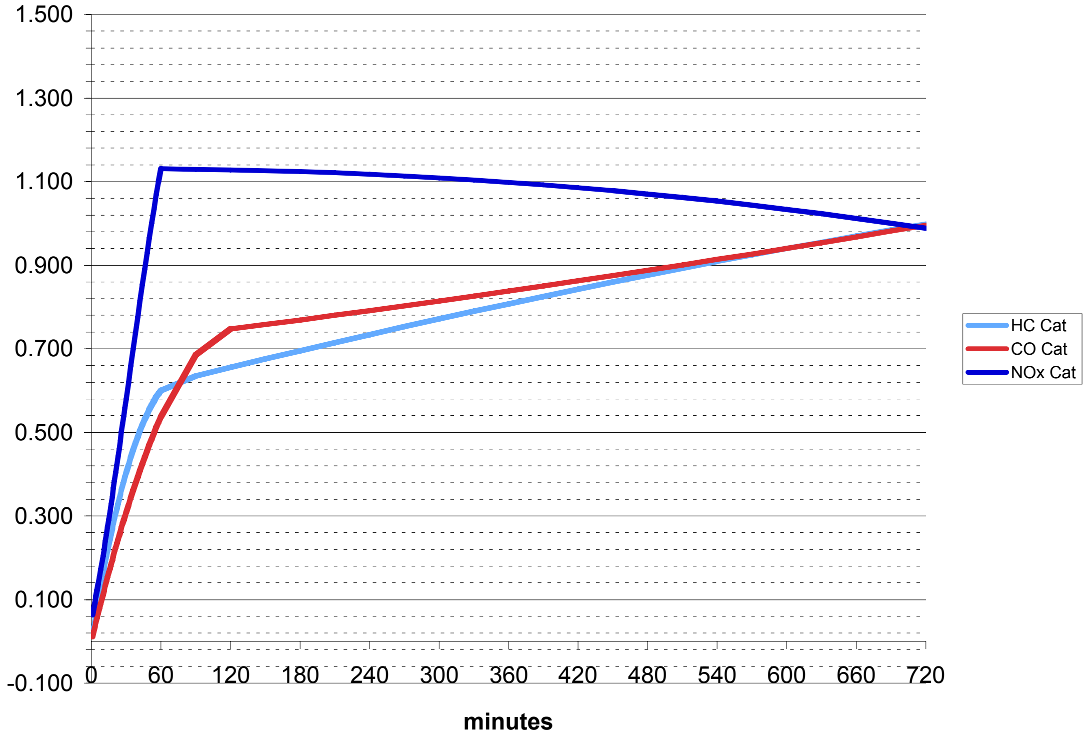
The first testing program used the FTP and a special start cycle, and tested one set of vehicles with soak periods of 0, 5, 10, 20, 30, 40, 50, 60,120, and 720 minutes. A second testing program that used a different set of vehicles added more soak periods between 120 minutes and 720 minutes, dropped the special start cycle, and used the FTP and the LA-92 cycle.

From the test data, CARB first defined the start emissions as the emissions during the first 100 seconds of vehicle operation (regardless of test cycle), and next developed the fraction of start emissions to cold start (i.e., 12-hour soak) emissions for catalyst vehicles as shown in Figure 5-4. EPA then used these curves and the cold start emissions developed in Figures 5-1 through 5-3 to determine the emissions at intermediate soak periods. It should be noted that the curves are based on a different definition of start emissions (i.e., emissions in the first 100 seconds), than EPA's definition (Bag 1 minus Bag 3 emissions, corrected for the hot start). EPA presented no information on how these two definitions actually correlate.

[6] "Methodology for Calculating and Redefining Cold and Hot Start Emissions", Documentation for EMFAC7G, California Air Resources Board.

[7] "Study to Define Cold and Hot Start Emissions Final Investigative Report", Arnold and Sabate, April 1997.

Figure 5-4. Fraction of Emissions during start to cold start



An example of using Figure 5-4 is as follows. Taking the 1990-1993 cold start HC emissions of 2.26 grams as estimated earlier, and estimating start emissions for a one-hour soak, using the equations that produced Figure 5-4, at 60 minutes, the ratio of start to cold start emissions is 0.600. Thus, the start HC emissions for a 60-minute soak for 1990-1993 vehicles is 0.6 * 2.26 = 1.356 grams.

The NOx emission line in Figure 5-4 shows that the ratio of start to cold start emissions reaches 1.0 at 50 minutes, and then exceeds 1.0, extending to just over 1.1, and then slowly declining back to 1.0 at 720 minutes. The levels of this ratio that exceed 1.0 mean that the start emissions exceed cold start NOx emissions. This is not surprising, since engines produce more NOx when they are warmed up than when cold. At 60 minutes, for example, the engine is still warm but the catalyst has cooled off considerably (although conversion of HC and CO is still significant at a 60 minute soak). So, NOx emissions are relatively high on start-up (higher than they would be for a cold start), but the catalyst is not reducing the NOx very much until it warms up. Thus, total NOx during this type of start can be higher than a cold start.

There are two concerns with EPA's method. The first is that the data used to develop the ratio of start emissions to cold start emissions is very old. It is based entirely on pre-Tier 1 vehicles. It should be based on a mixture of Tier 1, LEV, and Tier 2 vehicles, primarily the latter two vehicle types. The key question is how the start to cold start emission ratios that are displayed in Figure 5-4 would change over time with the introduction of newer technology vehicles. It is not known if these ratios would increase faster or slower, or be about the same. It is possible that the increased use of close-coupled catalysts on later model year vehicles would make these ratios increase at a slower pace, because heat in the engine may cause these catalysts to retain heat longer.

To investigate this issue, the start to cold start ratios were modified for HC, CO and NOx for 2004 and later light duty gasoline vehicles. The modified ratios were used to produce new binned start emissions for MOVES, and the model was run comparing start emissions from the default case to the modified cases. The results are shown in Figures 5-5 through 5-9.

Figure 5-5 shows the default start to cold start ratios with soak time for all three pollutants in the draft MOVES model. Figures 5-6 through 5-8 show the changes in these distributions to 80% and 110% of the default for HC, CO, and NOx, respectively. Figure 5-9 shows the impact of the modified distributions on start emissions from light duty gasoline vehicles in the summer in Chicago.

Figure 5-5

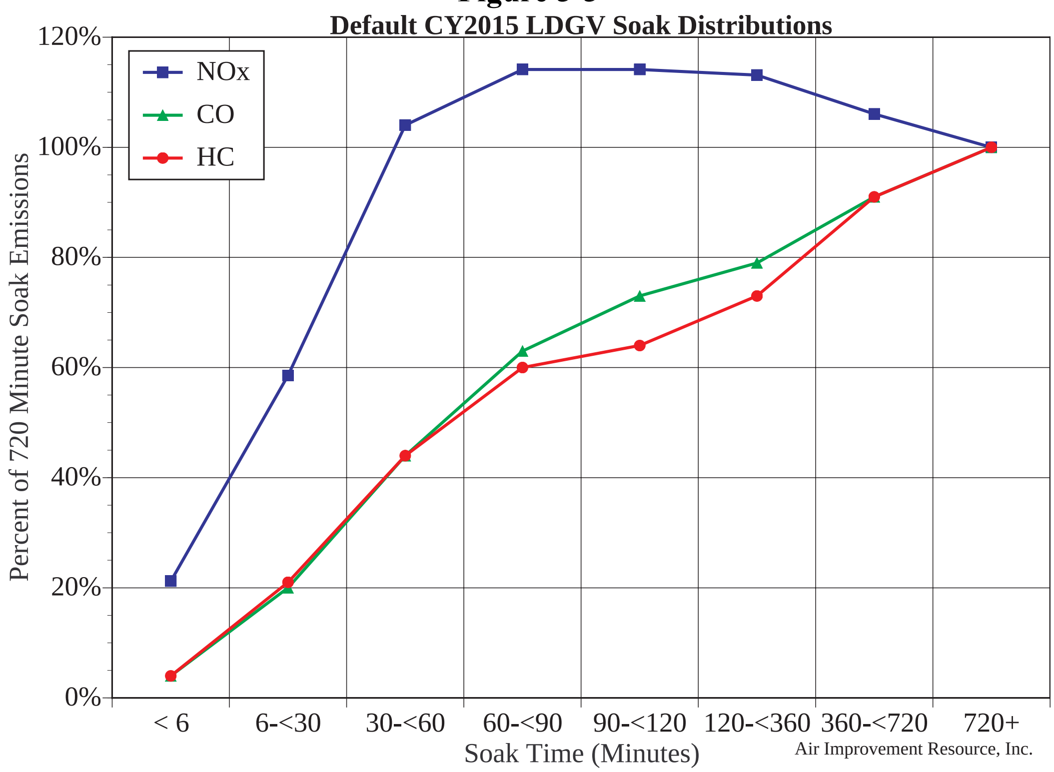


Figure 5-6

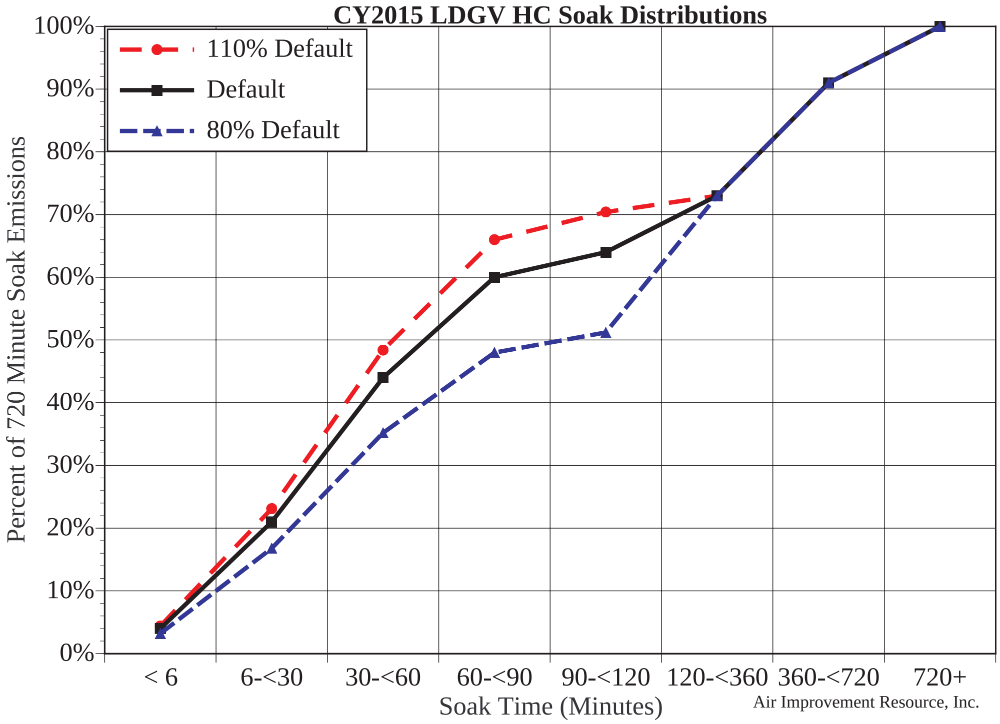


Figure 5-7

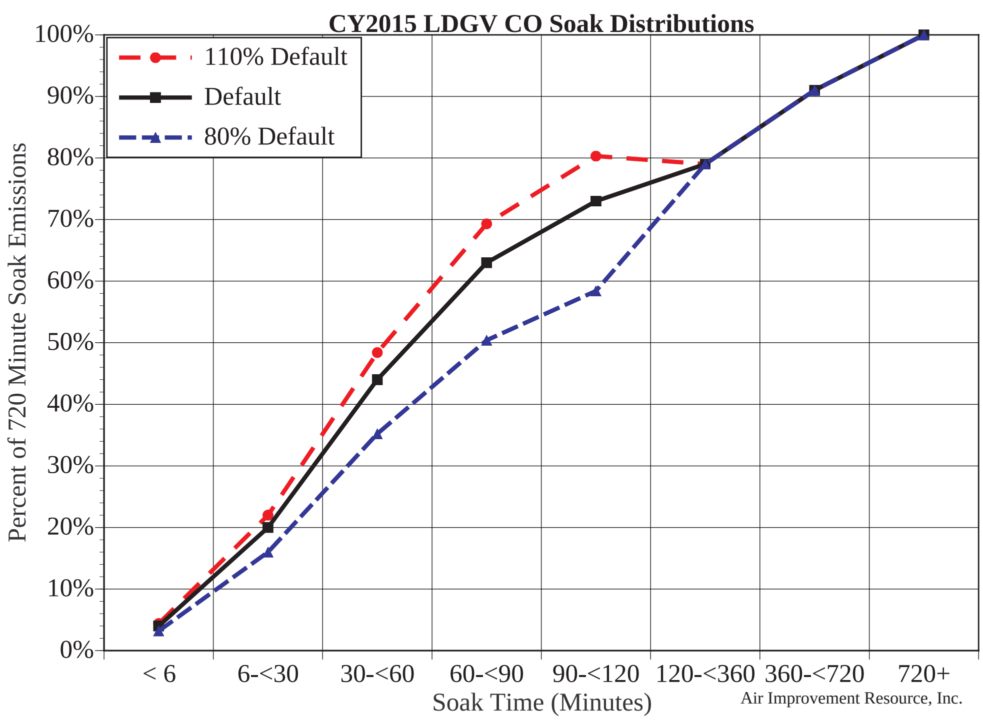


Figure 5-8

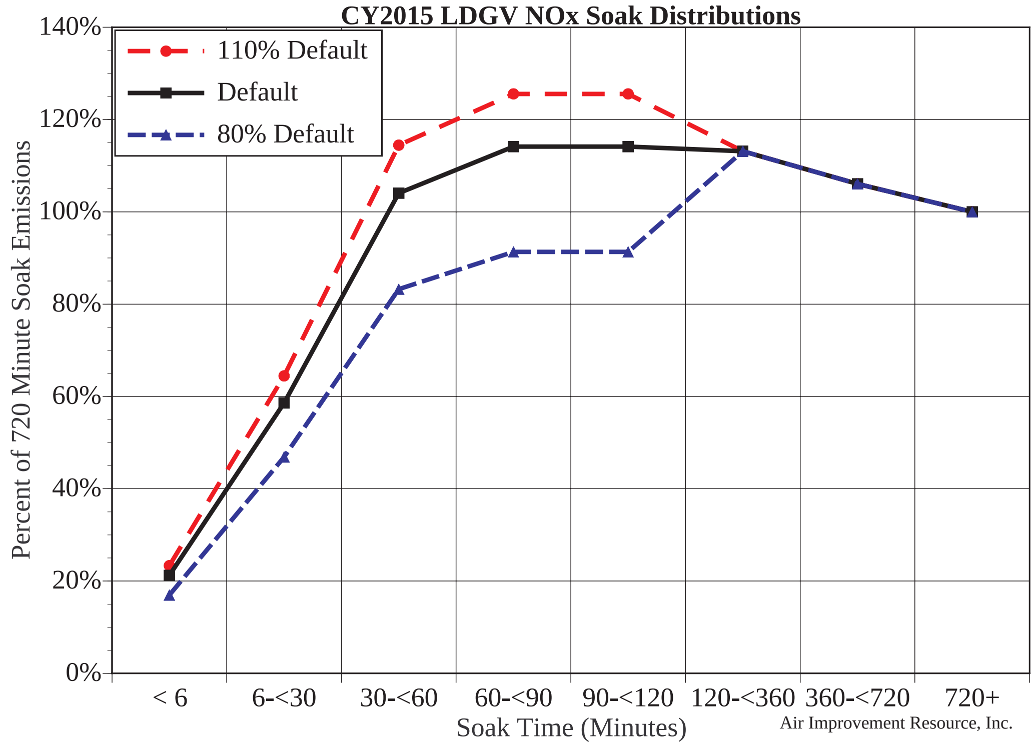

CY2015 LDGV NOx Soak Distributions
140%
120%
100%
80%
60%
40%
20%
0%
Percent of 720 Minute Soak Emissions
110% Default
Default
80% Default
< 6
6-<30
30-<60
60-<90
90-<120
120-<360
360-<720
720+
Soak Time (Minutes)
Air Improvement Resource, Inc.

Figure 5-9

Impact of Soak Distributions on CY2015 LDGV Start Emissions

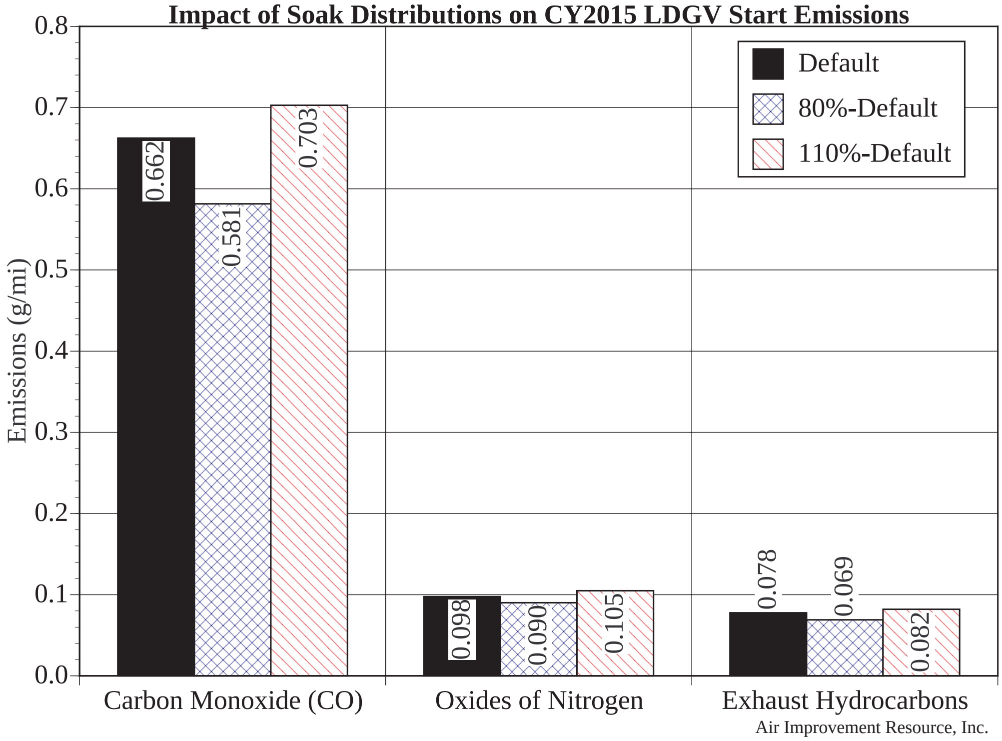


Figure 5-9 shows that when the start to cold start ratios are increased to 10% over the default, that emissions of all three pollutants increase. When the start ratios are reduced to 80% of their default level, then start emissions decrease from the default levels. The changes in emissions in the range of ratios tested are not dramatic. We do not have a method to check whether the range of ratios that were tested is a realistic range. Further testing of the start to cold start ratios for newer technology vehicles is warranted.

The second concern with EPA's method is that EPA is using the CARB start to cold start ratios versus time, where the CARB cold start emissions are defined as the emissions in the first 100 seconds. However, EPA is using these with its own cold start definition, which is the difference in Bag 1 and Bag 3 emissions, corrected for the hot start. EPA should re-examine the raw CARB data and develop the curves based on its definition, rather than on CARB's definition. How the curves would change if this were done was not evaluated.

A summary of the emissions data sources for cold start emissions is shown in Appendix B.

5.1.2 Deterioration in Start Emissions

Start emissions are corrected for deterioration, temperature, and fuel effects. Currently they are not corrected for I/M, although EPA plans to correct start emissions for I/M in the final model. This section discusses deterioration, and Section 8 discusses both temperature and fuel effects.

Start emission deterioration is estimated by first estimating running deterioration (discussed in the next section) from MOVES, then estimating both start emissions deterioration and running deterioration from MOBILE6, and using the ratio of MOBILE6 start emissions deterioration to running emissions deterioration to adjust the MOVES start emissions deterioration. This is shown in the following equation:

$$\text{Start } DR_{MOVES} = \text{Running } DR_{MOVES} * \text{Start } DR_{MOBILE6}/\text{Running } DR_{MOBILE6}$$

Where:

Start DR_{MOVES} = start deterioration factor by age for MOVES (ratio to 0-3 age)
Running DR_{MOVES} = running deterioration from MOVES
Start $DR_{MOBILE6}$ = start deterioration factor from MOBILE6
Running $DR_{MOBILE6}$ = running deterioration factor from MOBILE6

The MOVES model does not separate high and low emitters, so determining the running deterioration for MOVES is straightforward. MOBILE6, however, has deterioration for low and high emitters, and different low and high emitter fractions in I/M and non-I/M areas. Thus, EPA estimated increases in deterioration for both low and high emitters, and weighted them together using the MOBILE6 high emitter fractions versus mileage. This process is performed for both I/M and non-I/M areas. Three additional steps are then implemented.

- The MOVES model is used to simulate Bag 2 (running) emissions for all model years. This is accomplished by inputting the driving cycle of Bag 2 of the FTP into the model. The model then determines the VSP distribution, and utilizes that VSP distribution to estimate Bag 2 emissions.
- MOVES is then run by model year to evaluate the Bag 2 (running) deterioration by age group relative to the youngest group. The youngest group is the 0-3 year group, so the 0-3 year group is assigned the relative factor of 1.0, and higher age groups are assigned values higher than 1.0.

- The MOVES running deterioration factors versus age are then multiplied by the MOBILE6 start to running deterioration factors.

The start deterioration "reduction ratios" for HC and CO as determined from MOBILE6 data are shown in Table 5-3. There are no start deterioration reduction ratios for NOx, as start NOx is assumed to deteriorate at the same rate as running NOx. The ratios in Table 2 show that deterioration for starts is estimated to be less than that for running emissions. For example, start deterioration for age 8-9 year old vehicles for HC is estimated to be 41% of the running emissions deterioration.

Table 5-3. Start Deterioration Reduction Ratios for HC, CO		
Age Group (years)	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

As shown in Section 10, start mode emissions dominate HC exhaust emissions from gasoline vehicles, and are a significant contributor to CO and NOx emissions as well. It is a concern that the start deterioration methodology for MOVES borrows heavily from MOBILE6 emission projections, which are assembled much differently than for MOVES.

The authors are not convinced that this is the best method of estimating start deterioration. At least one concern is that the MOBILE6 ratio of start to running deterioration, based on older ideas about the growth in high emitters, is being used to determine the start deterioration emissions from MOVES.

Another method of estimating start deterioration is to first evaluate start deterioration directly from the manufacturer In Use Vehicle Program (IUVP) test data. Test data are available by bag from this program, and so Bag 1 minus Bag 3 emissions can be regressed versus age from this data. If there is a concern with this sample, one can also evaluate Bag 2 deterioration from these data, and compare it with the Arizona I/M data. If the Arizona data has higher deterioration than Bag 2 from the IUVP, then the start deterioration can be ratioed to MOVES (Arizona). But at least with this method, start emissions deterioration is measured directly, instead of modeled from MOBILE6. Of course it is recognized that taking deterioration from one location (Arizona) and applying it to the whole country is less than ideal.

5.2 Running Emissions

Research in previous years convinced EPA to estimate running emissions for MOVES by vehicle specific power (VSP) bin.[8] This allows users to modify the input cycle to MOVES and estimate emissions for a wide variety of operating modes and a wide range of geographical scales (macro-, meso- and micro-scale). Vehicle specific power is estimated with the equation shown in Figure 5-10.

Figure 5-10. Vehicle Specific Power Equation

Vehicle-Specific Power

VSP represents the vehicle's tractive power normalized to its weight, and calculated is a function of velocity, acceleration, weight and the Vehicles' road-load coefficients

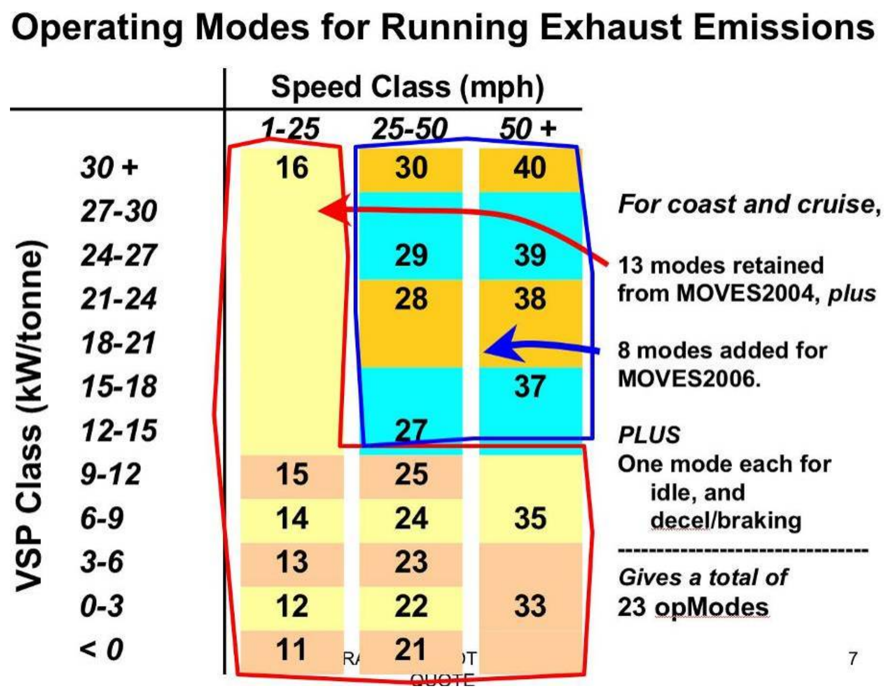
$$\mathrm{VSP}_t = \frac{Av_t + Bv_t^2 + Cv_t^3 + mv_t a_t}{m}$$

v = velocity, m/sec
a = acceleration m/sec2
m = weight (tonne)
A = rolling resistance (kW-sec/m)
B = rotating resistance (kW-sec^2/m^2)
C = aerodynamic drag (kW-sec^3/m^3)

The vehicle specific power bins used for MOVES are shown in Figure 5-11. There are 23 operating modes, and three different speed classes, 0-25 mph, 25-50 mph, and 50+mph.

[8] "EPA's Onboard Analysis Shootout: Overview and Results", EPA420-R-02-026, October 2002.

Figure 5-11. MOVES VSP Bins



After deciding on this method, there were a number of data sources to consider, including

- Surveillance data (lab tests of vehicles recruited in-use)
- I/M data
- Remote sensing data

Upon consideration of the pros and cons of each of these data sources, EPA decided to base the emissions primarily on I/M data, and supplement with surveillance data to fill in data gaps, or holes. Thus, the base emission rates for MOVES include I/M effects, and this created the need to develop a method for estimating emissions for non-I/M areas.

After examining a number of I/M programs, EPA decided that the best area to obtain I/M data is Arizona. This is the only I/M program currently performing somewhat random, full I/M240 and I/M147 tests on a limited sample of vehicles. Full IM147 (or IM240) tests are needed to adequately characterize emissions over a fairly wide range of VSP levels.[9]

By using the second-by-second speeds, the investigators examined the distribution of time spent by vehicles in the various VSP bins for both the I/M240 and I/M147 testing procedures. The results are shown in Figure 5-12. The figure shows that there is little to no vehicle operation for these tests in bins 40, 39, 30, and 29. There is some operation in Bins 27 and 28.

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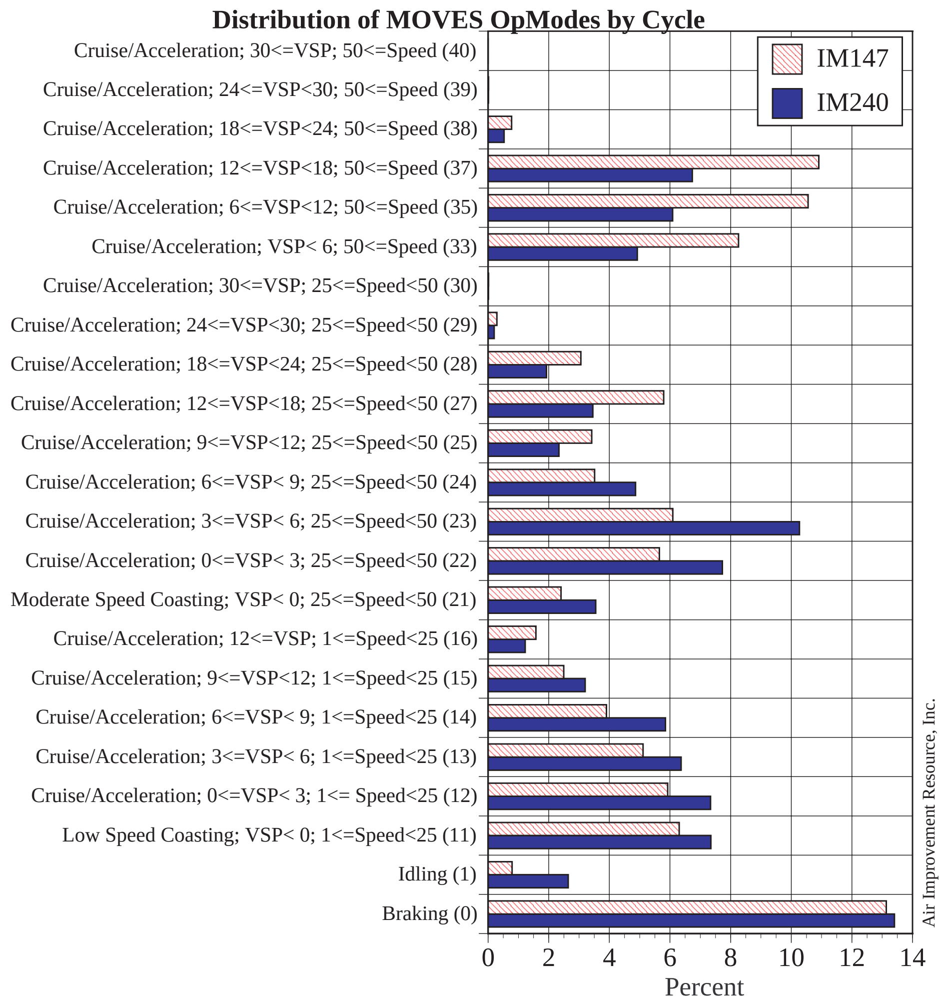
[9] Most areas doing transient testing conduct a full IM240 or IM147 only on vehicles that fail the test. Many passing vehicles are simply “fast-passed” and do not receive the full test.

Figure 5-12. Distribution of Time Spent by Operating Mode (VSP Bin)



Arizona I/M data were obtained by EPA for calendar years 1997-1999 and 2002-2005. Data were not available for 2000 and 2001 because the contractor did not collect the random data in those calendar years. The test procedure used in 1997-1999 was the I/M240, and the I/M147 was used in 2002-2005. Also, the I/M240 data collection from 1997-1999 was a random sample, but the data collection from 2002-2005 was a stratified random sample, where higher emitters were intentionally over-sampled. As a result, there is a need to correct the I/M147 data for this higher emitter over-sampling so that can be recombined with the I/M240 data.

Examination of the second-by-second emissions and speed data showed that there is often a 2-4 second lag between the sudden change in vehicle speed and the recorded emission change. This lag time is due to two factors: (1) the time it takes for the emissions to reach the analyzer, and (2) the analyzer response time. EPA time-aligned the key VSP and emission events in the Arizona I/M data to establish accurate cause/effect relationships. Each pollutant is time-aligned separately. It should be recognized that diffusion and turbulence during the transport time can reduce peak concentrations.

For the I/M147 data, EPA developed passing and failing weighting factors to weight each of the I/M147 emissions results. First, the true frequencies of vehicles failing or passing the I/M147 test were estimated in the overall Arizona sample by dividing the total passing and failing vehicles by the total number of vehicles tested. Next, EPA estimated the weighting factors for passing and failing vehicles by dividing the frequencies into 1.0. Finally, each data point was multiplied by the weighting factors, the product summed and divided by the sum of the weighting factors. This process is illustrated in the example below.

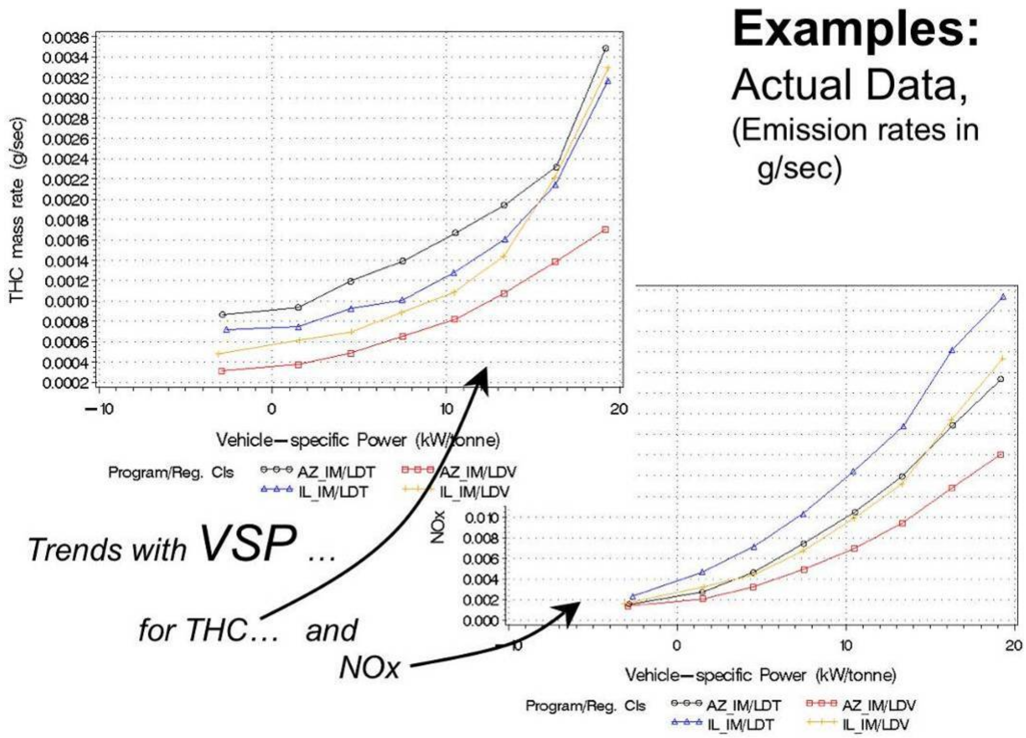
In this example, it is assumed that the overall fail rate is 5%, so the passing rate is 95%. Thus, all passing vehicles are weighted with 20 (1/0.05) and all failing vehicles are weighted 1.05 (1/0.95). If one has a sample of 14 vehicles, where the low emitters are at 2, and the higher emitters are at 10, then the overall sample average is 5.7. If the vehicles are weighted at 2 with 20, and the vehicles at 10 with 1.05, then the weighted average is 2.35, much closer to the lower value. These estimates are shown in more detail in Table 5-4. The investigators believe this is an appropriate method to weight the higher and lower emitters in the I/M147 data.

Table 5-4. Example Development and Use of Sample Weighting Factors

Vehicle Number	Data	Data * Wtg Factor	Weighting Factor
1	2	40	20
2	2	40	20
3	2	40	20
4	2	40	20
5	2	40	20
6	2	40	20
7	2	40	20
8	10	10.5	1.05
9	10	10.5	1.05
10	10	10.5	1.05
11	10	10.5	1.05
12	10	10.5	1.05
13	10	10.5	1.05
Sums		343.2	146.3
Un-weighted Average	6.0		
Weighted average	2.35	(2.35 is 343.2/146.3)	

Emissions are then estimated by VSP, model year group, and age. THC and NOx emissions versus VSP are shown in Figure 5-13 separately for the AZ and IL data sets. As expected, emissions on a g/sec basis increase with the higher VSP bins.

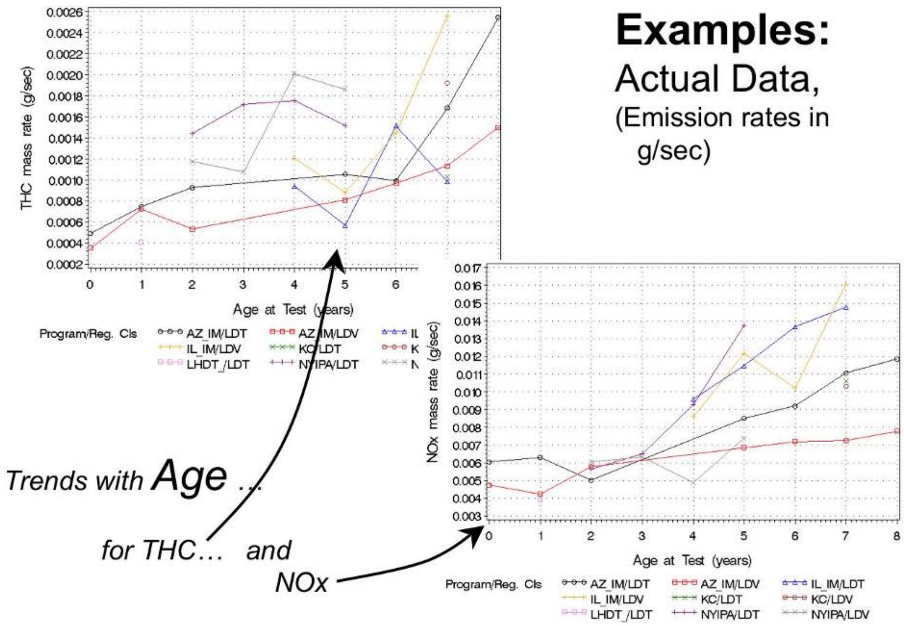
Figure 5-13. Emissions by VSP Bins



Emissions data from the Illinois I/M program are generally higher than for Arizona. The Arizona data in Figure 5-13 have been sample-weighted. EPA believes that the higher emissions in Illinois could be due to the sample of vehicles from Illinois not being random. Additionally, there is no I/M cutpoint for NOx in Chicago, so higher NOx emissions were expected in Illinois.

Emissions versus vehicle age are shown in Figure 5-14 for AZ, IL, and KC. In these examples, emissions increase with vehicle age.

Figure 5-14. Emissions versus Age



Figures 5-15 (NOx) and 5-16 (THC) show emissions versus age for various model years in the Arizona I/M data. The data are only shown through model year 2003, because the Arizona I/M program exempts the first four model years from the program. EPA illustrates two points – the drop in deterioration rates with the Tier 1 vehicles, and the flattening trend that appears to take place around the fifteenth year of life for most vehicles. The drop in deterioration rates is expected given improved fuel quality and vehicle technology, including aftertreatment and reduced oil consumption. The flattening trend has also been observed with remote sensing data; it is thought to be due to the fact that the vehicles that are remaining at this point are vehicles that are well-maintained and used less than the typically-used vehicles.

Figure 5-15. NOx Emission versus Age

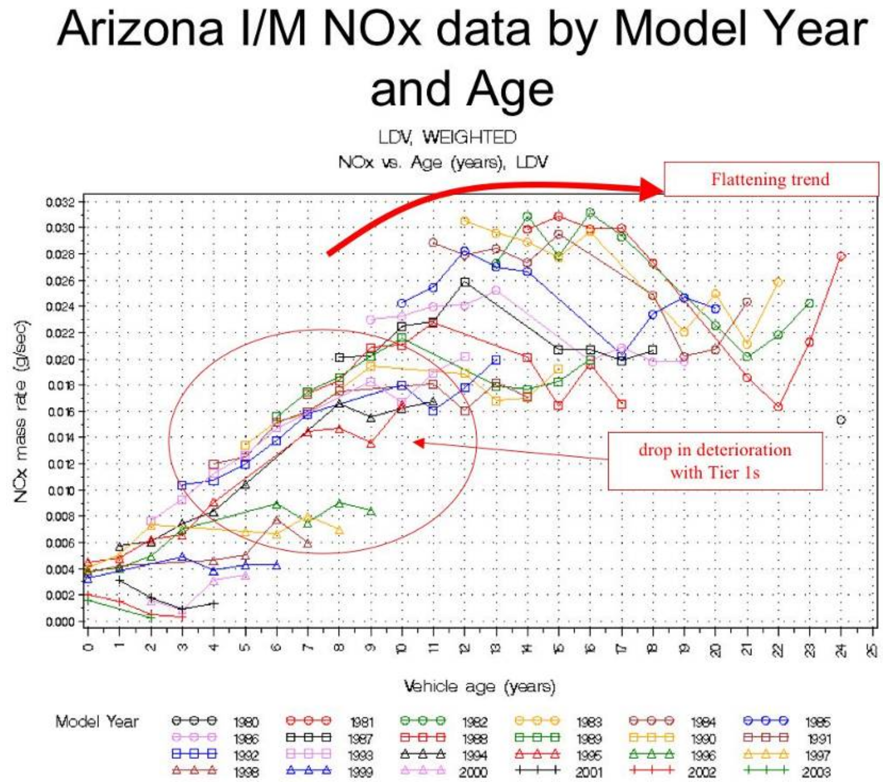
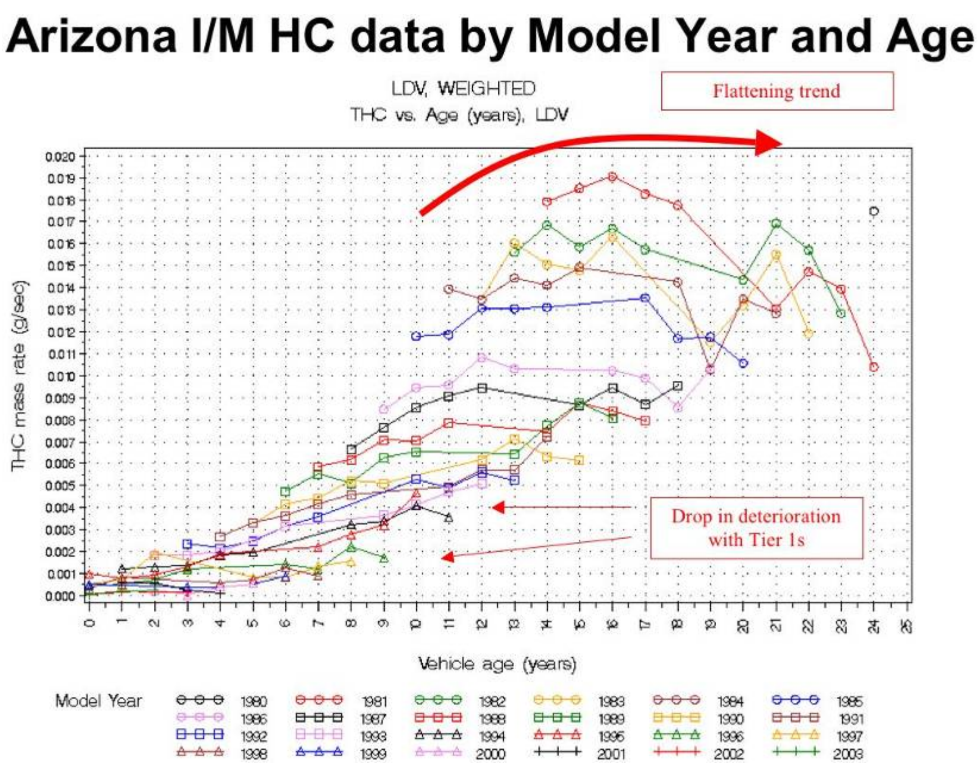


Figure 5-16. HC Emissions versus Age



The reader will note that the approach for estimating emissions based on aggregated I/M data is a significant departure from MOBILE6, not only in that I/M data are being used instead of in-use surveillance data, but also that "high emitters" are not being separated from "normal" emitters. The high and normal emitters are combined into "average" emissions that probably do not represent either emitter group. There are positive and negative aspects of this method, as follows:

Positive: On the positive side, the data speaks for itself. There is no need to determine how the fraction of high emitters increases or changes with age. If I/M programs are measuring the true fleet of vehicles, perhaps there is no need to separate high and normal emitters. Of course, there are vehicles not covered by the I/M program, which are the very oldest vehicles, as well as vehicles that are operating in the area but are not being re-registered in the area. They may be vehicles in need of repair (i.e., high emitters), where the owner cannot afford the repairs, so he or she postpones re-registering the vehicle. But if 95% of the vehicles are being measured, and the other 5% are not all high emitters, this method, even though it produces "average" emissions, should be adequate for most inventory purposes.

Negative: On the negative side, the method may give the wrong impression of what control programs are needed to address high emissions. MOVES users may conclude that the only method to reduce emissions in the future is with lower emission standards which affect the "average" emissions, which in truth only slightly reduce the emissions of normal vehicles, having little effect on inventories (the OBD requirements, however, have reduced the frequency of higher emitters). Secondly, I/M programs and other in-use programs such as remote sensing of high emitters are designed to find and repair high emitters, so the MOVES model should be designed accordingly to take these basic factors into account. Thirdly, the evaporative emissions model in MOVES is still built around the normal/high emitter concept (as discussed in Section 6), with its purge and pressure fail vehicles and "leaking" vehicles being the higher emitters of this process.

If EPA were to revise the exhaust model to be a normal and high-emitting model, it would need to revisit the Arizona I/M data at a minimum, determine a low/high emitter emission level, and develop separate emissions by VSP bin, vehicle class, and age group, for both normal and high emitters. This would double the source bins for exhaust emissions, leading to some model performance impact. In addition, EPA would need to determine the rate of growth of high emitters with vehicle age, and incorporate this as well.

5.2.1 2000 Model Year and Earlier Running Emissions

The running emission rates for model year 2000 and earlier light-duty vehicles are based on the Arizona I/M data. There are about 65,000 vehicles in the dataset, after accounting for screening and quality assurance measures.

One of the key issues is to determine how the running emissions of these vehicles deteriorate as a function of age. After trying different models, EPA chose a log-linear deterioration model, as shown in Figure 5-17. Figure 5-17 shows the increase in InTHC versus age in years for the different VSP bins. The higher VSP bins have higher emissions. This plot is just for model year 1996-1998 Tier 1 LDVs.

Figure 5-17. Log Linear Deterioration Model

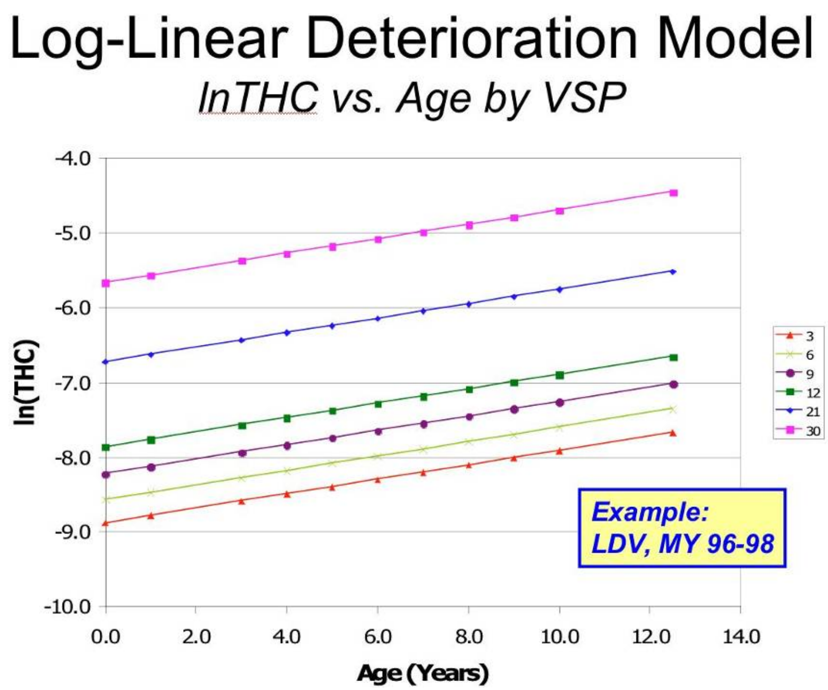
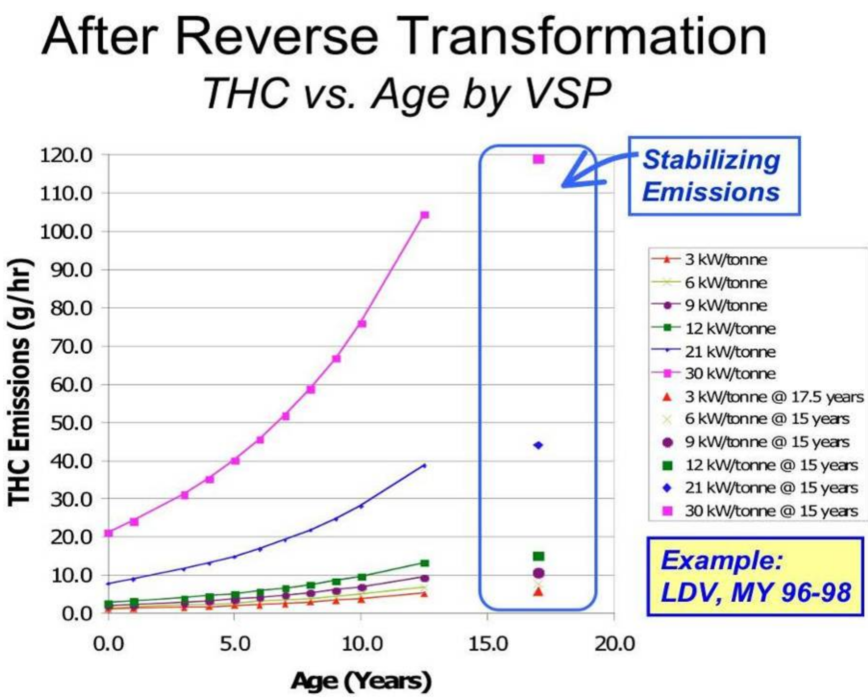


Figure 5-18 shows these emission rates after reverse transformation. With a log-linear model, the emissions increase slowly at first, and then more quickly at older ages. At about 17 years of life, EPA caps the emission rates from increasing any more, to more closely represent the "flattening trend" shown in Figures 5-15 and 5-16.

The curves in Figure 5-18 illustrate that the emissions from the higher bins increase much more quickly and to much higher levels than the lower bins. For example, the 12 kW/tonne bin emissions increase modestly, but the 21 and 30 kW/tonne bins increase dramatically because of their higher starting levels. The extent to which these higher VSP bins influence the overall emissions of the model year group is a function of the fraction of hours spent in each bin. However, as vehicles age, with a given fraction of hours in the higher bins, these higher bins will have a greater impact on overall emissions than when the vehicles are younger. A later section of this report (Section 5.3) illustrates that the higher VSP bins contribute the majority of emissions for light-duty vehicles, even though the amount of vehicle activity in these higher VSP bins is small.

In defending this approach, EPA points to I/M data on individual model year groups of LDVs from both the Arizona and Chicago I/M programs such as those shown in Appendix C. The charts in this appendix appear to show emissions increasing as a function of age and then leveling off for some model years. However, these data are for the full IM147 test, and are not for specific VSP bins. EPA has not shown conclusively that emissions from vehicles operating in the higher VSP bins increase in a log-linear fashion. In addition, the I/M data show not only a leveling off, but an eventual decrease in emissions. EPA's approach caps the emissions at a certain year, but there is no eventual decline that may be observed in the real world.

Figure 5-18. THC Emissions after Transformation



5.2.2 AIR's Evaluation of Exhaust Emissions Deterioration

To evaluate the concept of the log of emissions deteriorating with age, AIR obtained the Arizona I/M data sample used in EPA's analysis. AIR evaluated the full I/M cycle results versus age for three different model years to determine if these are consistent with EPA's analysis, and also evaluated the second-by-second results to determine if emissions in the higher VSP bins have the same deterioration characteristics as the lower VSP bins.

The model years chosen for this analysis are 1987, 1992, and 1997. The 1987 and 1992 model years represent two different ages of Tier 0 vehicles. The 1997 vehicles represent fully phased-in Tier 1 vehicles that have relatively high ages (compared to 1997-1999 Tier 1 vehicles).

The I/M data available from EPA are for calendar years 1997-1999 (3 years), and 2002-2005 (4 years). Figure 5-19 shows LDGV THC emissions versus age for the three model years. In this figure, the I/M147 results have not been weighted for the correct passing/failing percentages. There are gaps in the data where data were not collected for two years. Figure 5-20 shows LDGV THC emissions, where the I/M147 results has been weighted using the EPA weighting methodology. The weighting methodology significantly reduces emissions for the I/M147 results. The CO and NOx weighted results are in Figures 5-21 and 5-22.

Figure 5-19
LDGV THC Emissions versus Vehicle Age
Averages with Standard Errors, Aggregated

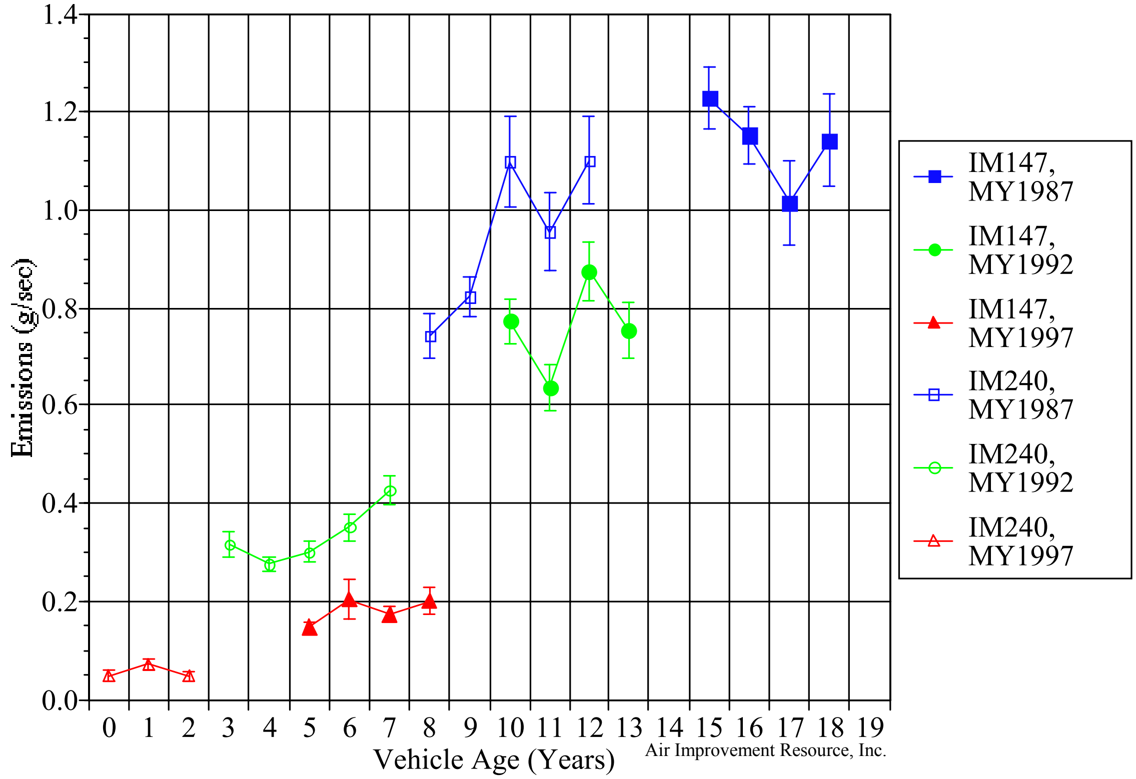


Figure 5-20

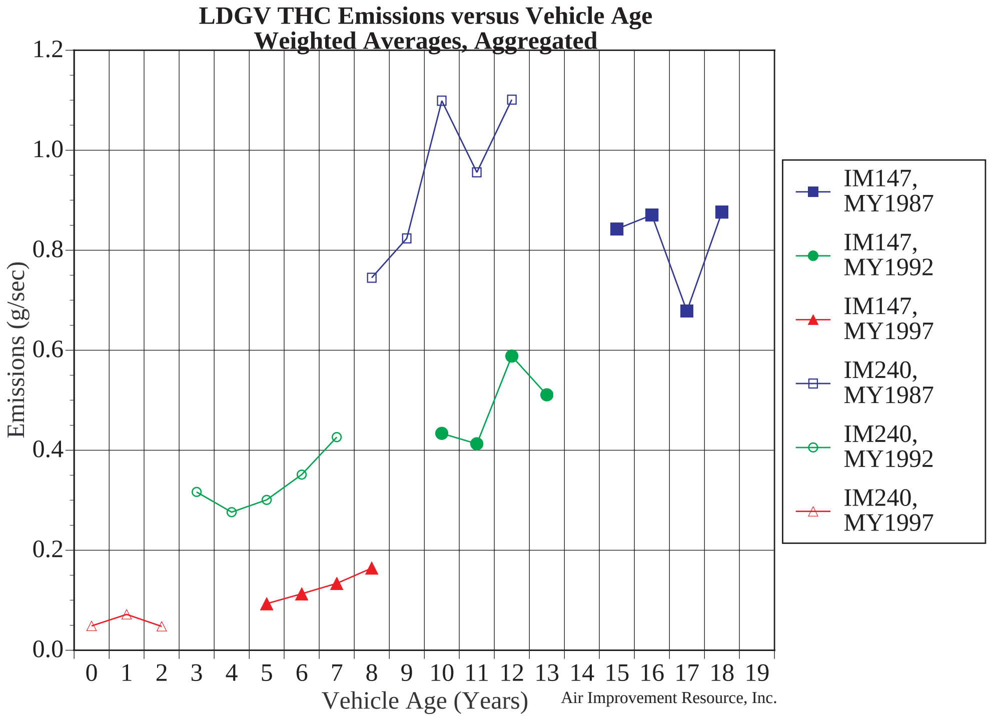


Figure 5-21

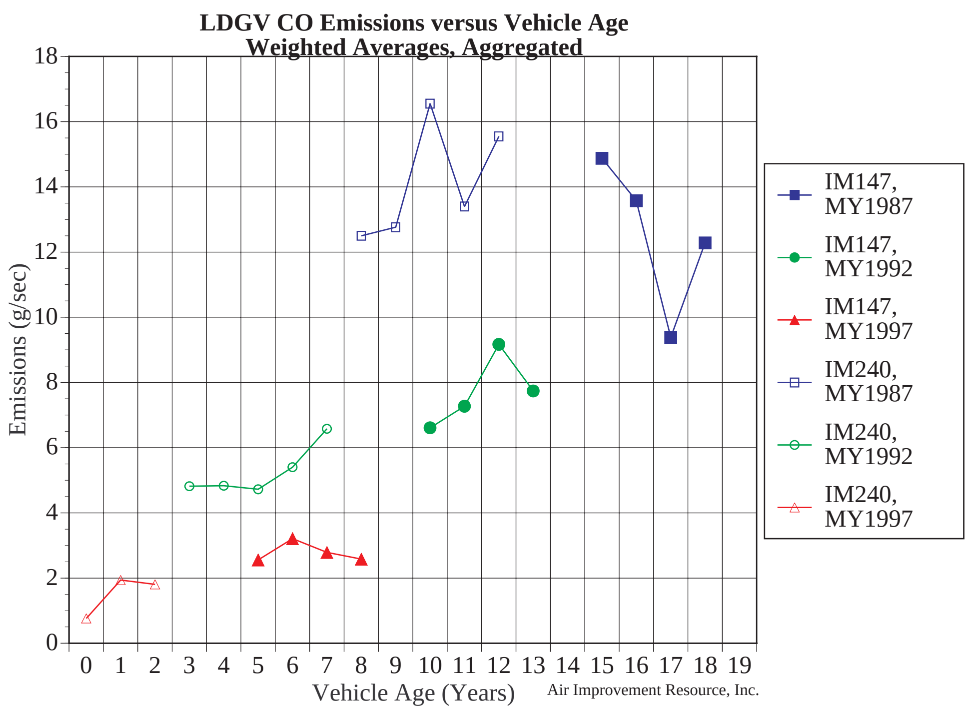
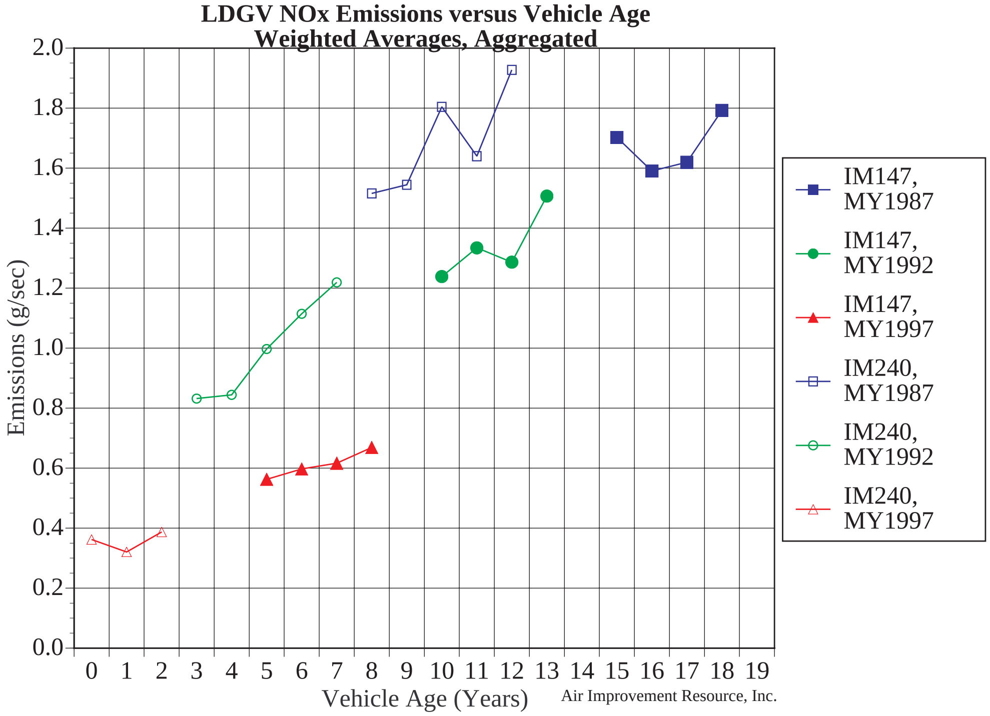


Figure 5-22



These plots of cycle emissions versus age for these three model years of LDGVs do not provide strong evidence of an age versus log emissions relationship. If one considers that the 1987 and 1993 vehicles are Tier 0 vehicles that are quickly disappearing from the road due to attrition, and focuses one's attention on the 1997 Tier 1 vehicles (which will be the next vehicles to disappear from the road, since they are all now over ten years old), the Tier 1 vehicle emissions appear to increase with age in a linear, rather than a log fashion.

To test this further, regressions were fit through the log of emissions versus age, and emissions versus age for LDGVs. The parameters for these regressions are shown in Table 5-5. The correlation coefficients for both log and linear increases in emissions are high, but for 1997 vehicles, the linear model appears to provide a better fit of the data although the two models are close.

Table 5-5. Regression Parameters for Log and Linear Emissions versus Age								
		Linear			Log			
Weighted Pollutant	MY	Slope	Intercept	R^2	Slope	Intercept	R^2	"Best Fit"
THC	1987	-0.01	1.01	0.061	-0.01	0.00	0.058	Linear
	1992	0.02	0.21	0.793	0.06	-1.43	0.816	Log
	1997	0.01	0.04	0.886	0.14	-3.04	0.884	Linear
CO	1987	-0.21	16.12	0.134	-0.02	2.81	0.146	Log
	1992	0.38	3.32	0.842	0.06	1.35	0.873	Log
	1997	0.22	1.32	0.726	0.12	0.21	0.654	Linear
NOx	1987	0.01	1.57	0.052	0.01	0.45	0.062	Log
	1992	0.06	0.69	0.896	0.05	-0.29	0.877	Linear
	1997	0.04	0.32	0.959	0.09	-1.11	0.940	Linear

Nonetheless, EPA does not use the total cycle results, but uses only the second-by-second results for these tests, and evaluates deterioration by VSP bin. In the next comparison, the data were split into high and low VSP, the results were averaged, and the change in emissions for high and low VSP versus age were evaluated. The results are shown in Figures 5-23 through 5-25 (for these vehicles the authors have connected the gaps for the two years that there is no data). Twelve kW/tonne was used as the cutoff for defining low and high VSP levels; this seemed to be a natural break when examining Figure 5-11. When viewing these plots, one should not extend the 1992 data to low ages by utilizing or visualizing the 1997 data at young ages; at young ages the 1992 data would have started at a higher level than the 1997 data.

Except for perhaps HC and CO for the 1992 model year, these figures do not show compelling evidence that log of emissions increase with vehicle age.

Figure 5-23

LDGV Average THC Emissions versus VSP Group and Age

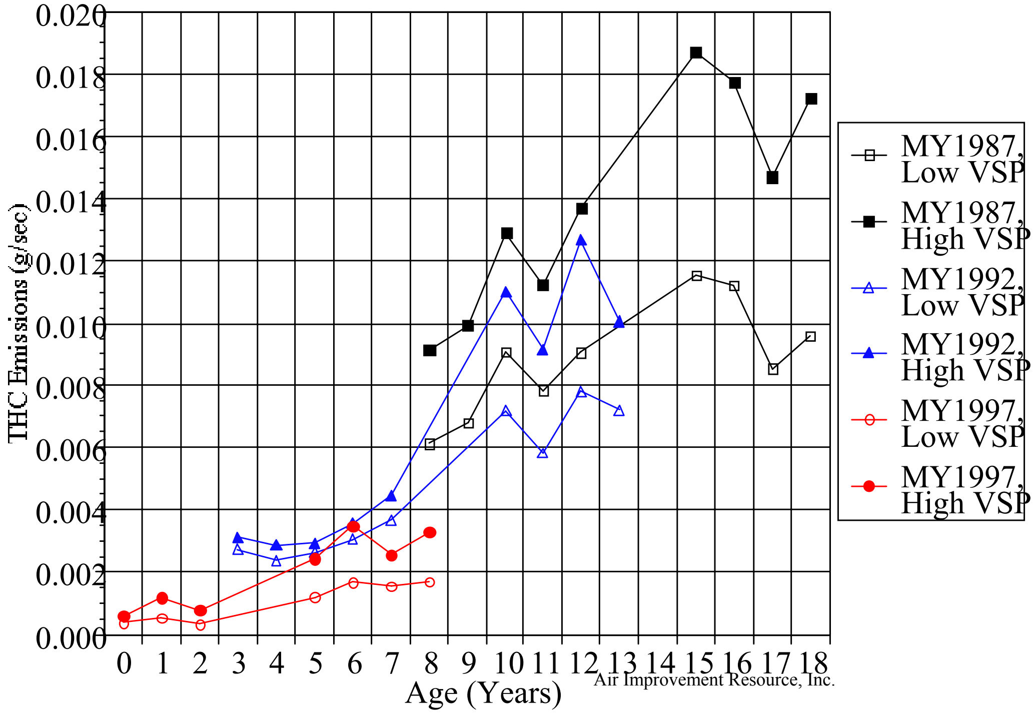


Figure 5-24

LDGV Average CO Emissions versus VSP Group and Age

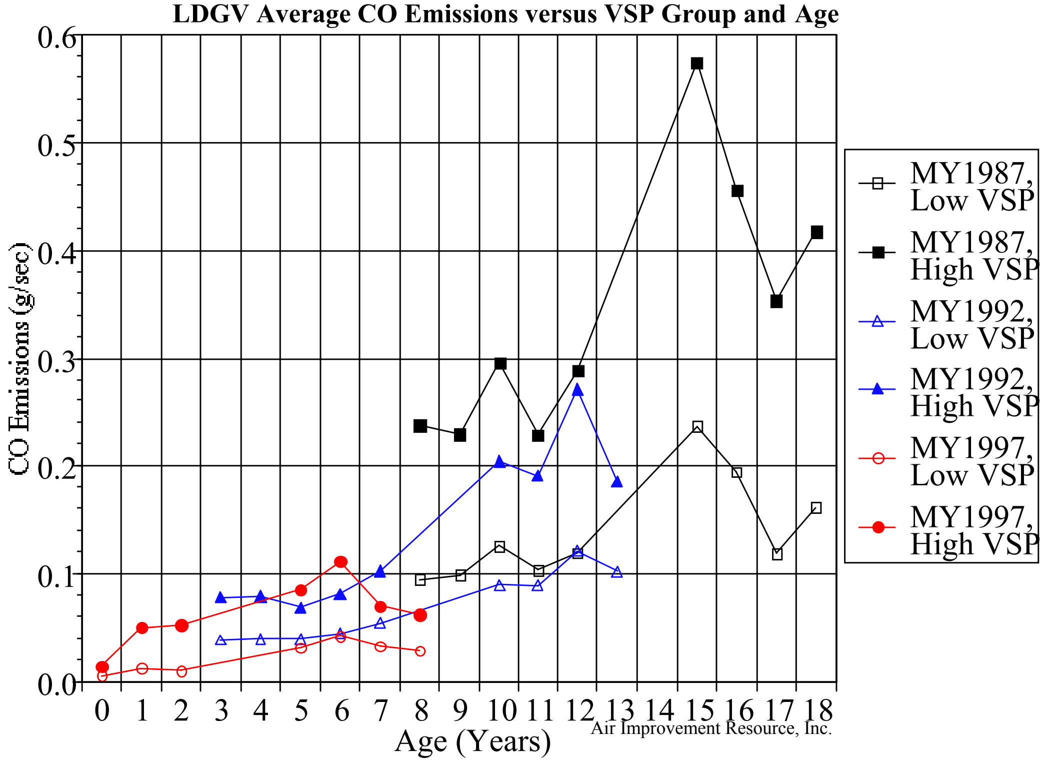
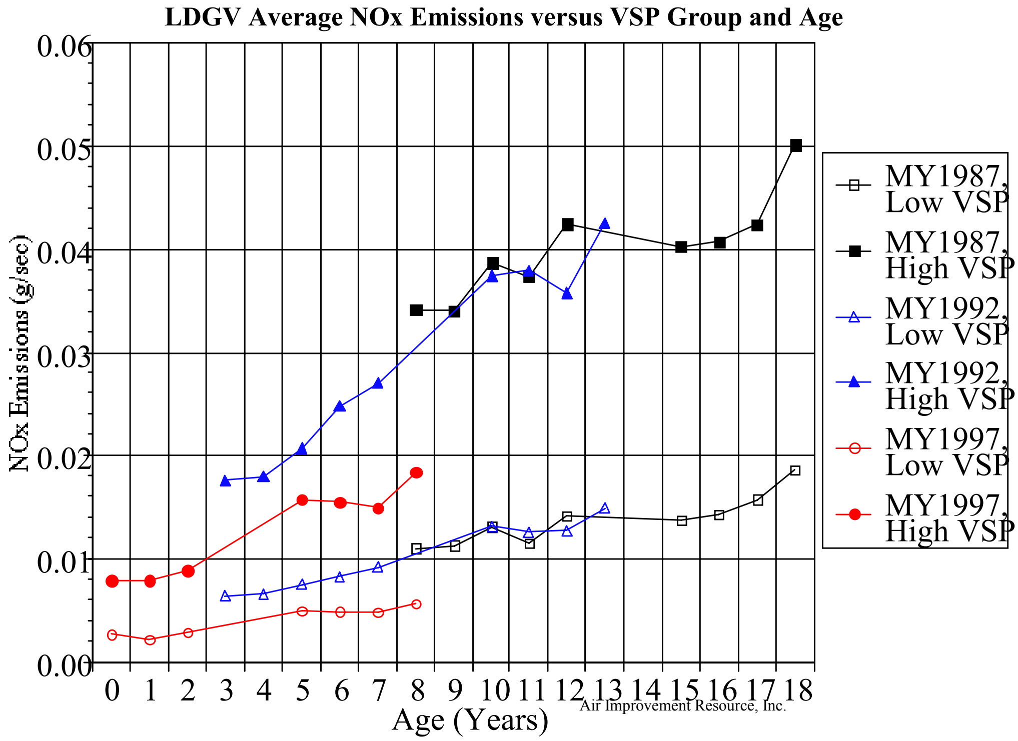


Figure 5-25



The authors conclude that for Tier 1 and later vehicles EPA should revise its deterioration model to be emissions versus age, rather than log of emissions versus age. Figures 5-26 through 5-28 show the differences in these two models for 1997 vehicles for the entire cycle. The two models both model the data appropriately. However, they have very different impacts beyond the current data. The log model results in much higher emissions at the higher ages. EPA does flatten the emissions curve around 17 years, but even if the curve is flattened at 17 years, there would still be very large differences in these predictions at ten years of age and greater.

Figure 5-26

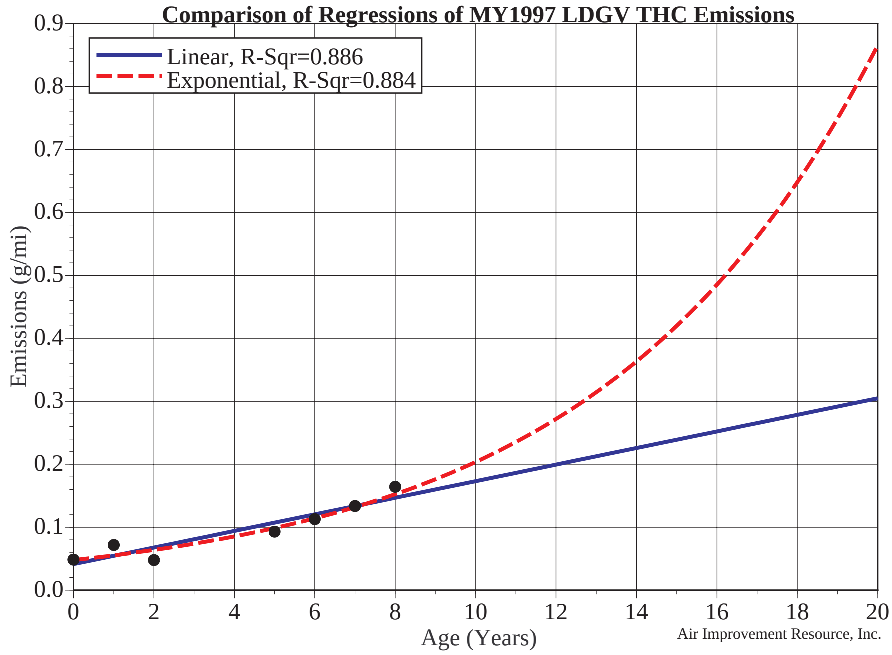


Figure 5-27

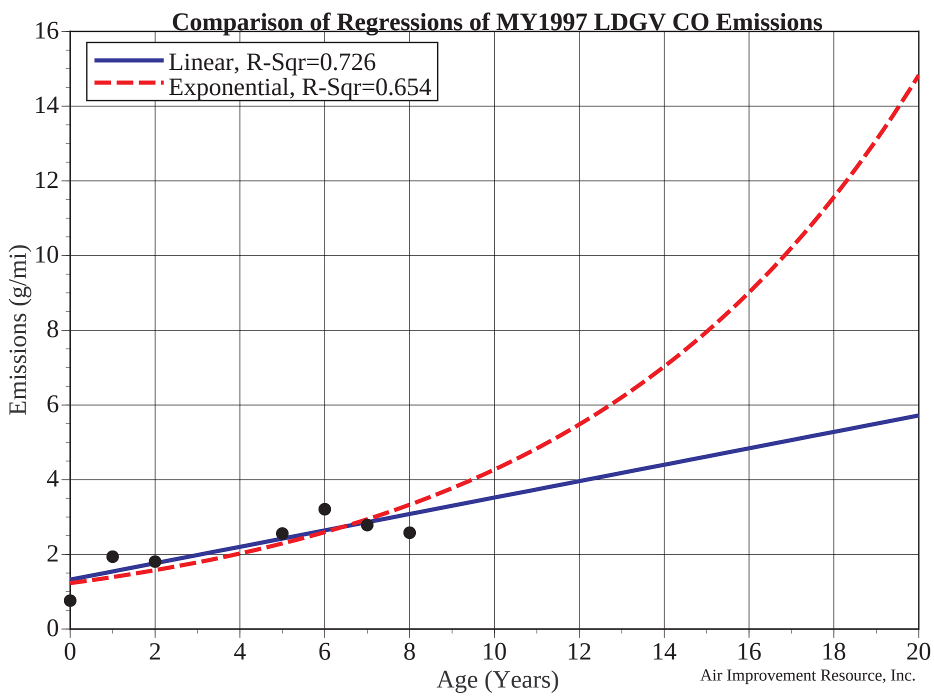
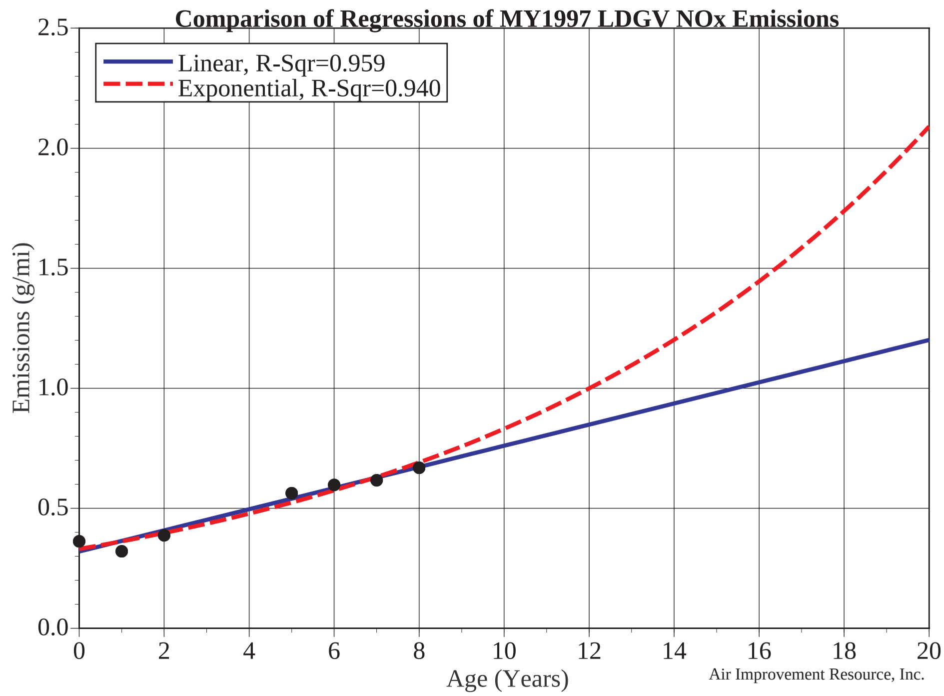


Figure 5-28



The log emissions model may be one of the major reasons why MOVES emissions are much higher than MOBILE6 in Section 4 (the three-city comparison).

5.2.3 2001+ Running and Start Emissions

At the time when EPA started analyzing exhaust emissions for use in MOVES, there were insufficient I/M data on 2001 and later model year vehicles. For 2001 and later model years (start emissions for 1996 and later), EPA has relied on the data from the In-Use Vehicle Program (IUVP). This is a program where manufactures recruit low and higher mileage in-use vehicles and test the vehicles, and send the data to EPA. EPA receives about 2,000 tests per year. Only bag data are available through this database, not second-by-second results. Thus, to develop second-by-second results for the 2001+ vehicles, EPA must start with the Tier 1 second-by-second emissions data from Arizona, and adjust these data to lower levels for 2001+ vehicles using the IUVP data with the following process:

1. Average the IUVP data by Tier and emissions standard bin
2. Develop phase-in assumptions (by Tier, model year, and vehicle class)
3. For both running emissions (defined as bag 2) and start emissions (defined as Bag 1 minus Bag 3), combine the IUVP data and phase-in assumptions to produce technology weighted emission rates for each model year. Next, estimate ratios of start and running emissions for the 2001+ model years to the respective Tier 1 (2000 model year) emissions. Then calculate running emissions by VSP bin in each model year by multiplying the model year 2000 emission rates by the previously estimated ratio for each model year. For MY 2001+ start emissions,

use the weighted average FTP starts directly in conjunction with the start fractions discussed earlier in Section 5.

4. Estimate “off-FTP” emissions
5. Apply deterioration

An example of step 3 for NOx emissions for LDGVs is shown in Table 5-6. Weighted start emissions are 0.12 g/start. The weighted running emissions are 0.00887 g/mi. The ratio of 2010 running emissions to 2000 Tier 1 emissions is 0.06. The 0.06 value is multiplied by each of the Tier 1 emissions estimates by VSP bin to produce running emissions by VSP for the 2010 model year.

Table 5-6. Example Weighting of Start and Running NOx Emissions for 2001+ Light-Duty Gasoline Vehicles (LDGVs)

Model Year - Standard	Cold Start	Running	Fraction
2000-Tier 1	0.983	0.149	1.0
2010 – Bin 5	0.172	0.011	0.595
2010 – Bin 4	0.098	0.007	0.035
2010- Bin 3	0.036	0.005	0.368
2010 – Bin 2	0.049	0.00005	0.002
Weighted start emissions (g/start)	0.12		
Weighted running emissions (g/mi)		0.00887	
Ratio of 2010 to Tier 1 (2000)			0.06

Some of the in-use results for NOx are shown in Figure 5-29. The upper two lines show the certification and useful life emission standards, and the composite line is the average of the test data (FTP). The emissions of these vehicles are well below their emission standards. Bin 5 is the average NOx for fully phased-in Tier 2 vehicles.

Figure 5-29. In Use Vehicle Testing program NOx Results

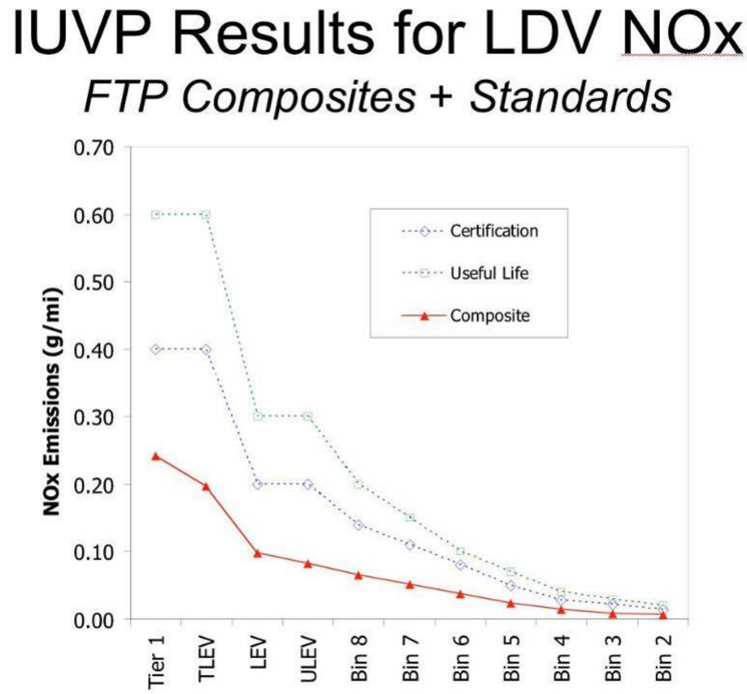


Figure 5-30 shows NOx results for both cold start (Bag 1- Bag 3) and hot running emissions for the IUVP data. Figure 5-31 shows the percent reductions from Tier 1 vehicles. The California LEV vehicles in this plot are all "LEV1" vehicles, and do not reflect the more recent "LEV2" standards that have much lower NOx. Nonetheless, the Tier 2 Bin 5 vehicles have much lower cold start and running emissions than Tier 1 and LEV1 vehicles.

Figure 5-30. Cold Start and Running Results for NOx

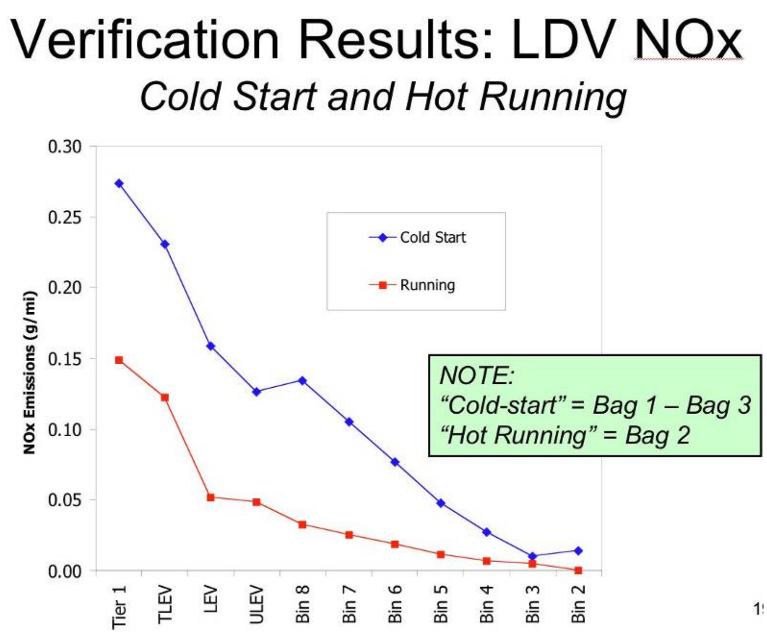


Figure 5-31. Percent Reductions from Tier 1

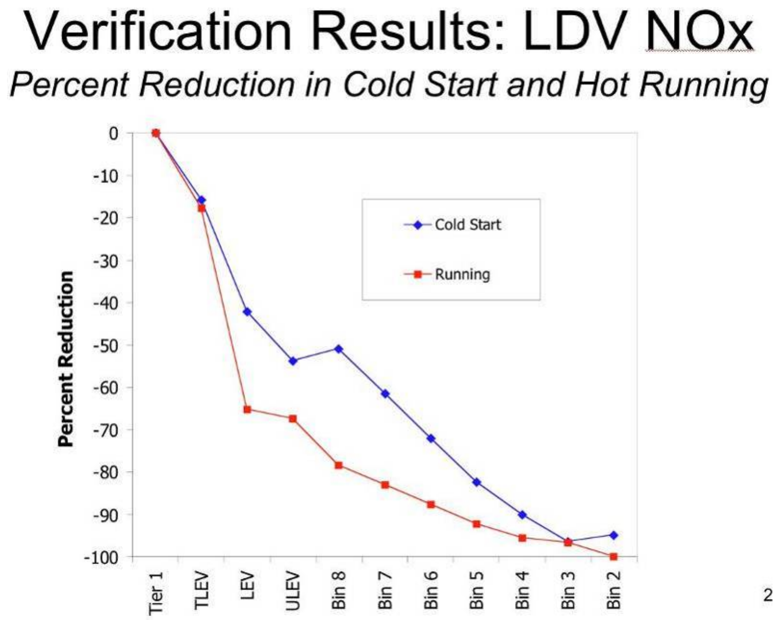


Table 5-7 shows the Tier 2 phase-in assumptions for LDVs and passenger LDTs (LDT1s and LDT2s). Table 5-8 shows the Tier 2 phase-in assumptions for commercial LDTs (LDT3s and LDT4s) being used in MOVES.

Table 5-7. Tier 2 Phase-in fractions for LDVs/LDT1s/LDT2s						
MYR	TLEV	LEV	T2 Bin 5	T2 Bin 4	T2 Bin 3	T2 Bin 2
2001	0.4	0.6				
2002	0.2	0.8				
2003		1.0				
2004		0.75	0.2	0.03	0.01	0.01
2005		0.5	0.45	0.03	0.01	0.002
2006		0.25	0.68		0.068	0.002
2007			0.65	0.048	0.3	0.003
2008			0.632	0.042	0.323	0.002
2009			0.628	0.04	0.330	0.002
2010+			0.595	0.035	0.368	0.002

Table 5-8. Tier 2 Phase-In fractions for LDT3s/LDT4s				
MYR	Tier 1	T2 Bin 8	T2 Bin 5	T2 Bin 4
2001	1.0			
2002	1.0			
2003	1.0			
2004	0.6	0.25	0.14	0.01
2005	0.36	0.5	0.13	0.01
2006	0.12	0.75	0.12	0.01
2007		0.87	0.12	0.01
2008		0.5	0.49	0.01
2009		0.5	0.49	0.01
2010+			1.0	

Figures 5-32 and 5-33 show the weighted average HC and NOx for LDVs/LDT1/LDT2s and for LDT3s/LDT4s based on the phase-in fractions from Tables 5-7 and 5-8.[10] For HC, emissions are slightly higher in model year 2010 for LDT3/LDT4 than for the lighter classes. However, for NOx, the averages for the two vehicle groups are very close in model year 2010. Therefore, one would not expect to see large differences between the car and light trucks classes in a fully-turned over Tier 2 fleet with respect to NMOG and NOx emissions, at least based on the underlying technologies assumed by EPA.

[10] LDT1s: 0-6,000 lbs gross vehicle weight (GVW), 0-3,750 lbs loaded vehicle weight (LVW), LDT2s: 0-6,000 GVW and 3,751-6000 lbs LVW, LDT3s: 7,001-8500 lbs GVW and 3,751-5,750 lbs LVW, LDT4s: 6,001-8,500 lbs GVW, >5751 lbs LVW.

Figure 5-32. MOVES NMOG Averages for Tier 2

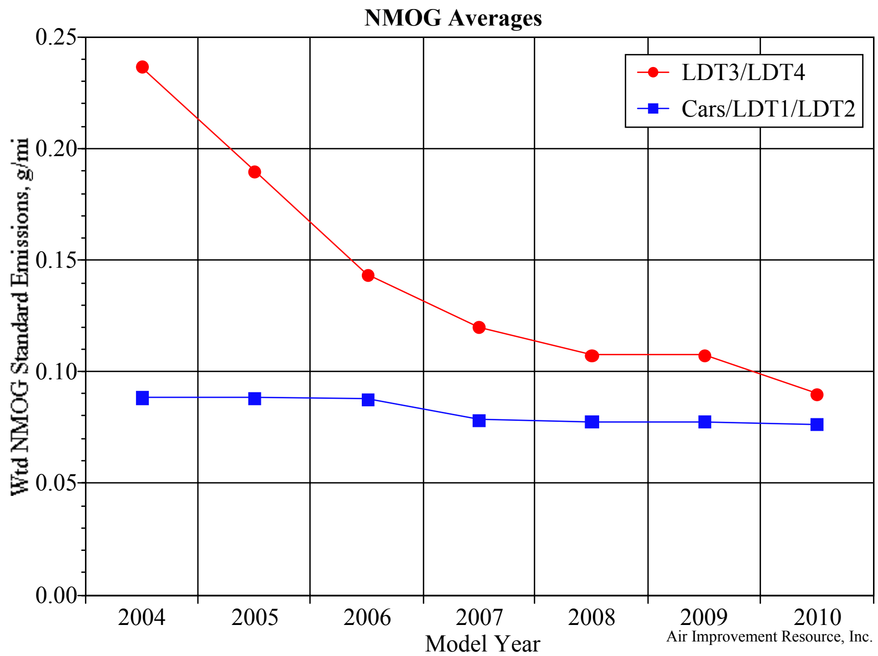
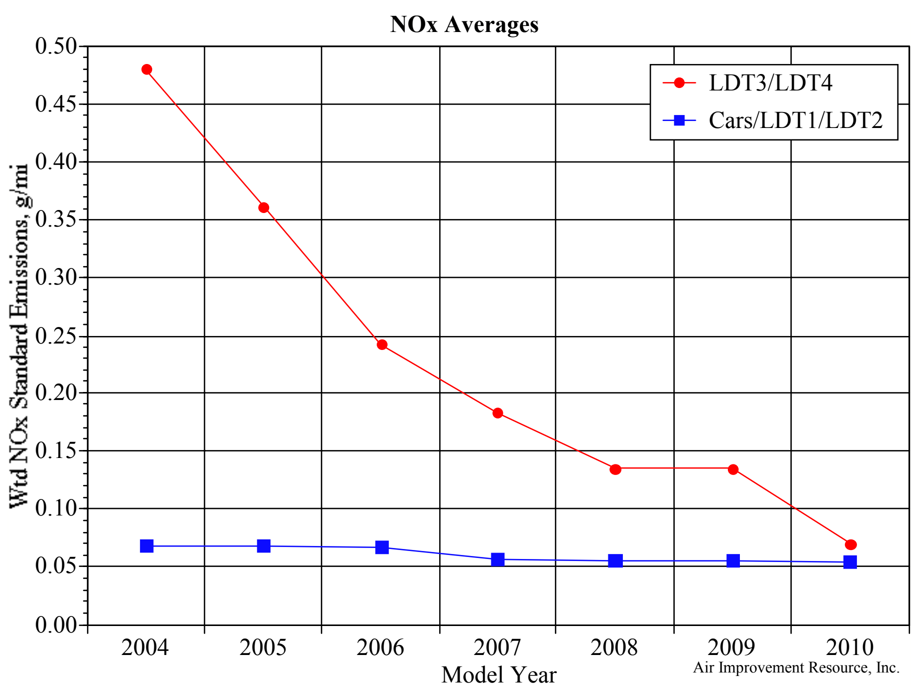


Figure 5-33. MOVES NOx Average for Tier 2



To check the emissions of Tier 2 LDTs versus passenger cars, the MOVES model emission rates by VSP bin for NOx are extracted from the model for the 2010 model year. The NOx emission rates for both vehicle groups are shown in Figure 5-34. The ratio of LDT emissions to LDV emissions by VSP bin are shown in Figure 5-35, along with the ratio of the NOx standards for model year 2010.

Figure 5-34

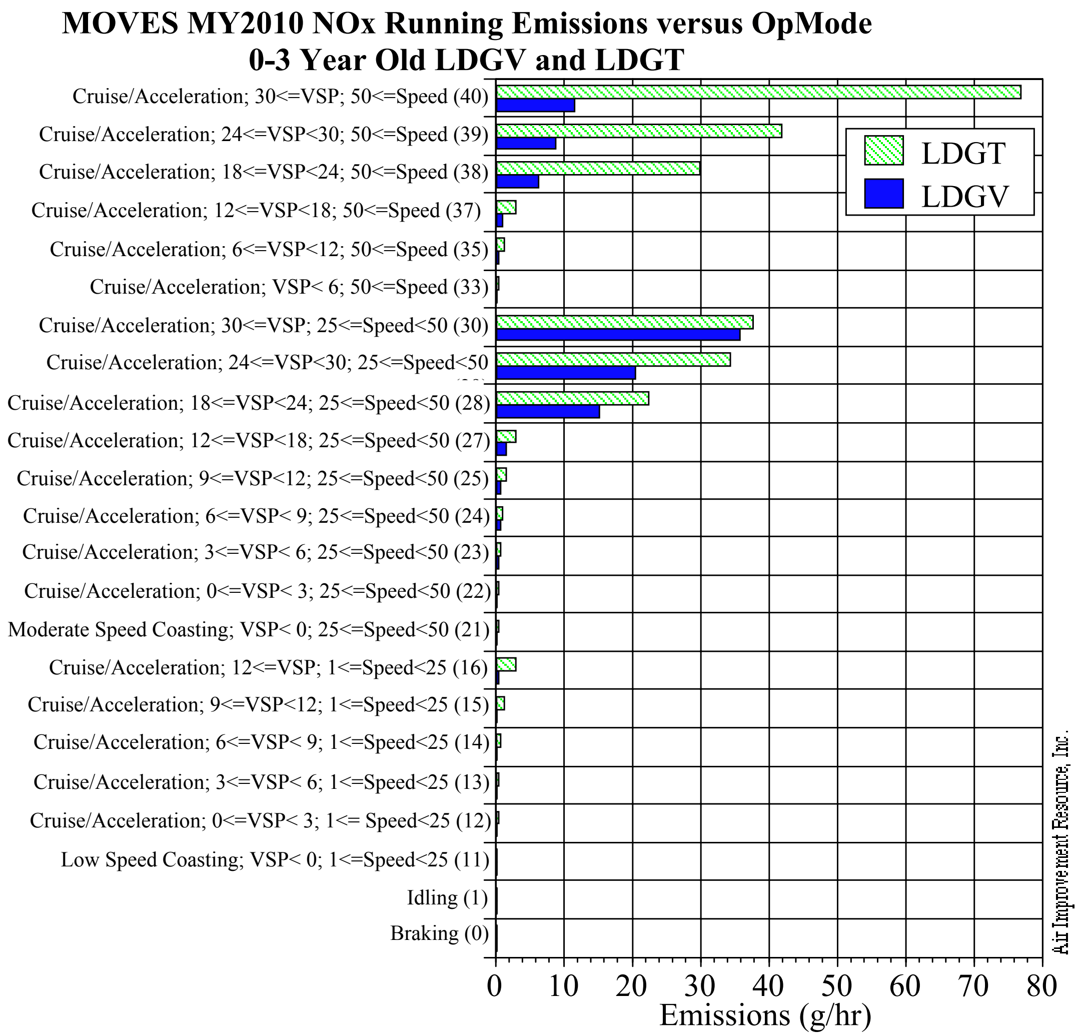
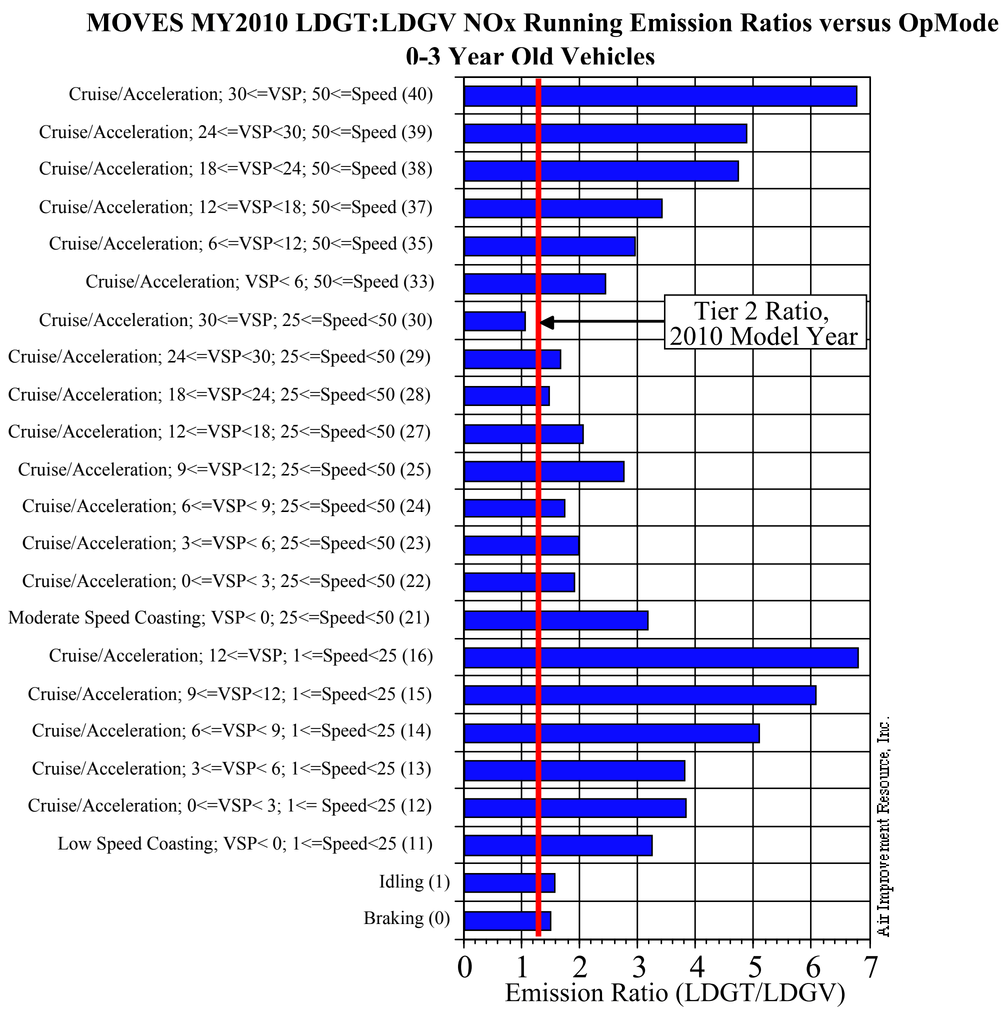


Figure 5-35 shows that the emissions of LDTs (all) are higher than LDVs for many VSP bins – notably 40, 39, 38, 24, and 16. It is difficult to determine the relative emission rates of some of the other bins because of the scale required in this plot. Figure 5-34 presents the ratio of LDT/LDV emissions for each VSP bin and clearly shows that the ratios of emissions in nearly all bins are substantially higher than the emission standards would imply (represented by the red line).

This is a serious concern with MOVES. The Tier 2 program provided for the same emission standards for cars and LDTs. Tables 5-7 and 5-8 indicate that cars and LDTs will meet Bin 5 or lower standards for the 2010 model year, therefore, they should have the same emissions by VSP bin, and clearly they do not. The authors theorize that this occurred is because EPA started with different car and LDT emissions for Tier 1 vehicles and then reduced both proportionally; thus, they are not equivalent when the emission standards are equivalent. The LDT emissions for 2010+ Tier 2 vehicles should be much closer to the passenger car levels.

Figure 5-35



The comparison above is for running emissions only. This issue needs to be examined for cold start emissions as well.

Figure 5-36 shows the percent reductions in NOx emissions for model year 2010 Tier 2 vehicles versus the 2000 Tier 1 vehicles, for age 0-3 vehicles. Cold start NOx is reduced by 86% and running NOx is reduced by 93%. Figure 5-37 shows the percent reductions in HC emissions for Tier 2 vehicles. Start emissions are reduced by 67% and running emissions by 90%.

These percent reductions are used to reduce the emissions for most VSP bins of Tier 0 vehicles to the Tier 2 levels. For the high-powered VSP bins, a different approach is used.

Figure 5-36. Tier 2 Reductions for NOx

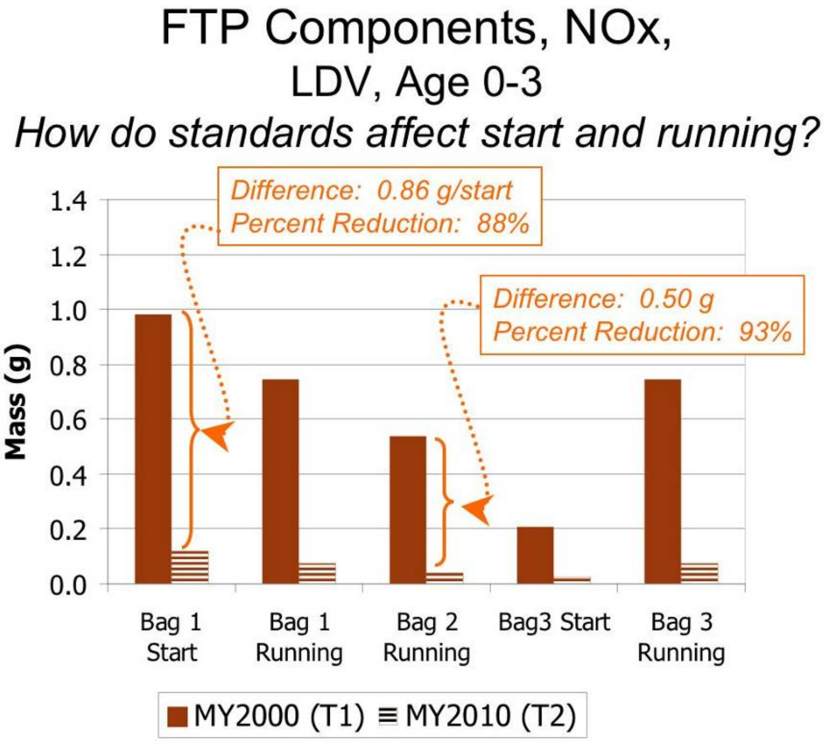
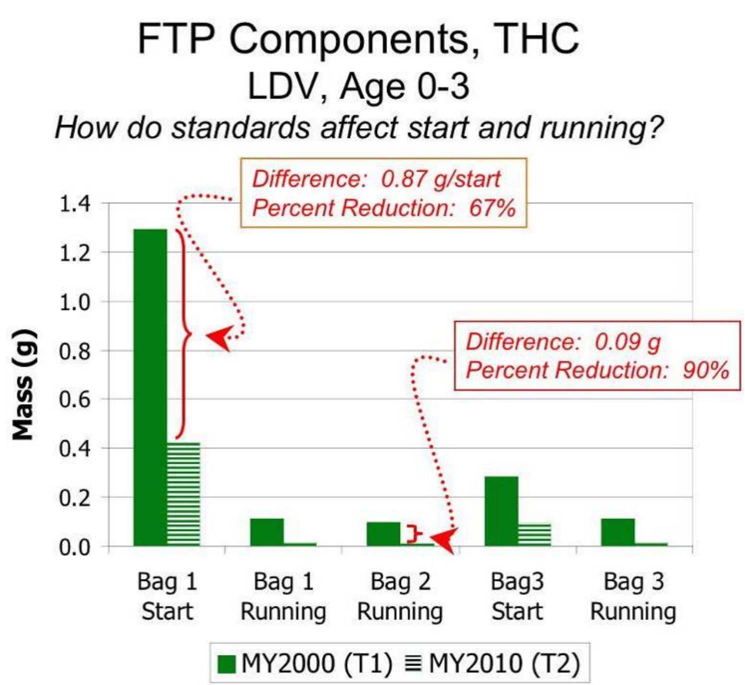
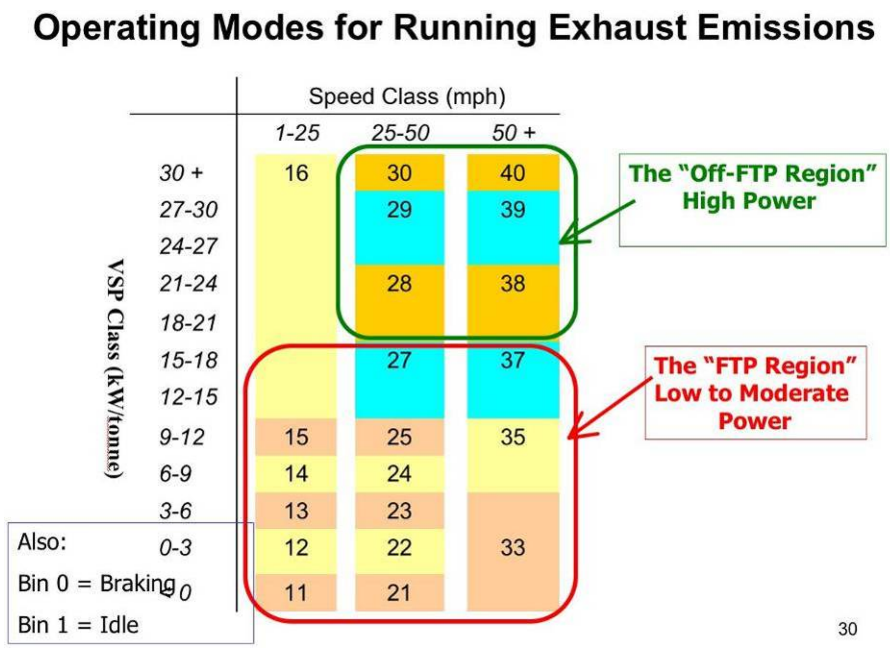


Figure 5-37. THC Reductions for Tier 2



The off-FTP region of higher power is represented by bins 28-30 and 38-40 shown in Figure 5-38. To determine the reductions in emissions from Tier 1 to Tier 2 for this region, EPA examined Arizona I/M147 data on 2001-2003 NLEVs versus the 2000 model year Tier 1 vehicles.

Figure 5-38. Illustration of High Power Region



The 2001-2003 model year NLEV reductions versus Tier 1 vehicles (based on 102 NLEV vehicles tested) are shown in Table 5-9.

Table 5-9. Reductions in Emissions for the Higher VSP Bins	
Pollutant	Percent Reduction
HC	65%
CO	45%
NOx	70%

These reductions are much less than the 94% and 90% for NOx and HC for Tier 2 vehicles for the running emissions, and are applied at the higher bins for all Tier 2 vehicles.

There are several concerns with this method of developing emissions for the higher VSP bins for Tier 2 vehicles, as follows:

1. The IM147 does not include adequate operation in the high VSP range to characterize these reductions. The IUVP data also includes the US06 test. At a minimum, EPA should examine the IUVP US06 data to more fully characterize reductions in the higher VSP bins. Prior to doing this, however, EPA would need to determine what fraction of US06 operation is in the higher VSP bins (since the second-by-second data

are not available from the IUVP data). This would give an indication of whether the changes in emissions on the US06 can be used as a surrogate to adjust the higher VSP bins.

2. The I/M data used by EPA to develop these reductions is a mixture of 2001-2003 passenger cars and LDTs. The supplemental FTP (SFTP) standards were not fully phased in within this time frame. Table 5-10 below shows the phase-in percentages for LDVs and LDTs. [11]

Table 5-10. SFTP Phase-in Schedule

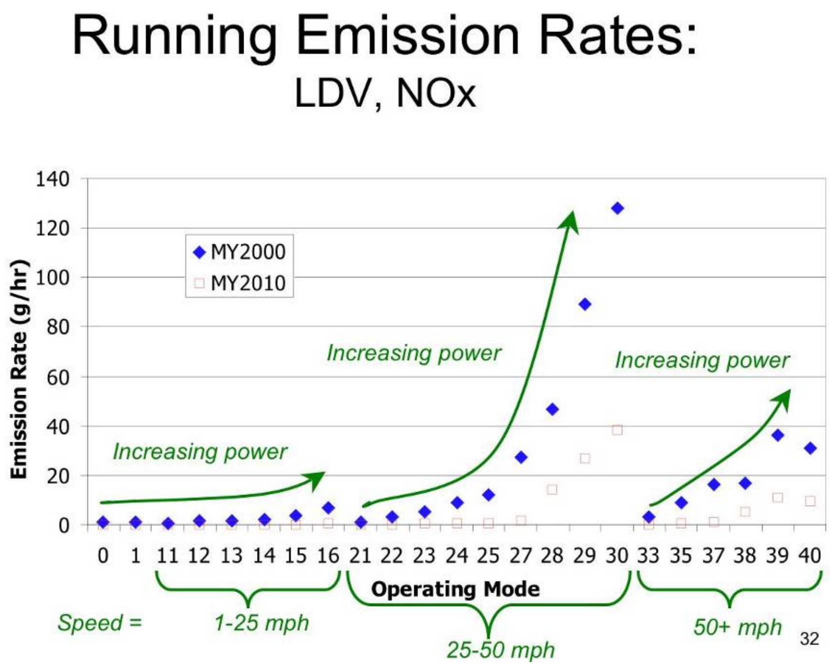
Model year	Car/LDT1, LDT2	LDT3, LDT4
2001	25%	0%
2002	50%	40%
2003	85%	80%
2004	100%	100%

The Arizona I/M data used by EPA to make the high VSP adjustments include a mixture of vehicles, some of which are certified to SFTP standards and some of which are not. By 2004, all vehicles are certified to the SFTP standards. In order to use this I/M data to make this adjustment, EPA would have to separate the SFTP certified vehicles from the non-SFTP certified vehicles, and determine separate percent reductions from each group. Then, these reductions need to be applied using the phase-in percentages supplied in Table 5-10. However, as indicated in the previous comment, a better way to do this would be to examine the IUVP US06 data for the same two groups of vehicles. Perhaps it is best to conduct both sets of analyses (of I/M data and IUVP data on US06) and compare the results.

Figure 5-39 shows the impact of the adjustments in NOx emissions (there are similar adjustments for HC and CO) for all bins from Tier 0 to Tier 2. The Tier 2 emissions are significantly lower in all bins. However, the highest bins for Tier 2 would most likely be even lower if the SFTP rules were appropriately accounted for.

[11] Federal Exhaust Emission Standards Implementation Schedule for Light-Duty Vehicles and Light-Duty Trucks, EPA420-B-00-001, February 2000.

Figure 5-39. Tier 2 Reductions for NOx



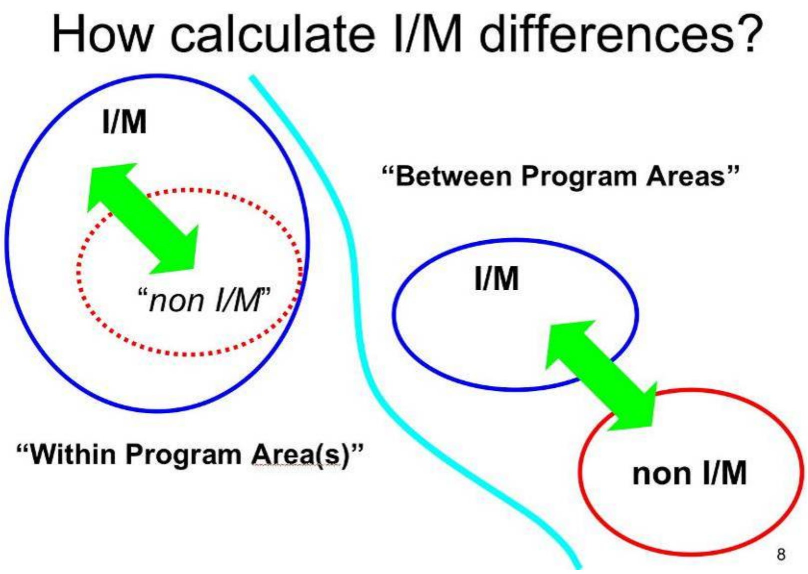
MOVES does not model hybrid vehicle emissions separately from non-hybrid vehicles. While hybrid vehicles generally have zero emissions during braking and idling operation, they meet the same emission standards as non-hybrid vehicles. Thus, while they have lower emissions during idle and braking than non-hybrids, EPA's position is that hybrids could have slightly higher emissions than non-hybrids during other VSP modes, so modeling hybrids' emissions separately is not a high priority for EPA at this time.

5.2.4 Non I/M Emissions

In MOBILE6, the base emissions were developed for non-I/M areas, and I/M "credits" were applied to these emission rates to obtain I/M emissions. In MOVES, it is exactly the opposite: the emissions are developed for I/M areas (from the Arizona I/M data), and then the non-I/M emissions must be developed from the I/M emissions.

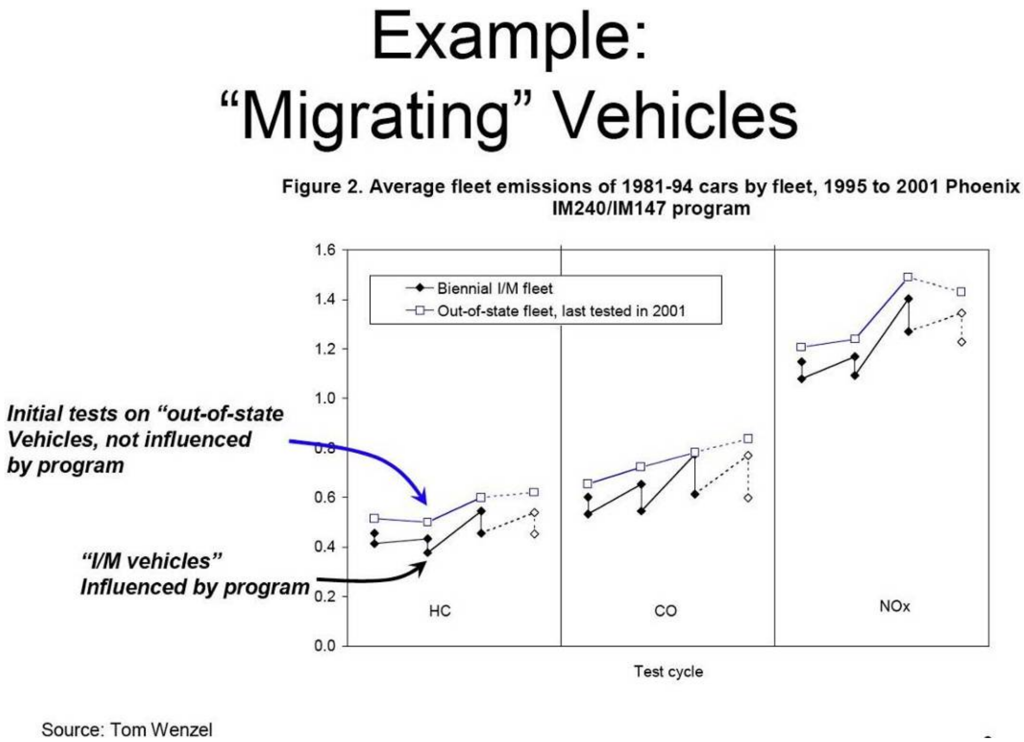
Concepts considered for developing the non-I/M emissions are illustrated in Figure 5-40. Theoretically, emissions can be developed within an I/M program area (left of figure) by comparing vehicles that have recently migrated into the area (and therefore were not subject to I/M) to vehicles that have been in the I/M area for some time. The second method is to compare emission rates in I/M and non-I/M areas directly. This second method requires extensive remote sensing data in both areas, since there are no I//M test results in non-I/M areas. The remote sensing data would need to be corrected for any fuel differences between areas and for deficiencies in the HC measurement methodology.

Figure 5-40. I/M Benefit Concepts



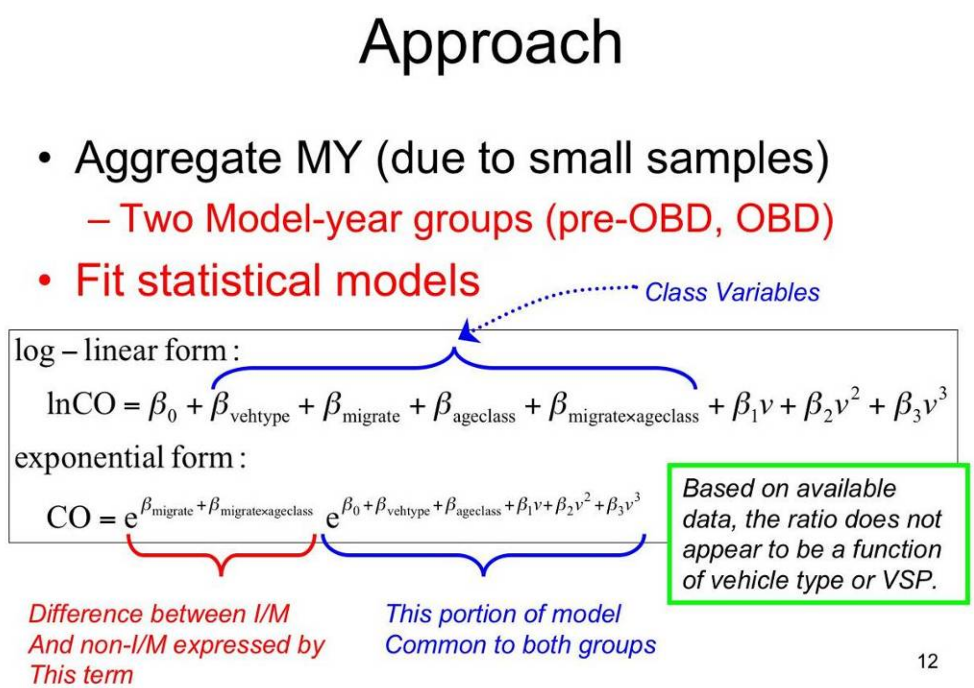
An example of the migrating vehicle approach is shown in Figure 5-41. From an analysis performed by Tom Wenzel, the figure shows average fleet emissions of 1981-1994 cars by fleet (migrating and indigenous) tested from 1995 to 2001 (biennially). The migrating vehicles, uninfluenced by the program, have somewhat higher emissions than indigenous vehicles subject to the Arizona I/M program.

Figure 5-41. I/M Benefits Using Migrating Vehicles



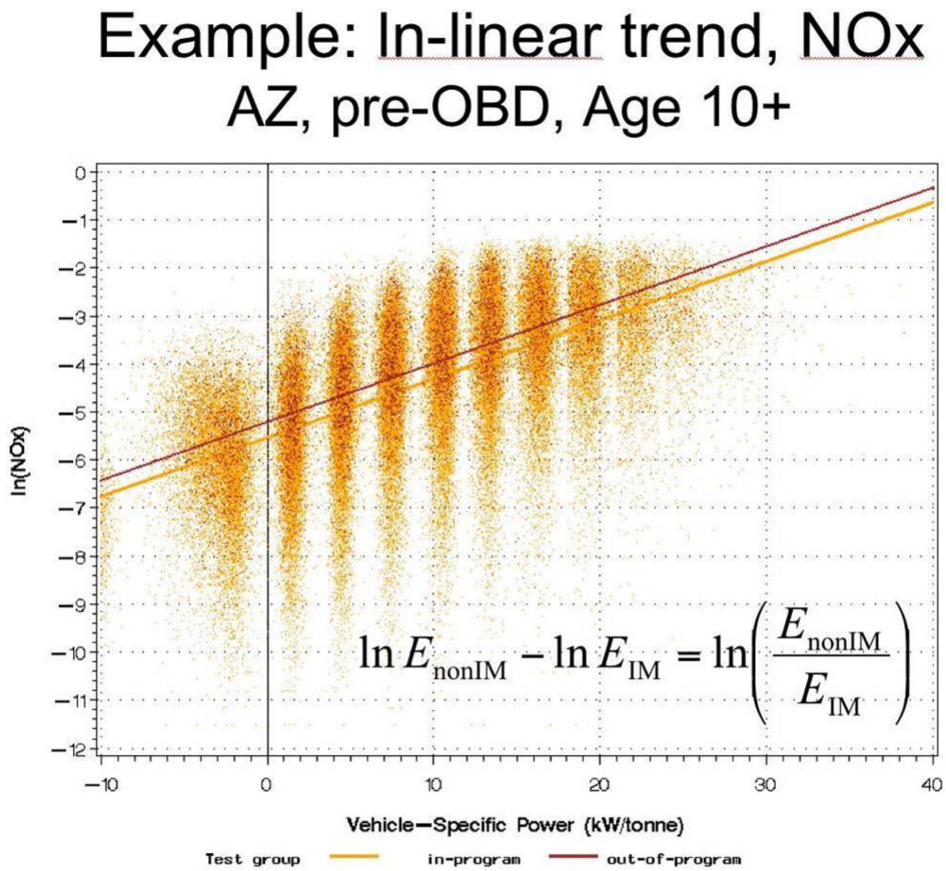
To further estimate the non-I/M emissions, EPA developed a multivariable regression of the log of emissions for OBD and non-OBD vehicles as shown in Figure 5-42.

Figure 5-42. Regression Approach for Estimating I/M Benefits



An example regression using this model is shown in Figure 5-43. The figure shows a small difference in the log of emissions between migrating and indigenous vehicles.

Figure 5-43. NOx Regression for I/M Benefits



EPA performed this analysis for both the Arizona and Chicago I/M program data. The results are shown in Figures 5-44 and 5-45. The two I/M programs show similar results for both OBD and non-OBD vehicles. It was somewhat surprising that OBD- equipped vehicles showed differences in emissions for both programs even in the 0-4 age year group. Both the Arizona and Chicago I/M programs exempt age 0-4 year vehicles (except for change-of-ownership vehicles), so there were probably very few vehicles on which to base the 0-4 age group estimates. This also explains the high variability of this group.

Figure 5-44. Non/I/M: I/M Ratios for Phoenix

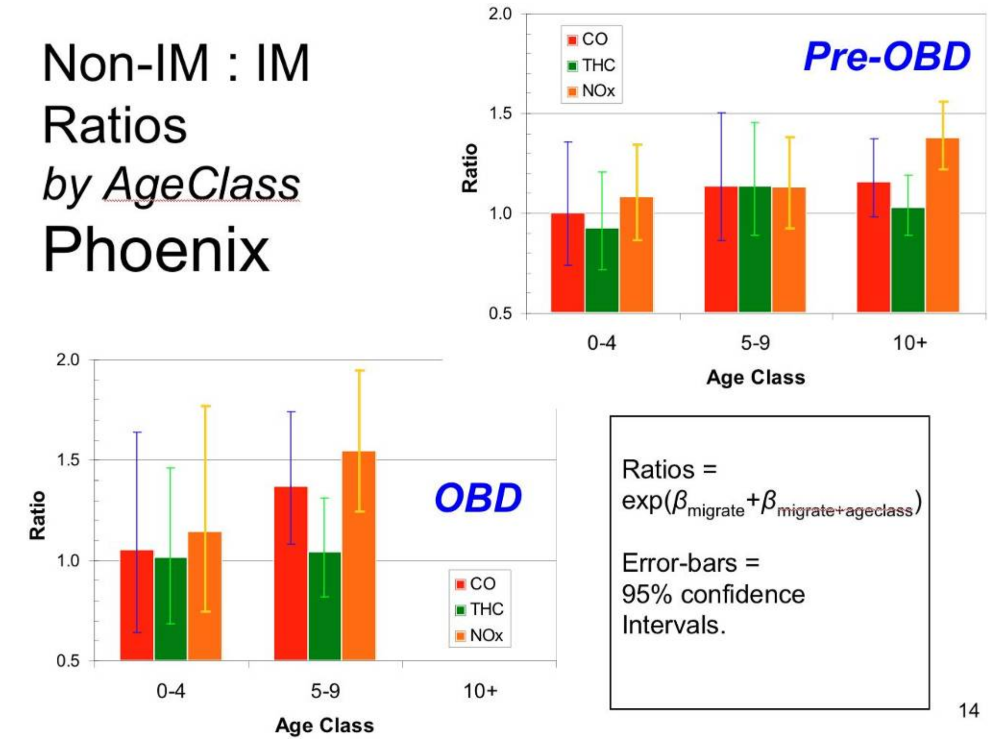
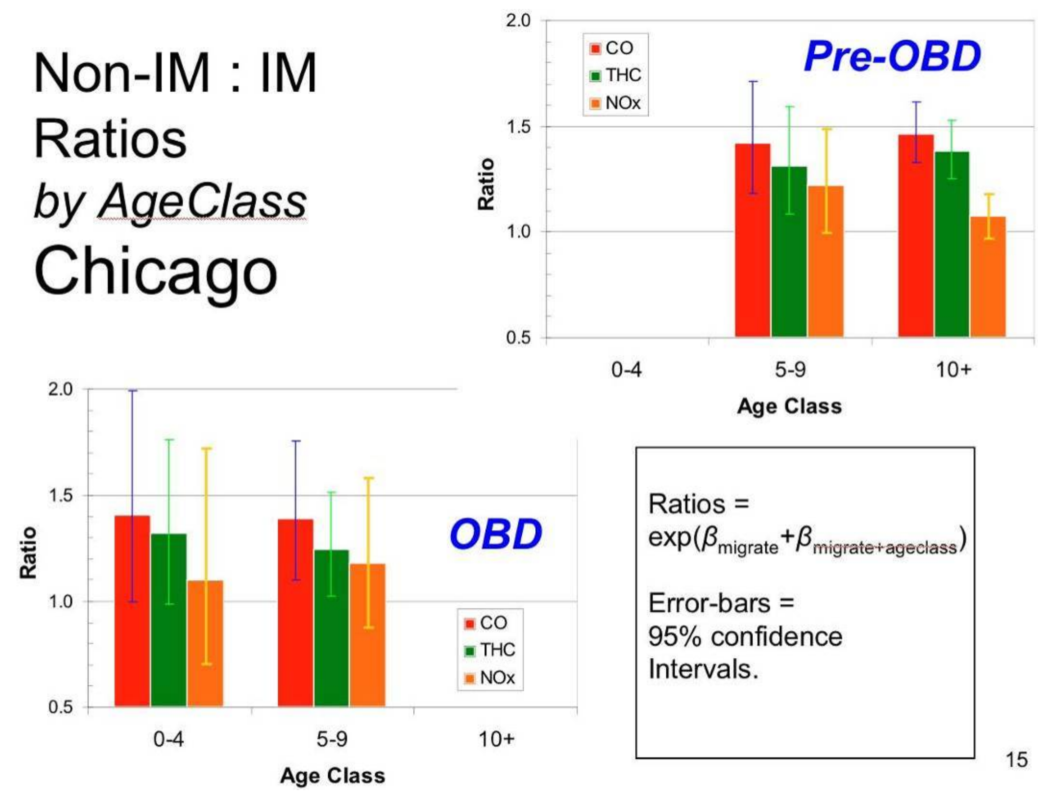


Figure 5-45. Non/I/M:I/M Ratios for Chicago



Analyzing data from both the Arizona and Chicago I/M programs, EPA determined that the ratio of non-I/M to I/M emissions is not a function of car versus LDT, or VSP, but is a function of vehicle age. Therefore, MOVES estimates non-I/M emissions by multiplying I/M emissions by a ratio of non-I/M emissions to I/M emissions by age. These ratios are shown in Figures 5-46 through 5-48. For CO, the emissions are generally about 20% higher than the I/M emission rates at ages six years and above. For HC, the non-I/M emission rates are about 30% higher, and the non-I/M emissions for NOx are about 20% higher at ages six years and above. The differences are much less for the 0-3 age vehicles, which are based only on the Arizona data because EPA could not determine why the Chicago 0-3 non-I/M emissions were so much higher than the I/M emissions. Still, it is surprising that there is any I/M benefit for 0-3 age year vehicles, nearly all of which would be under warranty.

For the final ratios, EPA did not differentiate between OBD and non-OBD equipped vehicles.

Figure 5-46. Non-I/M: I/M Ratios for CO versus Age

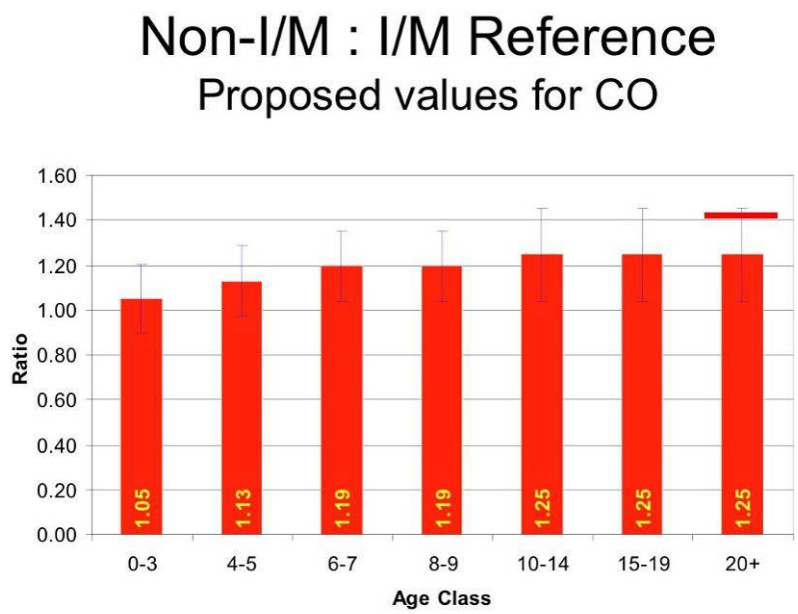


Figure 5-47. **Non-I/M: I/M Ratios for THC versus Age**

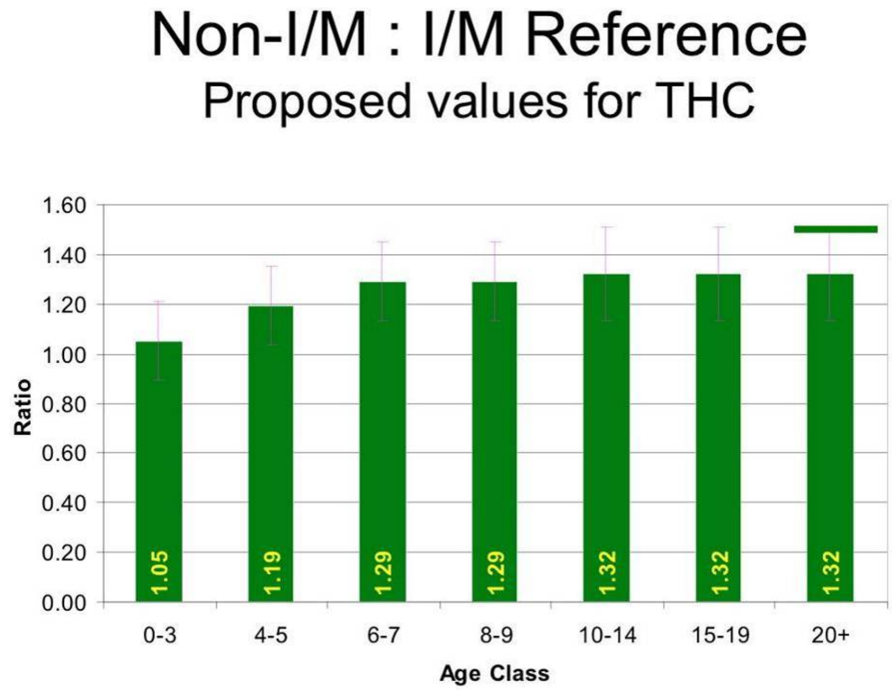
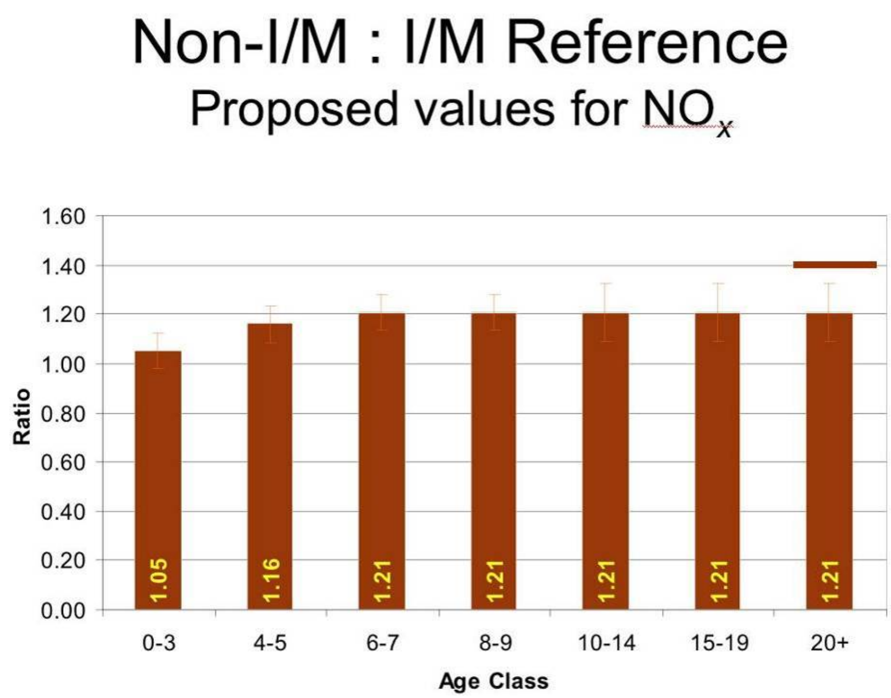


Figure 5-48. **Non-I/M: I/M Ratios for NOx versus Age**



EPA also developed a method for modifying the benefits of a particular I/M program for the characteristics of that I/M program. The method is to apply an I/M adjustment factor that weights the I/M and non-I/M reference emissions together as follows:

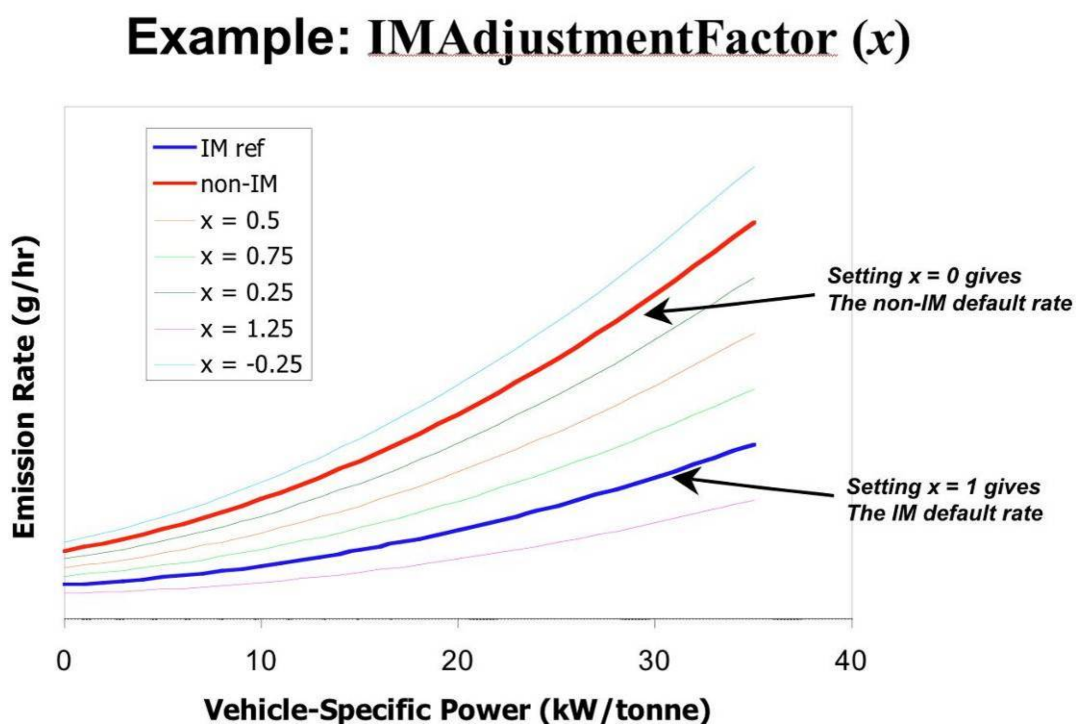
$$E_{prog} = x * E_{I/M} + (1-x) * E_{non\ I/M}$$

Where:

E_{prog} = emission rate for given program
$E_{I/M}$ = I/M reference emission rate (AZ program in 2003)
$E_{non\ I/M}$ = non I/M reference emission rate
x = I/M adjustment factor

The application of this concept is shown in Figure 5-49 below.

Figure 5-49. I/M Adjustment Factors



EPA uses MOBILE6 to develop the adjustment factors, based on the following MOBILE6 I/M inputs, relative to the I/M reference case (AZ I/M program in 2003):

- Pre-1981 model year stringency
- Waiver rates (combined with compliance rate)
- Specific I/M240 cutpoints
- Technician training credits
- I/M effectiveness factor
- I/M program start years
- Residual I/M effects

Thus, the basic I/M benefit is developed from the Phoenix I/M program and vehicles migrating into Phoenix, and then this is modified somewhat using MOBILE6. Therefore, this method assumes that MOBILE6 estimates the relative differences in different I/M programs correctly.

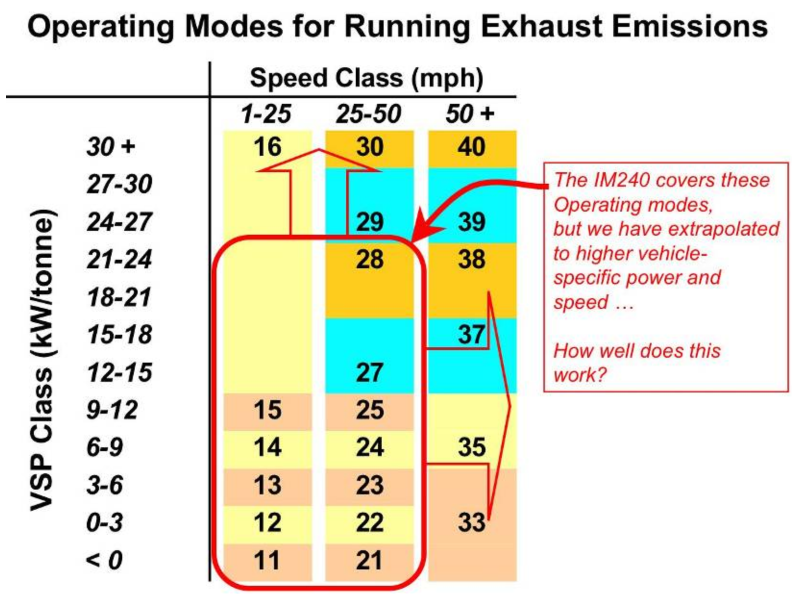
Overall, EPA indicates that I/M benefits are 20%-70% lower in MOVES than in MOBILE6. Reasons EPA identifies are:

- MOBILE6 assumed a 10% response rate to OBD outside warranty without I/M
- Since then, CRC performed a comprehensive survey of MIL response in non-I/M areas, and found response is over 90% [12]

5.2.5 Higher VSP Emission Effects

Figure 5-50 shows that the IM240 represents many, but not all VSP bins. To augment the IM240 data at the higher VSP bins, EPA identified other testing that was conducted on the US06 and the Modal Emissions Cycle (MEC), as shown in Figure 5-51. None of the vehicle model years tested as shown in Figure 5-51 are subject to the SFTP standards.

Figure 5-50. VSP Bins in MOVES



[12] CRC E-72, "Consumer Response to MIL Illumination", Final Report, Eastern Research Group, April 15, 2005.

Figure 5-51. Data Sources for Aggressive Cycles

Data on Aggressive Cycles
Model-Year group X RegClass

MYG	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

CO, NOx, and THC emissions for the different VSP levels are shown in Figures 5-52 through 5-54. The lowest emission lines are for the newer model years of LDVs and LDTs.

Figure 5-52. HC Emissions on Aggressive Cycles

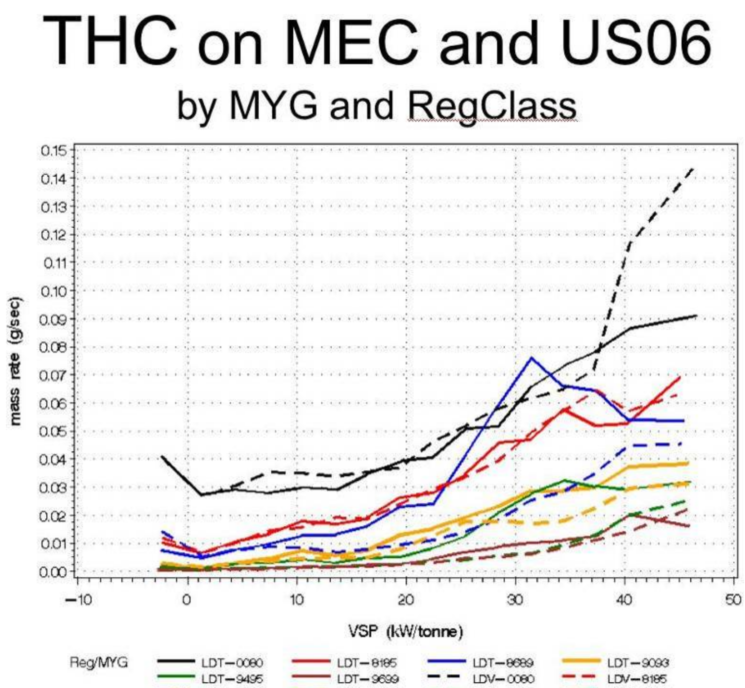


Figure 5-53. CO Emissions on Aggressive Cycles

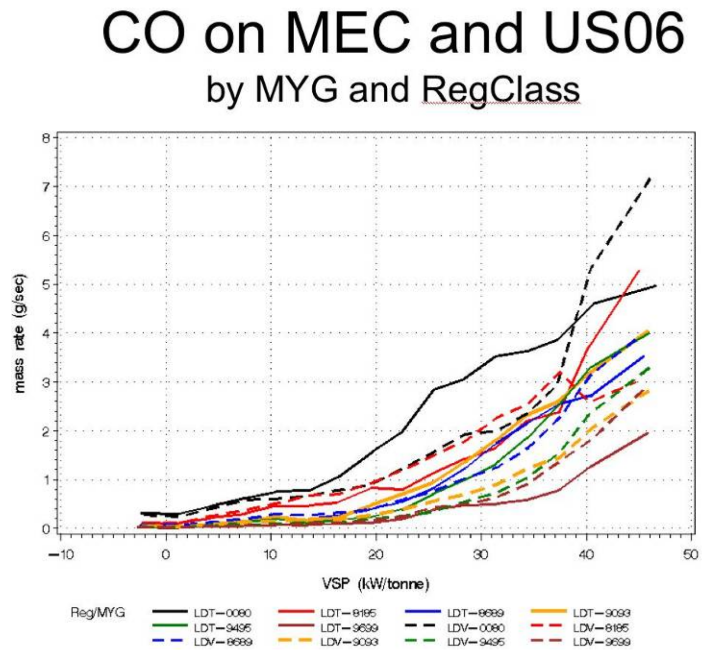
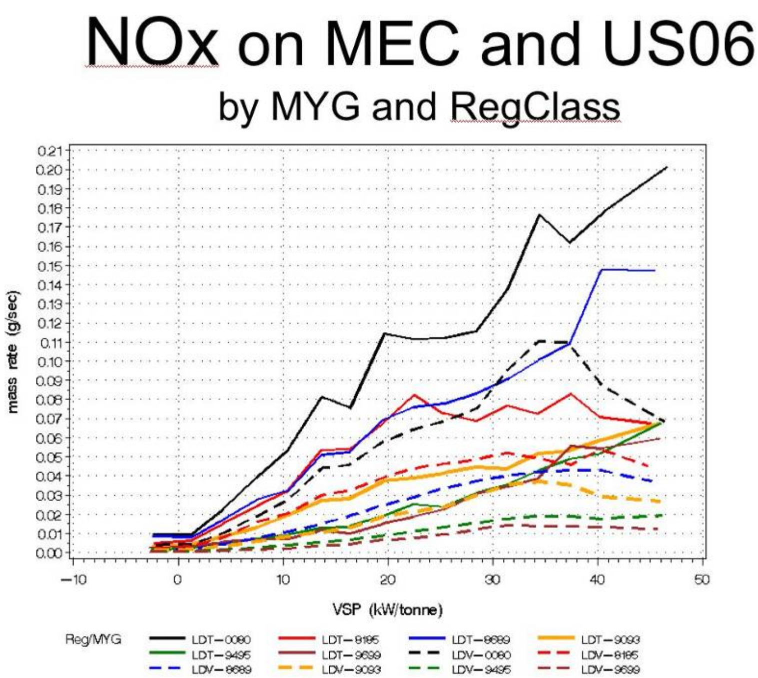


Figure 5-54. NOx Emissions on Aggressive Cycles



EPA determines the ratio of bins 28, 29, and 30 to bin 27, and bins 38, 39, and 40 to bin 37. The I/M 240 does include some operation in both bin 27 and bin 37 (see figure 5-10). The ratios are then compared to the ratios developed from using only the I/M240 data (called the "initial" ratio). The initial ratio is a candidate for adjustment if it lies outside the 95% confidence interval of the rate determined from the ratios. Adjustments for THC, CO, and NOx are shown in Figures 5-55, 5-56, and 5-57. In some bins the initial rates are adjusted upward, and in other cases they are being adjusted downward.

Figure 5-55. Adjusted THC Emissions

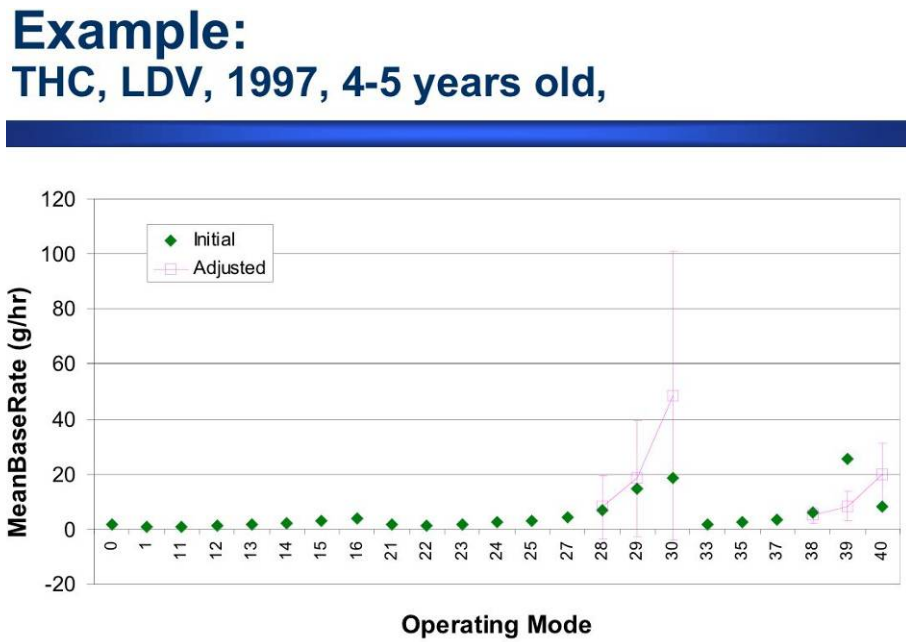


Figure 5-56. Adjusted CO Emissions

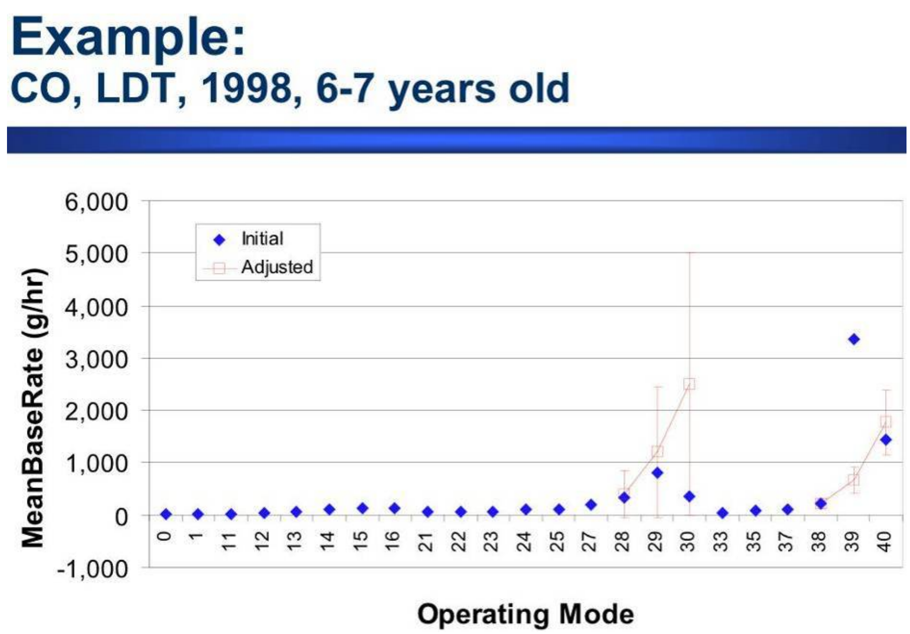
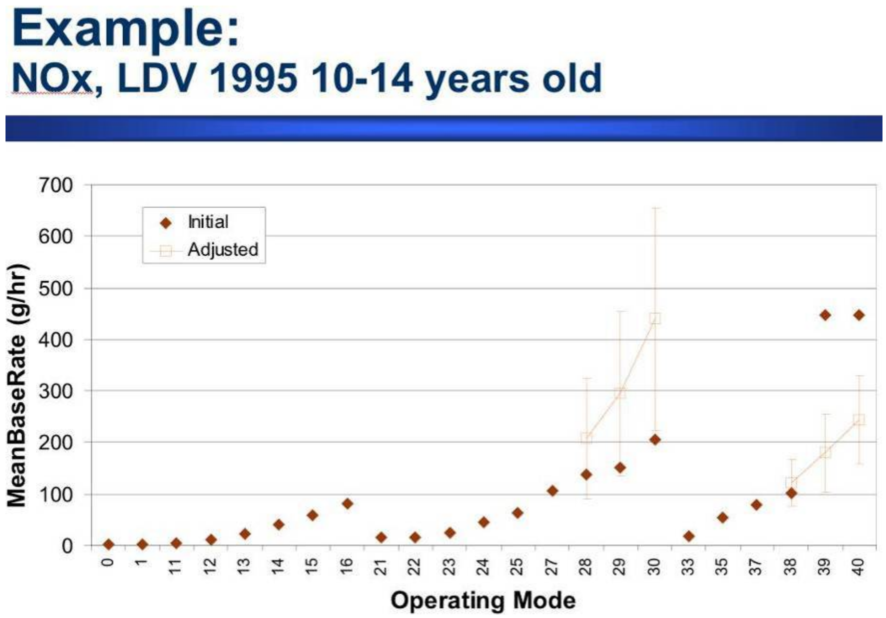


Figure 5-57. Adjusted NOx Emissions



It is commendable to improve the data in high VSP bins. However, as will be shown later in Section 5.3, these adjustment factors could be over-weighting emissions in the higher VSP bins, particularly for vehicles subject to SFTP emission standards.

5.2.6 MOBILE6 and MOVES Running Emissions Comparison versus Age

This section shows four figures prepared by EPA that compare MOBILE6 and MOVES running emissions. There is no EPA comparison of MOBILE6 and MOVES that included starts. The emissions for both models are estimated at the average speed of the FTP, or 19.6 mph. For MOBILE6, this meant running the model with an input speed of 19.6 mph. But for MOVES, this meant having to input the VSP distribution of the FTP into the model for all roadway types, since average speed is not an input to the model.

Figures 5-58 and 5-59 show NOx emissions of model year 2000 and 2010 light-duty vehicles. The 2000 model year are Tier 1 vehicles, and 2010 are Tier 2 vehicles. The emission rates are shown with and without I/M, and show MOBILE6 versus MOVES. For NOx for 2000 model year vehicles, the I/M and non-I/M rates are higher than MOBILE6 (the I/M rates for MOVES are significantly higher). For 2010 model year vehicles, however, the MOVES I/M and non-I/M rates are lower, especially the non-I/M emission rates. The indicated standard emission rates are for 50K miles. The useful life rates are 0.6 and 0.07 g/mi for Tier 1 and Tier 2, respectively.

Figure 5-58. NOx for Model Year 2000 (Tier 1)

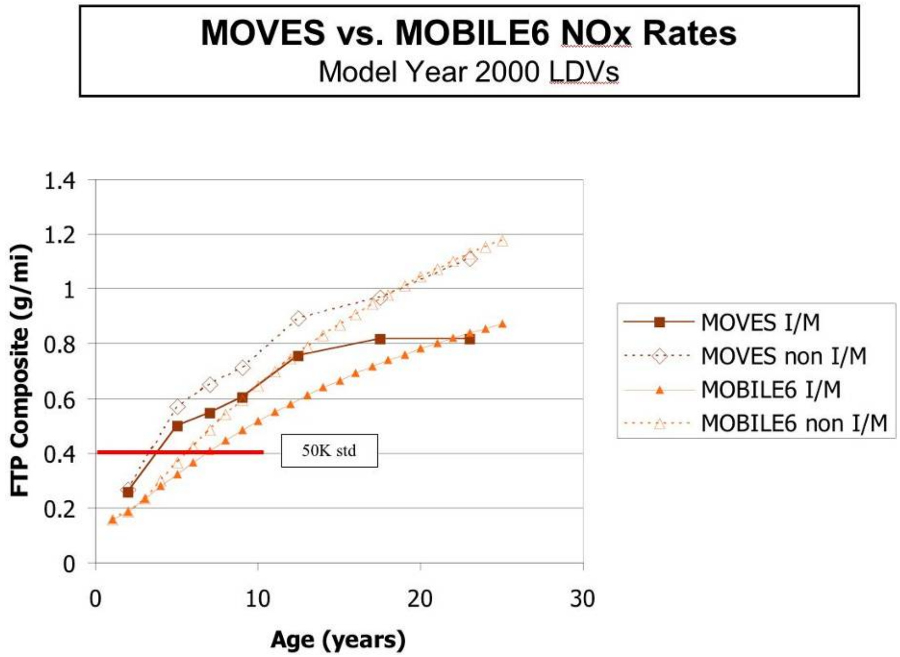
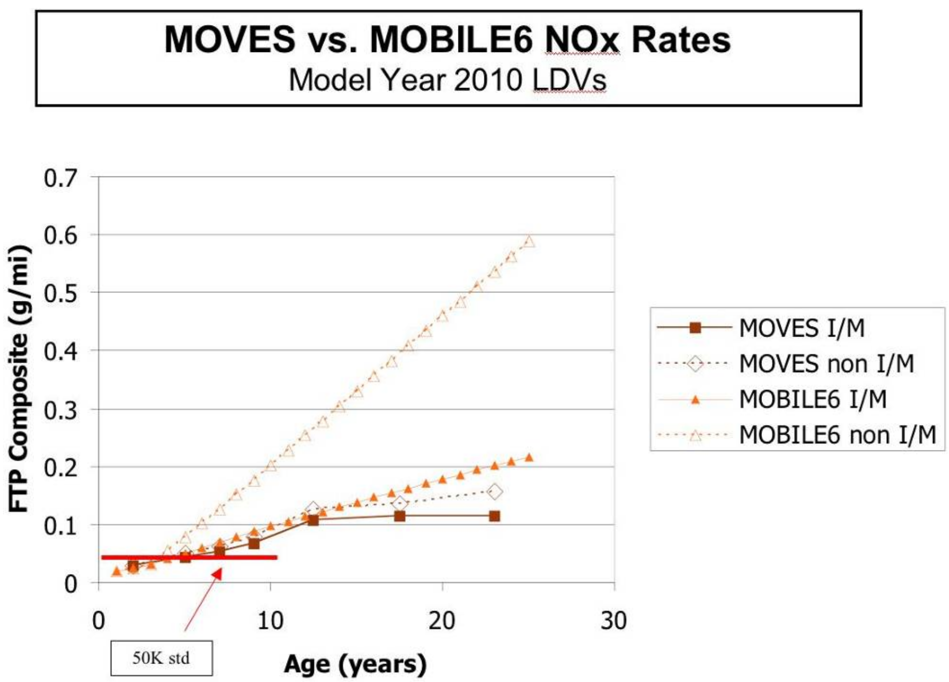


Figure 5-59. NOx for 2010 Model Year (Tier 2)



The HC emission rates are shown in Figures 5-60 and 5-61. For the 2000 model year, the non-I/M rates are somewhat lower for MOVES than for MOBILE6, and the I/M rates for MOVES are significantly higher. But for the 2010 Tier 2 vehicles, the MOVES rates are lower, again, especially the non-I/M emission rates. The 50K emission standards are indicated on the graphs. The NMHC useful life standard for Tier 1 vehicles is 0.31 g/mi and the useful life NMOG standard for Tier 2 vehicles is 0.09 g/mi.

Figure 5-60. THC for 2000 Model Year (Tier 1)

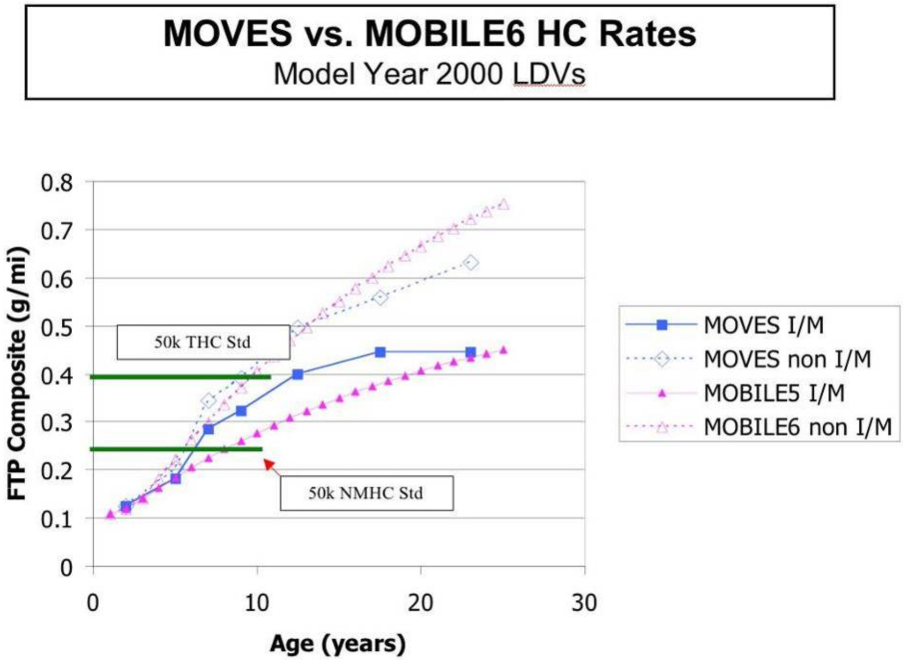
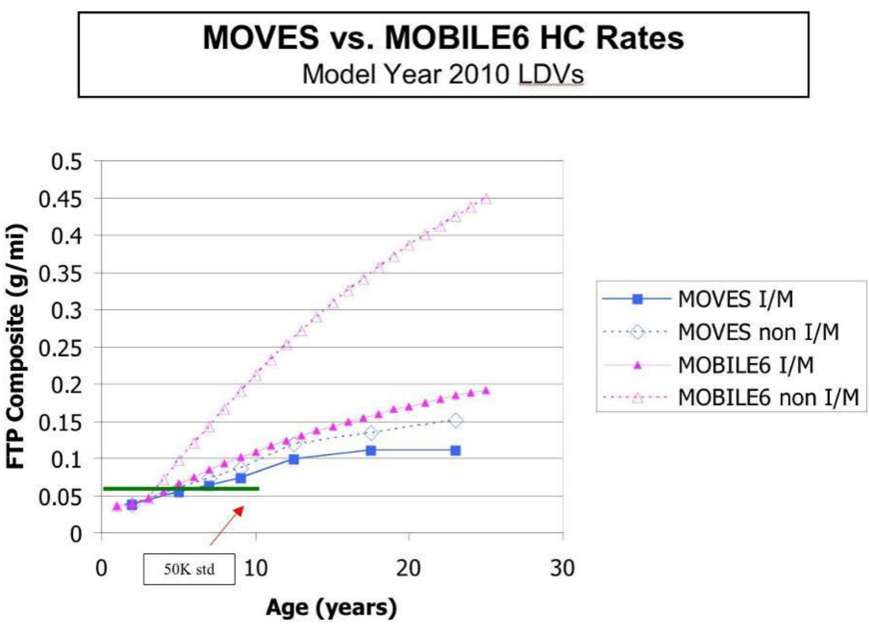


Figure 5-61. THC for 2010 Model Year (Tier 2)



5.3 Distribution of MOVES Emissions by VSP Bin

Since MOVES divides running vehicle emissions by VSP bin, it is important to examine the distribution of running emissions from MOVES by VSP bin. This analysis was conducted for Cook County, for summertime emissions in calendar year 2015, and for three vehicle types – passenger cars, light-duty trucks, and heavy-duty vehicles. The following 10-step process is used to estimate emissions by VSP bin using MOVES:

1. Run MOVES so that all the bundles are saved
2. Extract the source hours operating (SHO), operating mode distribution (OPMODEDISTRIBUTION), and source bin emission rate (SBWEIGHTEDEMISSIONRATE) files from each of the bundles
3. Combine all of the files of each type and eliminate all duplicate lines
4. Generate unique database keys for the three files, based on vehicle class, pollutant, model year, age, road type, link, and op mode.
5. Match up the source operating hours, operating mode, and mean base emission rate lines using these database keys
6. Calculate the weighted emissions via (SHO * opMode * meanBaseRate) for each line
7. Calculate the weighted opMode via (SHO * opMode) for each line
8. Compute the total weighted emissions for each vehicle class, pollutant and opMode
9. Compute the total weighted opMode for each vehicle class, pollutant and opMode
10. Compute the emission and opMode percentages

This analysis does not use any of the temperature or fuel correction factors. However, it does include emissions deterioration due to age. Emissions for all roadway types are combined. To limit the amount of emissions output and the runtime, emissions were estimated for one hour of the day – from 12:00 noon to 1:00 pm.

The results of this analysis are shown in Figure 5-60 through Figure 5-67. Each of these figures shows the emissions and operating mode distributions by VSP bins. The VSP bins are labeled by number and type. Idle and braking are at the bottom of each chart.

Figure 5-62 shows the distribution of THC emissions. The top four bins are 27, 28, 30, and 40. Forty-three percent of THC emissions are in these four bins, and 5% of the vehicle operation (source hours operating) is in these four bins. The largest single operating mode is braking (14%+ of source hours operating), and this bin contains only 4% of emissions.

Figure 5-63 shows the distribution of CO emissions. Fifty-two percent of emissions are in bins 28, 29, 30, and 40 (bin 27 also has significant emissions). Figure 5-64 shows the distribution of NOx emissions. Forty-seven percent of emissions are in these same four bins. Figure 5-65 shows the distribution of $PM_{2.5}$ emissions, where emissions are more evenly distributed by VSP bin, but bin 30 still has the highest emissions.

Figures 5-66 through 5-68 show the same HC, CO, and NOx figures for 2009 Tier 2 vehicles in 2015 (the $PM_{2.5}$ chart is the same for model year 2009 as for the 2015 fleet).

Many bins with significant source hours operating have very little emissions (bins 21-26, for example). Sixty-five percent of THC emissions are in bins 28, 29, 30, and 40. Figure 5-67 shows CO emissions, and 70% of emissions are in bins 28-30 and 40. Figure 5-68 shows that 69% of NOx is in bins 28-30, and bin 38.

Figure 5-62

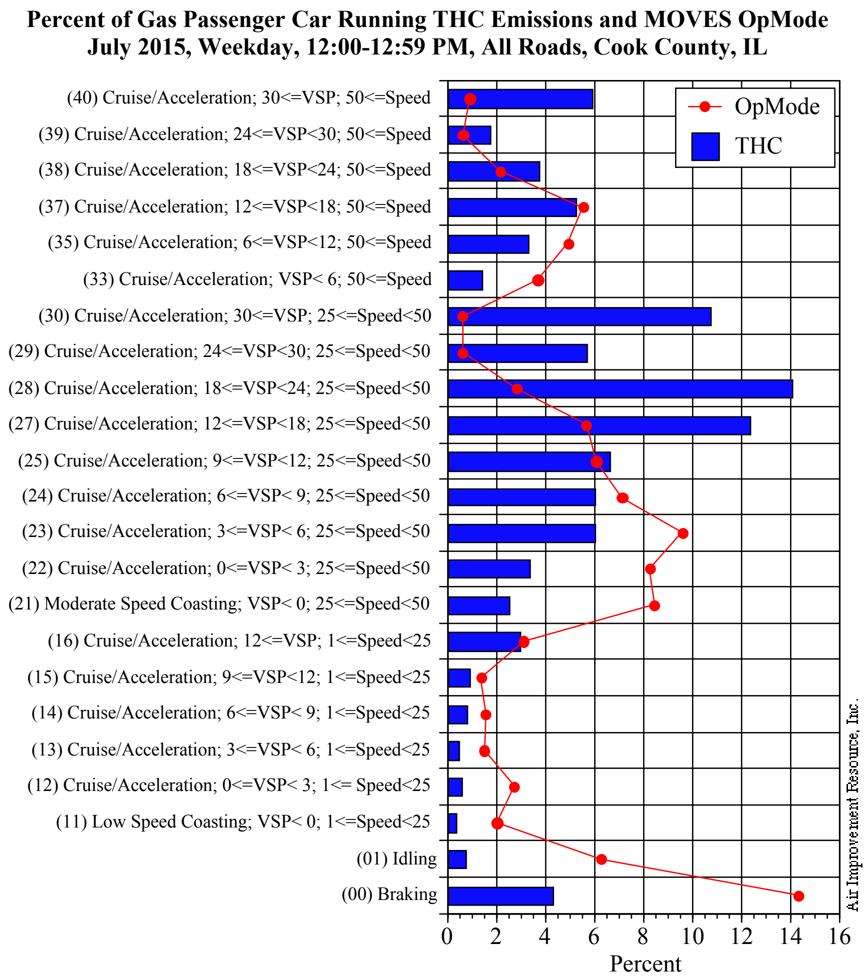


Figure 5-63

Percent of Gas Passenger Car Running CO Emissions and MOVES OpMode July 2015, Weekday, 12:00-12:59 PM, All Roads, Cook County, IL

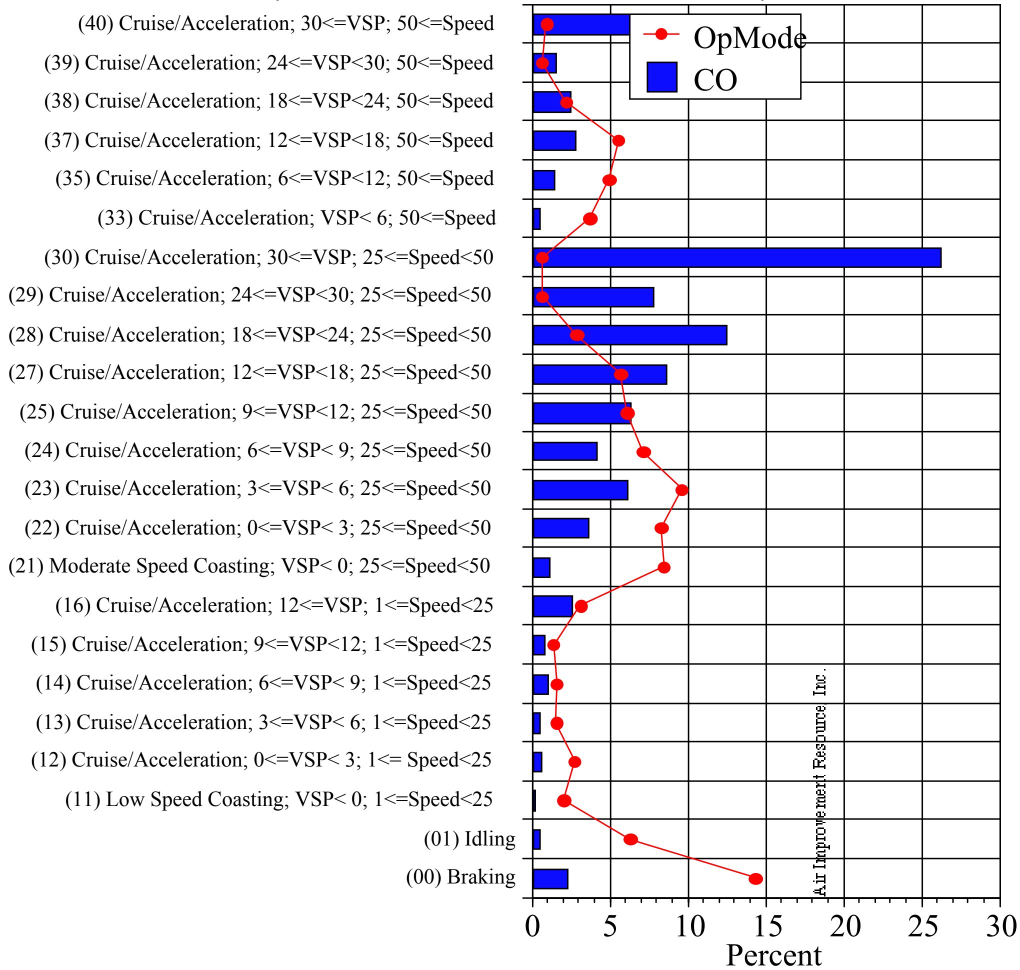


Figure 5-64

Percent of Gas Passenger Car Running NOx Emissions and MOVES OpMode
July 2015, Weekday, 12:00-12:59 PM, All Roads, Cook County, IL

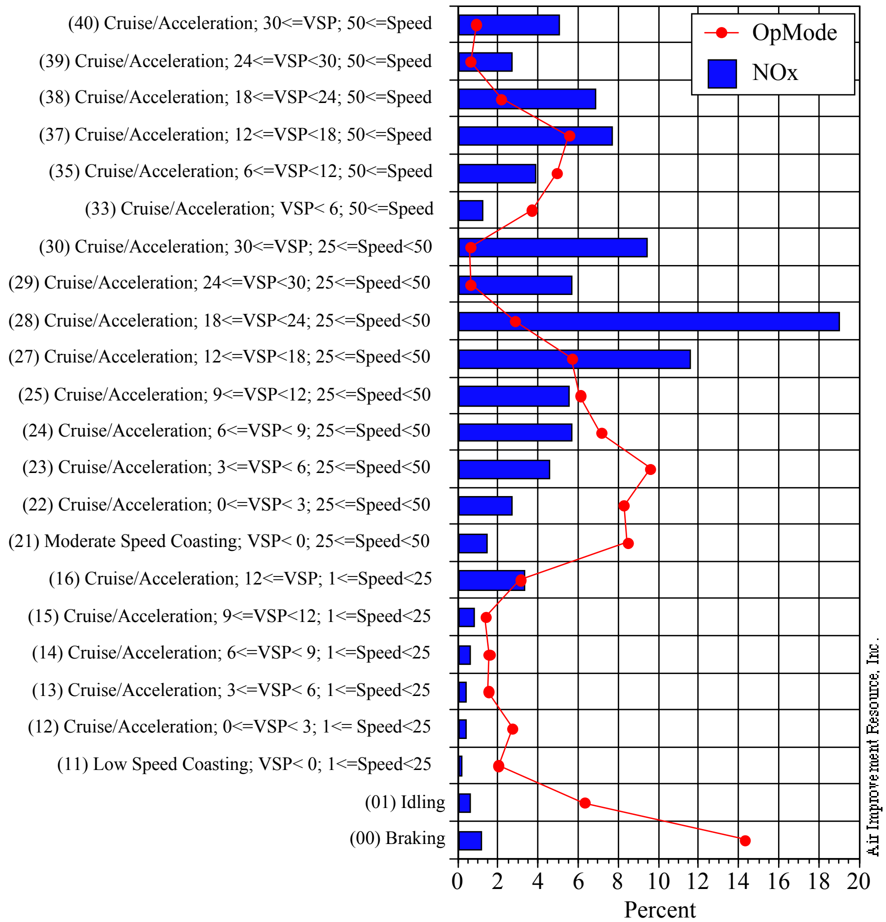


Figure 5-65

Percent of Gas Passenger Car Exhaust $PM_{2.5}$ Emissions and MOVES OpMode
July 2015, Weekday, 12:00-12:59 PM, All Roads, Cook County, IL

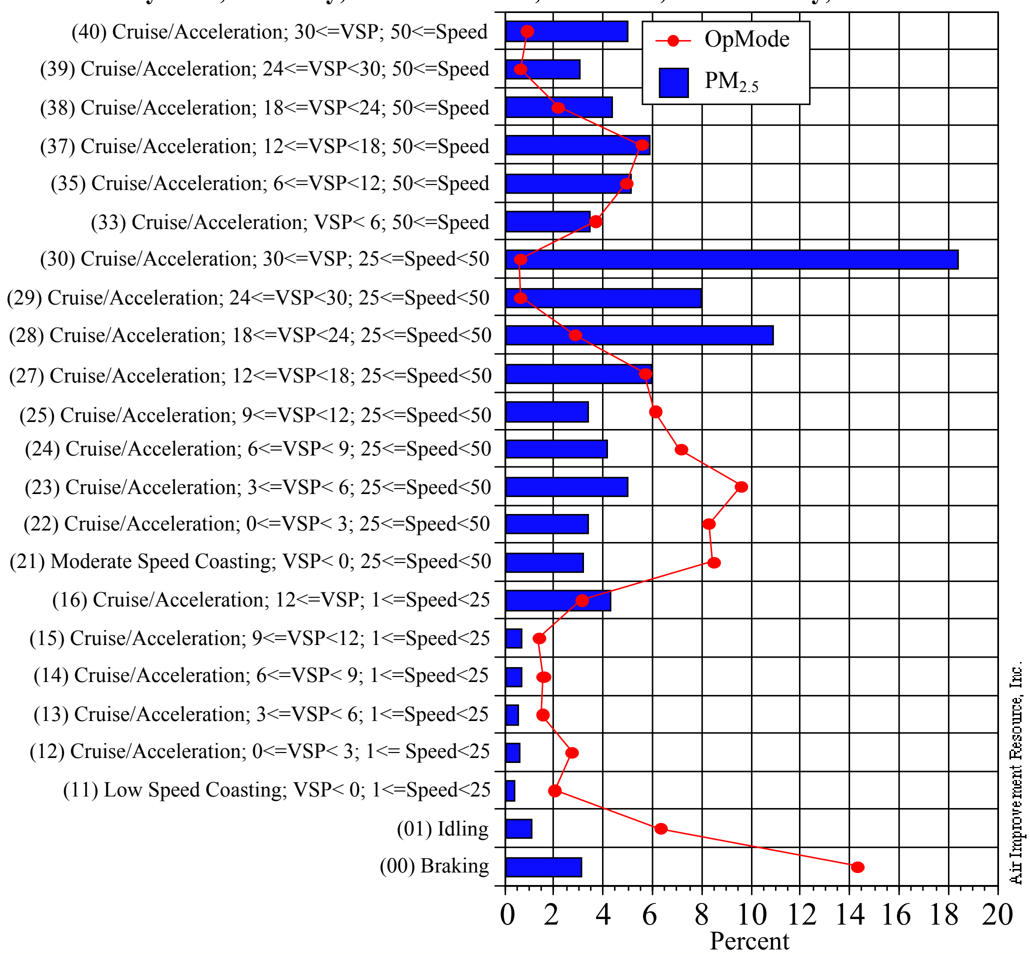


Figure 5-66

Percent of MY2009 Gas Passenger Car Running THC Emissions and MOVES OpMode
July 2015, Weekday, 12:00-12:59 PM, All Roads, Cook County, IL

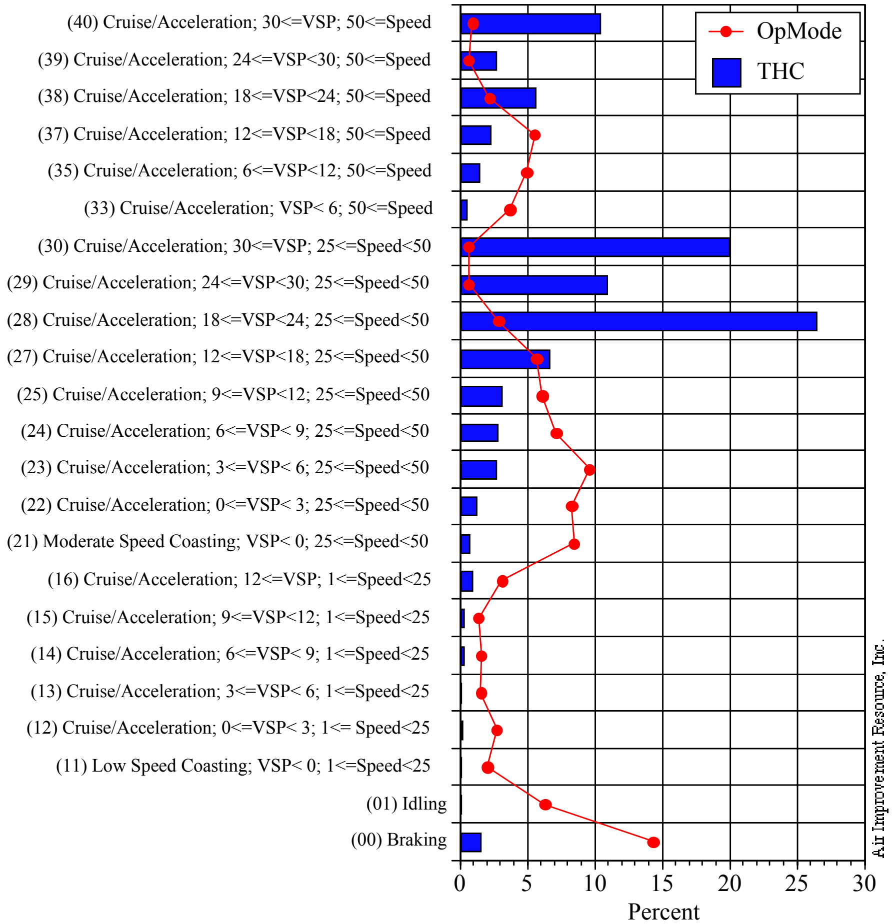


Figure 5-67

Percent of MY2009 Gas Passenger Car Running CO Emissions and MOVES OpMode
July 2015, Weekday, 12:00-12:59 PM, All Roads, Cook County, IL

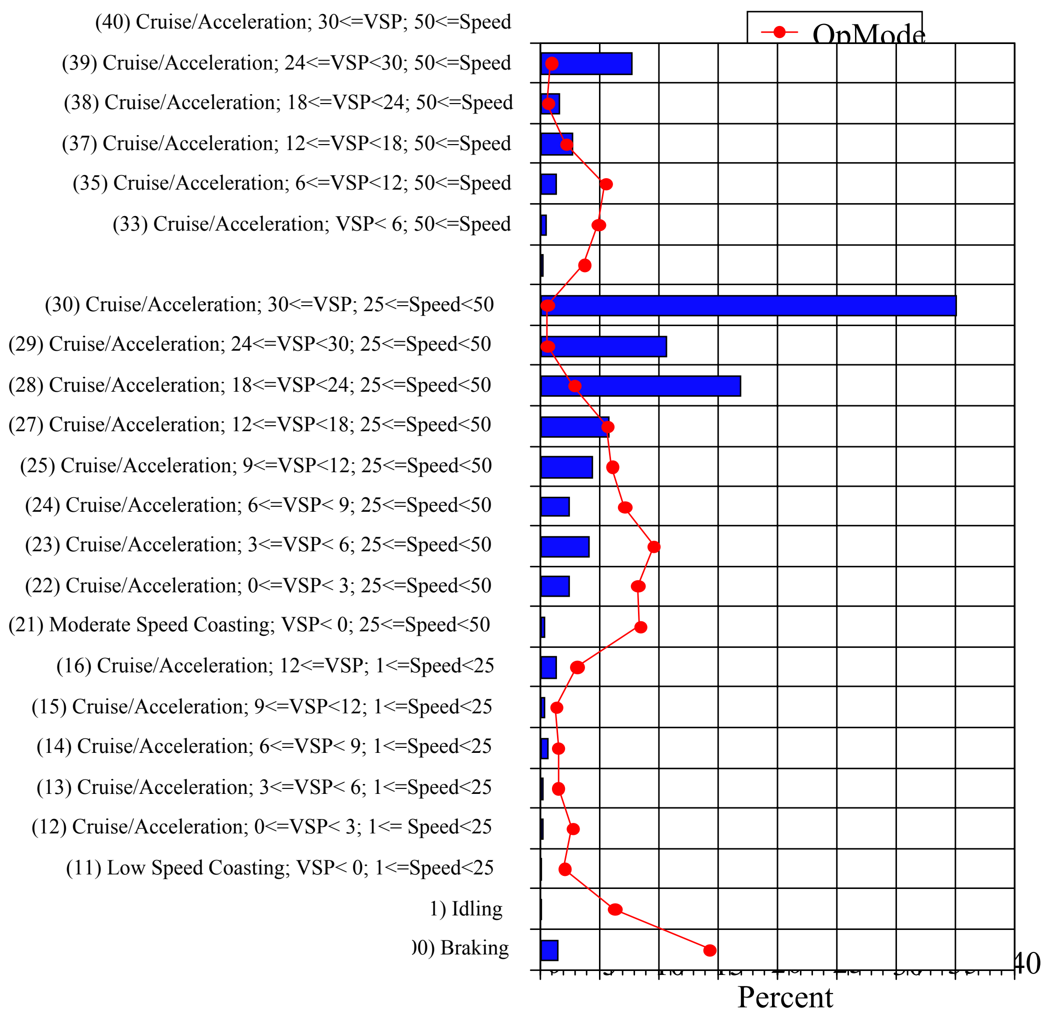
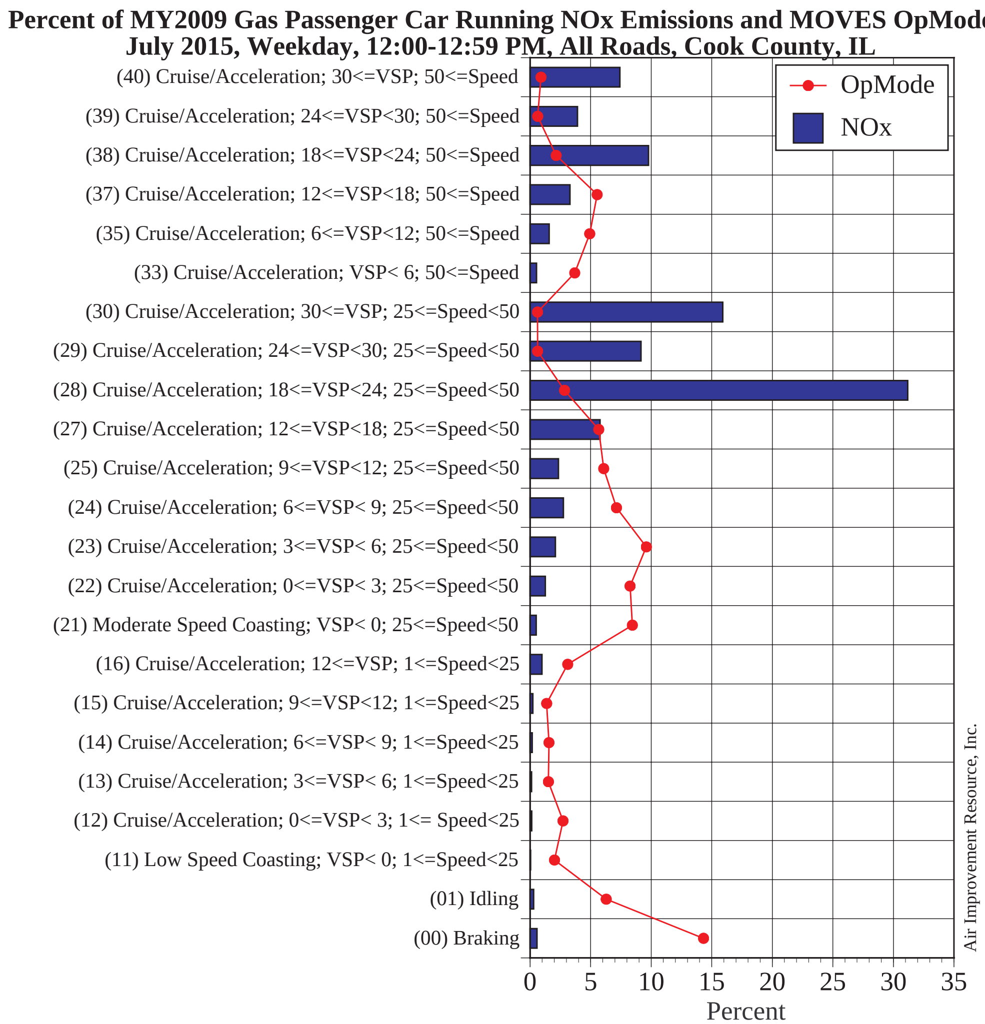


Figure 5-68



Similar charts for LDTs are shown in Appendix D. Charts from the evaluation of heavy-duty diesel vehicles are shown in Chapter 7.

This analysis shows that certain VSP bins produce most of the HC, CO, and NOx emissions. It also shows that this factor becomes more pronounced for the newest, lowest emitting vehicles, and there is very little actual vehicle operation in these modes. Put another way, 60-70% of the running emissions from Tier 2 technology vehicles is based on a relatively small testing program of vehicles with no SFTP controls. As a result, the data and methods that are used to estimate emissions in these higher VSP bins are extremely important, as is the necessity to properly include the benefits of SFTP controls.

5.4 Summary of Recommendations and Concerns on Exhaust Emissions

The recommendations from this analysis of the exhaust emission rate methods follow:

1. The start methodology developed by EPA relies on start emissions data developed by CARB on Tier 0 vehicles. Additional testing of LEV2 and Tier 2 vehicles should be conducted soon to check the ratio of start to cold start emissions.

2. The start deterioration method borrows heavily from the methodology used in MOBILE6. The frequency and emissions characteristics of higher emitters are explicitly modeled in MOBILE6 but are only implicit in MOVES; this could affect the outcome. EPA should determine the fraction of higher emitters in the data underlying MOVES, adjust MOBILE6 to this fraction, and then determine the start deterioration factor from MOBILE6 to use for MOVES.

3. EPA uses a log-linear deterioration model for determining running emissions as a function of vehicle age and VSP bin. However, there is little evidence that emissions as a function of vehicle age increase in a log-linear fashion for Tier 1 and later vehicles. The deterioration model for Tier 1 and later vehicles should be revised to a linear model.

4. MOVES estimates of emissions by VSP bin for Tier 2 light-duty trucks are significantly higher than those for cars, even though their emission standards are identical or nearly identical for these two vehicle categories. The emissions of light-duty trucks should be the same as for cars, if the emission standard is the same (the model will retain an activity differences).

5. The I/M 147 data should not be used to estimate emission reductions from NLEV and Tier 2 vehicles in the higher VSP bins, because there is little operation in the higher VSP bins for this test.

6. EPA's method of estimating emissions for Tier 2 and newer vehicles operating in the higher VSP bins does not currently take into account the full implementation of the SFTP rules. The emission estimates for the higher VSP bins should be modified accordingly.

7. There is also a concern that only a few VSP bins are "driving" the running emissions (especially for the newest technologies). There is little emissions data available to help fill these VSP bins, and hence the results are highly variable. This is a significant vulnerability in the current MOVES model.

8. Estimating the benefits of I/M programs requires more effort. EPA used innovative techniques to evaluate I/M benefits for the reference program (the Arizona I/M program in 2003), but these techniques predicted an I/M benefit in the first three years of vehicle life, which is somewhat questionable.

6.0 Evaporative Emissions – All Vehicles

This section is organized into the following subsections:

- Overview
- Fuel Tank Temperature
- Data Sources
- Permeation Emissions
- Liquid Leaks
- Refueling Emissions
- Non-Fuel Emissions
- Recommendations

The primary FACA workgroup material which underlies the authors' explanation of EPA's method for estimating evaporative emissions in MOVES is the following briefing:

"Approach for Modeling Evaporative Emissions in MOVES", Gururaja, June 26, 2007

6.1 Overview

In MOBILE6 and previous MOBILE models, evaporative emission rates were built around the evaporative test procedures, like the diurnal test, the hot soak test and the running loss test. In MOVES, the evaporative emissions are built around the evaporative processes, instead of the test procedures. These processes are shown in Figure 6-1, and include fuel vapor venting, fuel permeation, liquid leaks refueling vapor, refueling spillage, and non-fuel emissions.

Figure 6-1. Evaporative Concepts

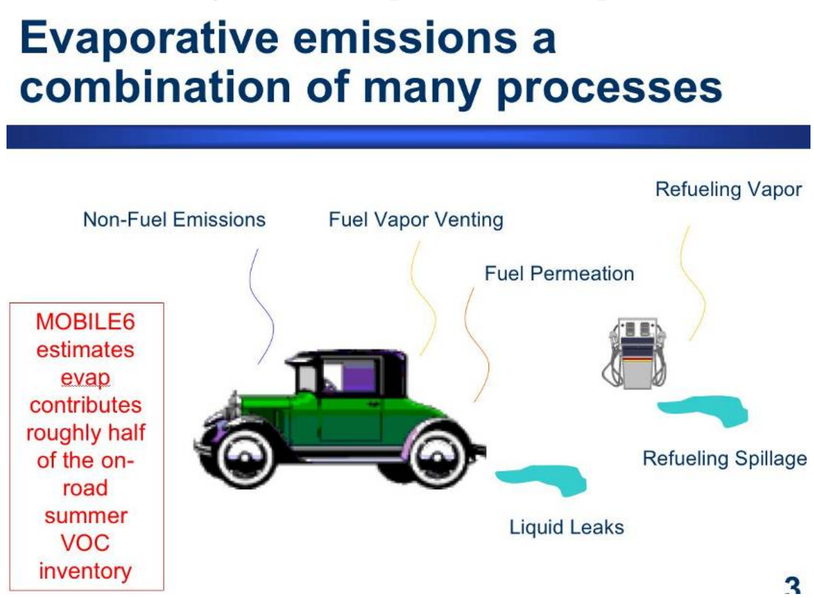
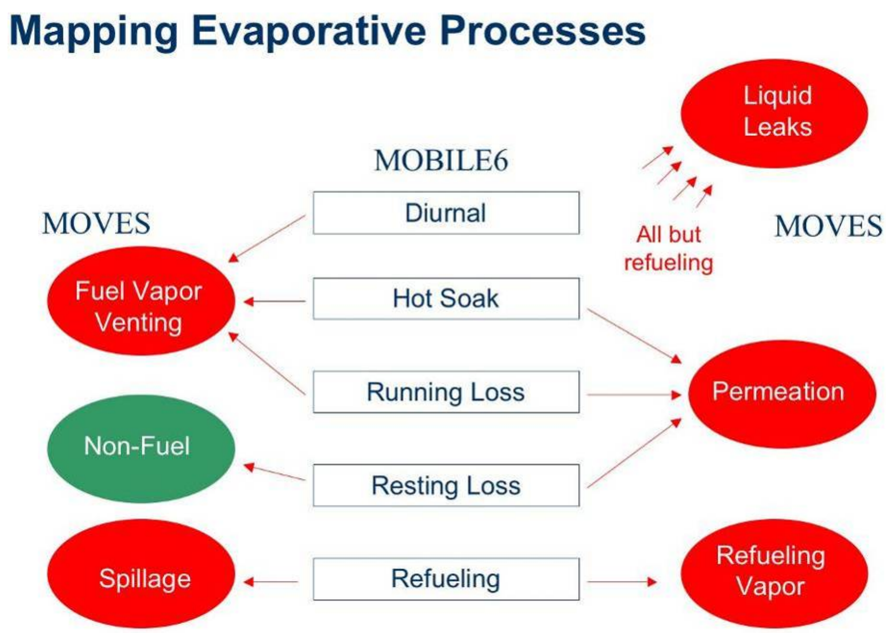


Figure 6-2 shows how the various MOBILE6 processes are mapped into the fundamental processes used in MOVES (it is suggested that a line be added from diurnal down to permeation as well). There were several key objectives in EPA's methodology:

- Use of the most recent data
- Better allocation of evaporative emissions by space and time
- Dynamically consistent activity information
- Explicit treatment of ethanol permeation

Figure 6-2. Evaporative Processes



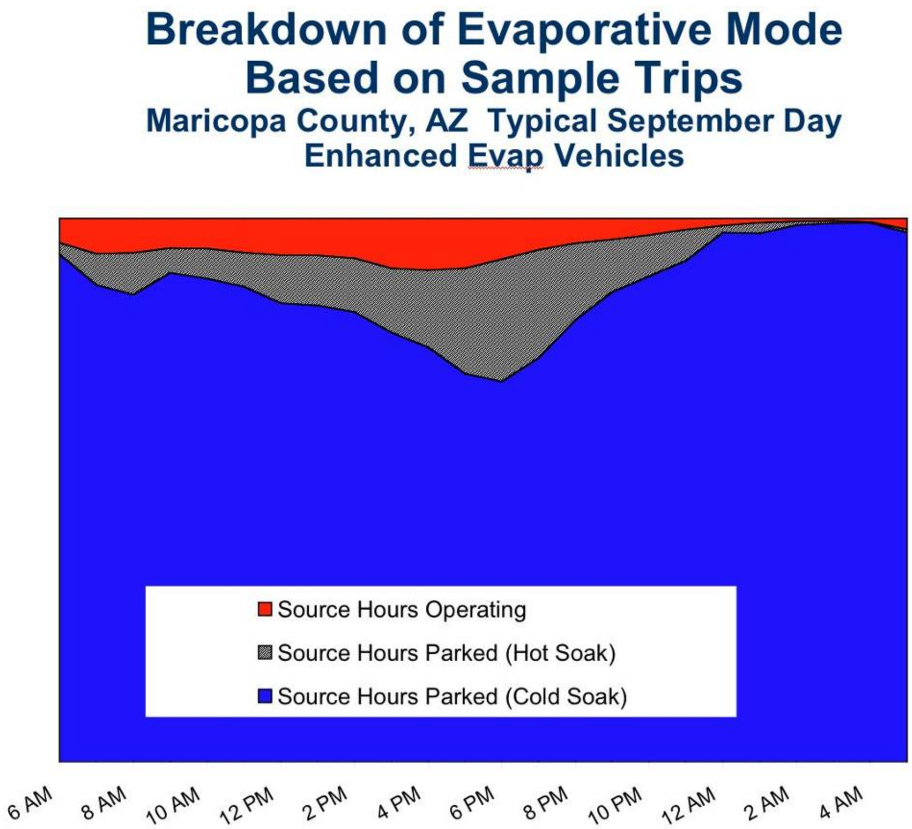
For estimating evaporative emission processes in MOVES, fleet average fuel tank temperature and emission are grouped by hour of the day, by vehicle class (LDV, LDT, HDV<14K, HDV>14K), and by model year. EPA is not attempting to model individual vehicle, second-by-second emissions or canister loading and purge cycles. EPA's approach follows many recommendations made by Haskew and Associates.[13] In this approach, the emissions are time-based instead of distance-based. The time basis for activity is "source hours parked", which is split into cold soak and hot soak modes, and "source hours operating", for the running mode. This activity is allocated independently of vehicle miles traveled. The distribution of hours parked (when, how long) is calculated within MOVES via sample trip data.

An example of this trip data is illustrated in Figure 6-3, which shows the fraction of source hours operating, source hours parked (hot soak) and source hours parked (cold soak) for enhanced evaporative vehicles in Maricopa County, AZ on a typical September day (in this plot, the y axis is percent). What is striking from this plot is that regardless of

[13] "A New Approach to Modeling vehicle On-Road Vehicle Evaporative Emissions", Haskew and Associates for EPA, June 2005.

time of day, most of the time is in parked condition; comparatively little hours are actually for vehicles operating. Therefore, the parked condition emissions should drive the overall evaporative inventories.

Figure 6-3. Example of Evaporative Activity



6.2 Fuel Tank Temperature

In estimating both permeation and fuel venting emissions using this method, knowledge of the fuel tank temperature is central. It is the main driver for both permeation and fuel vapor venting emissions and is a function of the day-to-day vehicle operating patterns. In MOVES, real-world fuel tank temperature values are estimated based on instrumented vehicle sample trip data averaged by: hour of day, mode (cold soak, hot soak, operating), and by vehicle/evaporative certification group (pre-enhanced LDV, pre-enhanced LDT, enhanced and later).

The theory utilized for predicting tank temperature during cold soaks is as follows:

$$dT_{tank}/dt = k\ (T_{air} - T_{tank})$$

Where:

dT_{tank}/dt = rate of change of tank temperature with time
K = constant
T_{air} = air temperature
T_{tank} = tank fuel temperature

The rate of tank temperature change is directly proportional to the difference in the ambient and tank temperatures. EPA experimentally determined “k” to match the Mobile Source Operations Division (MSOD) data for 60° to 84°F, 72° to 96°F, and 82° to 106°F diurnal temperature cycles. The value determined for “k” is 1.4.

To determine the change in fuel tank temperature during the “source hours operating” evaporative emissions mode, the MOVES model estimates temperature at the beginning and ending of each vehicle trip. The change in temperature is determined based on analysis of test data performed at an ambient temperature of 95°F. For pre-enhanced evaporative vehicles, this comes from the CRC E-35 data, for enhanced evaporative vehicles it comes from the certification temperature profiles, where the testing utilizes the combined Urban Dynamometer Driving Schedule (UDDS) and New York City Cycle (NYCC). [14] From these data, the increase in tank temperature for pre-enhanced evaporative LDVs is 35°F, for pre-enhanced LDTs the increase is 29°F, and for all enhanced evaporative (essentially 1996 model year and later) it is 24°F. The same value (24°F) is used for all cars and LDTs vehicles certified to lower Tier 2 evaporative standards. The maximum fuel tank temperature in the model is limited to 140°F. Since fuel volatility plays a major role in overall evaporative emissions, the MOVES model should consider using a volatility effects model that includes more parameters than just RVP – such as T5, T10, and T20.

The change in temperature is scaled to the key-on tank temperature using the following expression and indicates that the increase or change in tank temperature is lower as starting temperatures move higher: [15]

$$\Delta T_{tank} = 0.352 * (95-T_{tank,\ key\ on}) + \Delta T_{tank,\ 95F}$$

The key-off temperature is then estimated assuming a linear increase in temperature during the trip. The average fuel tank temperature during the operating mode is the average of the key off and key on temperatures. The fuel tank temperature during the hot soak mode is determined by the same basic method as the cold soak temperature.

As indicated earlier, the fuel tank temperature profiles discussed above are based on a log of trips from a group of instrumented sample vehicles. The values are used to estimate fractions of vehicles that are operating, hot soaking and cold soaking for each hour, on weekends and weekdays.

Figure 6-4 shows the fuel tank temperature profile for a vehicle with different control technologies (pre-enhanced car, pre-enhanced LDT, and enhanced car and LDT) operated in Washtenaw County (Michigan).

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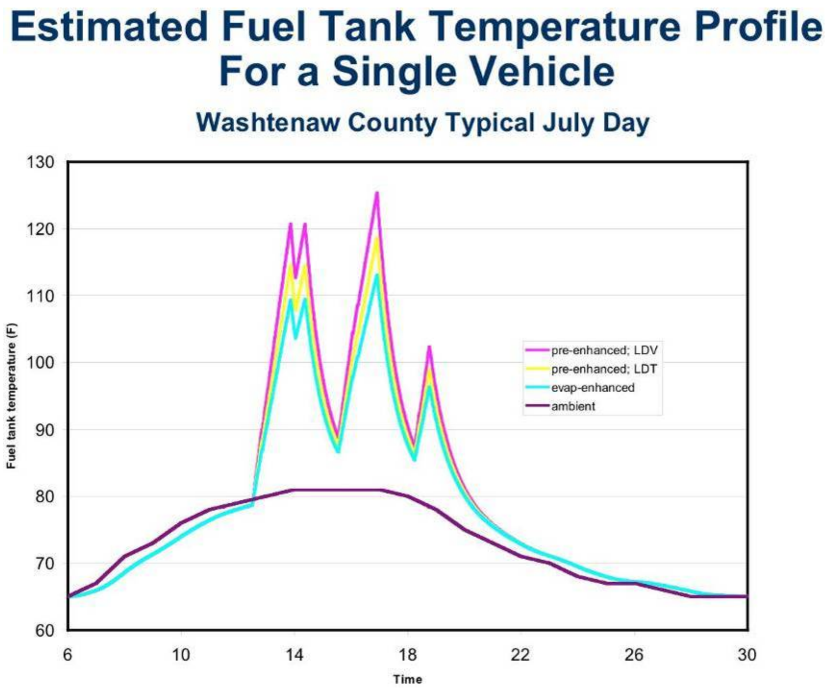
[14] CRC E-35, Report No. 612, “Running Losses from In-Use Vehicles”, Harold Haskew and Associates, February 1999.

[15] T. Cam, K. Cullen, and S. L. Baldus, Running Loss Temperature Profile, *SAE. 930078*, Society of Automotive Engineers, Warrendale, Pa 1993.

Figure 6-4. Tank Temperature Profile



6.3 Evaporative Emission Data Sources

EPA is using the following data sources for emissions:

Historical EPA testing (MSOD) - There are hundreds of tests on older vehicles in this database. Many of the tests are non-real-time testing hot soak and diurnal utilizing a 60 to 84°F temperature diurnal. But there also are a number of tests utilizing real-time SHED testing and diurnal temperatures of 60 to 84°F, 72 to 96°F, and 82 to 102°F. There are also running loss tests on vehicles utilizing multiple NYCC cycles.

CRC E-9 Real Time Diurnal Study – Conducted in 1996, there are 151 vehicles in this study: 51 from 1971 through 1977, 50 from 1980 through 1985, and 50 from 1986 through 1991.[16] Odometers range from 39,000 miles to 439,000 miles. Diurnal temperatures used are 72 to 96°F.

CRC E-35 Running Loss Study – Conducted in 1997, there are 150 vehicles in this study, the ambient temperature used during testing is 95°F, and the fuel Reid Vapor Pressure (RVP) is 6.8 psi.[17]

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[16] CRC Report No. 610, Project E-9. "Measurement of Diurnal Emissions from In-Use Vehicles", September 1998.

[17] CRC Report No. 611. Project E-35, "Measurement of Running Loss Emissions from In-Use Vehicles, Automotive Testing Laboratories, February 1998.

CRC E-41 Late Model In-Use Evaporative Emissions Hot Soak Study – Conducted in 1998, there are 50 vehicles in this study (30 passenger cars and 20 LDTs), from 1992 to 1997 model years.[18] The diurnal temperatures are 72 to 96°F, and running loss tests are conducted at 95°F. The running loss driving schedule is LA4-NYCC-NYCC-LA4. Fuel RVP is 6.5 psi.

EPA Compliance data – There are 77 vehicles tested in this program, utilizing a 2-day real-time diurnal test and a hot soak test. The RVP used is 8.8 psi. The diurnal testing used both a 72 to 96°F temperature and 65 to 105°F.

CRC E-65 and CRC E-65.3 – This study measured and reported the permeation emissions of ten different vehicle fuel systems with three different gasolines. Model years of the test vehicles ranged from 1978 to 2001. Emission measurements consisted of steady state measurements at 105°F and 85°F, and two-day diurnal measurements using the California test procedure (from 65° to 105°F). This project vented the tank vapor emissions outside the VT-SHED to allow an independent measurement of permeation.[19] E-65-3 added three new vehicles: a "near-zero" evaporative control vehicle, a "zero" evaporative control vehicle, and a flexible fuel vehicle (FFV). Six fuels are tested in this program: non-oxygenated base fuel, 5.7% ethanol by volume, 5.7% ethanol with increased aromatics, 10% ethanol by volume, 20% ethanol by volume, and E85 (85% ethanol by volume). [20]

CRC E-77 and E-77-2 – E-77 was a pilot study to determine if the mechanisms of evaporative emissions could be measured effectively, where the mechanisms were defined as leaks, diurnal displacement vapors, and permeation. [21] Nine vehicles were tested: three vehicles in the 1990-1995 model year pre-enhanced evaporative vehicles, five 1996-2000 enhanced evaporative vehicles, and one 2007 model year vehicle certified to Tier 2 evaporative emission standards. Vehicles were tested on both 7 and 9 RVP fuel. In E-77-2, eight of the nine vehicles were selected for additional evaluation on five gasoline blends, including three levels of ethanol (zero, 10 volume percent, and 20 volume percent). In addition, two of the vehicles were given a limited evaluation with implanted small leaks in the evaporative system. The evaporative testing consisted of four parts: a static permeation rate measurement at 86°F, a dynamic running loss permeation and canister loss measurement at 86°F, a hot soak following the dynamic test, and a two-day diurnal permeation and canister loss test.

6.4 Permeation and Soak Emissions

[18] CRC Project No. 622, Project E41-1 and E41-2, "Real World Evaporative Testing of Late-Model In-Use Vehicles", October, 1999.

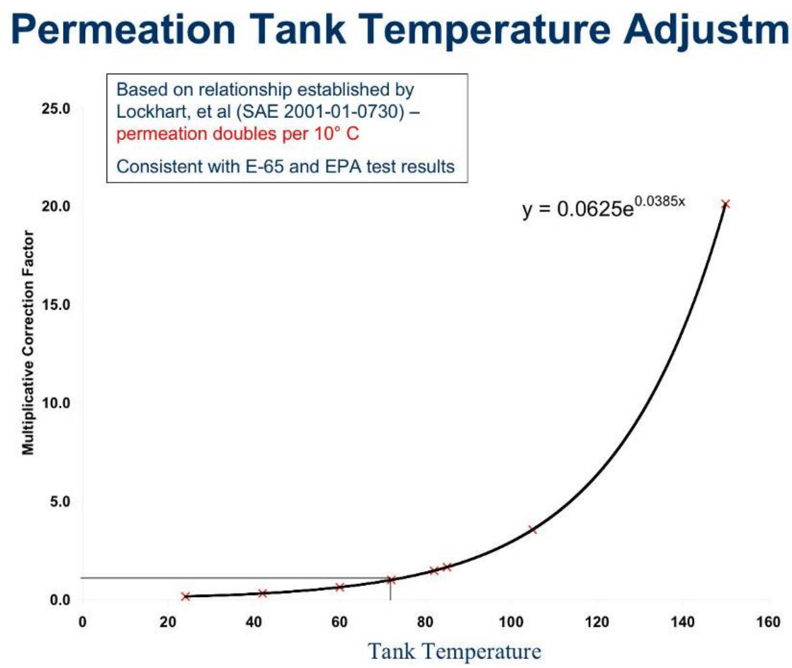
[19] CRC Project No. E-65, "Fuel Permeation From Automotive Systems", Final Report, Harold Haskew and Associates, September 2004.

[20] CRC Report No. E-65-3, "Fuel Permeation from Automotive Systems: E0, E6, E10, E20 and E85", Final Report, Harold Haskew and Associates, December 2006.

[21] CRC Report No. E-77, "Vehicle Evaporative Emission Mechanisms: A Pilot Study", Harold Haskew and Associates, June 24, 2008.

The base permeation emission rates are developed at a temperature of 72°F, and approximately represent the average emissions during the last six hours of the diurnal test.[22] The permeation rates are stratified by model year group and age group. The only adjustments to these data are for fuel temperature and the presence of ethanol. The fuel tank temperature adjustment is shown in Figure 6-5. Maximum fuel tank temperature in the model is limited to 140°F.

Figure 6-5. Tank Temperature Adjustment



For permeation emissions of future vehicles (Tier 2 evaporative), EPA makes two assumptions:

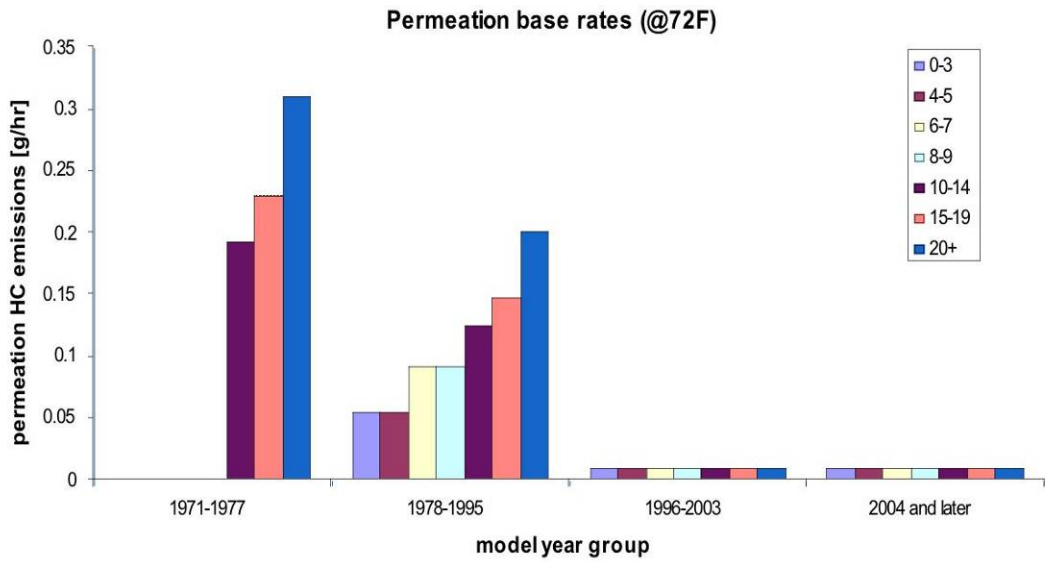
- There is no deterioration of permeation emissions on enhanced evaporative and later vehicles
- There is no reduction in permeation emissions for vehicles certified to Tier 2 or LEV II evaporative standards. The reductions in emissions for these standards are entirely attributed to reductions in fuel vapor venting. However, EPA indicated it will re-evaluate this latter assumption as new data became available (such as the data being generated in the ongoing CRC E-77 test program)

Figure 6-6 shows the permeation base rates at 72°F for vehicles of different model year groups and ages. As expected, the 1996 and later permeation emissions are much lower than the earlier model year groups because of the enhanced evaporative testing

[22] The testing temperatures in the last 6 hours of the federal test procedure decline from about 75.8°F to 72°F (an average of 73.7°F for last 6 hours), and EPA assumes that all the emissions during this time period of the test are permeation emissions and can be associated with a test temperature of 72°F. The E-77-2 testing is ongoing.

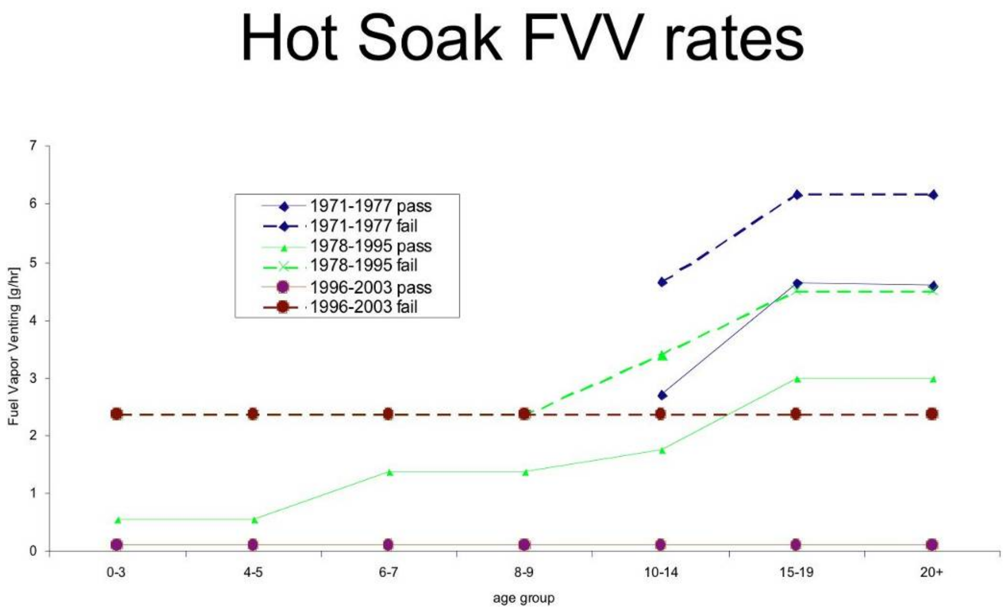
procedures. However, the authors feel that further investigation should be made of the fact that Tier 2 vehicles are not estimated to have lower permeation emissions than enhanced evaporative vehicles even though they have lower standards. In addition, vehicles in California states designed to the PZEV zero evaporative standard should have still lower permeation emissions; those vehicles do not appear to be represented in Figure 6-6 (there are PZEVs in non-California states as well).

Figure 6-6. Permeation Emission Rates for Different Model Year Groups and Ages



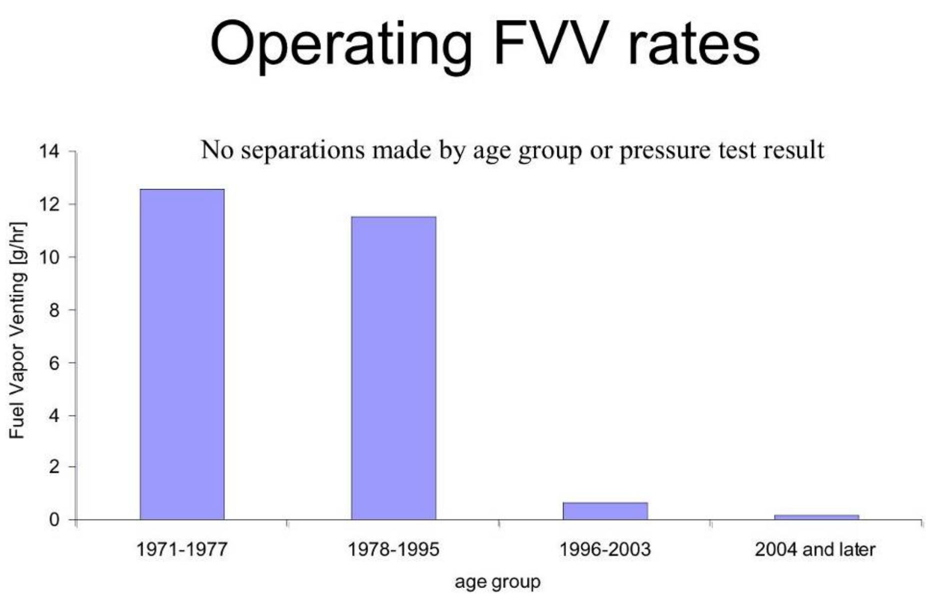
Fuel vapor venting emissions from hot soaks are estimated as the difference in total HC and permeation emissions for the hot soak test, and the operating fuel vapor venting emissions are estimated as the difference in total HC and permeation emissions for the running loss test. The hot soak fuel vapor venting emission rates in g/hr are shown in Figure 6-7. These rates are shown for both "passing" and "failing" vehicles, where pass and fail refers to whether the vehicle fails a pressure check. The hot soak emission rates for passing vehicles are very close to zero for those certified to enhanced evaporative emissions standards, but are about 2.3 g/hr for vehicles with a vapor leak.

Figure 6-7. Hot Soak Fuel Vapor Venting Emission Rates



Fuel vapor venting emission rates for the operating mode are shown in Figure 6-8. Unlike the hot soak vapor venting rates, EPA did not separate the operating fuel vapor venting rates by age group or pressure test result.

Figure 6-8. Operating Fuel Vapor Venting Emission Rates

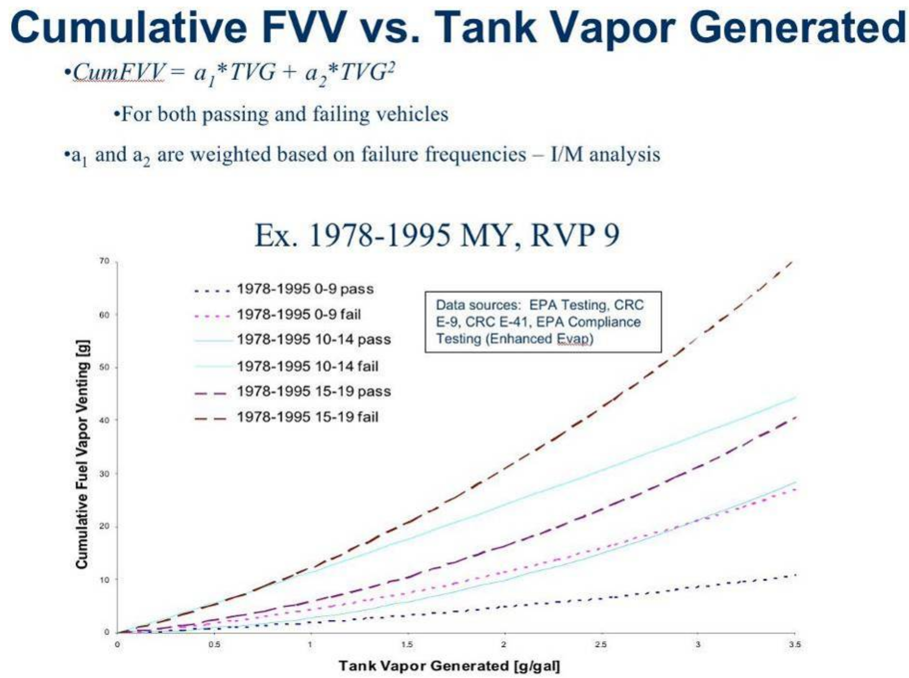


The cold soak emissions are based on the Reddy equation, which essentially expresses the tank vapor generated (TVG) as a function of the change in fuel tank temperature, fuel RVP and presence of ethanol.[23] An example of these emission rates is shown in Figure 6-

[23] R.S. Reddy, "Prediction of Fuel Vapor Generation from a Vehicle Fuel Tank as a Function of Fuel RVP and Temperature", *SAE 892089,* 1989.

9. The horizontal axis shows tank vapor generated in g/gal, and the vertical axis presents cumulative fuel vapor venting in grams. This plot is for 1978-1995 vehicles, with 9 RVP fuel.

Figure 6-9. Fuel Vapor Vented versus Tank Vapor

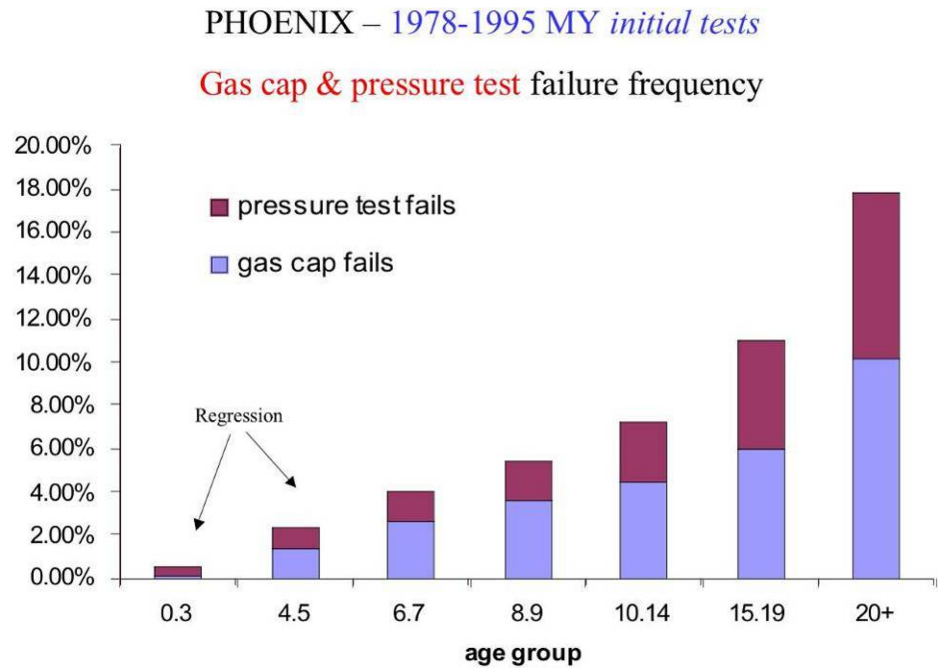


EPA believes that the limited data requires weighting pressure test pass/fail strata outside of MOVES to approximate representative emission rates. Therefore, EPA analyzed I/M data to determine pressure failure frequency. Three items are examined: gas cap missing, OBD malfunction indicators, and pressure test results. All three of these fails are assumed to have the same effect and tank vapor emissions as a pressure test failure. Since there is little overlap between gas cap, fill-pipe pressure tests, and OBD failures, all three failure types are combined for vehicles with OBD (1996 and later vehicles), and the first two failure types are combined for vehicles without OBD. The OBD codes used to determine evaporative failures are P0440, P0442, P0445, P0446, and P0447 for all vehicle makes, and additionally the P1456 and P1457 codes for Honda and Acura vehicles. Vehicles that had one or more of these faults were flagged as failing vehicles.

Combined gas cap and pressure test failures for initial tests are shown in Figure 6-10 from the Phoenix I/M program for 1978-1995 vehicles (not subject to OBD requirements). The rates start very small, but build to 17% or so for vehicles that are 20+ years old.

The fail rates in Figure 6-10 are the initial tests and do not represent the fail rates of the overall I/M program, since many vehicles are repaired. To estimate overall fail rates in the I/M program, EPA averages the before and after I/M fail rates for each model year and age group.

Figure 6-10. Gas Cap and Pressure Test Failure Rates versus Age



The initial evaporative test failure frequency in Phoenix is not a reflection of the failure rate in a non-I/M area, since presumably some vehicles being tested in Phoenix have already been identified and fixed by the I/M program. To determine failure rates in non-I/M areas, EPA examined failure rates in a restricted Phoenix sample of vehicles with license plates from states that do not have an I/M program.

The Phoenix non-I/M data are augmented with data from three other I/M programs. North Carolina decentralized I/M data are used to determine the non-I/M failure frequencies for OBD tests. Vehicles in North Carolina were flagged as non-I/M tests if they were tested before the official start of the program, in a new I/M county and were registered in that same county, or in a new I/M county and were registered in a non-I/M county or a county that did not start I/M within the last year. These data were compared to failure frequencies for vehicles in older I/M areas, where vehicles had been previously tested. From the North Carolina data, the average ratio of non-I/M to I/M OBD fail frequencies is 1.6. Colorado I/M data are used to determine non-I/M failure frequencies for gas cap tests. Vehicles were flagged as non-I/M tests if their registration state was a 100% non-I/M state, or if the registration county was a non-I/M county of Colorado. EPA compared the fail rates of the flagged vehicles to those of the full tested fleet. The ratio of these two frequencies was then applied to the Phoenix gas cap failure frequencies to determine non-I/M failure frequencies. For Colorado, the average ratio of non-I/M to I/M gas cap failure frequencies is 2.2.

Not all I/M programs contain the same evaporative tests. For example, the Phoenix I/M program performs a gas cap check, a fuel inlet pressure test, and the OBD check for OBD-equipped vehicles. EPA designated the Phoenix I/M program as the Reference I/M program, and developed an I/M scaling factor from the Phoenix and other I/M data. The

I/M scaling factor lets MOVES interpolate and extrapolate the non-I/M emission rates and I/M emission rates depending on the characteristics of the I/M program in each county.

The Phoenix I/M program was given a factor of 1.0, and non-I/M areas were given an I/M factor of zero. I/M factors for other areas are interpolated or extrapolated from these two levels. For example, the Tucson I/M program is an annual I/M program, although it does not include the fuel inlet pressure test, so the Tucson program was used to determine the I/M factor for an annual test versus the biennial test. Tucson's failure frequency for the combined OBD and gas cap test is lower than Phoenix (presumably because the test is annual), so it has higher I/M factor than Phoenix (about 1.2). Colorado has a biennial test like the Phoenix test, but does not require OBD compliance, so its evaporative failure rates are lower than for Phoenix, resulting in a lower I/M factor of about 0.42 compared to Phoenix.

EPA assumes that there is no deterioration of fuel vapor emissions on passing vehicles certified to enhanced evaporative and later emissions standards, and that the reductions in fuel vapor emissions from the LEV II and Mobile Source Air Toxics (MSAT) emission standards are based on the ratio to the standards. For example, the MSAT requirements reduce the evaporative standard from 2.0 g/test to 0.5 g/test, a 75% reduction. The 75% reduction in emissions is applied only to fuel vapor venting emissions, and not to permeation emissions.

Failing vehicles are assumed to have the same fuel vapor venting emissions as pre-enhanced vehicles. The failure rates are based on OBD evaporative fail rates.

6.5 Liquid Leaks

EPA is assuming that for future vehicles, gross leak emissions and the frequency of leaks are not affected by either evaporative emission standards or by OBD. To estimate liquid leaks during the cold soak, EPA segregated vehicles in the database, which were specified as liquid leakers, and defined leak emissions as the Total HC minus permeation minus fuel vapor venting (where the latter two terms are estimated as discussed in Sections 6.2 and 6.4). Hot soak and running loss leakers were identified as those having emissions greater than 10 g/hr, and these rates were estimated the same way as for the cold soak leaks. The following are the resulting liquid leak emission rates:

- Cold soak: 9.85 g/hr
- Hot soak: 19.0 g/hr
- Operating: 178 g/hr

The reason that hot soak and operating emissions can be higher than cold soak emissions is related to the increased pressures that the fueling system experiences during these modes which can result in faster leaks for a given leak orifice size.

EPA developed a regression of leak frequencies versus age from the API/CRC data for use in MOVES. The leak frequencies used in MOVES are shown in Table 6-1.

Table 6-1. Leak Frequencies versus Age	
Age	Percent of Leakers in Fleet
0-9 years	0.09%
10-14 years	0.25%
15-19 years	0.77%
20+ years	2.38%

The API/CRC study did not find any leaking vehicles in the 0-9 year age group, and yet, EPA has estimated the frequency at about 0.1% (one in one thousand). The sensitivity runs in Chapter 10 of this report show that even a frequency this low has a significant impact on evaporative emissions from the new vehicle fleet.

6.6 Refueling Emissions

These emission rates are based on the MOBILE6 method, and are split into spillage and vapor. Refueling emissions rates are based on fuel consumption and estimated for uncontrolled vehicles at 5.26 grams/gallon of dispensed fuel. All new gasoline vehicles are now equipped with onboard vapor recovery with an assumed effectiveness of control of 98%. The refueling rates also include spillage, which is estimated at 0.31 grams/gallon.[24] Spillage rates may need to be reexamined in the future if average fuel tank sizes change.

6.7 Non-Fuel Emissions

Non-fuel emissions are a combination of wiper fluid, tires, a/c refrigerant, upholstery and adhesives. These are not broken out separately, but are included in the total HC measurements.

6.8 Summary of Recommendations

The following are the recommendations from this section:

1. Permeation emissions are not reduced for Tier 2 and MSAT evaporative emission standards. Permeation emissions should be reduced in addition to reducing the tank vapor emissions. This should be evaluated as soon as possible with testing (this factor is being examined in the CRC E-77 program).
2. There do not appear to be evaporative emissions estimates for PZEVs in MOVES, which are sold in states with the California and Federal emission standards (in other words, all states).
3. Leak frequencies as a function of vehicle age should be updated with new test data as soon as possible. The existing data show that there are no leaking vehicles in the first nine years of vehicle age. Based on the existing data, the leak frequency should be reduced for these vehicles to zero.

[24] Refueling emissions spreadsheet “Refuelcalc.xls” obtained from EPA.

4. Finally, the evaporative emissions of new technology vehicles are very low, as expected. The increase in evaporative emissions is driven by two factors: (1) the estimated frequencies of leaking vehicles and their associated emissions, and (2) the estimated frequencies of vapor leaks and their associated emissions. CRC's current testing program to evaluate the frequency of liquid leaking vehicles will provide valuable information in estimating these frequencies. Also, the CRC E-77 program is investigating the emissions associated with vapor leaks in new technology vehicles. However, a much more complete review of the *frequency* of vapor leaks in-use in I/M and non-I/M areas by vehicle technology and age is needed. Issues to be examined are (1) if the current EPA method may be double-counting leaks for different tests that are being added together, and (2) if newer data from I/M programs exists for estimating OBD failures.

7.0 Heavy-Duty Emissions

This section is organized into the following subsections:

- NOx
- PM
- HC and CO
- Start emissions
- Extended idle
- Tampering and malmaintenance
- Crankcase emissions

The presentations to the FACA workgroup that are utilized in this section are the following:

"MOVES Status and Overview", Koupal, August 8, 2006

"MOVES Heavy-Duty PM", Hart, August 8, 2006

"MOVES 2008 Heavy-Duty NOx Results", Gururaja, December 15, 2008

"Crankcase Emissions in MOVES" (no author), December 15, 2008

"Draft Heavy-Duty Vehicle $PM_{2.5}$ Methodology for MOVES", Glover, December 15, 2008

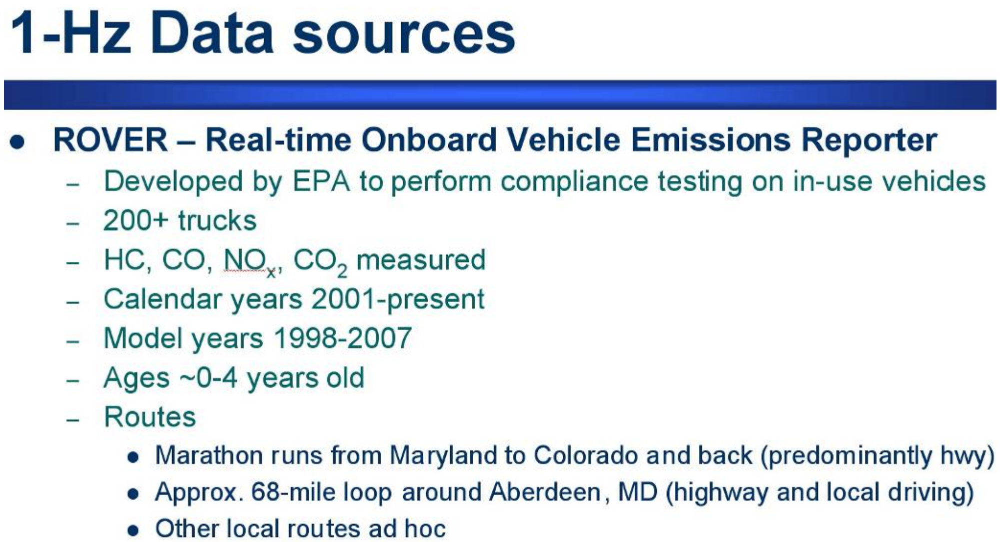
"Draft Heavy-Duty Vehicle Tampering and Malmaintenance Methodology for MOVES", Cullen, December 15, 2008

7.1 NOx emission rates

EPA is developing a methodology to estimate heavy-duty emissions of gaseous pollutants for use in MOVES based on two sources of 1-Hz data. However, only the results for NOx were presented to the FACA workgroup. Staff indicated to the FACA workgroup that the methodology for HC and CO is not as fully developed. The data EPA ultimately used to develop HC and CO emission rates in draft MOVES are described in Section 7.3 below.

The two data sources are described in the NOx presentation. The ROVER (Real-time On-road Vehicle Emissions Reporter) data set includes HC, CO, NOx and CO_2 data on over 200 trucks from model years 1998-2007 measured during long-haul runs from Maryland to Colorado and back along with data from local and highway driving in and around Aberdeen, Maryland. The data were obtained in calendar years 2001 to the present; further details are shown in Figure 7-1.

Figure 7-1. ROVER Data



The second data set on NOx and CO_2 from about 150 trucks was obtained at West Virginia University (WVU) from HD consent decree in-use testing (Figure 7-2). Model years 1994 to 2006 were included; the data were obtained over calendar years 2001 to 2006. Details of the consent decree testing are available.[25] A summary of the ROVER testing (measurement system, driving routes, sampling strategy, and vehicle recruitment) is also available.[26]

It is important to note that both data sets include vehicles that may have higher NOx due to calibration strategies employed by a number of heavy-duty engine manufacturers in the 1990s to reduce fuel consumption. These trucks with higher NOx are required to be "reflashed" upon engine rebuilding. It is unlikely that the fraction of reflashed trucks in these datasets would be as high as it will be in the next few years as many of them come in for their first rebuild. Therefore, EPA should estimate the fraction of engines rebuilt in the future and determine the NOx reductions that these trucks will experience. The resulting factor should be worked into the MOVES model. EPA should also recognize that reflashes have been taking place earlier than required, and factor that into the model as well.

[25] M. Gautam, N. Clark, G. Thompson, D. Carder, and D. Lyons, "Evaluation of Mobile Monitoring Technologies for Heavy-Duty Diesel-Powered Vehicle Emissions, West Virginia University Department of Mechanical and Aerospace Engineering report, March 9, 2000; M. Gautam, N. Clark, G. Thompson, D. Carder, and D. Lyons, "Development of In-Use Testing Procedures for Heavy-Duty Diesel-Powered Vehicle Emissions", West Virginia University Department of Mechanical and Aerospace Engineering report, March 20, 2000; G. Thompson, D. Carder, N. Clark and M. Gautam, Summary of In-use NOx Emissions from Heavy-Duty Diesel Engines, SAE paper 2008-01-1298.

[26] Jack, J. "U.S. Army Aberdeen Test Center Support of Heavy-Duty Diesel Engine Emissions Testing." http://www.epa.gov/ttn/chief/conference/ei15/session1/jack.pdf

Figure 7-2. WVU MEMS Data

1-Hz Data sources

- WVU – MEMS (Mobile Emissions Measuring System)
  - Used for HD consent decree in-use vehicle testing
  - ~150 trucks
  - NO_x and CO_2 measured
  - Calendar years 2001-2006
  - Model years 1994-2006
  - Ages ~0-7 years old
  - Fixed routes in WV and PA involving highway and urban driving

The WVU documentation includes a comparison of the ROVER instrumentation available at the time with the Mobile Emissions Measurement System (MEMS) developed and used in the consent decree testing. The ROVER instrumentation had several limitations in comparison with the MEMS system. For example, the ROVER instrumentation used an electrochemical cell that measured NO only with no apparent means of reducing NO_2 to NO. In addition, the system used cold sampling lines and a simple water trap that could absorb NO_2 and had no provision for continuously measuring ambient humidity. ROVER also had a substantially slower NOx response time than the MEMS. Another open question is the accuracy of the torque data from the ROVER program. In the WVU dataset, lug curve information for each engine tested was obtained from the manufacturer. In the ROVER data set, the source of the lug curve data for each tested engine is unclear and the method for determining accessory loads is not known.

In WVU testing, FTP cycle integrated brake-specific mass emissions of NOx measured by MEMS were within 0.5% of WVU's FTP laboratory data. Simultaneous measurements with the ROVER system yielded differences in brake-specific NOx mass emission rates as high as 7.9% between the ROVER and the laboratory data. On other operating cycles developed by WVU for the consent decree testing, brake-specific NOx mass emissions were within ±4% of the laboratory values for the two systems. While one might expect the ROVER system to underestimate NOx due to its inability to measure NO_2, the overall comparison with MEMS is reasonable. However, the ROVER system included an analyzer correction factor (the significance of which has not been explained) in the output file, perhaps to account for biases in the measurement.

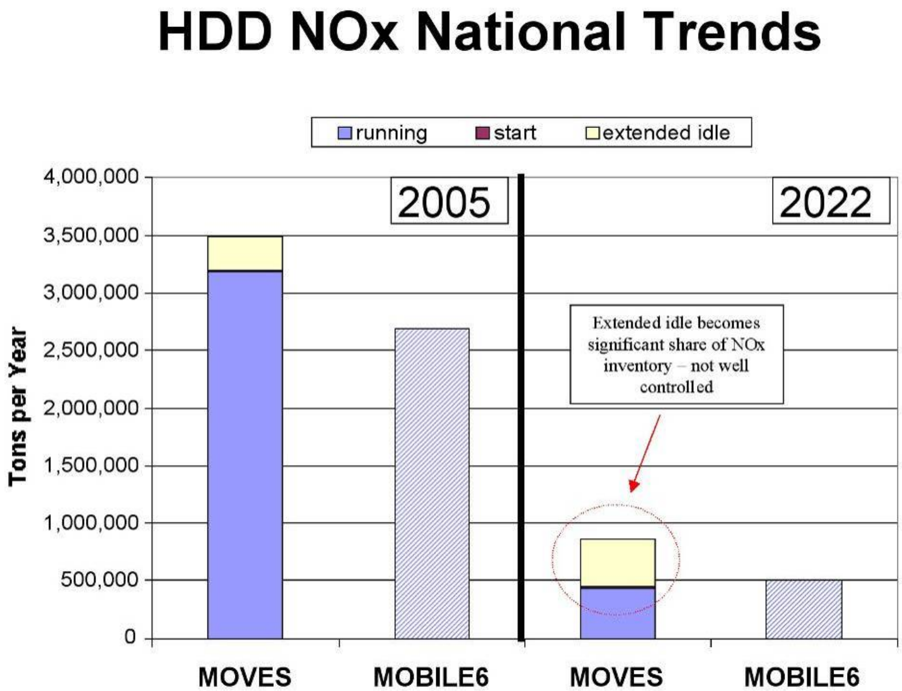
Since there are no standard procedures for the analysis of transient in-use on-road emissions, judging the adequacy, limitations, and uncertainty of the various data sets requires careful documentation and comparisons to established procedures. Since the MEMS data are well documented, a comparison between the MOVES emission factors derived from MEMS with those derived from ROVER would be instructive.

To determine NOx emission rates for each Vehicle Specific Power bin in MOVES, engine power is calculated from engine speed and engine torque (from the engine control unit) corrected for estimated accessory and driveline losses. The driveline loss is estimated at 10 % for all HD categories and the accessory loads were estimated from available data on typical operation as a function of speed and load. The accessory losses ranged from 6.6 kW for a MDH operating at low power to 10.5 kW for a HDT operating at high power. Consideration should be given to making accessory losses a function of ambient temperature.

Data from 308 trucks or buses were used in the analysis. Since the amount of data available differed among the trucks, the results for each vehicle were weighted equally in determining the emission rate in a given VSP bin. Separate base emission rates were determined for each regulatory class, model year grouping, and VSP bin. For model years before 1991 and after 2006, the data were apportioned by certification level and decreases in standards, respectively. As noted later in the presentation on tampering and malmaintenance, no age deterioration for NOx is assumed for non-aftertreatment engines. For aftertreatment technologies, separate estimates were made for lean NOx traps and urea SCR systems, with substantial increases in composite malfunction rates (16% and 13%) and in overall emission factors (87% and 72%) over the useful life in each case. The impact of HD OBD is not included in these estimates. The effects of OBD should be included in the heavy-duty emission rates as soon as possible.

Staff showed one comparison of the MOVES model output with MOBILE6 for calendar years 2005 and 2022, as illustrated in Figure 7-3 using the default activity data in the models.

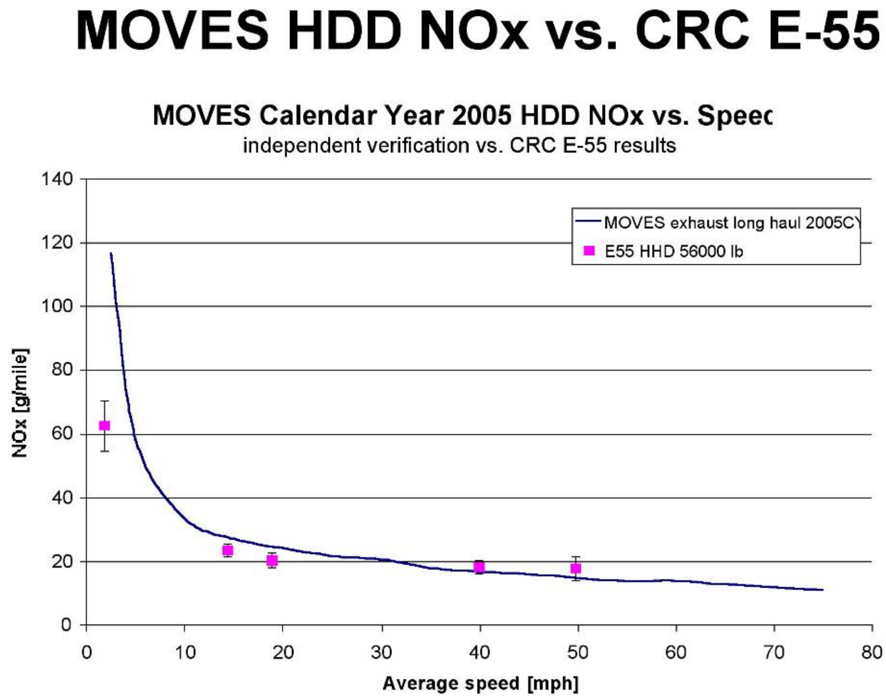
Figure 7-3. National NOx Trends



Running exhaust NOx in calendar year 2005 is increased by about 15 to 20% going from MOBILE6 to MOVES. The running exhaust in 2022 is reduced by over 80% compared to 2005 and is similar in the two models. Extended idle emissions, which are not included in MOBILE6, are estimated to be almost as large as running exhaust emissions in 2022. The basis for this estimate is discussed in section 7.5 below. Start NOx emissions are extremely small in the MOVES model. As discussed below in section 7.4, NOx start emissions for LHD vehicles are estimated to be 1.68 grams per cold start but, due to limited and conflicting data, NOx start emissions for MHD and HDV vehicles are assumed to be zero. A complete breakdown of heavy-duty emission components is presented and discussed in Section 10.

In Figure 7-4, EPA compares the MOVES model output for calendar year 2005 as a function of average speed with CRC E-55 results and shows reasonably good agreement, except at low speeds where the model overestimates emissions on the creep cycle used in the CRC E-55 study.

Figure 7-4. MOVES NOx vs. Speed



Drive schedules are used in MOVES to determine the distribution of time in various VSP bins. EPA's September 2009 draft activity report notes that Draft MOVES2009 employs 40 drive schedules, mapped to specific source types and roadway types.[27] The activity report summarizes the way driving schedules are used in draft MOVES as follows:

> Briefly, for each speed bin in the speed distribution, the MOVES model selects the two associated driving cycles with average speeds that bracket the speed bin's average speed. The Vehicle Specific Power (VSP) distributions determined for each bracketing driving schedule are averaged together, weighted by the proximity of the speed bin average speed to the driving schedule average speeds. In this way, the VSP distribution of any road type's speed distribution is determined from the available driving schedules.

The activity report also indicates that Medium-Duty and Heavy-Duty driving schedules are developed specifically for MOVES, based on work performed for EPA by Eastern Research Group (ERG), Inc.[28] The details of the low speed driving schedules in draft MOVES should be evaluated to determine the source of the disagreement between draft MOVES and the CRC E-55 results at low speeds.

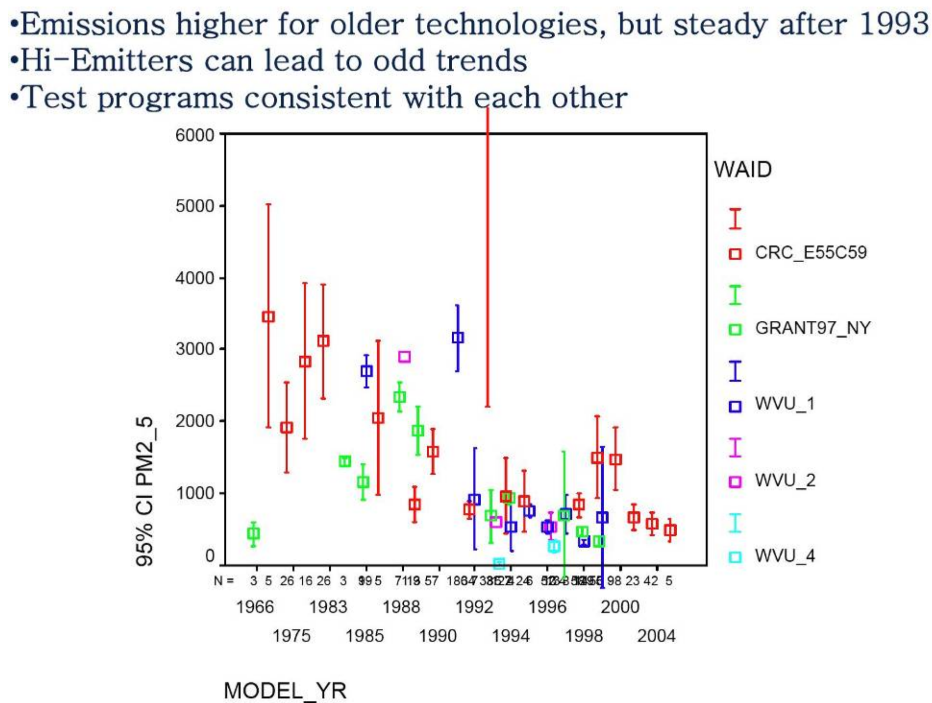
[27] Draft MOVES2009 Highway Vehicle Population and Activity Data, Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, EPA-420-P-09-001, August 2009.

[28] Eastern Research Group, Inc. (ERG), "Roadway-Specific Driving Schedules for Heavy-Duty Vehicles." EPA Contract 68-C-00-112, Work Assignment 3-07, August 15, 2003.

7.2 PM emission rates

In 2006, EPA made a presentation to the FACA workgroup on the plans for developing HD PM emission rates.[29] This presentation indicated that the Agency had data on PM emissions over several driving cycles for 100 trucks that, while not a random sample, reflected real-world deterioration and maintenance. The data set included 66 trucks from the CRC programs, 12 from WVU, and 22 from a New York Department of Environmental Conservation study. The 2006 presentation also noted that the data set is biased to older, potentially dirty trucks and that outliers may drive the results. The 2006 presentation included a plot of PM emissions versus model year noting higher emissions for older technologies (see Figure 7-5), generally flat emissions after 1993, and that high emitters can lead to odd trends.

Figure 7-5. PM Trends with Model Year



One of the limitations of the CRC program for use in MOVES is that continuous PM measurements are only available for the trucks included after phase 1. The 2006 presentation also summarized an EPA contract with WVU to recommend the best set of continuous PM data from the CRC E-55/59 program.

As described in the December 2008 presentation to the FACA workgroup, the diesel $PM_{2.5}$ emission rates in MOVES are developed from a sub-set of the 100 truck data. This consisted of 56 vehicles from the CRC test programs, for which there are real-time PM measurements. Since there were multiple cycle tests on each vehicle, EPA used over 480 real-time PM measurements on 56 trucks (23 MD and 33 HD) to develop the emission

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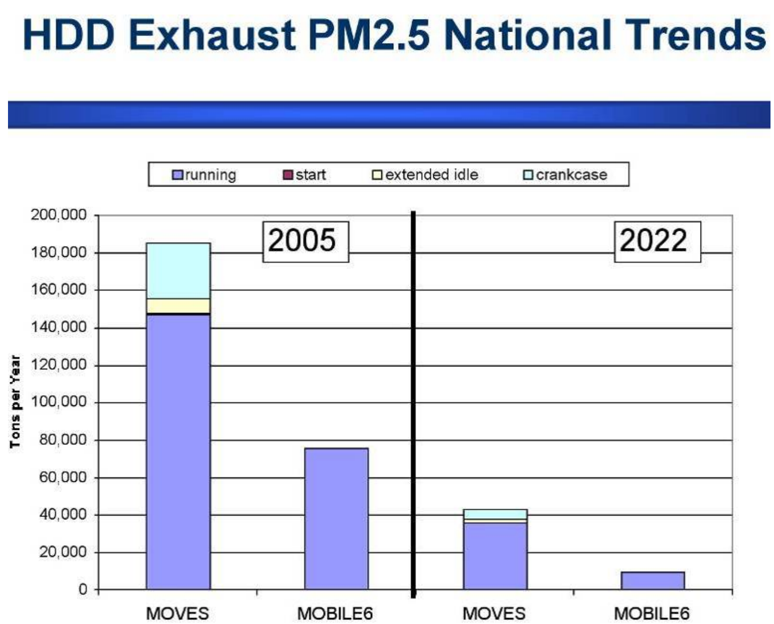
[29] MOVES Heavy-duty PM, August 8, 2006 EPA presentation to FACA.

factors for the various VSP bins, model years, and regulatory classes. The mass of PM is determined by the output of the real-time PM measurement as normalized by the $PM_{2.5}$ filter measurement. The power input to VSP is determined through vehicle speed and acceleration measurements from the chassis testing, Because of the limited data, EPA used various regression and other statistical techniques to fill in all the needed data bins. Additional details are available in a recent draft report on heavy-duty emissions in MOVES.[30]

The available data included trucks from model years 1969 to 2005. For 2007+ model years, EPA projected base emissions to be 10% of the 1998-2006 group data. This estimate appears to be very conservative, since testing of trap-equipped engines indicate that PM emissions are reduced by approximately 99%. PM emissions from 2007 and later model engines need to be tracked and used to revise MOVES. Based on the argument that no longitudinal data were available, EPA developed deterioration factors for tampering and malmaintenance that were described in a separate presentation.[31]

The 2008 PM presentation includes several comparisons with CRC E-55 data and with MOBILE6. Figure 7-6 indicates that draft MOVES will increase both estimated 2005 and estimated 2022 running exhaust PM emissions from heavy-duty engines dramatically. As for NOx, start emissions make a very small contribution to national emissions.

Figure 7-6. HDD PM National Trends



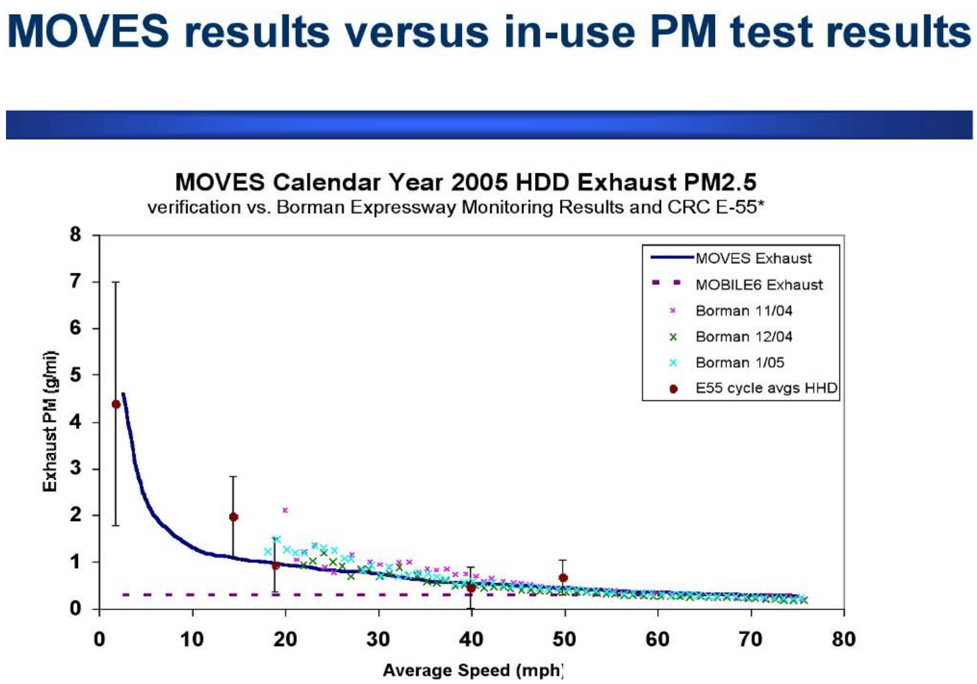
30 Development of Emission Rates for Heavy-Duty Vehicles in the Motor Vehicle Emissions Simulator (Draft MOVES2009), Draft Report, Assessment and Standards Division, Office of Transportation and Air Quality, U.S. Environmental Protection Agency, EPA-420-P-09-005, August 2009.

31 Longitudinal data are data on the same vehicles tested at different times of their lives.

There is a substantial contribution from diesel crankcase emissions in 2005 that is discussed in greater detail below. In addition to possible differences in activity-related inputs, there appear to be two main differences between MOBILE6 and draft MOVES. The first is that MOBILE6 did not include the effect of speed. This is shown in Figure 7-7, which is a plot of MOVES 2005 calendar year $PM_{2.5}$ output versus average speed with a comparison to (1) CRC E-55 data that was used in the development of the model, (2) emission data derived from a real-world expressway experiment, and (3) MOBILE6. MOBILE6 and MOVES emission rates are similar at average speeds of 40 mph and above but MOVES has much higher emission rates at lower speeds. By including speed effects, MOVES is an improvement over MOBILE6.

The emissions versus average speed curve in Figure 7-7 continues lower above 50 mph, while the data start to show a small increase in emissions. It is doubtful that PM emissions continue lower at higher speeds, since the aerodynamic friction component increases by the cube of the vehicle speed. This is an area that should be examined in more detail.

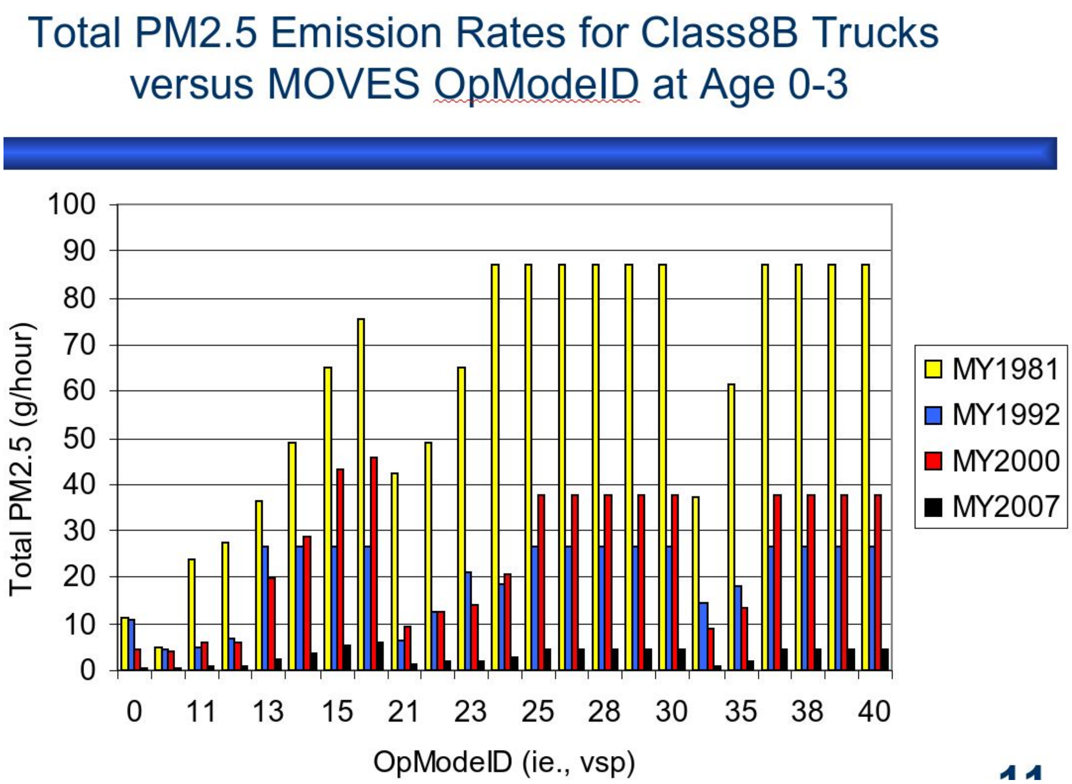
Figure 7-7. PM Emissions versus Speed



The second major difference is that MOVES includes a separate increment of emissions for tampering and malmaintenance. This factor increases PM emissions at useful life by 85% for 1994-1997 trucks, by 74% for 1998-2002 trucks, and by 48% for 2003-2006 trucks. Since the data used to develop the model is acknowledged in EPA's 2006 presentation to the FACA workgroup to reflect real-world deterioration and maintenance and to be biased to older, potentially dirty trucks, the need for a separate T&M factor can be questioned. The CRC program included a separate effort on T&M; the final CRC report noted that about 10% of the PM data is influenced by malmaintenance. Therefore, EPA should re-visit the need for a T&M correction and look for other data sets (tunnel studies, for example) to test the MOVES output against (with and without a separate T&M factor).

There are additional reasons to conclude that the draft MOVES PM factors (without the T&M factor) may be biased high. First, EPA noted in 2006 that outliers may drive the results, and there is evidence that this is the case. For example, as shown in Figure 7-8, the model year 2000 emission rates are higher than the model year 1992 emission rates for many of the VSP bins. Second, the PM emissions of the subset of 56 trucks used by EPA appear to be somewhat higher than the PM emissions from the complete 100-truck data set as noted in Figure 7-5, particularly in more recent years. Several high emitters in the CRC data are also apparent in Figure 7-8. One way to test this assumption is to statistically fit the emission rate versus MY data in Figure 7-5 using all 100 truck test results (with and without outliers removed) and compare that with the fit of the subset of data used to develop draft MOVES.

Figure 7-8. PM Emissions by Model Year



As noted above, the plot of emission rate versus speed includes data from a real-world experiment. The experiment, reported by Soliman, Jacko, and Palmer[32] measured $PM_{2.5}$ up and downwind of the Borman Expressway to determine the $PM_{2.5}$ caused by mobile sources on the expressway. Although this section of expressway experiences heavy truck

[32] A. Soliman, R. Jacko, and G. Palmer, "Development of an Empirical Model to Estimate Real-World Fine Particulate Matter Emission Factors: The Traffic Air Quality Model," J. Air & Waste Manage. Assoc., **56**, 1540-1549 (2006).

traffic, the resulting emission factor includes mobile-source related emissions beyond those from running exhaust of HD trucks. For example, crankcase emissions, tire and brake emissions, light-duty gasoline exhaust emissions, and any fine particles included in the road dust re-suspended by the traffic are all included in the emission factors plotted in Figure 7-7. Therefore, the expressway data do not provide an appropriate test for the magnitude of the MOVES HD running exhaust emission factors.

The draft MOVES model reports $PM_{2.5}$ emissions according to two subtypes: elemental carbon (EC) and organic carbon (OC). The final model will also include a generic sulfate emission factor. Details of the EC/OC split are provided in Appendix A.5 of the August 2009 Heavy-Duty report (EPA-420-P-09-005). The EC fractions used in draft MOVES for pre-2007 model year trucks (i.e., before diesel particulate filters (DPFs) were standard) vary according to regulatory class and MOVES operating mode bin. They typically range from 25 percent at low loads (low VSP) to over 90 percent at highly loaded modes. The primary dataset used in the analysis came from Kweon, et al. (2002).[33] Kweon, et al. (2002) measured particle composition and mass emission rates from a single-cylinder research engine based on an in-line 2.3-liter turbo-charged direct-injection six cylinder Cummins N14-series engine, with a quiescent, shallow dish piston chamber and a quiescent combustion chamber. Since the split between EC and OC is an operationally defined measurement, staff chose to report EC and OC as measured by thermo-optical reflectance (TOR), the method EPA uses for $PM_{2.5}$ speciation in the ambient monitoring network.

For 2007 and later model year, DPF-equipped vehicles, a different methodology is used, since it is believed that virtually all of the particulate that is emitted from the tailpipe will be OC and that only a modest fraction will be EC. It should be noted that aftertreatment devices used in 2007 greatly reduce OC. The remaining OC measurement is complicated by problems with adsorption of gas-phase OC on filters and difficulties in interpreting tunnel blanks. Hence the OC emission rate from these engines is highly uncertain. Based on the data in SAE paper 2002-01-0432,[34] EPA chose an EC fraction of 0.0861 and an OC fraction of 0.9139 which is applied to all regulatory types and operating modes for 2007 and later diesel trucks and buses. Because of the limited nature and extent of the data used to develop EC/OC splits in MOVES, the current methodology should be considered provisional until additional data can be gathered and analyzed.

7.3 HC and CO emissions

Although the methodology for HC and CO was not presented to the FACA Workgroup, the August 2009 Heavy-Duty report provides some detail. That report notes that the on-road ROVER data used for the NOx analysis is not used since the program used the less accurate non-dispersive infrared (NDIR) technology instead of flame-ionization detection

[33] Kweon, C.-B., D.E. Foster, J.J. Schauer, and S. Okada, 2002. Detailed chemical composition and particle size assessment of diesel engine exhaust. *SAE Technical Paper Series*, 2002-01-2670.
[34] Lev-On, Miriam, et al. "Chemical Speciation of Exhaust Emissions from Trucks and Buses Fueled on Ultra-Low Sulfur Diesel and CNG", SAE 2002-01-00432, March 2002.

(FID) to measure HC. In addition, the WVU MEMS program did not collect HC and CO data. Therefore, to keep HC and CO data sources consistent, EPA used chassis test programs exclusively for the analysis of HC and CO emissions. Data were gathered from the CRC E-55/59 program, the Northern Front Range Air Quality Study, a New York Department of Environmental Conservation study, and WVU testing done for the Agency. Although data from a total of 131 1960 to 2002 HHD trucks, 57 1960 to 2002 MHD trucks, and 30 1960 to 2002 buses are available, only the data in the most prevalent regulatory class and model year group combination is used. Thus, for example, of the 131 HHD 1960 to 2002 trucks, only the data from the 58 trucks that were tested at 0 to 3 years of age are used in the analysis. The HC and CO emission rates are adjusted for tampering and malmaintenance as discussed in section 7.6. It is difficult to judge the appropriateness of the HC and CO emission rates, since insufficient detail is provided in the August 2009 Heavy-Duty report. Nevertheless, a more complete analysis of the full dataset compiled by EPA should be done to evaluate the extent of deterioration in the dataset. Such an analysis would also examine differences between in-use and certification vehicles to determine whether a T&M adjustment is appropriate.

7.4 Start emissions

In a similar manner to the light-duty vehicle start methodology, start emission increases for Light Heavy-Duty vehicles are developed from separate bags containing cold and hot start emissions over the FTP and ST01 cycles. Data from 21 vehicles, ranging from model years 1988 to 2000, were analyzed. Due to the limited number of vehicles, no distinction was made for model year or age of the vehicle. The average start emissions increases in grams for light heavy-duty vehicles in MOVES are 0.13 for HC, 1.38 for CO, and 1.68 for NOx.

Data for HHD and MHD trucks are much more scant. The August 2009 Heavy-Duty report indicates that EPA simulated cold start and warm starts on a 2007 Cummins ISB on an engine dynamometer. EPA also evaluated data from the University of Tennessee, which tested 24 trucks with PEMS at different load levels during idling. In these tests, the NOx difference between cold and hot start tests is positive in some cases and negative in others. Based on these limited tests, EPA set HHD and MHD HC and NOx cold start increments at zero and CO at 16 grams.

There is also scant data on heavy-duty truck $PM_{2.5}$ emissions for the start process since typically, heavy-duty vehicle emission measurements begin on fully warmed up vehicles. EPA tested one heavy-heavy-duty 2004 model year engine on the FTP in both hot and cold start conditions. The average difference in $PM_{2.5}$ emissions on the FTP cycle (using two tests for cold start and four replicate tests for hot start) is 0.1099 grams.
MOVES uses this value (0.1099 g of $PM_{2.5}$ per start) for 1960 through 2006 model year vehicles. A value of 0.01099 g is used for 2007 and later model year vehicles allowing for a 90% reduction due to DPFs.

As with light-duty vehicles, MOVES adjusts the start rates for soak times. Since no data are available for heavy-duty vehicles, EPA applied the same methodology and soak fractions used for light-duty emissions. Although HD start emissions are not a major contributor to inventories, start emissions should be re-evaluated once additional data are available.

7.5 Extended idle emissions

EPA includes another category of HD emissions in MOVES, extended idle, which applies only to diesel long-haul combination trucks. Extended idling, also referred to as "hoteling," is defined by EPA as any long period of discretionary idling that occurs during long distance deliveries by heavy-duty trucks. Details are provided in the August 2009 Heavy-Duty report and August 2009 Activity report. EPA collected data on extended idle emissions from a variety of sources that involved extended idle tests as an adjunct to other laboratory tests or from separate field tests using portable measurement systems. The data are separated by truck and bus and by idle speed and accessory usage to develop emission rates for different combinations of the variables. An earlier EPA analysis had concluded that factors that affect engine load, such as accessory use, and engine idle speed are the important parameters in estimating the emission rates of extended idling.

The studies focused on three types of idle conditions. The first is a curb idle, or low engine speed (<1000 rpm) and no air conditioning. The second is an extended idle condition with higher engine speed (>1000 rpm) and no air conditioning. The third is an extended idle condition with higher engine speed (>1000 rpm) and with air conditioning on. The results are summarized in Appendix A.3 of the August 2009 Heavy-Duty report. For inclusion in draft MOVES, EPA chose the emission rates from the third category: high idle rate with air conditioning in the “on” position, based on the assumption that drivers always increase idle speeds and use accessories extensively during extended idles.

The following table shows the extended idle emission rate inputs in g/hour.

Table 7-1. Extended Idle Emissions for HDD Vehicles (g/hr)

Model years	NOx	HC	CO
Pre-1990	112	108	84
1990-2006	227	56	91
2007 and later	201	53	91

For PM, the limited data on extended idling in Appendix A.3 indicates rates from 2 to 4 grams/hour. However, EPA has not yet included a separate PM extended idle emission rate in MOVES. Regular curb idle emission rates for PM are included in draft MOVES for extended idles.

The August 2009 Activity report indicates that MOVES uses data from Lutsey, et al. on the distribution of truck hoteling times along with data on truck usage as a function of time of day to distribute the extended idle emissions in MOVES.[35] Lutsey, et al. carried

[35] Lutsey, Nicholas, Christie-Joy Brodrick, Daniel Sperling, and Carollyn Oglesby. "Heavy-Duty Truck

out a survey of 365 long-haul truck drivers at large truck stops around the country. The truckers self-reported 5.9 hours of hoteling for every 10 hours of long-haul truck driving.

There are three major issues with the extended idle analysis in draft MOVES. First, EPA chose the highest emission rates for HC, CO, and NOx from the three categories of emissions testing they evaluated. This assumes that all hoteling utilizes high idle speeds and high accessory loads at all times. The Lutsey, et al. survey that EPA relies on for the distribution of hoteling time includes data on the distribution of idle speeds. Lutsey, et al. report:

> …when respondents were asked the idle speed of their engines, the average response was about 870 rpm, with responses fairly evenly distributed from 600 to 1,200 rpm and small peaks around 650 and 1,000 rpm.

The data are shown in Figure 2 of Lutsey, et al. Lutsey, et al. also refers to a 2001 California pilot survey that had similar responses with a mean engine speed of 850 rpm and peaks in the distribution of responses greater and less than the mean.[36] Thus, there is evidence that truckers utilize a range of idle speeds when hoteling and adjust the speed depending on the ambient environment and accessory use. EPA should use that evidence to revise the extended idle emission rates to accurately reflect the distribution of engine idle rates and accessory use. Since the emissions for all three gaseous pollutants for low idle-no A/C are less than half the rates used in MOVES as reported in Appendix A.3 of the August 2009 Heavy-Duty report, revising the rates to represent the real distribution of hoteling speeds and loads would substantially reduce extended idle emissions in MOVES. For example, the 188 tests reporting NOx emissions at low idle and no A/C average 94 grams/hour while the 31 tests at high idle and A/C on average 227 grams/hour.

Second, EPA assumed that 2007 and later trucks with advanced control systems would revert to 1990 to 2006 idle emission rates after one hour of extended idling without any data. EPA made this assumption on the basis that there is no requirement to address extended idling emissions in the emission certification procedure.

Third, most, if not all, of the major inventory-impacting states have adopted anti-idling laws and/or regulations that effectively preclude any extended idling (even from sleeper trucks) in the future. Currently, however, most states exempt sleeper berth trucks from idling laws. In addition, there is a trend of increasing availability and use of plug-in facilities for sleeper trucks. The MOVES model needs to take these regulations and use patterns into account.

Idling Characteristics - Results from a Nationwide Truck Survey." Annual Meeting of the Transportation Research Board, January 2004.

[36] Lutsey, N., C. J. Brodrick, D. Sperling, and H.A. Dwyer. Markets for Fuel-Cell Auxiliary Power Units in Vehicles: Preliminary Assessment. In *Transportation Research Record: Journal of the Transportation Research Board, No. 1842,* TRB, National Research Council, Washington, D.C., 2003, pp. 118–126.

7.6 Tampering and malmaintenance

EPA provided a separate presentation on the tampering and malmaintenance assumptions in draft MOVES that includes substantial detail on the frequency of occurrence and emission impacts of various failure modes. The staff indicated that it used information from CARB's EMFAC2007 modeling, as well as input from EMA/AIR comments along with several additional data sources and EPA internal engineering judgment to develop its assumptions. In several instances, EPA chose lower occurrence rates than CARB had assumed and more in line with EMA/AIR comments. Although EPA had developed tampering and malmaintenance rates for NOx on non-aftertreatment systems, staff indicated that the values are not used in MOVES because the rates did not correlate with the CRC E-55 data. The EPA presentation includes information on specific failure modes for NOx aftertreatment.

In addition, substantial information is provided on failure modes for PM, HC and CO. Figure 7-9 summarizes the percent increases in emissions due to tampering and malmaintenance for the various pollutants and model year groups. EPA is assuming no effect of T&M on NOx prior to model year 2010 because there is no NOx aftertreatment controlling NOx before that time. In addition, while EGR is common on 2003 and later engines, EPA determined that it is very unlikely that EGR is tampered with.

The T&M factors are applied as follows: EPA applied the zero-age rates through the emissions warranty period (which, in terms of vehicle age, varies with regulatory class) and then increased the rates linearly up to useful life. The final T&M increases are held constant at the levels in Figure 7-9 beyond the useful life age. Importantly, EPA indicated that the T&M analysis was completed prior to the final rule for OBD phase-in for heavy-duty trucks. Thus, the T&M factors in draft MOVES do not account for the impact of the OBD rule. [37]

[37] EPA plans to incorporate OBD with a 33% effectiveness rate for the final MOVES model.

Figure 7-9. Percent Increases in Emissions Due to Tampering/Malmaintenance

Overall Emission Factors

PM				
1994-97	1998-2002	2003-2006	2007-2009	2010+ No OBD
85%	74%	48%	50%	50%

NOX					
1994-97	1998-2002	2003-2006	2007-2009	2010+ HHDT	2010+ LDT
0%	0%	0%	0%	87%	72%

HC				
1994-97	1998-2002	2003-2006	2007-2009	2010+ No OBD
308%	302%	152%	154%	33%

CO				
1994-97	1998-2002	2003-2006	2007-2009	2010+ No OBD
308%	302%	152%	154%	33%

For pre-OBD vehicles and for HC, CO, and PM, T&M effects should not be added to these vehicles if the base vehicles from which the emission rates were developed already reflect deterioration and T&M. The CRC and ROVER data seem to fit that requirement. In addition, T&M adjustments are highly suspect because of their reliance on estimates of the frequency of occurrence of various problems, estimates of how long the problem persists before being repaired, and estimates of the increased emissions during the interval. CARB developed its T&M analysis in EMFAC to apply to certification level emissions in order to estimate in-use emissions. The base emission rates in MOVES have been developed from in-use trucks that are acknowledged by EPA to include some high-emitting trucks. In fact, EPA acknowledges in the August 2009 Heavy-Duty report that the inclusion of a separate T&M factor may result in double-counting T&M effects.[38]

7.7 Crankcase emissions

EPA is adding crankcase emissions from diesels into MOVES; MOBILE6 included crankcase emissions from gasoline vehicles but did not include any crankcase emissions from diesels. Although Positive Crankcase Ventilation (PCV) systems have been on gasoline vehicles since the 1960s, and on non-turbocharged diesels since 1968, they have only been on turbo-charged diesels since 2007. Thus, diesel crankcase emissions will show up as a new category of emissions in previous or current calendar years, but they will be a decreasing issue in the future, as a larger fraction of the fleet is equipped with PCV systems.

Crankcase emissions are being modeled as a fraction of exhaust emissions, for start, running and extended idle modes. The key assumption for non-PCV vehicles is that PM

[38] August 2009 HD report (Reference 30) at page 20.

crankcase emissions are 20% of exhaust PM. For PCV-equipped vehicles, the key assumption is a 4% failure rate since emissions for a working PCV valve are considered zero.

The 4% failure rate is not appropriate for OBD-equipped vehicles since, as detailed in the new HD OBD rule, EPA requires extensive monitoring for malfunctions of the crankcase ventilation system. [39]

The assumption that PM crankcase emissions are 20% of exhaust PM appears to be based on very limited data. The Zielinska, et al. paper noted in the list of data sources contains data on crankcase and exhaust emissions for only two school buses operating in a typical use. [40] In addition, the composition of crankcase PM is very different from exhaust PM, with crankcase high in OC and low in EC and ultra-fine particles, based on the Zielinska, et al. measurements and the 2005 Clean Air Task Force study. While referred to as blow-by, the emissions from the road draft tube are dominated by engine oil and unburnt fuel/air mixture present due to surface quenching rather than exhaust that blows-by the piston rings.

7.8 Distribution of Emissions by VSP Bin

This section evaluates the distribution of emissions by VSP bin for heavy-duty vehicles from MOVES in a manner similar to the method used for light-duty vehicles in Section 5. The distribution of THC, CO, and NOx emissions are evaluated for both the 2015 fleet of on-road long haul trucks, and also for the 2011 model year in 2015. The 2011 model is chosen because it is the first model year of line haul trucks that have full phased-in very low PM and NOx emission standards. Like the light-duty vehicles, the evaluation focused on Cook County, Chicago, in the summer between the hours of 12 Noon and 1 pm on a typical weekday.

THC, CO, and NOx, and $PM_{2.5}$ emission distributions for the fleet are shown in Figures 7-10 through 7-13. They also show the distribution of time by VSP bin. Unlike the light-duty vehicles, which show high emissions in some bins with very low operating times, the heavy-duty vehicle emissions distribution fairly closely follows the operating mode distribution. This would seem to indicate a moderate amount of variation in emissions by VSP bin for these vehicles. For the $PM_{2.5}$ emissions in Figure 7-13, both elemental carbon and organic carbon are shown. There are differences in these two distributions: there appears to be more organic carbon and less elemental carbon associated with lower VSP bins in each speed range, and higher elemental carbon and lower organic carbon associated with the higher VSP levels.

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[39] CFR §86.010-18(i)(2).

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

Figure 7-10

Percent of Diesel Long Haul Running THC Emissions and MOVES OpMode
July 2015, Weekday, 12:00-12:59 PM, All Roads, Cook County, IL

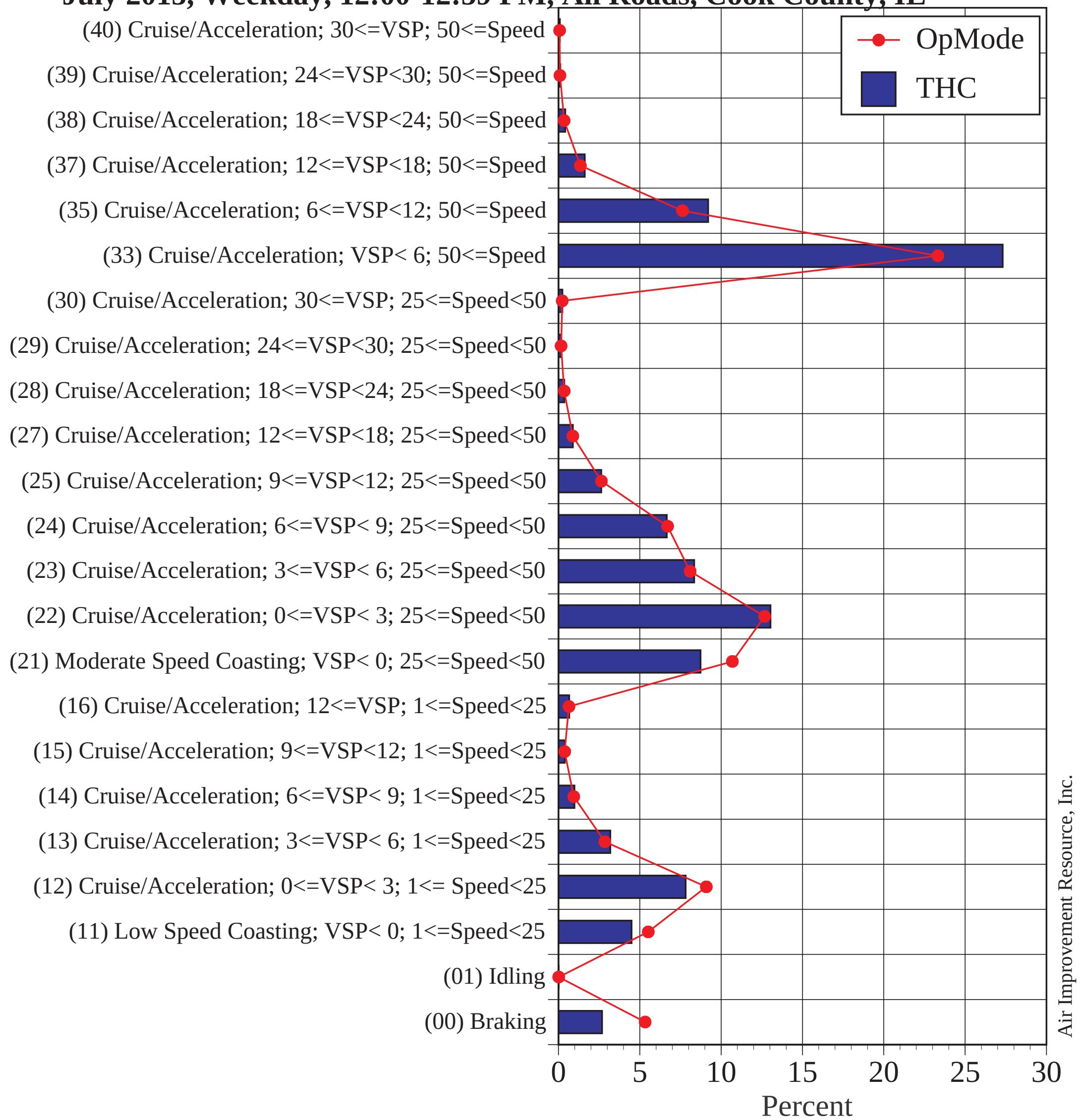


Figure 7-11

Percent of Diesel Long Haul Running CO Emissions and MOVES OpMode
July 2015, Weekday, 12:00-12:59 PM, All Roads, Cook County, IL

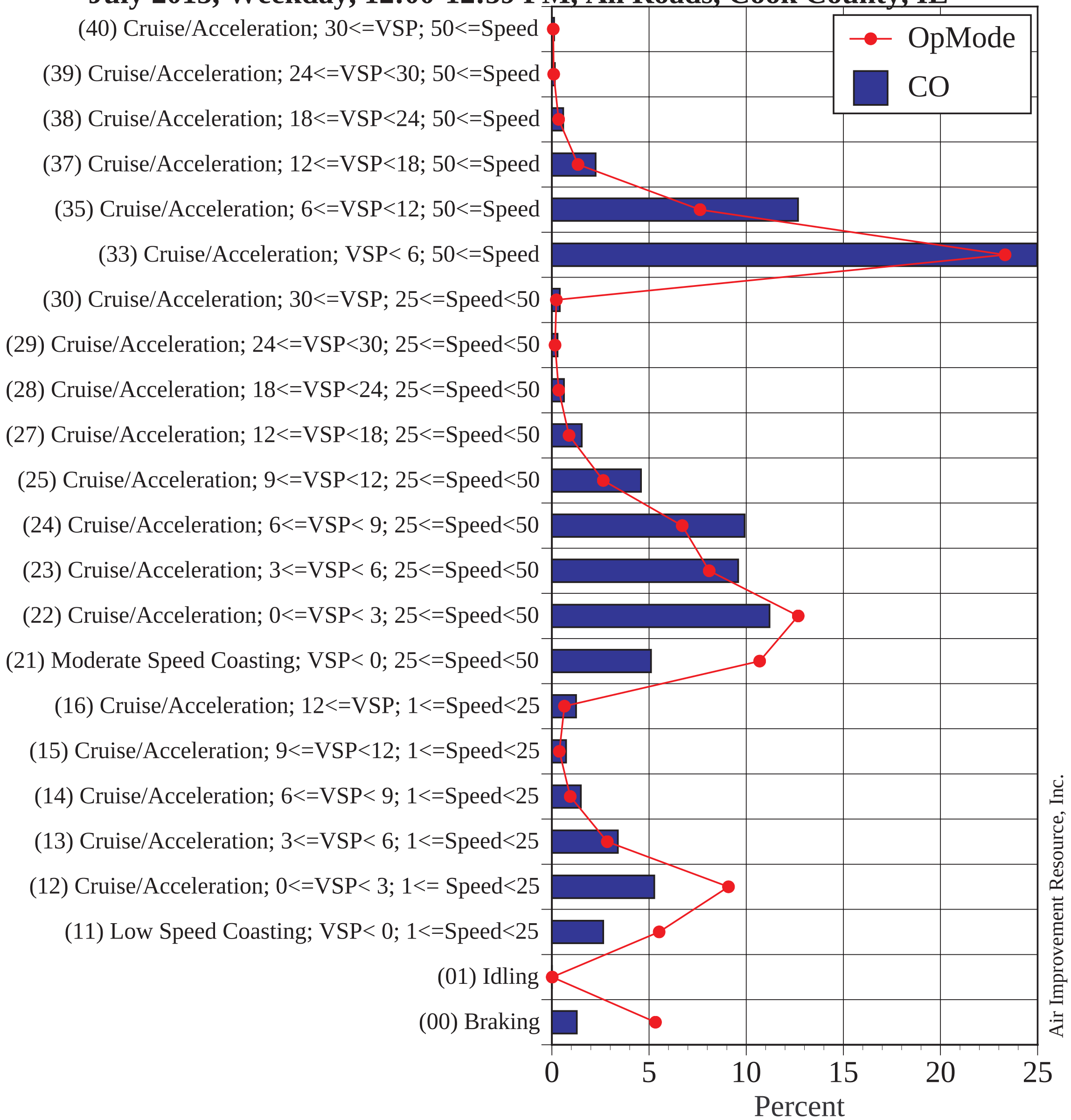


Figure 7-12

Percent of Diesel Long Haul Running NOx Emissions and MOVES OpMode
July 2015, Weekday, 12:00-12:59 PM, All Roads, Cook County, IL

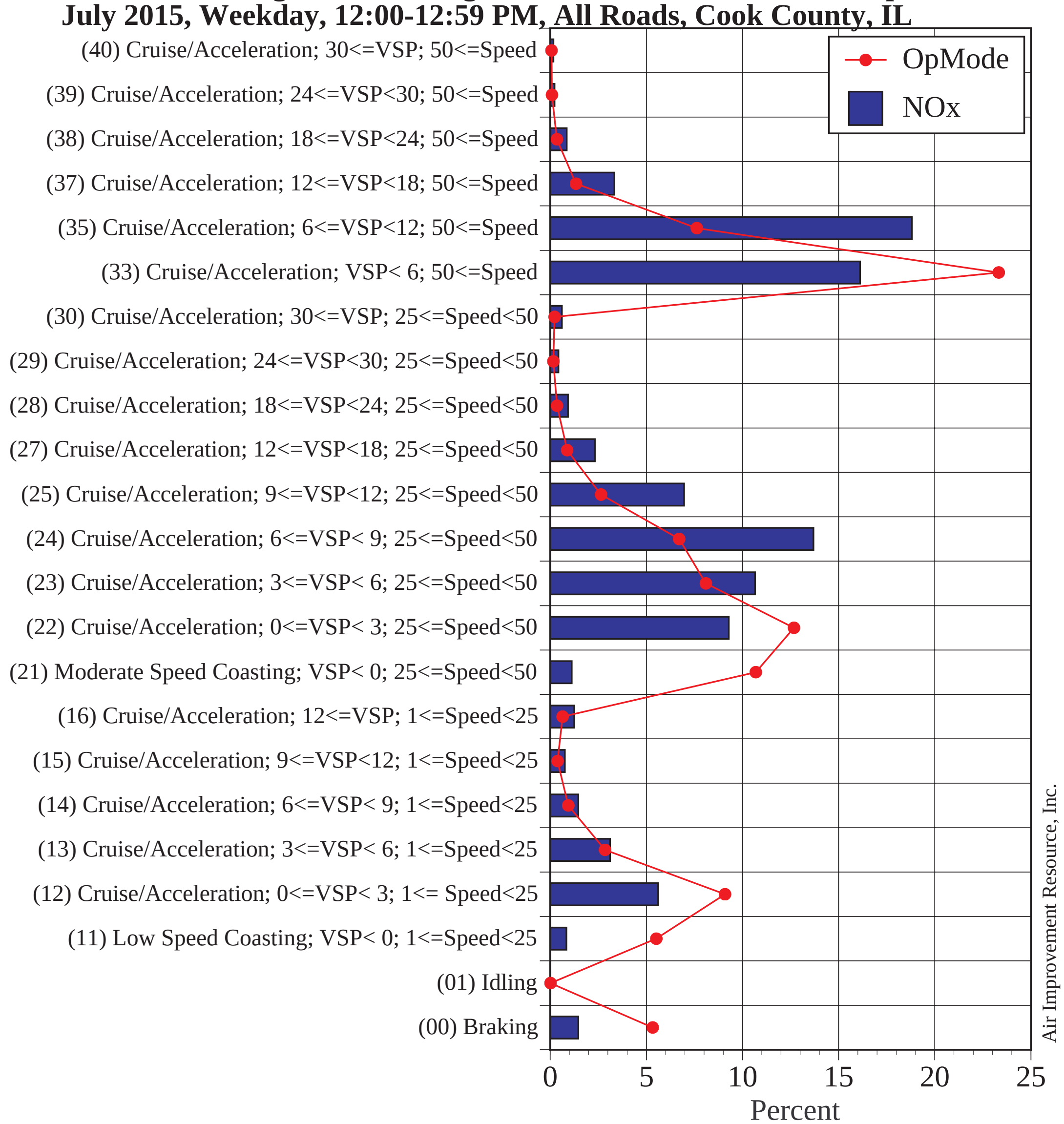
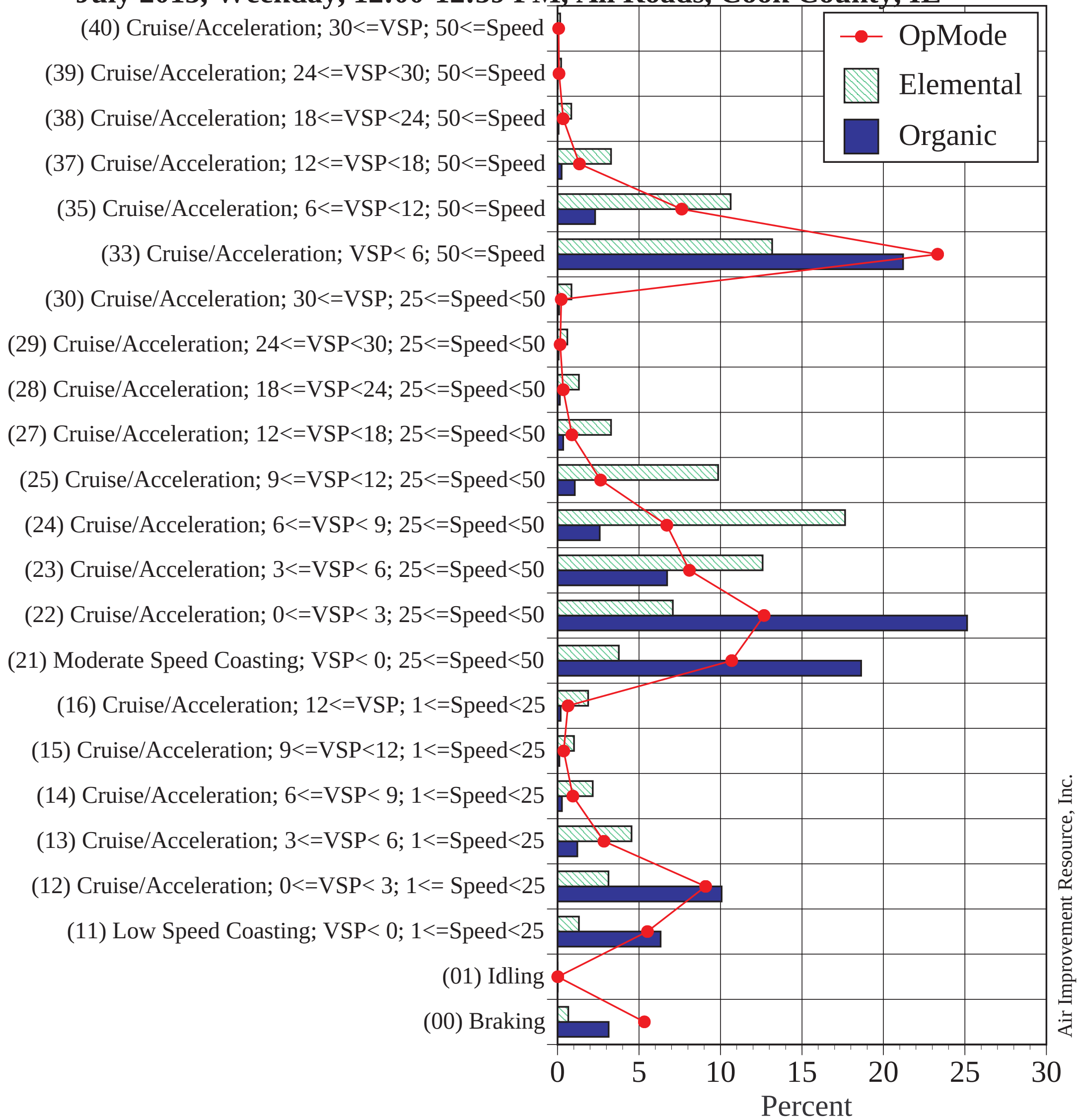


Figure 7-13

Percent of Diesel Long-Haul Truck Exhaust PM2.5 Emissions and MOVES OpMode July 2015, Weekday, 12:00-12:59 PM, All Roads, Cook County, IL



Figures 7-14 through 7-16 show emissions distributions by VSP bin for 2011 model year line haul trucks in 2015 ($PM_{2.5}$ is omitted because the distribution for model year 2011 is the same as for the fleet in 2015). The distribution of emissions by VSP bin for the 2011 model year is very similar to the fleet in 2015.

Figure 7-14

Percent of MY2011 Diesel Long Haul Running THC Emissions and MOVES OpMode July 2015, Weekday, 12:00-12:59 PM, All Roads, Cook County, IL

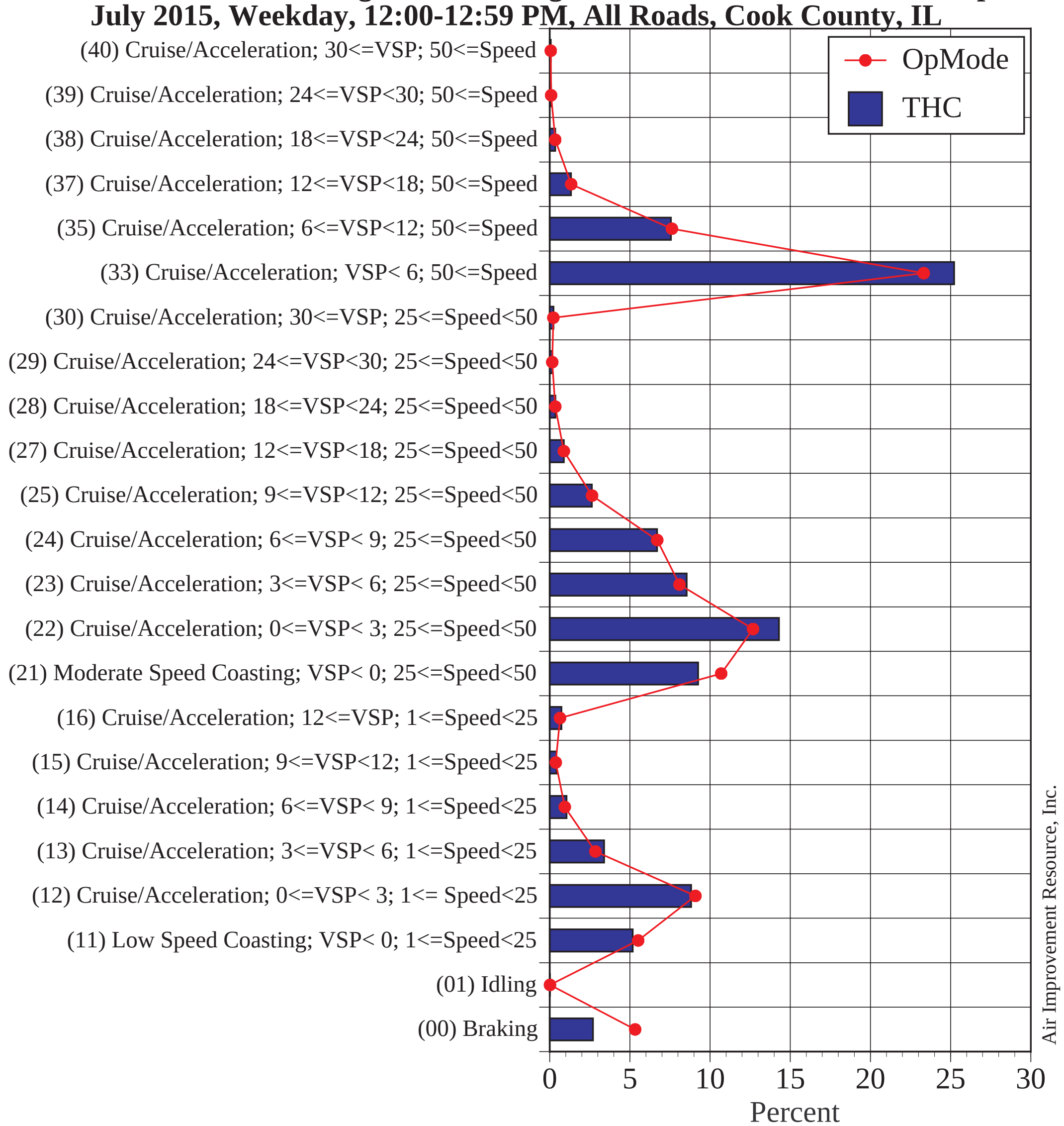


Figure 7-15

Percent of MY2011 Diesel Long Haul Running CO Emissions and MOVES OpMode
July 2015, Weekday, 12:00-12:59 PM, All Roads, Cook County, IL

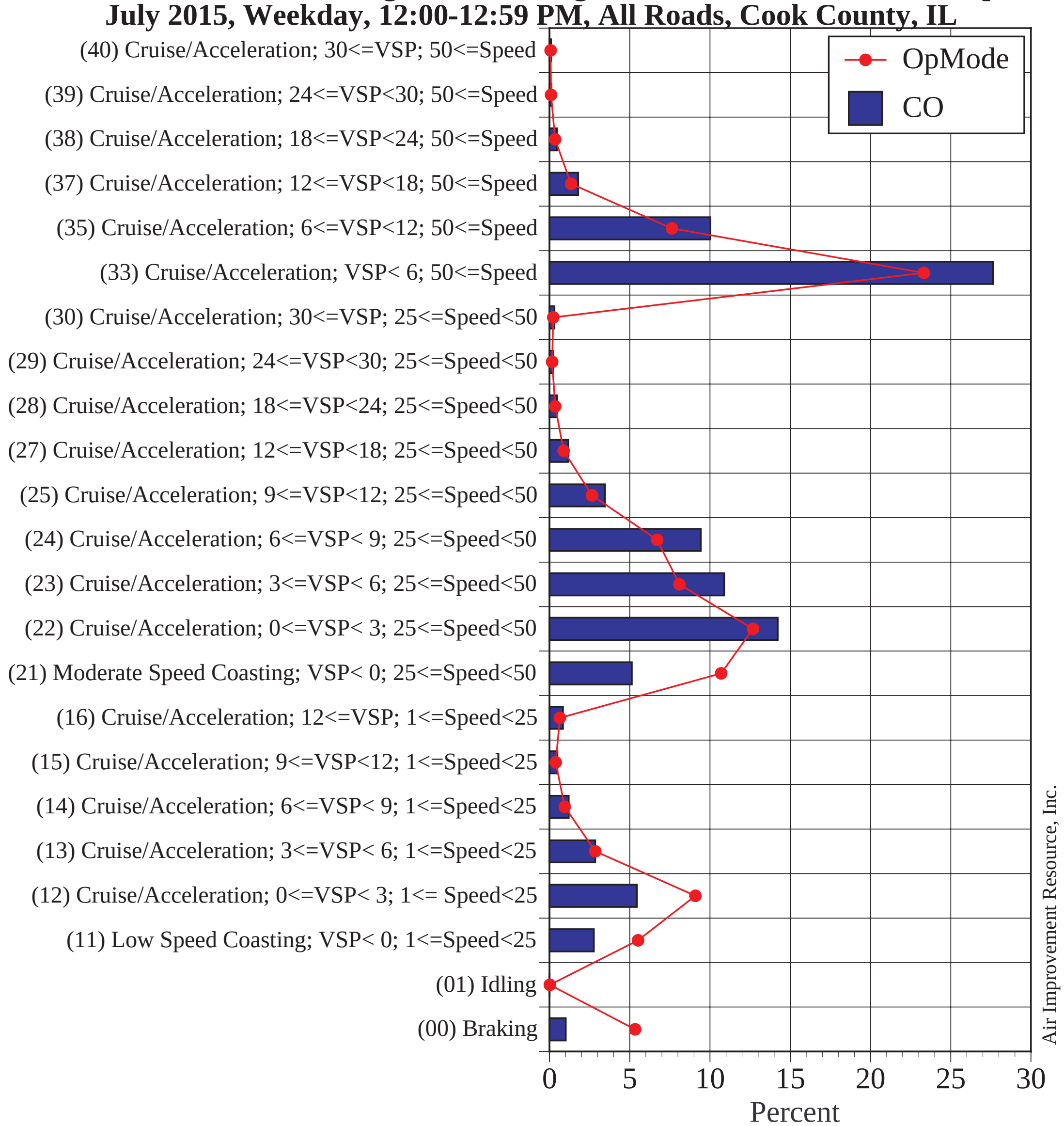
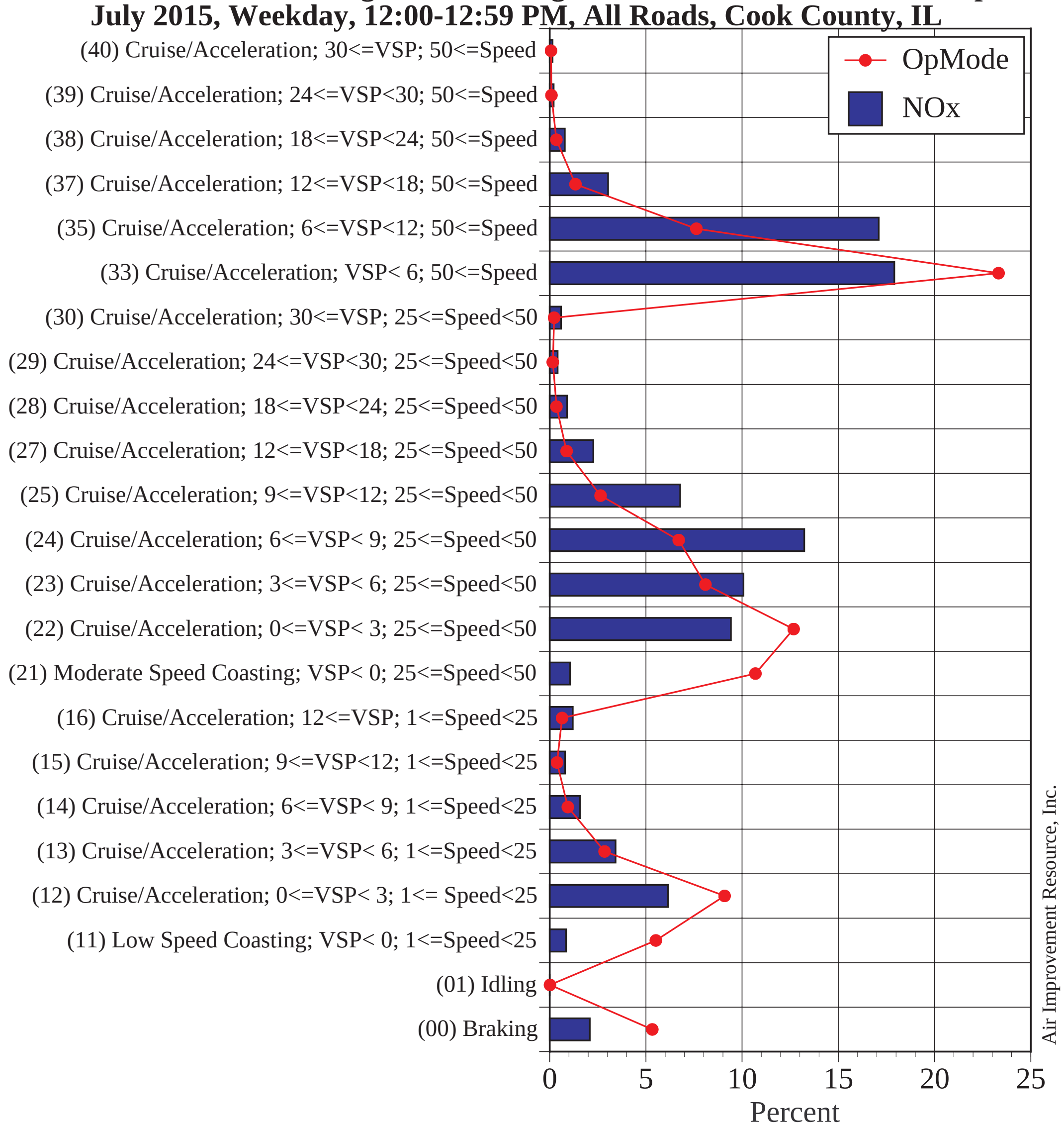


Figure 7-16

Percent of MY2011 Diesel Long Haul Running NOx Emissions and MOVES OpMode July 2015, Weekday, 12:00-12:59 PM, All Roads, Cook County, IL



7.9 Summary of Recommendations

Recommendations for the heavy-duty emissions follow:

1. EPA should include the effects of heavy-duty reflash programs that are done at the time of the first engine rebuild as well as those that have occurred earlier than required.

2. The impacts of OBD regulations should be included in the final MOVES model.

3. Idle emissions from diesels should incorporate state regulations controlling extended idles, and also the use of "hoteling" facilities. The draft model assumes all hoteling is done at high idle, where survey data indicate a mixture of high and lower idle operation. EPA should revise the emission rates for idle to reflect a split of high idle and lower idle operation, instead of all high idle operation.

4. Tampering and malmaintenance effects should not be included for HC, CO and PM emissions for pre-2007 model year vehicles, because the in-use tests that EPA bases the emission rates on already have some level of T&M. Alternatively, EPA should determine the level of T&M in the base emission rates and subtract this from the MOVES emission rates without T&M to avoid double-counting a T&M effect.

5. PM emission rates from 2007 and later model years need to be assessed; and most likely reduced from the assumptions used in the current version of MOVES. OC/EC ratios need to be measured and updated as well.

8.0 Correction Factors

This section is divided into the following subsections:

- Fuel Correction Factors
- Temperature Correction factors

There are no speed correction factors in MOVES as there are in MOBILE because speed is handled by the VSP binning approach. There are humidity correction factors in MOVES, but they are the same as in MOBILE so they will not be reviewed here.

The EPA FACA workgroup presentations utilized in this section include the following:

"Fuels and Fuel Effects in MOVES2006", Beardsley, August 8, 2006

"Analysis of CRC E-65 Data", AIR, Inc. August 16, 2007

"Approach for Modeling Evaporative Emissions in MOVES", Gururaja, June 26, 2007

"Air Toxic Emissions in MOVES", (no author), October 27, 2008

"Developing Draft Temperature Effects for Light-Duty Emission Rates for MOVES", Brzezinski and Landman, December 13, 2007

8.1 Fuel Correction Factors

This section discusses the fuel correction factors in MOVES, and is further divided into the following sections:

- Background
- Definition of fuel composition in MOVES
- Fuel Effects – Exhaust HC, CO, NOx Emissions
- Toxics
- Diesel Fuel Effects
- Biodiesel, CNG and E85

8.1.1 Background

The information in this section is based on material presented by EPA at a number of FACA meetings over the past few years, discussions with EPA staff at a meeting in Ann Arbor, and private communications with various EPA staff members.

EPA originally incorporated fuel effects into the MOVES model by adopting a binning approach for fuel properties. This approach is used in order to reduce the computing resources required to run the model. Fuels are defined categorically by type and subtype into hundreds of different bins intended to cover the range of compositions found in commercial gasolines.

Recently EPA has modified this approach and adopted an analytical approach to fuel effects in gasoline vehicles. Equations calculate the fuel effects relative to a base gasoline. The equations chosen are from previous EPA efforts and are meant to provide the best current estimate of fuel effects on exhaust emissions. EPA acknowledges that more recent data exist, and that more are currently being developed. They intend to update the equations in a more comprehensive way when current programs are completed, as part of the analysis mandated by EPACT.

The various models used or mentioned by EPA are:

EPA Complex Model – This model was published in 1993 and is based on emissions from 1990 technology light-duty vehicles. It includes exhaust emissions of VOC, NOx and toxics; and evaporative emissions of HC, including benzene (there is a separate Complex Model for CO). Normal emitters and high emitters are treated separately. The output of the model compares emissions of a test fuel to the Base Fuel defined in the Clean Air Act Amendments of 1990.

EPA Predictive Model – This model was originally created to evaluate the California request for a waiver to adopt CO_2 emission regulations. It is based on more recent data than the Complex Model but only predicts VOC and NOx. The model was revised when EPA evaluated the Renewable Fuel Standard (RFS).[41]

ARB Predictive Model (2006 version) – This model predicts emissions of a test fuel compared to ARB Phase 3 gasoline. It estimates exhaust emissions of NMOG, NOx, and toxics; and evaporative emissions of NMOG and benzene. The time period of the estimate is 2015, and all light-duty vehicles are considered in developing the model. No distinction is made in the model between normal and high emitters.

MOBILE6/MOBILE6.2 – This model is a precursor to MOVES and predicts inventory emissions for mobile sources. It is intended to be used by states in developing and evaluating plans to control emissions. Contained within MOBILE6 is a fuel module that estimates how emissions change as a function of RVP (Evaporative), sulfur, and ethanol (exhaust). The sulfur portion of MOBILE6.2 is described in an EPA report. [42]

[41] "EPA420-R-07-004, Regulatory Impact Analysis: Renewable Fuel Standard Program", USEPA, OTAQ, Assessment and Standards Division, April 2007.
[42] EPA420-R-01-039, "Fuel Sulfur Effects on Exhaust Emissions, Recommendations for MOBILE6",Venkatesh Rao formerly of EPA, July 2001.

8.1.2 Definition of Fuel Composition in MOVES

EPA determined fuel composition for each county in the U.S. for every month for every year that is modeled by MOVES (1990, 1999-2011, 2012+). The actual data used by EPA is almost always collected from only certain cities in January and July of every year. The intermediate months are interpolated by EPA based on the actual data and ASTM volatility regulations. In any given county and time, there may be multiple fuel formulations, which are associated with market share values.

Fuel properties are determined as follows:

1. 1990: Used Clean Air Act Amendments (CAAA) baseline fuel adjusted for ASTM RVP requirements. Sulfur was adjusted from the 1990 CAAA baseline levels based on the sulfur levels in the 1999 survey and the national average sulfur level in 1999.

2. 1999-2003: NMIM (National Mobile Inventory Model) survey data, with "RVP adjustments consistent with Renewable Fuel Standard (RFS) Notice of proposed Rulemaking (NPRM) baseline."

3. 2004-2006: NMIM values for sulfur, other values "consistent with RFS NPRM baseline."

4. 2007-2011+: Ethanol use "consistent with RFS NPRM scenario." For areas that are predicted to use significant amounts of E85, EPA estimated the average ethanol content. Emissions are based on this content.

One issue with the fuel properties is that EPA may not have used all available data, e.g., the fuel surveys compiled by TRW. The algorithms to define fuel properties should be reviewed in detail.

8.1.3 Fuel Effects - Exhaust Emissions of HC, CO, NOx

Fuel effects on exhaust emissions from gasoline vehicles are calculated from various models developed by EPA as shown in Table 8-1.

Table 8-1. Summary of Fuel Effects Methods		
Fuel Parameter	Emissions Component	Model used
Sulfur	HC, CO, NOx	MOBILE6.2 sulfur model. Includes short term, long term and irreversibility effects
RVP, E200, E300, Aromatics, Olefins, Oxygenates	HC, NOx	Pre-1994:EPA Predictive Model 1994 & newer: No effects
	CO	EPA Complex Model (all model years)

For areas with multiple fuels, MOVES calculates adjustment factors for each fuel and then averages the adjustment factors by sales. MOVES does not adjust emissions for other fuel or lubricant borne species such as P, Fe, or Mn.

8.1.3.1 Sulfur

The sulfur relationship is described in the Rao report cited previously. A number of important issues are identified, as follows:

- Use below 30 ppm
- Form of sulfur response
- More recent data
- Longer term effects
- Irreversibility
- Start and running emissions
- Normal/high emitters

Use below 30 ppm: According to Rao, the sulfur relationship for MOBILE6.2 is valid above 30 ppm only, because emissions data for LEV technology were available at the time only above 30 ppm. Most current sulfur levels are well below 30 ppm and will continue to be so in the future. The application of the MOBILE6 relationships and equations is uncertain in the range of interest. This is especially important considering the form of the relationship chosen for MOBILE6 (see below).

Mathematical Form of Sulfur Response: The form of equation chosen for MOBILE6 is log-log (for Tier 0 and LEV/ULEV vehicles), as shown below:

$$\ln(\text{Emissions}) = A * \ln(\text{Sulfur}) + B$$

This form differs from the one generally used when analyzing emissions data and in other models (e.g. EPA Complex Model, ARB Predictive Model), which is based on the log of emissions and a linear or quadratic term for fuel variables (log-linear or log-quadratic).

$$\ln(\text{Emissions}) = C * \text{Sulfur} + D$$

In the log-log type of equation. the response becomes very steep as sulfur levels approach zero. Since the original equation is not valid below 30 ppm, its use in this range may provide incorrect estimates and guidance on current and future emissions.

It should be noted that the decision to regress against the log of emissions stems from statistical considerations, not necessarily because it is a better fundamental form of equation derived from first principles of chemistry and kinetics. Using log of emissions provides a variable that has a more normal distribution of variance than the emissions themselves. This is commonly the case when there is a large range in the values of the variable. The Auto/Oil program made extensive use of this technique and it has gained acceptance in the industry for estimating fuel effects on emissions.

The most recent full analysis of sulfur effects was conducted during the development of ARB's updated Predictive Model in 2006. These efforts included new data as described below. ARB concluded that in the region of interest - below 30 ppm - the best representation of the sulfur response is a log-linear relationship. During the public discussions, Uihlein presented an analysis showing this type of model is appropriate down to extremely low levels. [43] An Alliance of Automobile Manufacturers (AAM) analysis also concluded that the response is linear or close to linear in the region of interest.[44]

Figure 8-1 provides a good example of this issue. It shows the relative NOx emissions for three cases, with emissions normalized to 1.0 at a sulfur level of 30 ppm. The first case (EPA PM LEV/ULEV) is a plot of the equation that EPA used in the Predictive Model to estimate the short term effect of sulfur on NOx emissions in LEV and ULEV light-duty vehicles (Rao report, page 23, Table 17). It is based on a log-log equation and is highly non-linear below 30 ppm. (The lowest sulfur level plotted is 1 ppm.) The responses of LEVs and ULEVs are particularly important because in the future, these vehicle technologies (and Tier 2) will dominate the population. The second case (ARB PM) is from ARB's Predictive Model. While the ARB Predictive Model is not approved for use above 30 ppm, the plot shows that the form of equation is well-behaved up to 120 ppm. The third case is based on the AAM analysis. The quadratic form of the AAM regression equation is shown to illustrate that even a non-linear equation does not necessarily exhibit the steep drop-off that is contained in the log-log equation. The WSPA analysis discussed above would show similar results, but the coefficients of the regression equation were not published.

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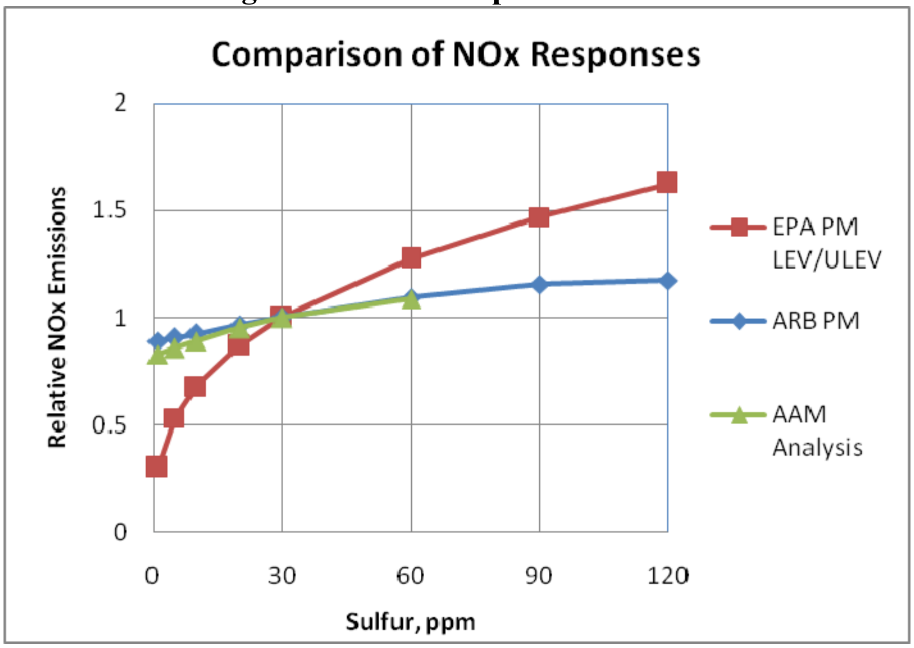
[43] James P. Uihlein, presentation at ARB Fuels Workshop, Sacramento, California, January 26, 2007, available at http://www.arb.ca.gov/fuels/gasoline/meeting/2007/012607wspaprstn.pdf

[44] Ellen Shapiro, presentation at the ARB Fuels Workshop, Sacramento, California, March 23, 2007, available at http://www.arb.ca.gov/fuels/gasoline/meeting/2007/032307_aam.pdf

Figure 8-1. NOx Response to Sulfur



More recent data: Since the Rao report, a number of programs have measured fuel effects at even lower sulfur levels, and these have been analyzed in a number of programs and publications. Two programs are the CRC E-60 program and the AAM/AIAM Industry Low Sulfur test program. [45,46]

Based on the experience developing the ARB Predictive Model, inclusion of these data in the database might change the slope of the response somewhat but the overall response would still be linear, as discussed above.

Long-term effects: The Rao report defined long-term effects for fuels with high sulfur levels. These effects are based on data comparing high sulfur fuel (350/450 ppm) with low sulfur fuel (30/40 ppm) in six 1998/99 LEV models. The long-term effects are applied only to LEV vehicles for sulfur levels above 30 ppm. It seems that this correction should depend on the sulfur level used, although this is not explicitly stated in the report.

In the future, when sulfur levels will be lower than 30 ppm, it is not clear how the long-term effect will be calculated. The data in Rao's report show no long-term effect at 30 or 40 ppm sulfur. That is, emissions did not degrade at all when mileage was accumulated at low sulfur levels. This would imply that if even lower sulfur levels were used, there might be a long-term benefit, in addition to the short-term benefit of lower sulfur use. While it is probably premature to include this effect in the sulfur response model, EPA should consider collecting more data in this area.

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[45] "The Effect of Fuel Sulfur on NH_3 and Other Emissions from 2000-2001 Model Year Vehicles", available at www.crcao.org.

[46] AAM/AIAM Industry Low Sulfur Test Program (Sulfur/Oxy Program) – available at http://www.arb.ca.gov/fuels/gasoline/carfg3/aam_prstn.pdf.

Irreversibility: Rao based the calculation of irreversibility on data collected by API suggesting that the latest technology vehicles do not show complete reversibility when low sulfur fuels are used following exposure to high sulfur fuels. EPA calculated an irreversibility factor for post-1994 cars based on the maximum sulfur level seen by a vehicle in its lifetime.

For the programs that measured irreversibility, data measured short-term effects only, and it is not clear that one batch of high sulfur fuel will increase emissions for the vehicle's entire lifetime.

A second issue to consider is that the irreversibility data are based on extremes in sulfur content, such as 30 ppm and 300 ppm. These levels will not be seen in the future, and EPA must necessarily assume that the effect will be linear when the differences between maximum and average sulfur levels are much smaller. More data in this area is also desirable.

Modeling start and running emissions: There are separate coefficients for start and running emissions for older technology (pre-LEV), but no distinction is made for LEV and newer technology. The lack of difference for newer technology reflects the fact that Rao did not carry out an analysis for this group.

For Tier 0 and Tier 1 vehicles, the coefficients for HC are almost two orders of magnitude lower for start emissions than for running emissions. For CO, the coefficient is negative for start emissions. This means that start emissions actually decrease when sulfur is increased. For NOx, the coefficients for start emissions are larger than for running emissions, meaning that the effect is bigger. The technical reasons for the CO and NOx results are not clear, and this approach should be adopted with caution. Overall, the regression results suggest that there may be some problem with the data or with the method used for splitting emissions into start and running fractions.

In the absence of a full data analysis, it is difficult to evaluate the validity of the assumptions for newer technology. Rao recognized this issue in his report, and did not differentiate between start and running emissions for new technology. In future years, when HC and CO emissions are almost all during start mode, this assumption may overestimate the sulfur effect on exhaust emissions.

Normal emitters/High emitters: Normal and high emitters are modeled separately. The only data available for high emitters are from two EPA programs with Tier 0 vehicles. (There are other data available from the AQIRP although these are apparently not used.) Rao suggested using the same relationships for Tier 1 and newer technology. The NOx response for high emitters is assumed to be 60% of the response of normal emitters; the HC and CO response is assumed to be the same for both classes.

In light of the paucity of data, it is appropriate to ask whether EPA's approach is valid for some portions of the fleet and not others, and whether the results from the older vehicles can be extrapolated to more modern vehicles that were designed to operate on low sulfur fuels.

8.1.3.2 Oxygenates

In their analysis of the Renewable Fuel Standard, EPA discussed a number of recent programs that measured the impact of oxygenates on exhaust emissions. These included programs by CRC (E-67), ExxonMobil, Toyota, AAM and AIAM. EPA carried out a statistical analysis of the results and concluded that most of the data supported a reduction in CO emissions when oxygenates were added. Results for HC and NOx were characterized as "not…sufficiently consistent" to confidently predict the impact of oxygen. This analysis is used as the rationale for including only oxygenate-CO effects for Tier 1 and newer vehicles.

This approach is somewhat questionable, especially for NOx emissions, since, as EPA noted, five out of six studies found that ethanol blends increased NOx emissions from LEV and later vehicles. One approach that EPA could adopt is to use the results of the analysis that they conducted and apply it to Tier 1 and newer vehicles. This would provide the best estimate, although not as precise as one would desire.

Considering the importance of ethanol in future gasoline composition, this is an area where more data collection would be helpful to future regulatory programs.

8.1.3.3 Other fuel parameters

In the current version of MOVES, the EPA Complex Model is used to model the relationships between fuel properties and CO emissions for all model years, and the EPA Predictive Model models HC and NOx emissions for model years 1993 and older. This is the approach taken by EPA in their RFS analysis.

The EPA Predictive Model is an updated version of the Complex Model that was originally developed to evaluate California's request to waive the RFG oxygen requirement. For the analysis of the RFS, EPA conducted an additional analysis of data on Tier 1 and newer vehicles to understand whether fuel impacts had changed. They included new studies such as CRC E-67 that evaluated T_{50}, T_{90} and oxygen content. No recent data are known which evaluated the effect of aromatics, olefins and RVP on exhaust emissions. CRC Project E-74b evaluated the effects of RVP and oxygen content on exhaust emissions. [47]

EPA conducted a statistical analysis of these programs and concluded that these parameters probably did affect exhaust emissions. They did not, however, carry out the full analysis of these newer data combined with the older data that would allow them to update the EPA models. They also recognized that there were few, if any, data on high

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[47] "Effects of Vapor Pressure, Oxygen Content, and Temperature on CO Exhaust Emissions", CRC Report No. E-74b, May 2009.

emitters for Tier 1 and newer vehicles. Faced with a choice of either assuming the same effects in Tier 1 and newer vehicles as in older vehicles, or assuming no effects, they chose the latter approach. The exceptions are the oxygenate effects on CO emissions and the sulfur effects on all exhaust emission components, both of which are described above.

The use of metal-based gasoline additives is not widely found in the United States. MOVES does not provide a means for assessing the impact of using metal-based gasoline additives such as manganese (Mn) or Iron (Fe), or non-metallic additives such as phosphorus (P). It may be necessary for EPA to provide guidance on how to model their impacts if their use is of concern to states or other entities. Reflecting the present lack of data to sufficiently incorporate such effects, data would need to be collected and reviewed to assess the effect of these additives.

8.1.3.4 Newer Technology

EPA has not included responses for most fuel parameters in the current version of the MOVES model. EPA recognized that they have not carried out a full analysis of recent data. ARB carried out such an analysis in development of the latest version of the Predictive Model, but their analysis and results may not be compatible with the approach that EPA has taken in the past.

Some recent data has been considered by EPA in their analysis of the ARB ethanol waiver request and the RFS mandate, but these dealt mostly with oxygen and the oxygen effects on CO have been incorporated into MOVES.

8.1.3.5 Modeling of Normal and High Emitters

EPA's current definition of a high emitter (for the purpose of estimating fuel effects) is a vehicle whose emissions are either greater than two times the standard for either HC or NOx, or greater than three times the standard for CO. While the contribution to the emissions inventory from high emitters is estimated to approach 50%, the amount of useful data on high emitters is considerably less than that available for normal emitters. The amount of data on fuel effects in Tier 1 and newer high emitters is even smaller than for Tier 0 and older vehicles. In addition, the statistical variability for high emitters is much higher than for normal emitters. These factors make it difficult to draw sound statistical conclusions from high emitters with respect to their responses to changes in fuel properties.

On the other hand, there are some valid reasons to model high emitters as a separate category, including:

1. Inspection/Maintenance programs may affect the number of high emitters in the future and could change the fuel responses.
2. Vehicles with inoperative catalysts would have different responses to some fuel parameters such as sulfur.

3. Vehicles with rich stoichiometry would have different responses to fuel parameters such as oxygen.

The ultimate decision about how to model the vehicle fleet should depend on both the statistical considerations and on the regulatory programs that might be impacted by the choice of approach.

There has been a great deal of discussion comparing the approach taken by EPA for dealing with high emitters with other approaches that deal with all vehicles tests in one group. EPA should carefully consider this issue when updating the Complex Model.

8.1.3.6 Summary of Issues for Fuel Effects on Exhaust Emissions

Sulfur - The model chosen for sulfur is not appropriate for use below 30 ppm, the region of future interest, and may significantly overestimate the impact of sulfur at these concentrations. In addition, there are important questions about modeling of high/normal emitters, and start/running emissions.

Oxygenate Effects – Data on the effect of oxygenates exist for newer technology, but are not incorporated into the model. The impact of the new data on the model is not clear.

Lack of fuel effects in Tier 1 and newer vehicles - The implications of EPA's choices mean that in the future, fuel composition - except for sulfur and oxygenates – will play a decreasing role in estimates of exhaust emissions inventories. This will continue until EPA has been able to update the Complex/Predictive Model, which is one of the tasks mandated by the Energy Policy Act of 2005 (EPACT).

Approach to modeling high and normal emitters - EPA's approach of splitting out the fuel effects for normal and high emitters suffers from a lack of quality data, and should be re-evaluated when the Complex Model is updated.

8.1.4 Fuel Effects In LDV/LDT - Evaporative Emissions

Fuel factors that affect evaporative emissions are RVP and ethanol. MOVES calculates the total amount of vapor generated by the fuel system using the Reddy equation, which requires knowledge of the RVP and tank temperature. Tank temperature calculation is discussed elsewhere in this report. In-tank RVP is estimated by considering RVP values of gasoline sold at service stations and subsequent weathering after purchase. Vapor emissions, or Fuel Vapor Venting, are a function of tank vapor generation.

As a result of the non-linear blending behavior of ethanol, if ethanol and non-ethanol gasolines mix, the resulting RVP can be equal to or higher than the RVP of either fuel individually. This commingling is taken into account in MOVES, but EPA did not provide enough details to fully evaluate the modeling.

The second component of evaporative emissions consists of liquid leaks, and these do not depend on fuel properties.

The third component of evaporative emissions is permeation, and this is affected by ethanol content. Permeation refers to the diffusion of fuel components through the fuel system materials such as elastomeric fuel lines and fuel tank walls. The largest body of data on permeation is from CRC's E-65 project. That project showed that ethanol in gasoline increases permeation emissions significantly. In the FACA meetings, EPA proposed that gasoline with 5.7% ethanol has permeation emissions that are 46% higher than gasoline with no ethanol and that gasoline with 10% ethanol increases permeation emissions proportionally (+79%). Based on more recent CRC data, EPA revised the permeation effect with 10% ethanol to be equal to permeation with 5.7% ethanol.

For enhanced evaporative emission control systems, MOVES estimates that E10 increases permeation emissions by 232%.

The fourth component of evaporative emissions is refueling emissions. EPA indicated in a meeting with CRC that refueling emissions are affected by changes in RVP in the same way as they are in MOBILE6.

8.1.5 Toxics Emissions

The following toxic compounds are defined by EPA: benzene, formaldehyde, naphthalene, MTBE, acetaldehyde, ethanol, 1,3 butadiene, and acrolein.

Emissions of toxics are calculated using the following formula:

Toxics Emissions = TR (Toxics Ratio) x THC Emissions

TR is calculated using methodology found in MOBILE6.2, which is based on the EPA Complex Model. The ethanol TR is based on MSAT data. [48] Evaporative emissions of toxics are mostly benzene and ethanol, and these are based on the same data as gasoline exhaust toxics.

Naphthalene emissions are a fraction of the PM10 emissions, defined in MOBILE6.2. Acrolein emissions are a fraction of VOC emissions as defined in MOBILE6.2. Ethanol emissions are a fraction of the fuel ethanol content.

The calculation of toxics emissions in the manner described above is a simplification consistent with the availability of data. Estimating toxics emissions from newer technology makes use of the Complex Model to calculate THC emissions for all technologies, and then applies a constant ratio to calculate individual toxic compound emissions. If emissions of individual toxic compounds are important from a regulatory

[48] "Draft Regulatory Impact Analysis: Control of Hazardous Air Pollutants from Mobile Sources", EPA420-D-06-004, February 2006.

perspective, then EPA should carry out an in-depth analysis of the data available and the best approach for modeling and developing predictive equations.

A large toxics program was carried out jointly by EPA and the Automobile Industry. [49] Nine vehicles of “approximately Tier 2 Bin 5 technology” were tested with five fuels. Four fuels varied in RVP, benzene, and sulfur, and the fifth fuel was a California RFG. Exhaust measurements were made of ten tailpipe toxics as well as HC, CO, NOx and CO_2. A complete statistical analysis was conducted and statistically significant differences were measured. These data represent the largest body of data on toxics emissions from Tier 2 LDVs. The results generally agreed with previous published data for earlier technology vehicles although the quantitative effects of fuel changes as a function of technology were not included in the analysis.

These data are not incorporated into the current version of MOVES. It is likely that their inclusion would not change the model drastically. However, it would be appropriate to update the toxics portion of the model to include these Tier 2 data when possible.

8.1.6 Diesel Fuel Effects

The only diesel fuel parameter considered in the MOVES model is sulfur; this affects only the sulfate portion of PM emissions. The form of this relationship was not specified by EPA at the FACA meetings. The impact of biodiesel components such as fatty acid esters is not included in the model.

A number of attempts have been made to develop a diesel emissions model.[50,51] EPA staff felt that these models are not suitable for use in this version of MOVES.

8.1.7 Biodiesel, CNG, and E85

Compressed natural gas vehicles included in the model are transit buses only. Currently they are assumed to have the same emissions as gasoline buses. E85 vehicles are assumed to have the same emissions as gasoline vehicles on E10, at all ethanol volume percents. The correction factors for biodiesel (B20) are shown in the table below, and were developed from “recent EPA testing.”[52]

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[49] “Joint EPA-Automobile Industry Tier 2 Vehicle Fuel Effects Test Program Final Report”, December 15, 2006, EPA-HQ-OAR-2005-0036-1160.pdf, available at www.regulations.gov

[50] EPA420-P-01-001, "Strategies and Issues in Correlating Diesel Fuel Properties with Emissions", U.S. Environmental Protection Agency Office of Transportation and Air Quality. July, 2001; EPA420-P-02-001

[51] “A Comprehensive Analysis of Biodiesel Impacts on Exhaust Emissions”, U.S. Environmental Protection Agency Office of Transportation and Air Quality, October 2002

[52] AIR is still trying to determine the data these are based on, and whether they apply at B20 or B100. The reference for these at this time is “MOVES2009 Fuel Effects Update”, for MOVES Review Workgroup, September 14, 2009.

Table 8-2. Biodiesel Factors Used in MOVES	
Pollutant	Percent Change
HC	-14.1%
CO	-13.8%
NOx	2.2%
PM	-15.6%
Napthalene	-15.6%
OPther air toxics	-14.1%

8.2 Temperature Correction Factors for Exhaust HC, CO, NOx

This section discusses the temperature correction factors for exhaust HC, CO, and NOx. EPA identified several sources of test data for developing temperature correction factors for MOVES, as shown below:

- MSOD data (FTPs mostly used for MOBILE6 and US06 tests)
- Kansas City program (LA-92s used)
- Two small EPA testing programs (ORD and SwRI)
- MSAT data (Tier 2 vehicles and LEVs)

From these data, EPA selected only tests from vehicles that are tested at multiple temperatures. EPA initially selected the following model year groups:

- 1960-1980
- 1981-1982
- 1983-1985
- 1986-1989
- 1990-2004
- 2005+

However, in response to comments, EPA expanded the 1990 and later model year groups to:

- 1991-1993
- 1994
- each year individually (from 1994-2020), until 2021+

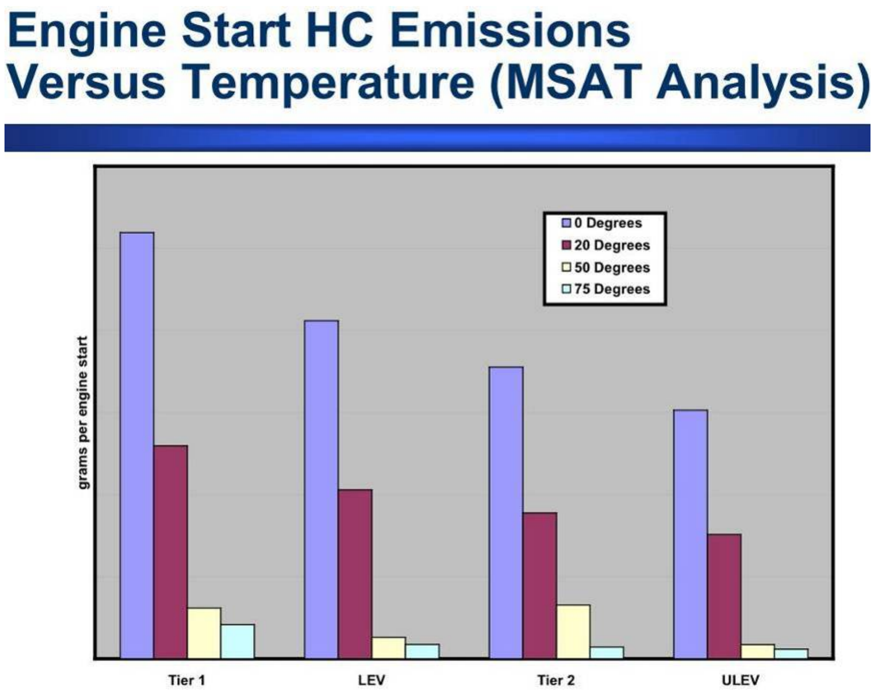
For the analysis phase, EPA estimated cold start emissions for each test by subtracting Bag 3 from Bag 1 (both FTP or LA-92). EPA also estimated hot running emissions, and then performed regression analysis on each model year group. The preliminary analysis showed little variation in the running emissions of HC, CO, and NOx, so EPA's proposal in MOVES is that there are no running emission temperature adjustments for HC, CO, and NOx. The authors believe this is appropriate.

In MOBILE6, CO emissions at temperatures below 75°F were modeled with an additive adjustment that increased as the temperature decreased. HC and NOx in MOBILE6 at lower temperatures are modeled with a multiplicative adjustment. For MOVES, EPA decided on an offset adjustment for all three pollutants. The reasons for this are:

- Additive adjustments make sense when the additional emissions are not necessarily proportional to the base emission rate, and the quantity of additional emissions is much larger than the base emission rate, and
- Additive adjustments prevent small changes in the base emission rate from causing massive changes in the temperature-adjusted emissions

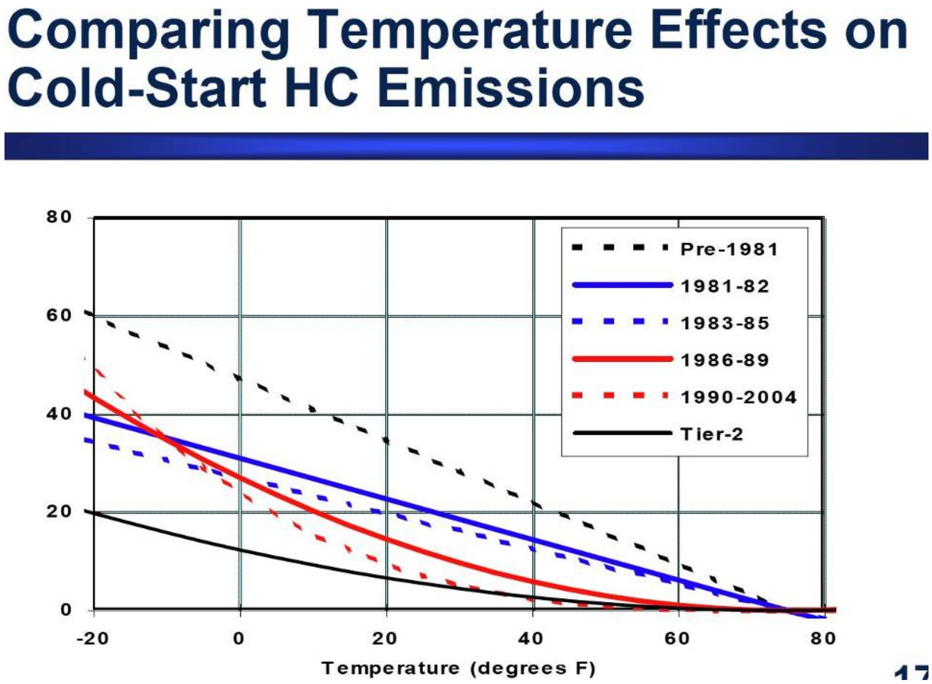
Figure 8-2 shows start HC emissions from the MSAT analysis for Tier 1, LEV, ULEV, and Tier 2 vehicles. The base emissions at 75°F are extremely low compared to the increase in emissions from 75° to 20°F, especially for LEV, Tier 2 and ULEV.

Figure 8-2. Engine Start Emissions



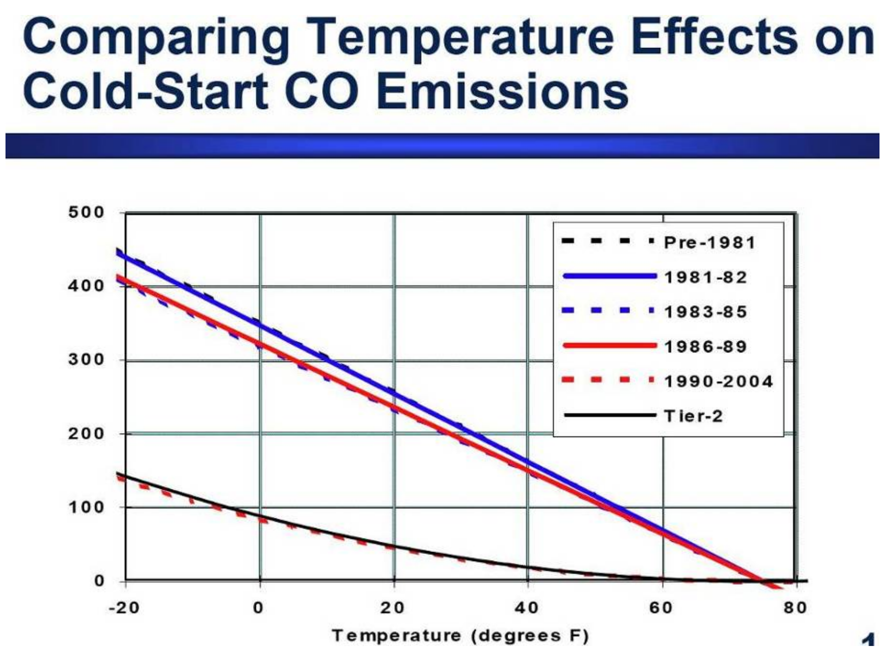
EPA's analysis of the cold temperature effects for HC, CO, and NOx is shown in Figures 8-3, 8-4, and 8-5. For HC at 20°F, the increase in HC for Tier 2 vehicles is about 8 grams from the level at about 75°F. For the earlier vehicles, it is much higher at 10-35 grams. The Tier 2 vehicles in this case are those that do not meet the MSAT cold HC standards. Grouping all 1990 – 2004 vehicles together may mask the fact that later model vehicles in this time period used close-coupled catalysts and other strategies to reduce light off times.

Figure 8-3. HC Temperature Effects (grams/cold start)



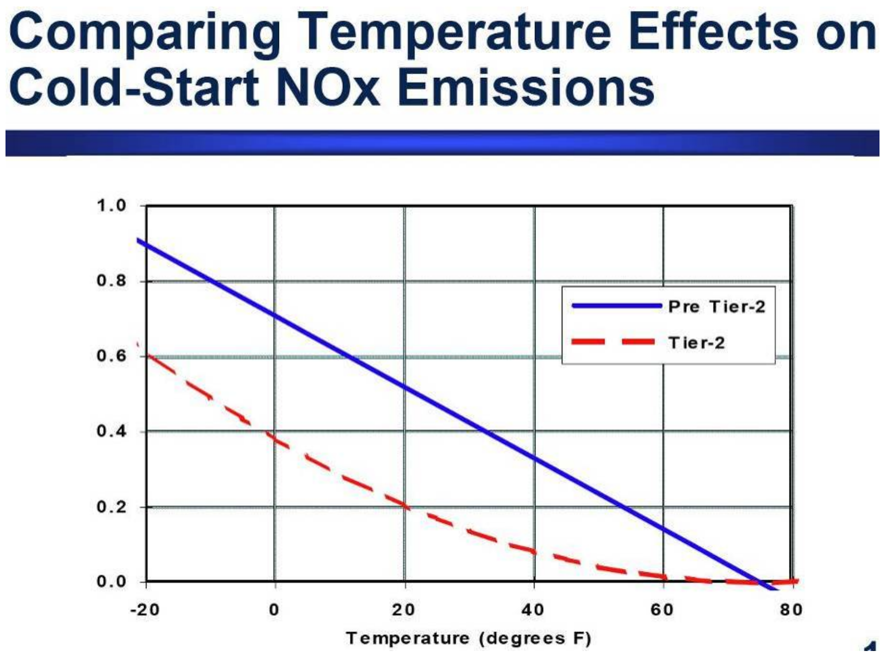
CO increases for both 1990-2004 and Tier 2 vehicles are estimated to be around 50 grams at 20°F. Emissions from the earlier model years are much higher. It is interesting that the CO emissions increases from Tier 2 vehicles are not lower than Tier 1 vehicles, when the Tier 1 vehicles have a much higher HC standard. It would have been expected that the Tier 2 vehicles would have lower CO increases than Tier 0 and Tier 1 vehicles. This is an area that should be investigated further.

Figure 8-4. CO Temperature Effects (grams/cold start)



NOx emission increases are shown in Figure 8-5. At 20°F, the Tier 2 NOx increase is 0.2 grams, and for the earlier model years is 0.5 grams or so.

Figure 8-5. NOx Temperature Effects (grams per cold start)



There are two factors to note about the cold start temperature correction factors. One is that EPA did not evaluate whether these temperature correction factors change as vehicles age: the temperature correction factors are assumed to be the same for all vehicle ages. The Kansas City data would seem to be a good data source to try to examine the age issue for temperature correction factors. Second, the CRC E-74b testing program, the results of which were released in May 2009, tested vehicles at 75°F and 50°F (see reference 48). These data could be used to check the temperature correction factors for MOVES, or could be incorporated into the other data sources for the purpose of estimating the temperature correction factors.

8.2.1 Cold Weather CO Requirement

The cold weather CO requirement for the 1994 and newer model year LDVs and LDTs limits the composite FTP CO to 10.0 g/mi at a temperature of 20°F. However, many of the vehicles used to estimate the cold temperature correction factors are certified to this requirement, so the effects of the 10 g/mi cold CO standards do not need to be incorporated separately.

8.2.2 Cold Weather HC Requirement for MSAT Rule

The MOVES model does include the MSAT rule low temperature HC standards. The MSAT-2 rule included a limit on low temperature NMHC emissions for light-duty and some medium duty gasoline-fueled vehicles. For passenger cars and for light trucks with

GVWR up to 6,000 lbs, the composite FTP NMHC emissions at 20°F should not exceed 0.3 g/mi. For heavy light trucks the composite FTP emissions at 20°F should not exceed 0.5 g/mi. The phase-ins for these standards are shown in Table 8-2.

Table 8-2. Phase-in Percents for MSAT-2 HC Standards at Low Temperature[53]

Model Year	LDVs/LDTs (up to 6,000 lbs)	HLDTs/MDPVs (6-8,500 lbs)
2010	25%	0%
2011	50%	0%
2012	75%	25%
2013	100%	50%
2014	100%	75%
2015	100%	100%

To incorporate these standards, EPA evaluated the emissions of one Tier 2 passenger car and three LDTs. The average NMHC emissions of these vehicles at 75°F are 0.02 g/mi for the passenger car and 0.04 g/mi for the LDTs. EPA then assumed that these vehicles would just meet the 0.3 and 0.5 standards, such that the passenger cars and trucks less than 6000 lbs GVWR would experience a 0.28 g/mi increase in HC emissions (0.3-0.02) and the heavier trucks and medium duty passenger cars would experience an increase of (0.5-0.04) 0.46 g/mi. In other words, even though there is a margin with respect to the NMHC standard for these vehicles at 75°F, EPA assumed no margin at 20°F. EPA then estimated the HC increases per degree decrease in temperature.

Our main comment is that to estimate these increases, EPA should have included about the same percent margin at 20°F as is indicated at 75°F. Just as manufacturers design with a margin at 75°F, they will do the same at 20° F. If one assumes a 30% margin at 0.3 and 0.5, this would be levels of 0.21 g/mi and 0.35 g/mi. The NMHC increases therefore would be 0.21-0.02 = 0.19 g/mi and 0.35 – 0.04 = 0.31 g/mi, respectively. This seems a more appropriate way of accounting for the lower temperature HC standards than the method used by EPA. Since the PM cold start corrections are estimated from correlations with the HC cold start corrections, this would affect the cold start PM temperature correction as well. The authors are certain this would have a significant effect on start HC and PM emissions at cold temperature for vehicles certified to these cold HC standards.

In addition, the cold HC standards will also reduce cold temperature CO emissions as well, although a method of accounting for this impact has not yet been developed.

Additional information from EPA on the cold temperature correction factors is presented in Appendix E.

8.3 Summary of Recommendations

[53] Federal Register: February 26, 2007,Volume 72, Number 37)

The following are the recommendations on the correction factors:

1. The sulfur response below 30 ppm used in MOBILE6 should not be used for MOVES. A new response should be developed from the available data under 50 ppm.

2. EPA should review the NOx effects of ethanol, since there are data available for this right now.

3. EPA should utilize the Kansas City data to determine whether temperature correction factors change with vehicle age. Also, the CRC E-74b testing program data could be used to further check the MOVES cold start correction factors and volatility adjustments (RVP).

4. The Tier 2 cold temperature response should be lower than for Tier 1 vehicles. In addition, the MSAT rules should reduce CO emissions as well as HC emissions.

5. The method used to develop HC temperature correction factors for the MSAT rule should be revised to include a compliance margin at 20° F to be consistent with the margin currently being utilized at 75°F.

9.0 PM Emissions – Light-duty Gasoline Vehicles

This section is divided into the following subsections:

- Exhaust Particulate Matter (PM)
- Non-Exhaust PM
- Recommendations

The following presentations to the FACA workgroup are utilized in the development of this section:

"Kansas City Particulate Matter Exhaust Emissions Study from Light-Duty Gasoline Vehicles – Part 1, Preliminary Results", Fulper, Nam, August 16, 2007

"Kansas City Particulate Matter Exhaust Emissions Study from Light-Duty Gasoline Vehicles – Part 2, Preliminary Analysis", Fulper, Nam, August 16, 2007

"Kansas City Particulate Matter Exhaust Emissions Study from Light-Duty Gasoline Vehicles – Part 3, Preliminary Hot Running Real Time PM Results", Fulper, Nam, August 16, 2007

"MOVES Gasoline PM Deterioration Rates", Nam, April 24, 2008

"Modal Analysis of Real Time Particulate Matter Data from Light-Duty Gasoline Vehicles in Kansas City", Nam, April 24, 2008

9.1 Exhaust PM

MOVES is designed to inventory gasoline vehicle exhaust PM where the PM is defined by current measurement methods. EPA recognizes that the current measurement methods may not reflect real-world PM emission rates. EPA stated in the April 24, 2008 FACA presentation by Ed Nam entitled "Modal Analysis of Real-Time Particulate Matter Data from Light-Duty Gasoline Vehicles in Kansas City" that:

> "PM is a dynamic pollutant that is constantly being influenced by its environment therefore its formation is constantly changing both in the exhaust stream and in the ambient air. Our tests are a snapshot using specific measurements under specific laboratory and thermodynamic conditions. Real-world PM may differ significantly."

Currently, gasoline exhaust PM from light-duty vehicles is measured by collecting a filter sample from exhaust diluted in a single stage dilution tunnel. Tunnels typically are operated in a constant volume mode at a flow rate that gives an average dilution ratio of approximately 10:1 for an FTP cycle. Current procedures specify that the sample filter be held at a temperature of 47 ±5 °C. Cold temperature testing requires that the tunnel dilution air be heated sufficiently to keep the relative humidity below 100%. Dilution air is filtered to remove background PM. These conditions are obviously quite different from real world conditions where the exhaust is rapidly diluted by a factor of a thousand at ambient temperature in an atmosphere with significant concentrations of PM.

MOVES calculates exhaust PM mass emission rates for $PM_{2.5}$ and PM_{10}. It does not calculate mass emission rates for smaller particle mass size fractions and it does not calculate particle number emission rates. It also does not include an estimate of secondary PM due to atmospheric transformations of PM precursors such as SO_2, NOx, NH_3, or HC.

9.1.1 Kansas City PM Exhaust Emissions Study from Light-Duty Gasoline Vehicles

Exhaust PM emission rates from gasoline vehicles have always been significantly below the emission standards. Hence, there has been limited interest in making measurements of in-use PM emission rates until recently. The most recent study was done in Kansas City (KC).[54] Since the results of that study are the primary basis for the data in MOVES, it is briefly described.

The KC study was planned to measure PM mass emission rates and the regulated emissions from in-use light-duty gasoline vehicles. Kansas City was selected as the largest non-I/M metropolitan area in the U.S. It does not use reformulated gasoline. Vehicles were recruited by telephone solicitation using a previous survey and registration data. The goal was to make vehicle recruitment as random as possible. Two hundred and forty vehicles were recruited for testing in the summer of 2004, and 240 vehicles were recruited for testing during the winter of 2005. Of the winter vehicles, 43 were vehicles that had been tested in the summer. Recruitment was designed to populate four age bins for light-duty gasoline cars and the same four bins for trucks. Recruitment in the summer failed to adequately fill the oldest age bins (pre-1981). Hence these vehicles were not well-represented in the summer testing.

Testing was done on a portable chassis dynamometer housed in a warehouse with no temperature control, i.e., vehicles were tested at ambient. All vehicles were tested as received on the LA92 (unified cycle) after preconditioning on a local road route. Vehicles were soaked a minimum of 12 hours before testing. Filter samples were collected for the three phases of the test cycle. Fuel and oil samples were collected from vehicles. Analysis of the fuel showed that most vehicles used non-oxygenated gasoline. Results were compared to previous studies and it was concluded that they were similar. It should be noted that no very high PM emitting vehicles were found during the summer tests. Only four vehicles had PM emission rates higher than 100 mg/mi. Previous programs had tested vehicles with considerably higher PM emission rates. It is unknown if these didn't exist in the KC fleet or if 240 vehicles sample was too small of a sample to

[54] Kansas City PM Characterization Study, Final Report, EPA420-R-08-009, April 2008. Prepared by Eastern Research Group, revised by EPA. "Analysis of Particulate Matter Emissions from Light-Duty Gasoline Vehicles in Kansas City", EPA420-R-08-010, April 2008.

catch the highest emitters. Overall, it was found that 50% of the PM emissions came from 13% of the vehicles on a fleet weighted basis. The study did compare results from a subset of vehicles whose owners originally refused to participate to those who didn't refuse initially. No difference was found in the emission rates. Extensive speciation was conducted on samples from 50 of the vehicles.

9.1.2 Modal PM Emission Rates

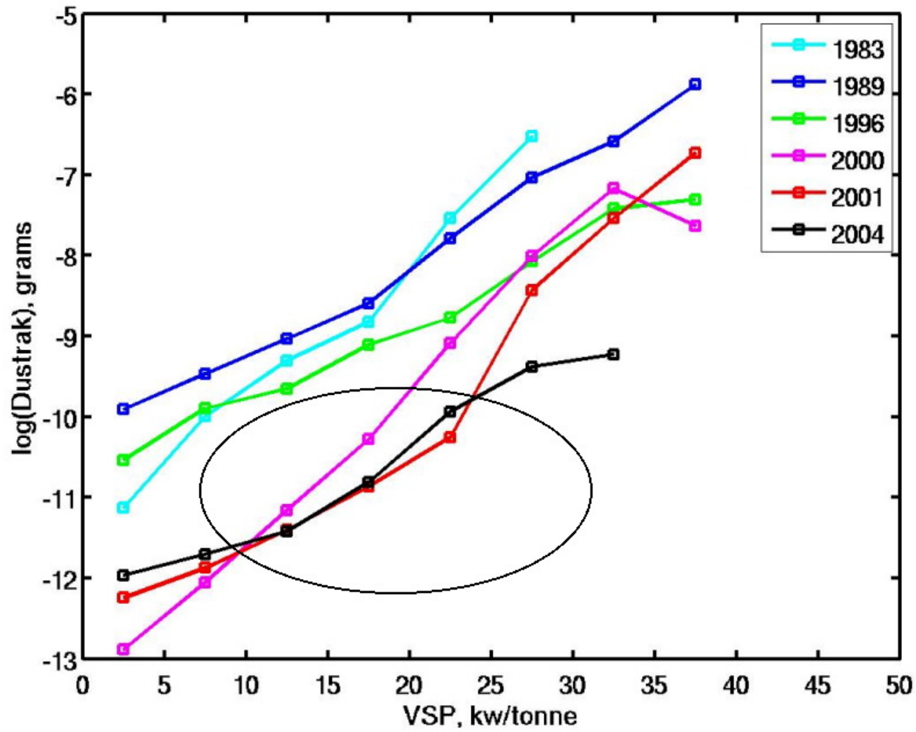
Real-time PM mass emission rate data are required to determine the binned MOVES rates. The Kansas City study is the only large-scale program that has attempted to collect real-time mass emission rates from in-use gasoline vehicles. Three instruments were used during dynamometer testing, the quartz crystal microbalance (QCM), the DustTrak, and the Dataram. The QCM senses changes in mass deposited on a vibrating quartz crystal. One problem is that water and semivolatile compounds can be deposited on the crystal at one point during the driving cycle, but then lost due to evaporation at another time. In this program, the QCM was quite noisy and needed 10-second averaging. Hence, it is only used as a check on the other instruments. Nephelometers use light scattering to detect particles suspended in a gas stream. The Dataram, a two-wavelength nephelometer, had poor correlation to filter data and occasional unexplained large swings in its response. Hence, it is not used. The DustTrak is a single wavelength, side scatter nephelometer. Out of the three instruments the DustTrak gave the most rational real-time response to PM in the KC study and hence, was selected as the primary instrument to be used to apportion the filter mass to VSP bins.

It must be emphasized that the response of nephelometers is highly dependent on particle size and significantly dependent on composition. Strong light absorbing particles give a much larger response than weak light absorbing aerosols in the size range typically found for exhaust PM. Previous studies have suggested that the DustTrak can be within a factor of 2 of the filter mass. If there is no variation of the particle size distribution or the composition of the particles during an emission test, then the DustTrak should give an accurate means of apportioning the mass to vsp bins. Undoubtedly, some variation does occur. However, this is currently the best available method and the only data available for binning PM emissions.

Examination of individual vehicle real-time PM data is instructive. A strong spike in PM emissions at 840 sec (rapid acceleration to approximately 65 mph in the cycle) is commonly present. This is found to be useful in time alignment of the data from various instruments. For some vehicles, high PM during the cold start appeared to extend into bag 2. At the time of this report, this PM was being counted as part of the hot running PM emissions, although including it in the cold start portion of the PM was being considered. Forty tests had a broad (time-wise) release of PM that had no elemental carbon. It appeared to be the release of oil caused by the heating up of the exhaust and sampling system. Other vehicles had broad-based OC emissions later in the tests while under load and appeared to be oil burners. Not surprisingly, vehicles with these broad-based PM emissions did not have the same VSP emission rate profile as the general fleet.

The VSP emission rates developed are shown in Figure 9-1. The plot indicates that the PM emission rate increases exponentially with increasing VSP and that the response is similar for different model years, with the possible exception of 2001 and 2004 model years (see the circled area).

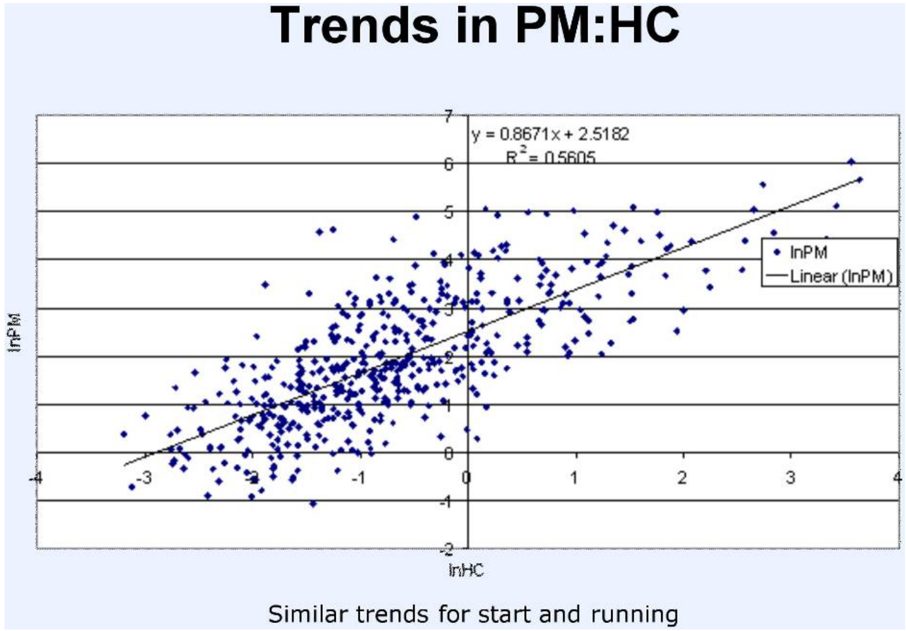
Figure 9-1. Relationship between PM emissions as determined by the DustTrak and VSP



9.1.3 PM Zero Mile Levels and Deterioration Rates

Deterioration rates are best determined from longitudinal studies, i.e., those studies that follow a set of vehicles over time. Unfortunately such studies are not available for PM. In the absence of longitudinal studies, EPA developed a model that uses zero mile emission rates (ZMLs) and then uses deterioration estimated from Kansas City results along with guidance obtained for HC deterioration rates. HC deterioration rates show ZML offsets between model years, deterioration rates that have parallel slopes in log space, and a leveling out after 12 years. The logic behind being guided by HC deterioration rates is that there is a reasonable correlation ($R^2 = 0.56$) between PM emission rates and HC emission rates as shown in Figure 9-2. It is recognized that this correlation is far from ideal, but is considered the best available data for guidance at this time.

Figure 9-2. Relationship between HC and PM emission rates from vehicles measured on the LA92 in the Kansas City study.

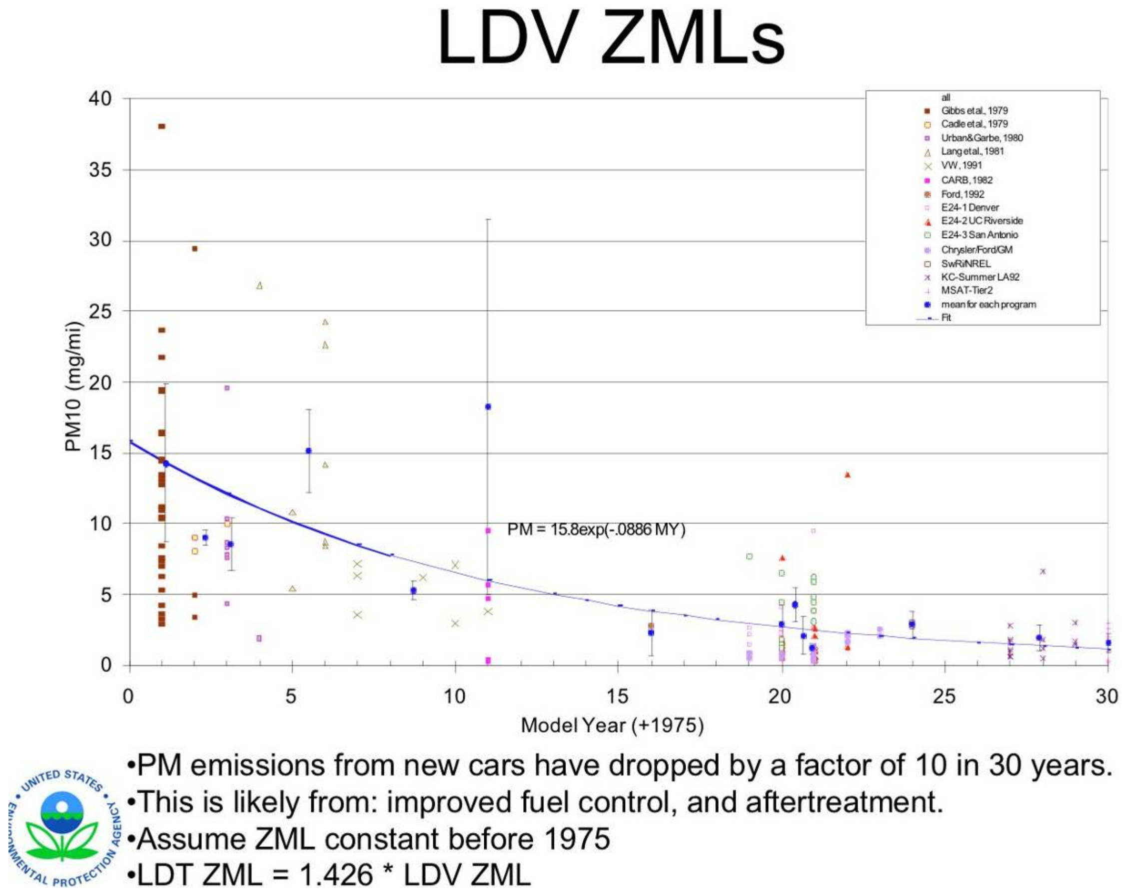


ZMLs were obtained from 15 studies conducted between 1978 and 2006. A curve is fit to the ZML data back to 1975. It is assumed that ZMLs for pre-1975 vehicles are the same as those in 1975. ZMLs were found to have dropped by a factor of 10 over this period. It should be noted that both fuels and test methods have changed during this time.

While it is appropriate to use the ZMLs for older vehicles in retrospective studies, the older vehicle ZMLs are probably overstated for the present time due to several factors. First, a large fraction of the PM from cars equipped with oxidation catalysts and those with dual bed catalysts and air pumps was sulfuric acid and the associated water. Reduction in fuel sulfur should have reduced these emissions. In their analysis, EPA removed the mass of sulfate from the reported PM emission rates, but did not consider the mass of the associated water. Second, many studies in the past did not use a cyclone to exclude larger particles that can originate from catalyst attrition, exhaust system rust, or from re-entrained deposits in the exhaust system. Since cyclones are now used, ZMLs would be expected to be somewhat lower today. Also, heated filter holders were not used. Heating might reduce the PM, although this hasn't been tested. Offsetting this is the fact that filter materials have changed. Older filter materials had higher positive interference (artifact formation) than the currently used Teflon membrane filters. Two of the studies had results using the LA92 while the rest were based on the FTP. EPA concluded that the results are not influenced by any difference in emissions between the two cycles. Furthermore, it is assumed that $PM_{2.5}$ emission rates are 90% of PM_{10} rates. This may not be correct for older studies if large particles are present.

Figure 9-3 shows the data and the resulting curve fit. Note that EPA assumes LDT ZMLs are 1.42 times those of LDVs. This value comes from an EPA analysis of the data from the studies used to generate the ZMLs (see FACA presentation "MOVES Gasoline PM Deterioration Rates", Nam, April 24, 2008). In terms of vehicle model year, 1975 is indicated on the graph as year 0.

Figure 9-3. Zero mile emission rates of exhaust PM as a function of vehicle model year.



Deterioration rates are developed by fitting slopes to log transformed data going from the ZML to data from Kansas City vehicles older than eight years. It was decided to model deterioration rates of future vehicles as being the same as those in MY 2002 since EPA has found no evidence to suggest that ZMLs drop for newer vehicles. Note that EPA assumes that PM emission rates continue to deteriorate out to 20 years, as compared to 12 years for HC. This conclusion was drawn from the Kansas City data shown in Figure 9-4. It is not clear how much of the deterioration in Figure 9-4 is due to vehicle age and how much is due to differences in the technology of the vehicles. Currently, there is no way to separate these factors. Presumably a longer deterioration period can be justified by the likelihood that PM will continue to increase due to increased oil consumption as vehicles accumulate more VMT. However, this has not been proven. Eventually, if hybrids become a significant fraction of the fleet, their PM emissions will have to be factored into the model. Currently, there are no efforts to do so.

Figure 9-4. PM emission rates determined in Kansas City as a function of vehicle model year.

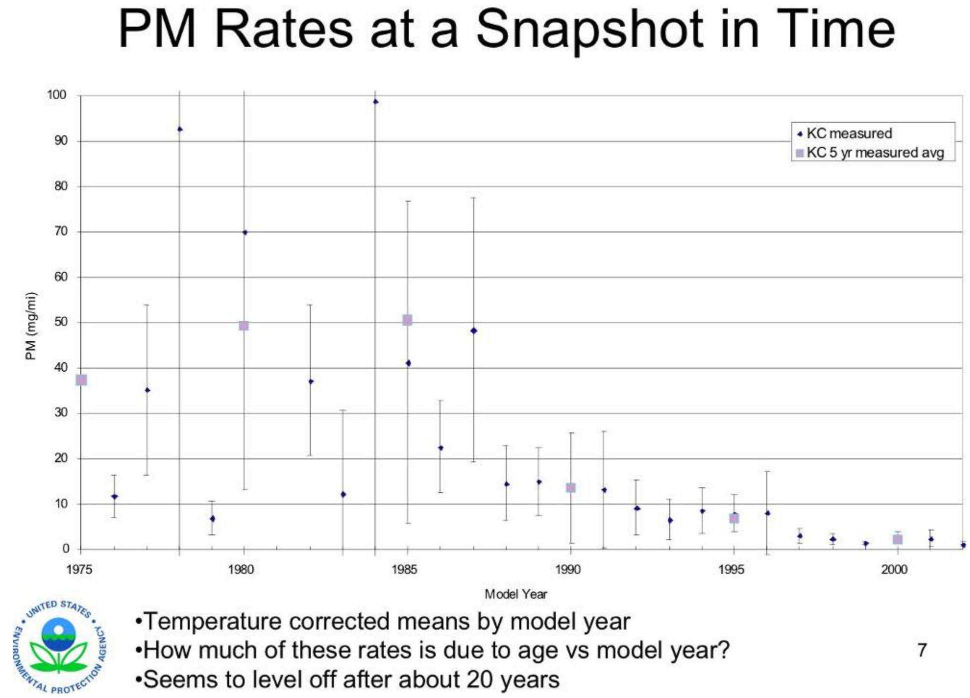
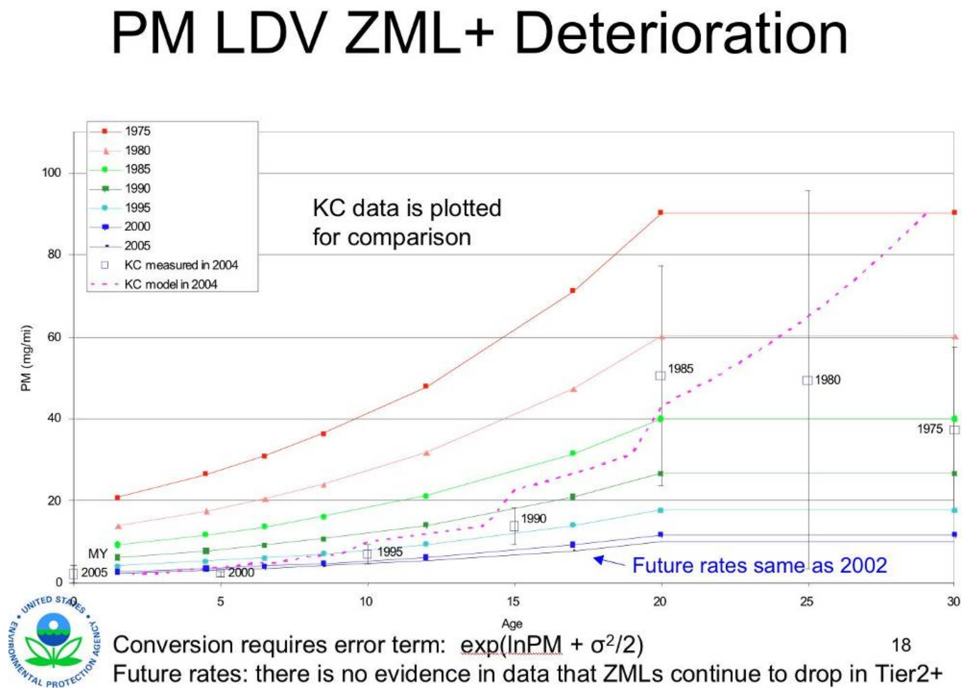


Figure 9-5 shows the resulting deterioration model along with the model that fits the data obtained in the Kansas City Study. The model projects that in 20 years current vehicles will have increased their average PM emission rates from 2 mg/mi to 10 mg/mi under standard test conditions, i.e. approximately 75°F.

Our analysis of HC, CO, and NOx emission rates for Tier 1 and later vehicles indicates that there is little evidence that there is a log-linear relationship between emissions and age.

Figure 9-5. Model for PM emission rate deterioration as a function of vehicle age



9.1.4 Temperature Effects

Three studies were examined to estimate low temperature impacts on PM emissions rates. The Kansas City study had average temperatures of 45°F in the winter and 76 °F in the summer. Data were examined for the vehicles tested in both seasons and also for the control vehicle that was tested 22 times during the study. Quality checks limited the number of matched pair vehicles with valid cold start data to 33. The control vehicle was a 1988 Ford Taurus. The EPA ORD program tested eight 1987 – 2001 MY vehicles on the FTP at five temperatures ranging from -20°F to 75°F.[55] EPA/OTAG tested 4 Tier 2 vehicles on the FTP at three temperatures from 0°F to 75°F in support of the MSAT regulations.[56] Decreasing temperature raised the PM emission rates 3.3, 3.8 and 3.5% per °F for the three data sets. Figure 9-6 shows the comparison for the best fit through the Kansas City data to the other two recent studies. The exponential slope for the fit is -0.0334. The slopes for cold and hot running are -0.0381 and -0.0299, respectively. Comparisons to earlier studies were not presented. The fits for cold and hot running from Kansas City are being used in MOVES to model the temperature response.

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[55] "Characterization of Emissions from Malfunctioning Vehicles Fueled with Oxygenated Gasoline," (Parts I, II & III) Fred Stump, Silvestre Tejada, and David Dropkin, National Exposure Research Laboratory, U. S. Environmental Protection Agency and Colleen Loomis, Clean Air Vehicle Technology Center, Inc.

[56] "VOC/PM Cold Temperature Characterization and Interior Climate Control Emissions/Fuel Economy Impact," Draft Final Report Volume I, EPA Contract 68-C-05-018, Work Assignment No. 0-4, SwRI Project No. 03.11382.04, Alan P. Standard, October 2005.

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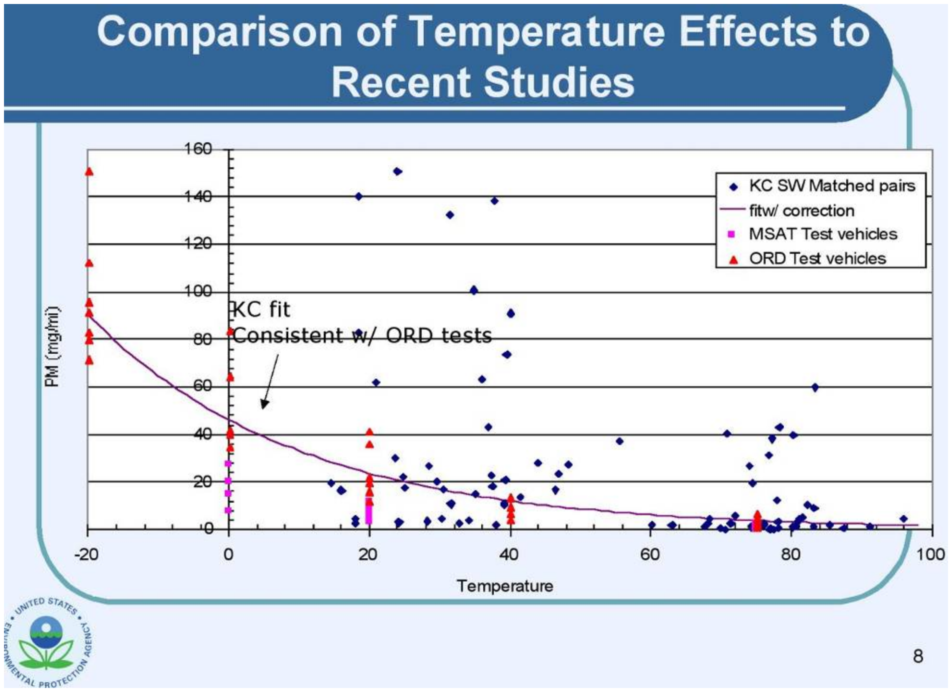
Figure 9-6. Effect of ambient temperature on the PM emission rate from gasoline LDVs



The fact that the Kansas City temperature response is similar to the two EPA studies suggests this correction approach is robust. However, it should be noted that the highest emitter included in this analysis for Kansas City for the summertime had a PM emission rate of 60 mg/mi. Much higher PM emitters have been identified in earlier studies. It isn't clear that a vehicle with high PM emissions due to oil consumption would have the same temperature response as a properly functioning vehicle. While such vehicles aren't common, they probably make a significant contribution to the fleet average PM emission rate.

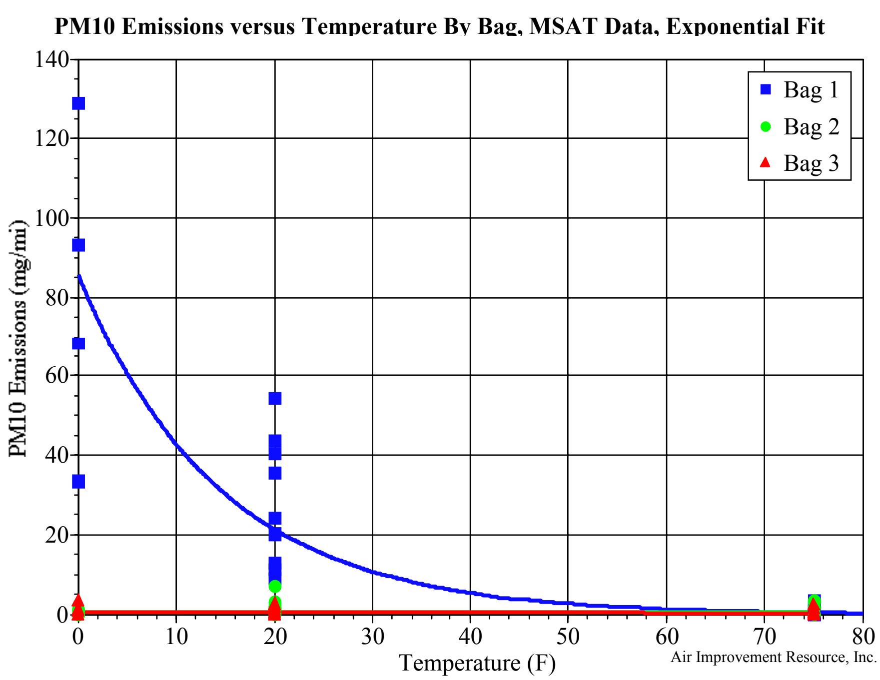
The winter portion of the Denver Northern Front Range Air Quality Study (NFRAQS) study conducted in 1997 tested vehicles at 60°F and at ambient temperature, which averaged between 31° and 40°F for the various vehicle categories[55b]. The results are shown in Table 9-1 for phases 1 and 2 (bags 1 and 2) for the 4 vehicle age classes and smokers, which were vehicles with visible exhaust smoke. Note that average PM emission rates for hot running emissions (phase 2) did not increase with decreasing temperature for the smokers and the older vehicles, although they did increase for the 1986-90 vehicles by a factor of 2.3 and for the 1991-96 vehicles by a factor of 7.0. Even cold start shows a vehicle age or emission rate effect. While the PM emission rate increased by a factor of 8.6 for the newer, low emitting vehicles, the increase for smokers and older vehicles is much lower, ranging from 1.59 to 1.97.

[55b] S. Cadle, P. Mulawa, E. Hunsanger, K. Nelson, R. Ragazzi, R. Barrett, G. Gallagher, S. Lawson, K. Knapp and R. Snow, "Light-Duty Motor Vehicle Exhaust Particulate Matter Measurement in the Denver, Colorado Area", JAWMA, 49, 1999

Table 9-1. Average FTP PM emission rates by phase (bag) at 60°F and a temperature in the 31° to 40°F range							
Vehicles	Number of Vehicles	Phase 1, 60°F, mg/mi	Phase 1, Cold, mg/mi	Phase1 Ratio Cold/60	Phase 2, 60°F, mg/mi	Phase 2, Cold, mg/mi	Phase 2 Ratio Cold/60
1971-80	15	147	290	1.97	28.2	26.1	0.93
1981-85	16	82.6	159	1.92	20.3	19.0	0.94
1986-90	14	27.9	70.6	2.53	7.3	16.9	2.3
1991-96	9	9.4	81.3	8.64	1.9	13.4	7.0
Smoker	15	742	1179	1.59	298	231	0.78

A fit to the effect of temperature on cold start PM emissions for all vehicles from Kansas City showed good agreement to the cold start data from the matched vehicles. However, the hot running data for all vehicles had a very modest increase in PM emission rate with decreasing temperature with an R^2 of 0.054 while the matched vehicle analysis indicated a strong temperature dependence. The reason for this disagreement has not been explained at this time. Because of this discrepancy, AIR evaluated the temperature impact by bag for the MSAT vehicles (all Tier 2 vehicles). The results are shown in Figure 9-7.

Figure 9-7. Effect of temperature on the PM emission rate from the MSAT vehicles.



The R^2 values for a linear fit to the data are 0.54 for Bag 1, but only 0.004 for Bag 2. Hence this limited data set does not appear to support the hot stabilized temperature correction derived from the matched Kansas City data. It is likely that the reason for the Bag 2 temperature effects in Kansas City are due to the shorter Bag 1 of the LA-92 used in Kansas City combined with the fact that the LA-92 bag 1 driving cycle is less aggressive than that of the FTP. Thus the LA-92 Bag 1 test could have resulted in some start effects carrying over into running operation. If an FTP test had been used instead of the LA-92 in KC, it is possible there would have been no effect of colder temperature on Bag 2 PM emissions.

The HC, CO, and NOx cold start correction factors are based on the FTP, where EPA found no temperature effects in Bags 2 and 3 for these pollutants. EPA should examine the second-by-second data for Bag 2 of the LA-92 in KC to determine if the temperature effects disappeared after the first minute or two of Bag 2. If so, EPA should make the PM temperature correction factors for running emissions consistent with HC emissions.

9.1.5 Effect of MSAT-2 Rule

The MSAT-2 rule requires a reduction of exhaust NMHC emissions from LDVs and LDTs at 20° F to a maximum of 0.3 grams per mile. While the MSAT-2 rule does not limit cold weather emission of PM, the Regulatory Impact Analysis document noted the strong linear correlation between NMHC and $PM_{2.5}$ emissions. Hence, it is expected that the rule will reduce PM. While no temperature effect has been found for HC running emissions, the foregoing discussion showed that EPA believes there is an effect for running PM emissions, while the authors believe this is still an open question. Thus, EPA has incorporated a reduction of both cold start and hot running PM emissions as a function of temperature into MOVES for 2010 and later vehicles. As the temperature drops from 72° to 20°F, the previous PM modeling gives an increase of a factor of 11.1 for cold starts and 5.22 for hot running. Since the MSAT-2 rule is expected to decrease NMHC by 30%, a 30% reduction is also applied to PM at 20°F. Hence for vehicles meeting the MSAT-2 rule PM will increase by a factor of 7.78 for cold starts and 3.66 for running emissions.[57] The exponential multiplicative adjustment factors for the effect of temperature on PM are adjusted to provide the expected effect at 20° F. MOVES uses the ARB soak adjustments for HC emissions at soak times shorter than that in the traditional cold start. Since HC and PM cold start emissions correlate, the same adjustment factors are used for PM.

9.1.6 Fuel effects

EPA has stated that they will include fuel effects on PM emission rates when such data becomes available. While some previous studies have indicated that oxygenated fuels can reduce PM emissions, especially during cold starts, EPA's examination of the data led them to conclude that there are no proven effects. It seems logical that an oxygenate

[57] If EPA modifies he HC corrections to include a margin at 20° F as indicated in section 8 of this report, this will affect the PM corrections as well.

in the fuel could reduce PM emissions during cold start. This would be most important during low temperature cold starts. It is not known if adaptive learning might correct for this in newer vehicles. Additional tests are warranted. While initial discussion with EPA indicated that there would be no fuel S effect on PM included, subsequent sensitivity testing discussed below in section 10.8 shows that a modest effect was included.

9.1.7 Particulate Matter Speciation

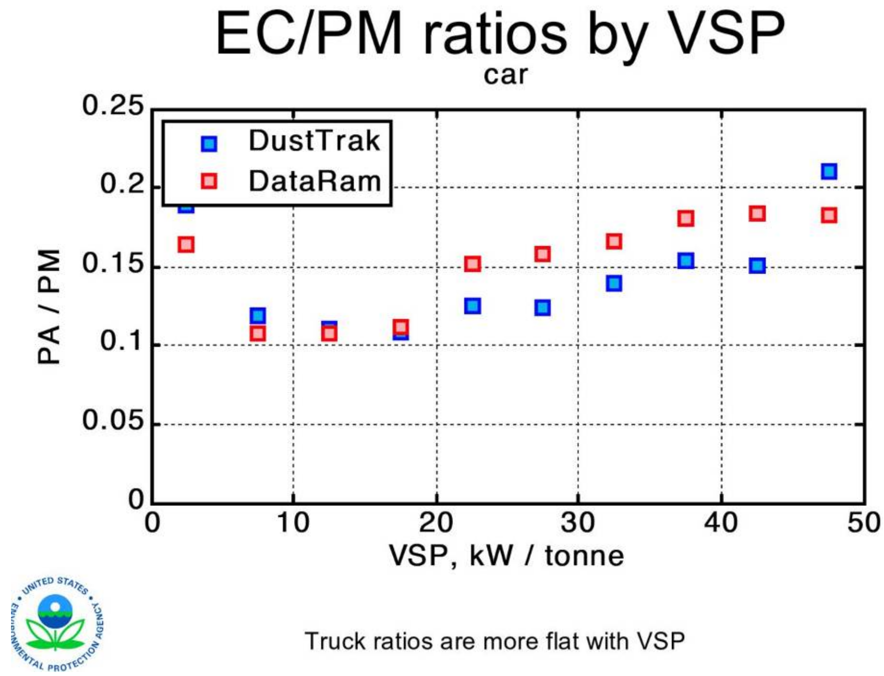
MOVES separates the PM into elemental carbon (EC) and organic carbon (OC) fractions. Sulfate emissions have also been added to the model recently; the authors have not seen any of that information. While it is recognized that there are other constituents in the PM, they will not be separated at this time. The Kansas City study measured the elemental composition of the PM from a subset of vehicles. It was found that the measured elements other than carbon accounted for 3.6% of the cold start PM and 11.9% of the hot running PM. OC and EC measurements were made for a subset of the Kansas City vehicles using the thermal optical reflectance (TOR) method. Since this is a batch method it provides data on a bag basis. No comparisons were provided to similar batch data generated in earlier studies.

Elemental carbon is measured continuously during all vehicle tests using a photoacoustic analyzer. In this analyzer light from a laser beam is absorbed by particles which heat the surrounding air, thereby creating a pressure pulse that is measured. Since elemental carbon in exhaust is black, it is highly absorbing. The assumption is that the absorption efficiency for exhaust EC does not vary with operating conditions or coatings of other material such as organic carbon, hence they can be calibrated for black carbon. Good correlations have been found with TOR results on vehicles with fairly high PM emission rates. An examination of the Kansas City data lead EPA to the following conclusions:

- The EC/OC ratio is different for cold start and running PM emissions.
- The EC/OC ratio varies with vehicle mass. This can only be handled in MOVES by using different values for cars and trucks.
- The EC/PM ratio is highly sensitive to load.
- There is no strong model year dependence on the EC/PM ratio (there is a lot of scatter in the data).
- The EC/PM ratio does not have a temperature dependence for cold start, but hot running does show an increase in the ratio with increasing temperature (there is a lot of scatter in the data and the temperature range went from 20° to 90° F).

Use of the photoacoustic analyzer allowed EPA to examine EC emission rates on a modal basis. No methods are available for simultaneous real-time OC measurements, nor were any other species measured in real-time. Hence OC is determined by the difference between EC and the PM mass. It is recognized that this classifies the non-carbonaceous material as OC. Given the continuous EC measurements, it is possible to determine EC/PM ratios by bin. Figure 9-8 shows the results for cars using PM mass derived from the DustTrak and the Dataram. Since EPA states they are using the DustTrak data, it is assumed that the DustTrak EC/PM ratio is used in MOVES.

Figure 9-8. Elemental Carbon to PM mass ratios by VSP bin



<u>Soak time and IM effects</u> - EPA plans to use the HC soak time data to model the effect of soak time on cold start PM emissions. Presumably temperature effects will be scaled using the same factors as used for a regular cold start. Clearly, this is not ideal, but given the absence of data there is little choice other than not to include soak time PM effects. A sensitivity analysis should be performed at low temperature to determine the magnitude of the impact. If it is significant, then data should be obtained for PM/soak time effects. At this time EPA has reported that there is no information regarding the effect of I/M on PM. EPA could have chosen to model I/M effects as being the same as I/M HC effects, as they did with soak time and the HC low temperature standard. But in this case they decided to be conservative. It is easy to envision a disconnect between high HC and high PM emissions that would invalidate a PM I/M effects model based on HC. For example would engine misfire create an increase in PM? Would a malfunctioning catalyst significantly increase PM? In the absence of data, it is probably best not to model I/M PM effects.

9.2 Non-Exhaust PM Emission Rates

In addition to exhaust PM, vehicles emit particles from the wear of brake systems and tires. The turbulent wake from vehicles also re-suspends road dust, some of which is generated from wear of the road surface. However, MOVES does not include re-suspended road dust.

9.2.1 Brake PM Emission Rates

EPA used results from two relatively recent brake wear studies to develop brake PM emission rates.[58] They took into account the number and types of brake systems, the variability of brake wear rates due to the composition of brake pads, the fraction of the total wear material that becomes airborne particulate and its size distribution as well as the effect of braking intensity. Results were based on two thirds of the braking occurring with the front brakes on light-duty vehicles, 60% of the worn material becoming airborne, and 10% of the total PM being $PM_{2.5}$. Emission rates are not adjusted for the weight of the vehicle. A curve is fit to a plot of $PM_{2.5}$ emission rate versus the vehicle deceleration rate. The resulting curve is combined with activity data to obtain estimates of brake PM emission rates for the FTP and for an average of activity data obtained for LA and Kansas City. $PM_{2.5}$ and PM_{10} emission rates for LA were 2.0 and 20.0 mg/mi, respectively. The average of LA and KC were 2.2 and 17.6 mg/mi, respectively. Heavy heavy-duty truck brake wear emission rates were taken from a recent roadside study that estimated a brake wear $PM_{2.5}$ emission rate of 12.1 mg/mi.

MOVES assumes that any activity in a bin associated with deceleration at a rate higher than expected for a coast down is due to braking. Most braking activity occurred in bins 0, 1, 11, 21 and 33. At the present time there is no correction to brake wear emission rates due to road grade. Therefore, one should be very careful accepting the results for micro-environments where there is a significant road grade.

9.2.2 Tire PM Emission Rates

Presently EPA is estimating tire wear emission rates on the basis of one recent top-down study and one recent bottom-up study. These studies gave an average tire wear PM emission rate for $PM_{2.5}$ and PM_{10} of 2.0 and 9.0 mg/mi, respectively, for four wheeled passenger vehicles. Emission rates for other classes of vehicles will be scaled to account for the number of tires on the vehicles. EPA recognizes that tire wear does not occur at a constant rate due to cornering, etc., but currently has no data or means to account for this inconsistency in MOVES. MOVES varies the rate as a function of vehicle speed, which provides a rate that is nearly constant on a distance basis. As with brake wear, care should be given to interpreting the results for microenvironments.

9.3 Summary of Recommendations

PM emission rates data have a lot of uncertainty since the acquisition of real-time data requires the use of surrogate instruments whose accuracy is not well known. The following are the recommendations for PM emissions in MOVES:

1. The combined MSAT and Kansas City data on matched pairs does not support a cold temperature adjustment for running emissions. Results from other studies such as NFRAQS should be included in the analysis, with special regard to high PM emitters.

[58] See the poster entitled "Brake and Tire Wear Emissions in MOVES" by Edward Nam, Sujan Srivastava, David Brzezinski, and Matti Maricq given at the 17th CRC On-Road Vehicle Emissions Workshop, March 26-28, 2007.

2. EPA should check adjustments to ZMLs for older vehicles to account for current fuels and measurement methods.

In addition, the following testing or analysis programs would be helpful in the longer term.

- Effect of low temperature HC standard on PM is based on HC emissions. At some point, this assumption should be checked with data, especially for running emissions.
- Improved real-time PM emissions data base with more data and improved instrumentation.
- Data base broader than one location – i.e., Kansas City, which is now almost 5 years old.
- Effect of fuels, especially oxygenated fuels, should be monitored.
- Effect of soak time on PM emissions rather than using a HC surrogate.
- Temperature effects are very important. More data are needed, especially for hot running emissions.
- Improved brake and tire wear emissions data and apportionment of tire wear emissions based on driving condition.

10.0 MOVES Sensitivity Runs

To further evaluate MOVES, the study team performed over 200 runs of the model in different conditions. From these 200+ runs, they pulled out certain information of interest and created bar plots to illustrate the emissions impacts of various elements of the MOVES modeling methodologies explained in earlier sections of this report. For example, they wanted to evaluate the impact of deterioration on emissions. To accomplish this, they set the emissions of all age bins equal to the emissions of the youngest bin, and ran the model, and compared the emissions with the default model.

For all of the MOVES model runs, they have examined the emissions in g/mi from Cook County, IL for January (winter) and July (summer) for several years: 2008, 2015, and 2020. Summer diurnal temperatures assumed are 67° F to 86°F. Winter diurnal temperatures are 19° F to 32°F.

The emissions are plotted in g/mi. Comparisons of the model results for these two seasons allows an evaluation of the temperature impacts. Comparison of the model results across the selected calendar years allows an evaluation of fleet turnover effects.

If a geographical area other than Cook County had been evaluated, the emissions comparison could be somewhat different than shown in this section. For example, a southern city (e.g., Atlanta) would have shown less variation in emissions between summer and winter. In a non-I/M area, emissions could have been somewhat higher, but the primary conclusions from these sensitivity runs would have been the same as for Cook County.

In most cases, evaporative emissions have been disaggregated into the following components: refueling, permeation, vapor, and leaks. The exhaust emissions are disaggregated by running and starts (both cold and hot starts together).

Emissions of VOC, NOx, $PM_{2.5}$, and CO are shown for three general cases: default, no I/M, and no deterioration. The method used to remove deterioration is to set the emissions for all ages equal to the emissions for the 0-3 year age bin. The method used to simulate no I/M is simply to turn the I/M parameters off. As mentioned in Section 5, the draft MOVES model does not have an effect for I/M on start emissions. EPA plans to include this effect for the final MOVES model, so the benefits of I/M would be somewhat higher than shown in the following plots.

The Chicago I/M program includes an exhaust and evaporative OBD check for 1996+ vehicles, IM240 and idle tests for pre-96 vehicles and pre-1981 vehicles, respectively. There is a gas cap check for all 1968 and later vehicles, the I/M240 test for mid-age vehicles, and an OBD check for the newest vehicles. The newest four years of vehicles are exempt from the I/M program.

The summer and winter temperature variations being modeled in MOVES are shown in Appendix F.

Emissions are presented for a number of vehicle types:

- Passenger cars only in 2008, 2015, and 2020
- Passenger cars, passenger trucks and light commercial trucks
- Heavy-duty diesels
- 1995 model year (80%+ Tier 1)
- 2008 model year (full Tier 2)
- Comparison of car and truck emissions
- Effects of high VSP correction factors
- Tier 2 vehicles compared to current vehicles

10.1 Passenger Car Only, Default, I/M, No Deterioration

Passenger car evaporative emissions are shown in Figures 10-1 (winter) and 10-2 (summer). Emissions are shown for three cases – default, no-I/M, and no deterioration. The no-I/M emissions are determined by turning off the I/M parameters, and the no deterioration emissions are estimated by setting the emissions in higher vehicle age bins to the lowest age bin (0-3 years). Evaporative emissions shown are refueling, venting, permeation, and leaks.

For winter, evaporative emissions for the default case start at about 0.12 g/mi in 2008 and are reduced to 0.05 g/mi by 2020. There are small reductions in refueling and permeation emissions, and large reductions in venting emissions. The no I/M case is only slightly lower for all three years than the I/M case. The no deterioration case is significantly lower than the other two cases. Interestingly, the no deterioration case still has a leak component of 0.01 g/mi. This is due to a very small fraction of vehicles assumed by EPA to be liquid leakers in the first three years of their lives, even though the data used to develop the liquid leaker rates showed no liquid leakers under nine years of age.

Summer evaporative emissions are much higher than winter emissions due to higher temperatures. For the default case, emissions start at about 0.2 g/mi in 2008 and drop to 0.075 g/mi by 2020. There is no difference in leak emissions between winter and summer. The no deterioration case for 2020 has emissions that are about 20% less than the default case.

While the winter emissions are less than the summer emissions, it is surprising that the winter evaporative emissions are as high as they are. There would be some emissions from liquid leakers and refueling emissions, but permeation emissions and vapor venting emissions should be very small, even if the fuel tank is heated somewhat in the winter through vehicle operation. It is doubtful, however, that it would be heated as much as it is in the summer for the same length of trip.

Figure 10-1. VOC Evaporative Emissions in Winter

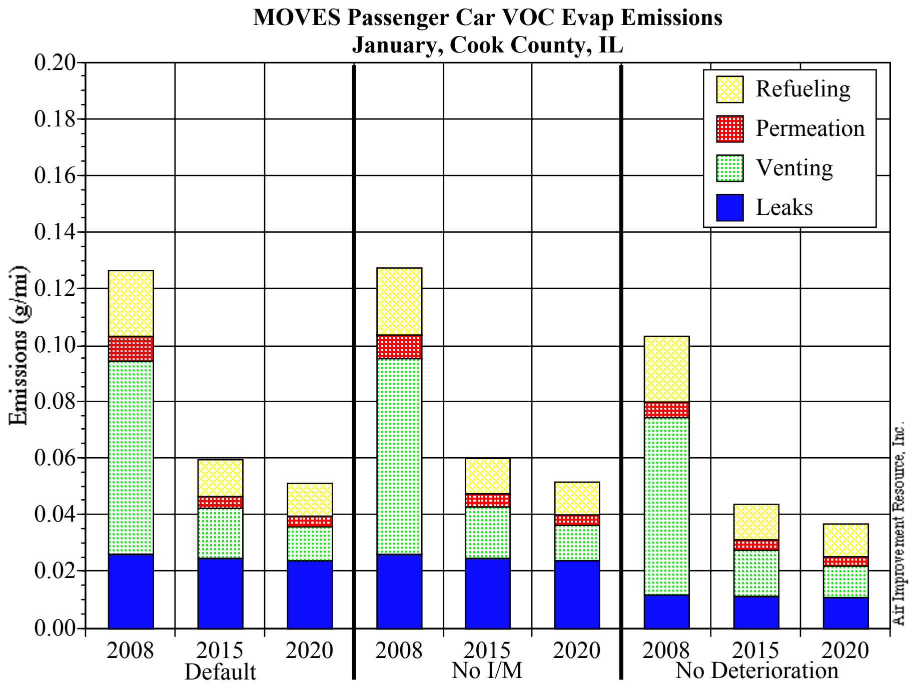
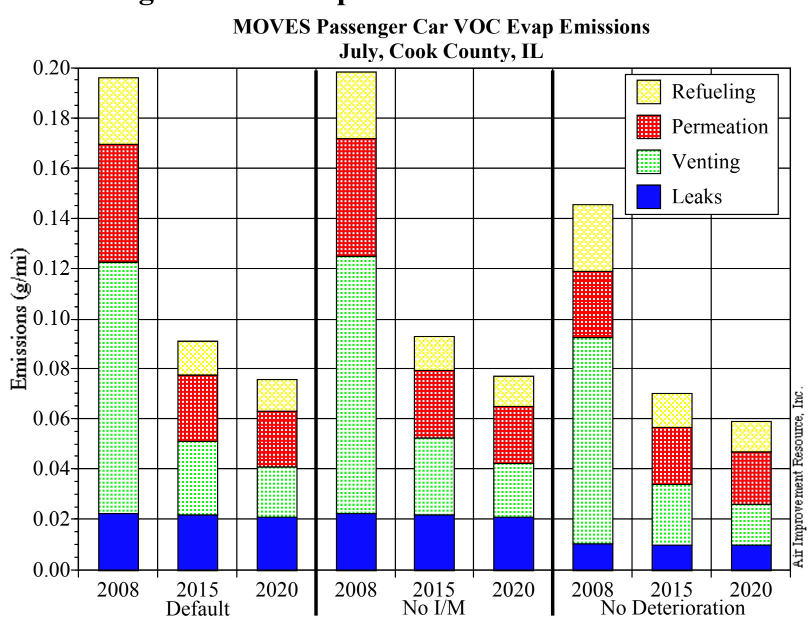


Figure 10-2. Evaporative Emissions in Summer



For summer, there are large reductions in venting emissions with time and much smaller reductions in permeation emissions. This is because EPA assumes that the Tier 2 evaporative standards do not reduce permeation emissions. This assumption should be verified through testing.

VOC exhaust emissions are shown in Figures 10-3 (winter) and 10-4 (summer). For the default case for winter, emissions start at 0.8 g/mi and are reduced to 0.20 g/mi (75%) by 2020. Nearly all of the emissions are projected to be start emissions, as opposed to running emissions. There is a small effect of I/M, and VOC emissions in the no deterioration case in 2020 are about 20% lower than in the default case.

One theory of so-called "high emitters" is that these vehicles experience higher emissions during the running portion of operation much more than during the start. However, if this is true, then Figure 10-3 seems to indicate a small effect of high emitters, because the increase in emissions between the no deterioration and default cases is almost all in the start mode, and very little in the running exhaust mode.

VOC emissions during the summer are much lower than the winter. In the default case, fleet emissions in 2020 are about 0.07-0.08 g/mi. Emissions in 2020 for the winter case are about three times higher than the summer. Emissions in 2008 in the winter are less than three times higher than the emissions in the summer. This indicates that EPA expects that temperature correction factors for later model year vehicles are somewhat higher than for the earlier model years. This is somewhat surprising, because the model includes 2009 and later vehicles that are subject to cold temperature HC emission standards which should reduce the temperature sensitivity of HC emissions. If the HC temperature correction factors are improved as recommended in Section 8, this would probably fix this concern.

Figure 10-3. Exhaust VOC Emissions in Winter

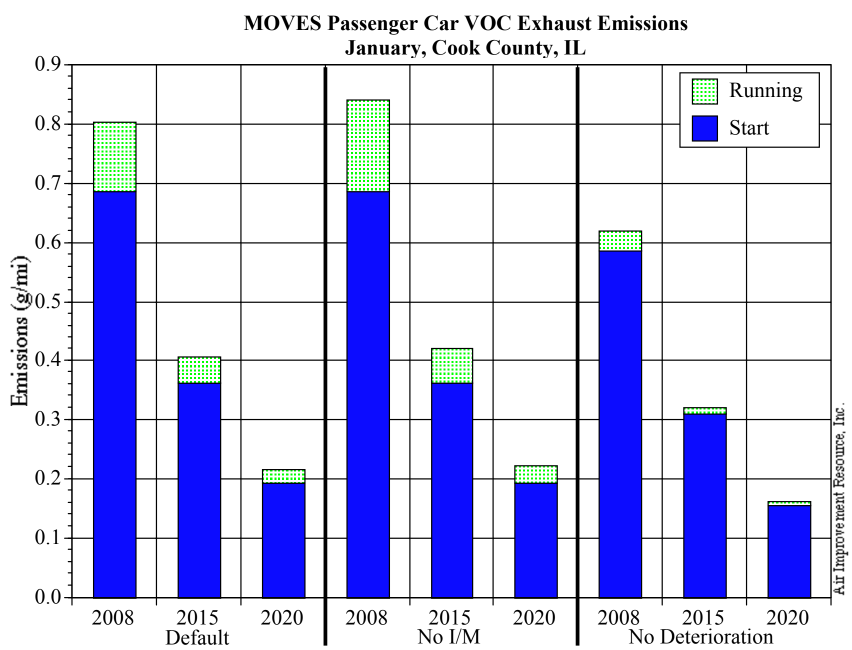
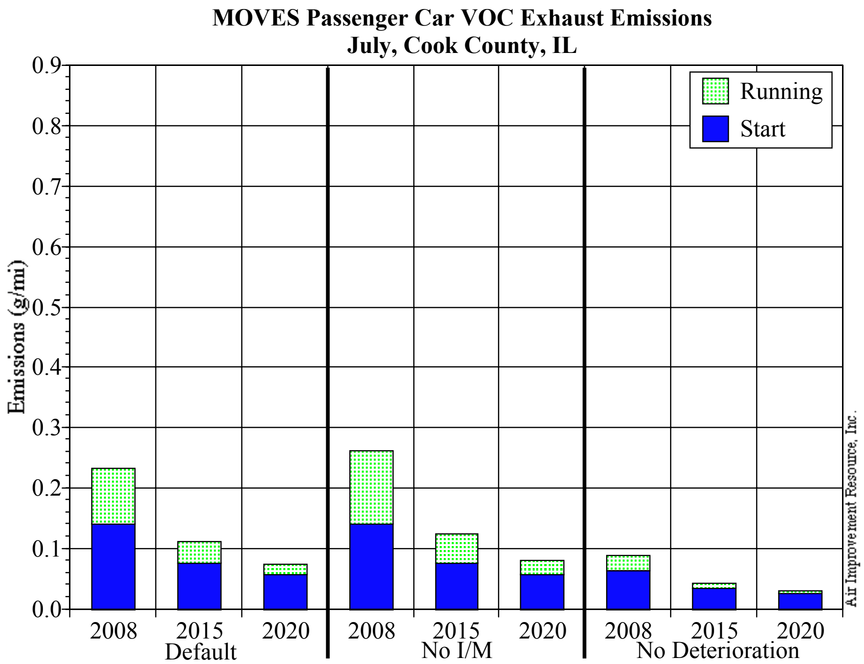


Figure 10-4. Exhaust VOC Emissions in Summer



For Figures 10-3 and 10-4, the data in Table 10-1 show the percent of emissions that are due to starts.

Table 10-1. Percent of Exhaust VOC Emissions in Start Mode						
Scenario	2008		2015		2020	
	Jan	Jul	Jan	Jul	Jan	Jul
Default	85.5	60.6	89.4	69.1	76.4	76.4
No IM	81.5	53.6	86.2	62.2	70.8	70.8
No Deterioration	94.9	73.0	96.7	81.6	86.6	86.6

NOx emissions are shown in Figures 10-5 (winter) and 10-6 (summer). Winter emissions are only a little higher than summer emissions, due to somewhat higher start emissions. Winter NOx emissions for the default case are 0.65 g/mi in 2008, and are reduced to 0.18 g/mi by 2020. Summer NOx emissions for the default case are just over 0.5 g/mi in 2008, and are reduced to 0.12 g/mi by 2020. For the no deterioration case in the summer, 2020 emissions are about 0.05 g/mi. The difference in the default and no deterioration case in 2020 shows both higher start and running emissions, which is different than shown for VOC. NOx emissions for the default (I/M) case are lower than the no I/M case. Chicago has no NOx cutpoint for its transient IM test, but does have an OBD I/M program for 1996 and later vehicles, and this appears to be finding some vehicles with higher NOx emissions.

Figure 10-5. NOx Emissions in Winter

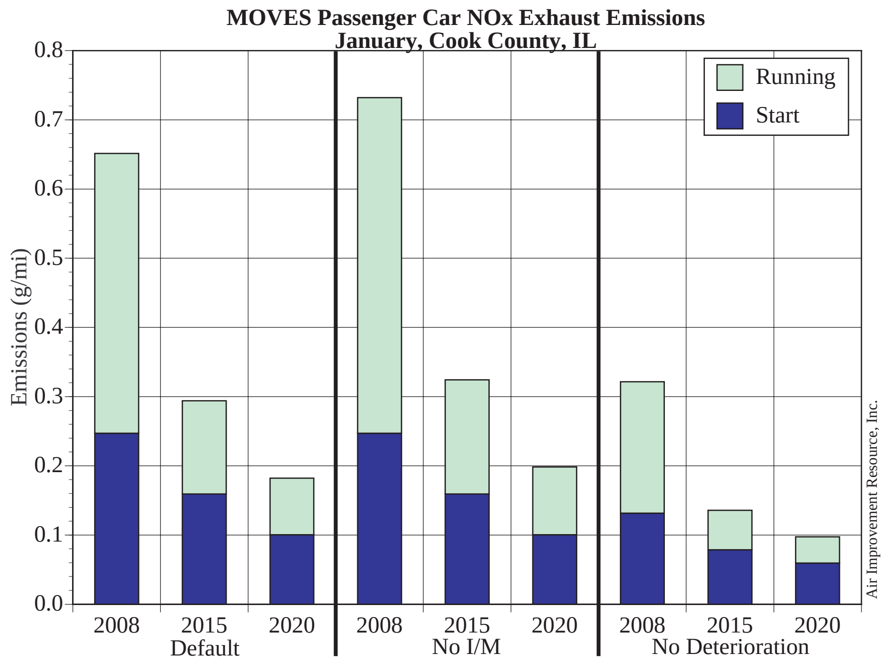
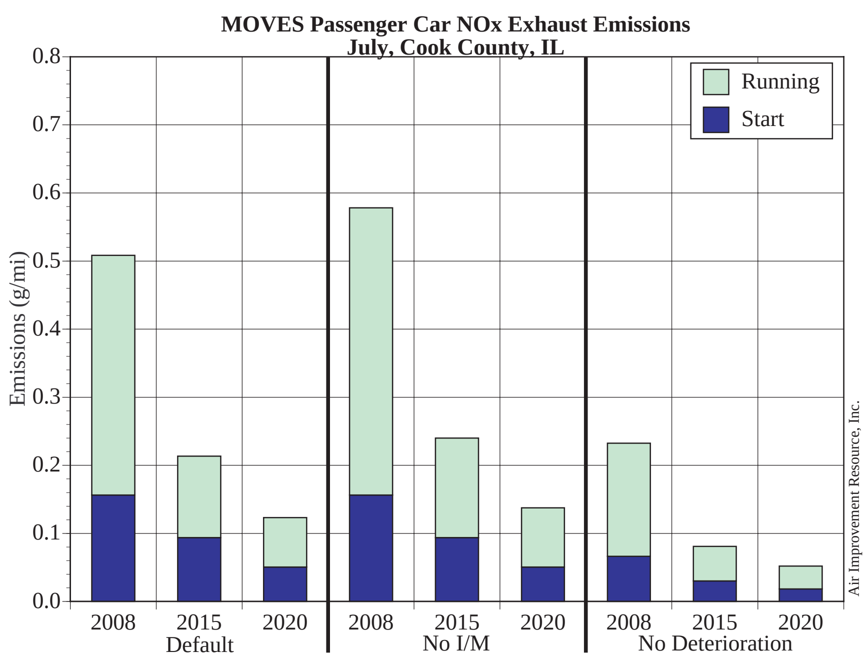
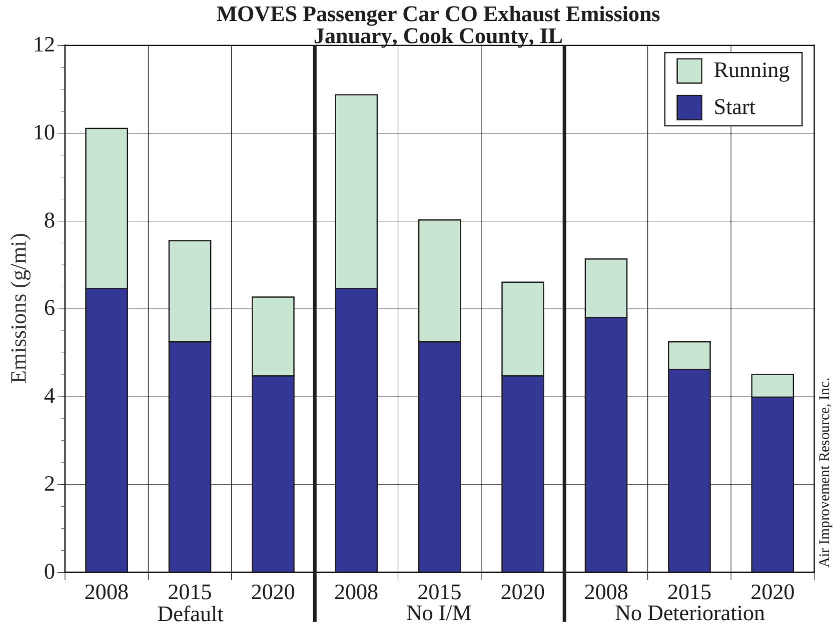


Figure 10-6.NOx Emissions in Summer



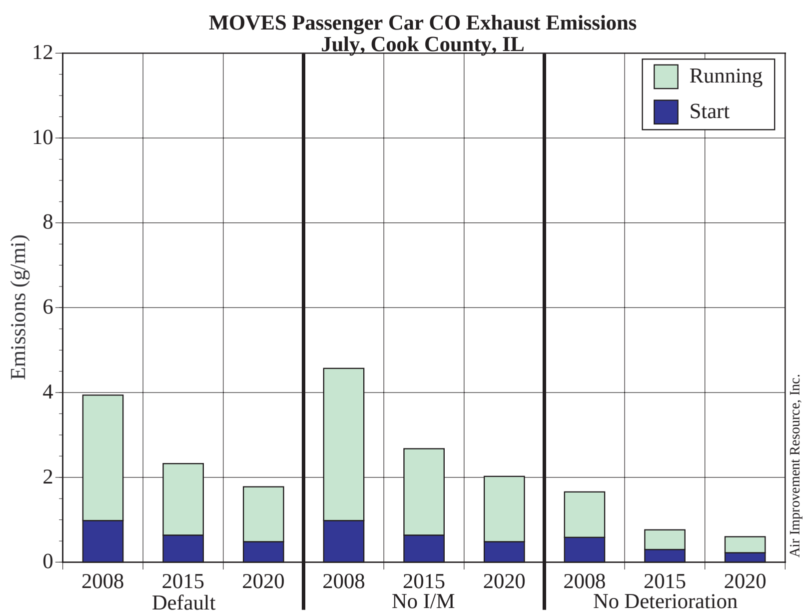
CO emissions are shown in Figure 10-7 (winter) and 10-8 (summer). In the winter, most of the CO emissions occur during the start mode, similar to VOC. Winter CO emissions start at 10 g/mi and are reduced to 6 g/mi by 2020. There is a small impact of I/M on winter CO emissions.

Figure 10-7. CO Emissions in Winter



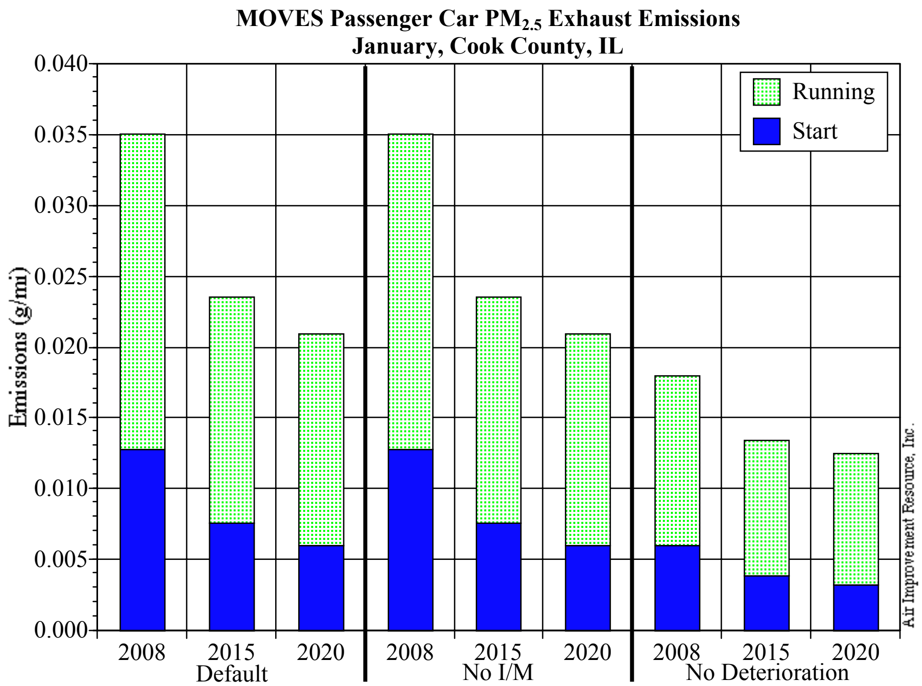
Summer CO emissions start at 4 g/mi in 2008 and are reduced to under 2 g/mi by 2020. Summer emissions are lower than winter because of reduced start emissions, not reduced running emissions. Most of the summer CO emissions in 2020 are running emissions, instead of start emissions. This is an important difference from exhaust VOC, where most of the summer VOC is start emissions. The reasons for this difference in VOC and CO emissions require further study, because they probably should be similar to each other.

Figure 10-8. CO Emissions in Summer



$PM_{2.5}$ emissions are shown in Figure 10-9 (winter) and 10-10 (summer). Winter PM emissions start at 0.035 g/mi in 2008 and are reduced to 0.02 g/mi in 2020. In the winter, most of the emissions are running emissions. EPA assumes that I/M has no effect on $PM_{2.5}$ (EPA may re-examine this issue in the near future).

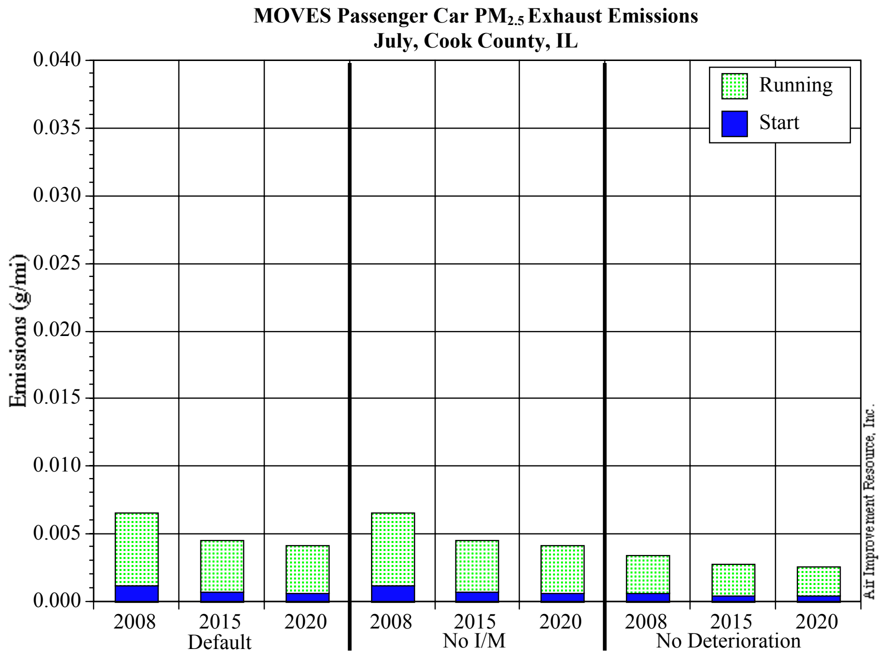
Figure 10-9. PM Emissions in Winter



Summer PM emissions are much lower than winter. Summer emissions start at 0.006 g/mi in 2008, and are reduced to 0.004 g/mi in 2020. Running emissions are assumed to be much higher than start emissions.

For $PM_{2.5}$, the model output shows that running emissions increase dramatically with temperature. This is different than for HC and CO, where only the cold start emissions increase; running emissions are estimated to be constant with temperature.

Figure 10-10. PM Emissions in Summer



10.2 Light Truck versus Car Comparisons

Figures 10-11 through 10-20 show evaporative VOC, exhaust VOC, NOx, CO, and $PM_{2.5}$ comparisons for passenger cars, passenger trucks, and light commercial trucks. All of these plots are for the default case, for both winter and summer.

Nearly all the plots show slightly to moderately higher emissions for passenger trucks and light commercial trucks than for cars. For evaporative emissions, this is expected because of the somewhat higher evaporative standard for trucks based on their larger fuel tank sizes. However, it is curious why leaks appear to be higher for cars than for light trucks in calendar year 2020 (both winter and summer). The reason for this has not yet been determined. The second item of note is that the NOx emissions for the truck categories show much higher emissions compared to passenger cars than for the other two gaseous pollutants, VOC and CO. Section 5 showed that the NOx standard averages for cars and light trucks are very similar in model year 2010. Finally, with regard to $PM_{2.5}$, it is not clear why the emissions of passenger trucks are higher than commercial trucks.

Figure 10-11. Evaporative Emissions in Winter

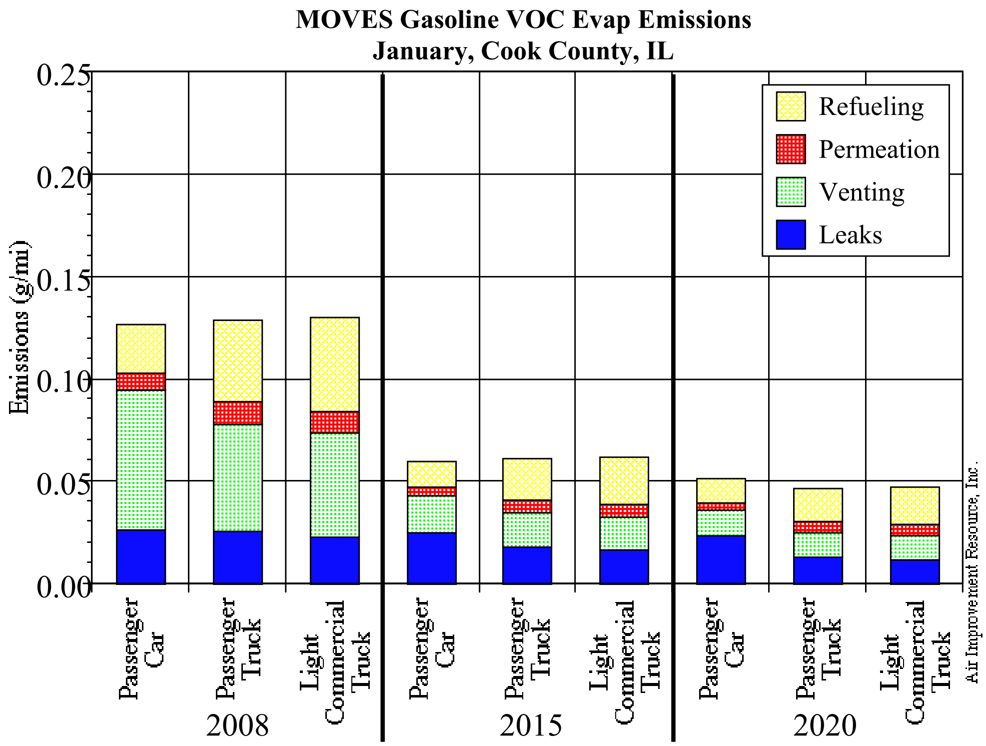


Figure 10-12. Evaporative Emissions in Summer

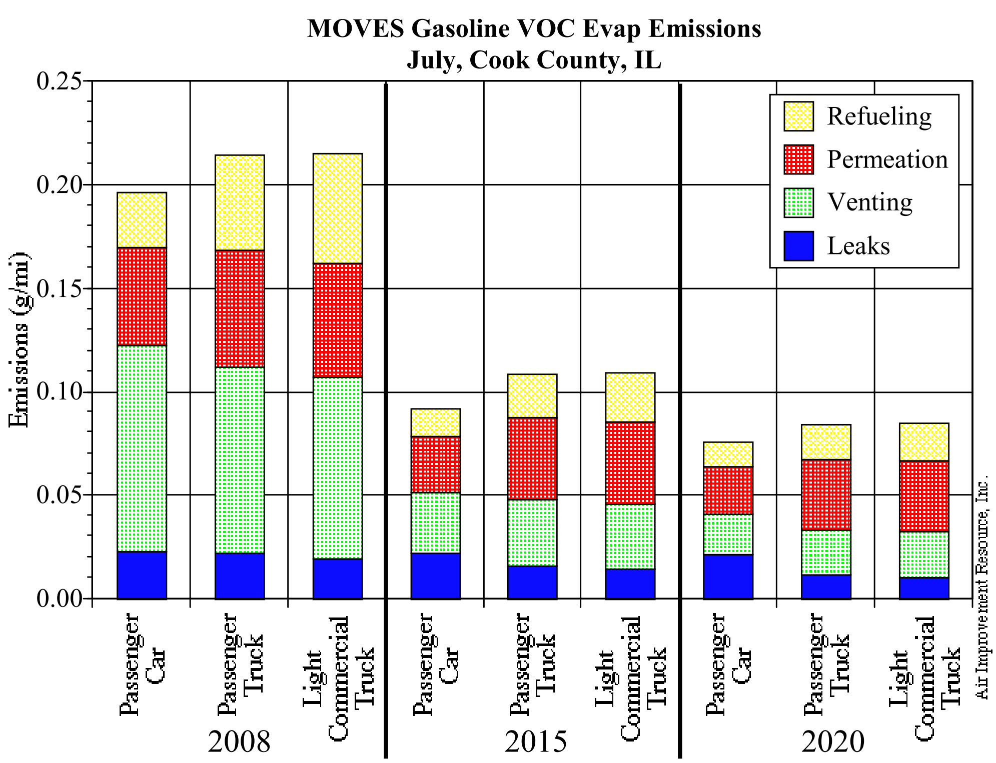


Figure 10-13. Exhaust VOC in Winter

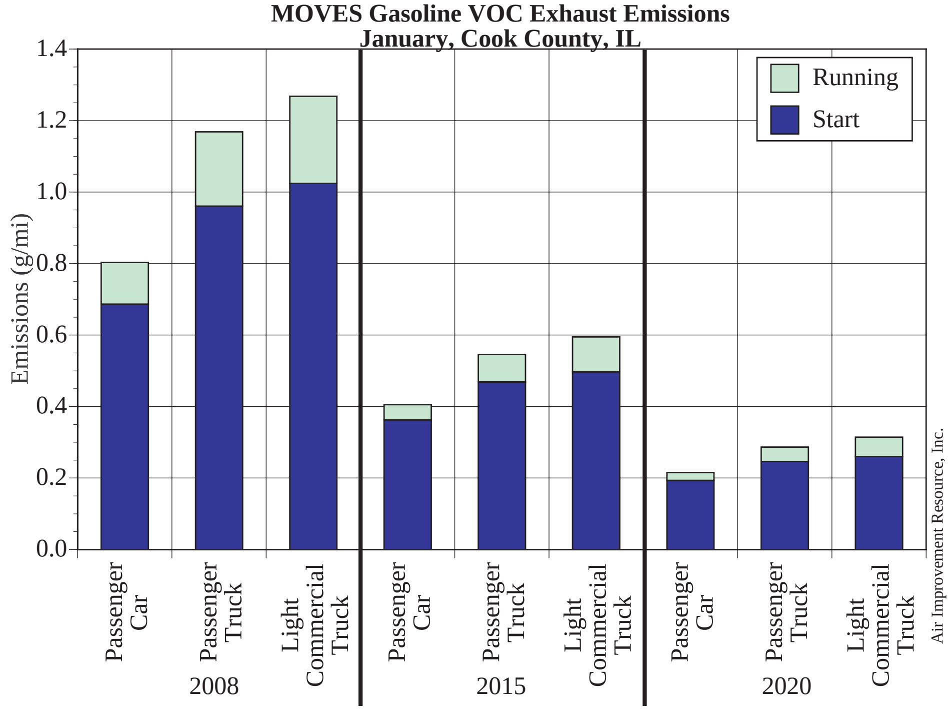


Figure 10-14. Exhaust VOC in Summer

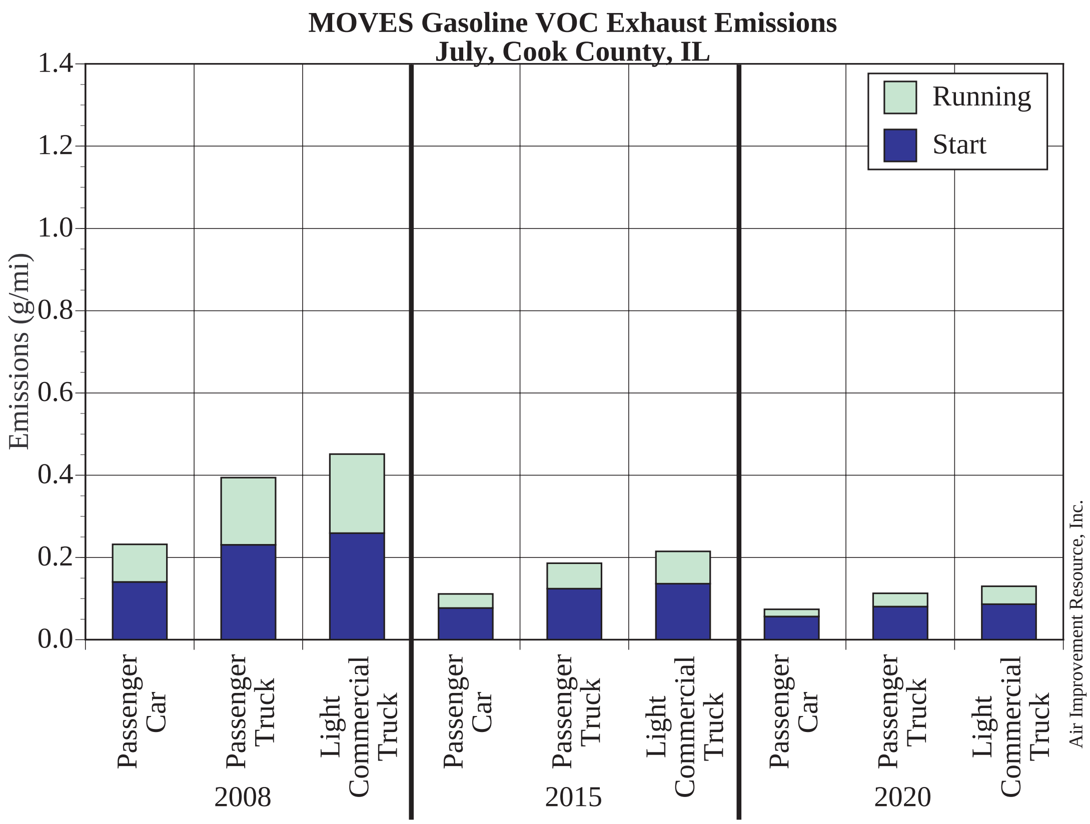


Figure 10-15. CO Emissions in Winter

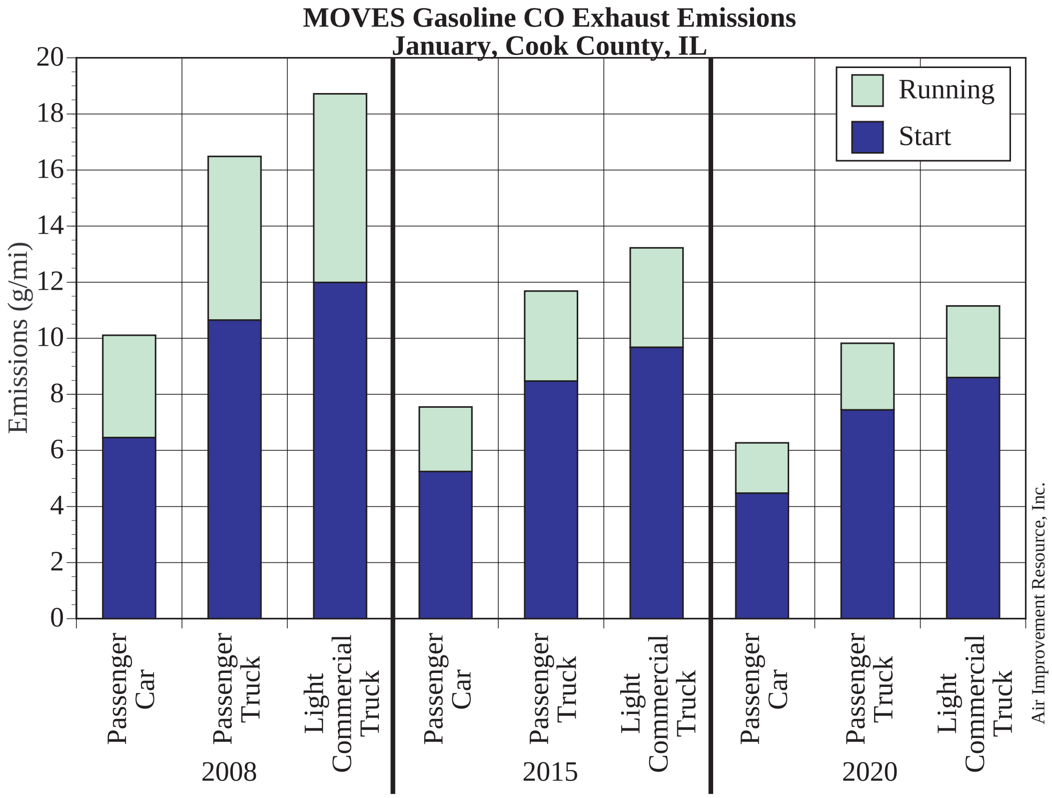


Figure 10-16. CO Emissions in Summer

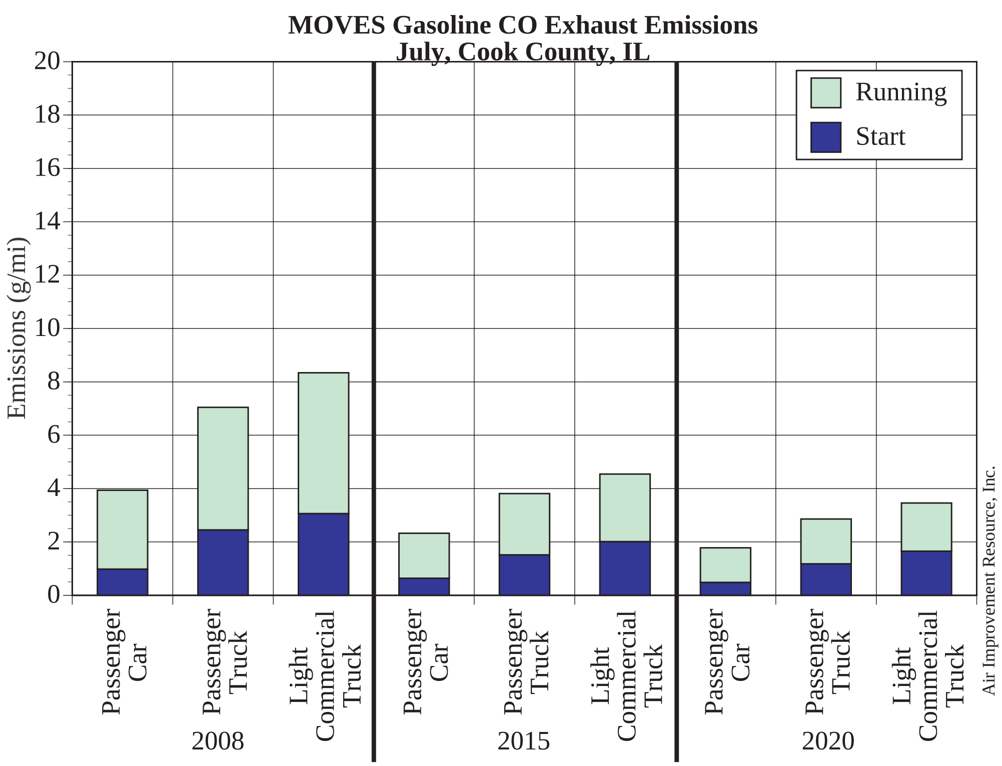


Figure 10-17. NOx Emissions in Winter

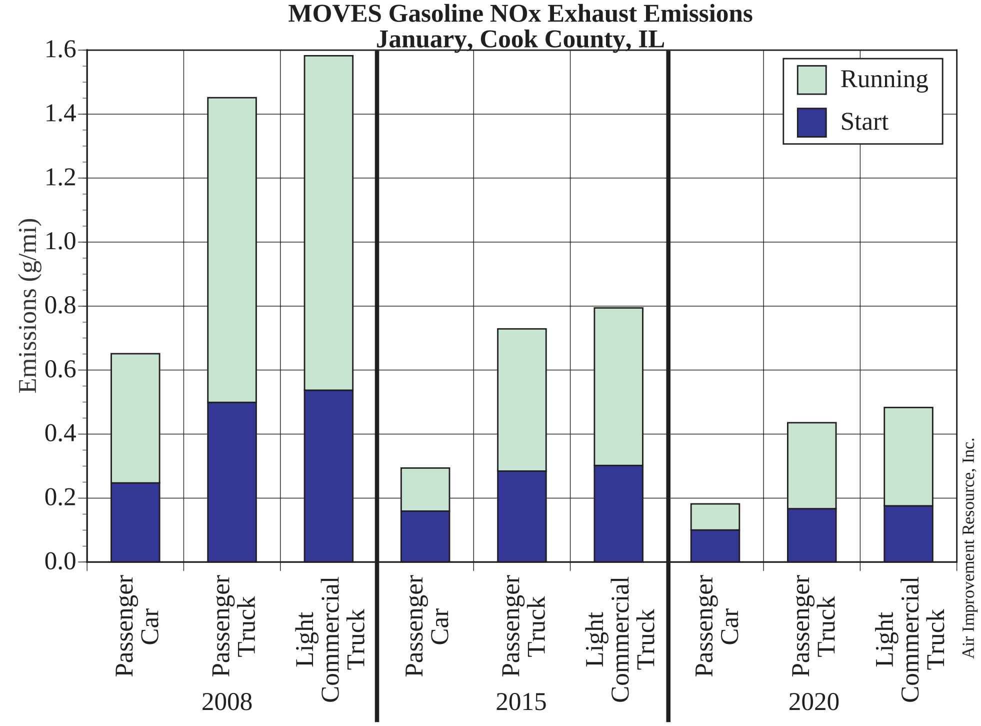


Figure 10-18. NOx Emissions in Summer

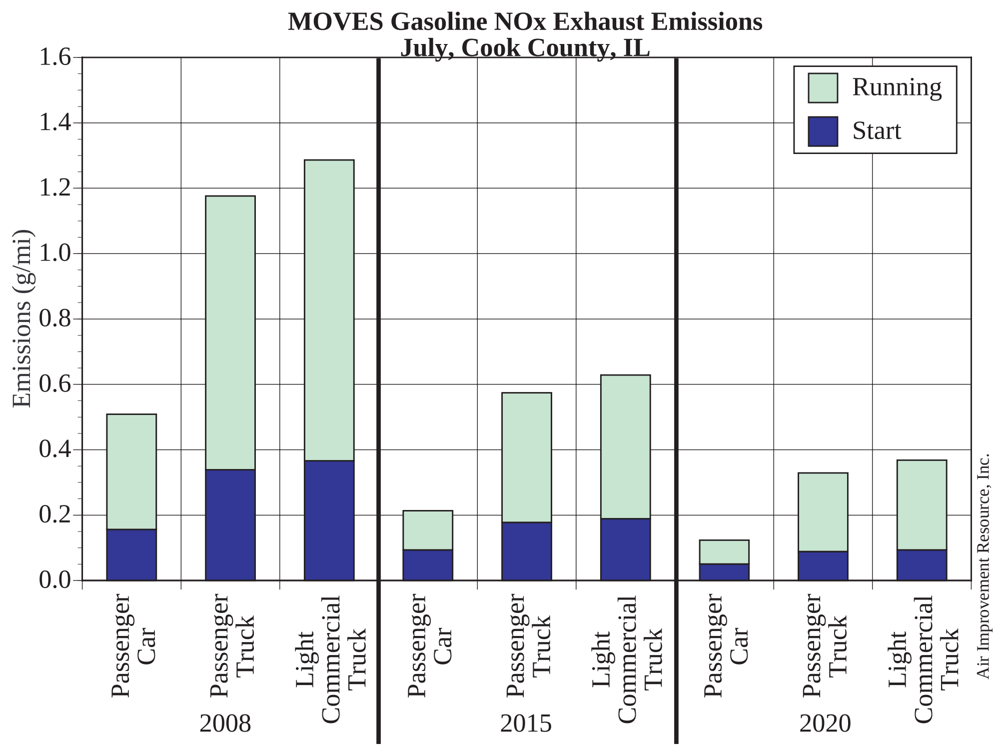


Figure 10-19. PM Emissions in Winter

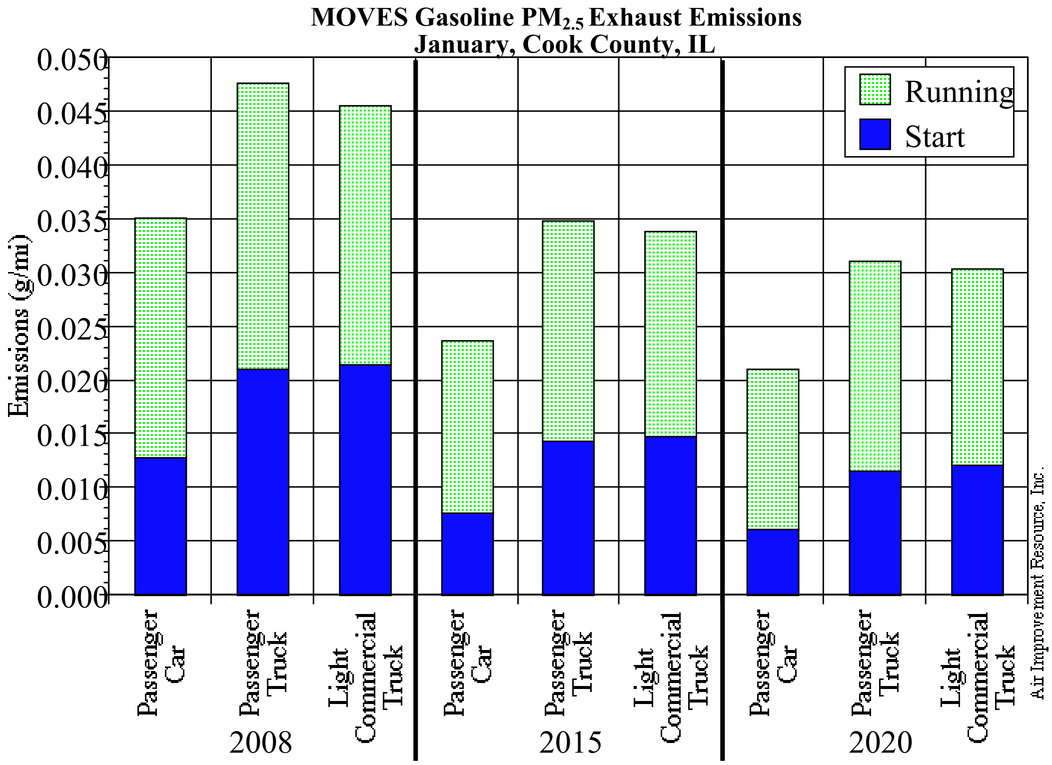
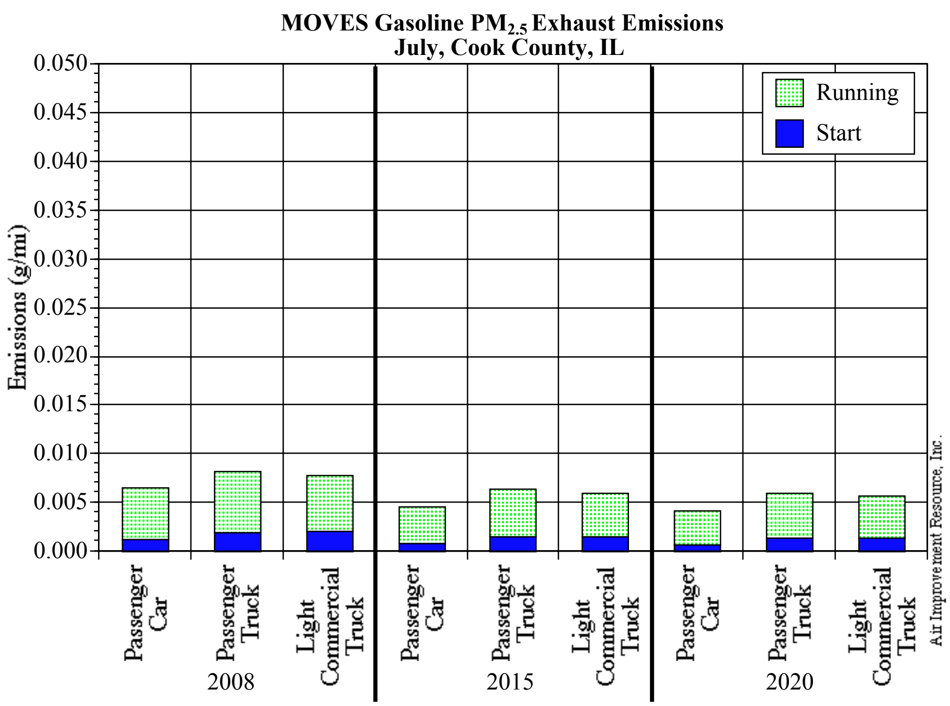


Figure 10-20. PM Emissions in Summer



10.3 Heavy Duty Diesel Trucks

Trends in heavy-duty truck emissions are shown in Figures10-21 through 10-28. All emissions show steep declines from 2008 to 2020 as the result of the phase-in of reductions in PM standards in the 2007-2009 timeframe and HC+NOx standards in 2010 model year. As expected, most of the emissions are running emissions. There are some start emissions for VOC in the winter, for CO emissions in the winter and summer, and a very small amount of NOx start emissions in the winter.

Figure 10-21. Diesel Truck VOC in Winter

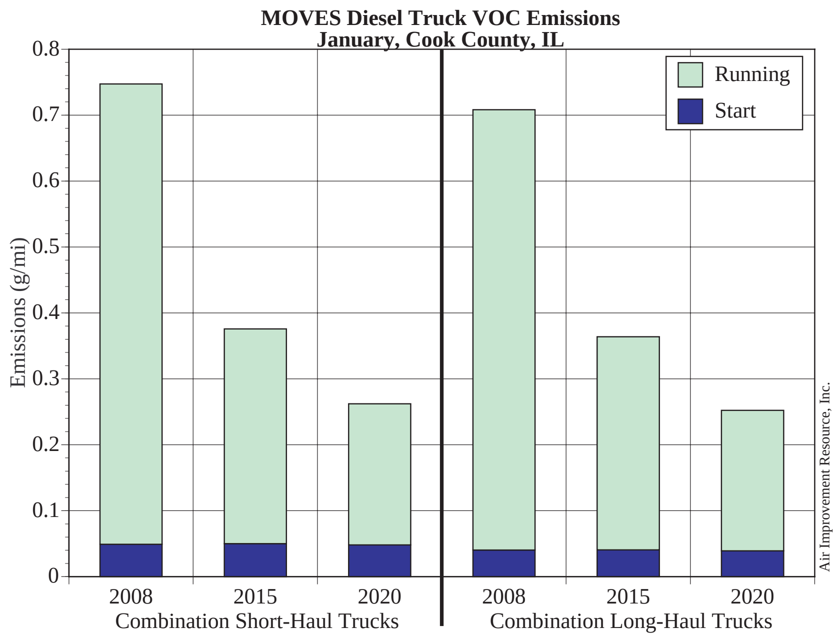


Figure 10-22. Diesel Truck VOC in Summer

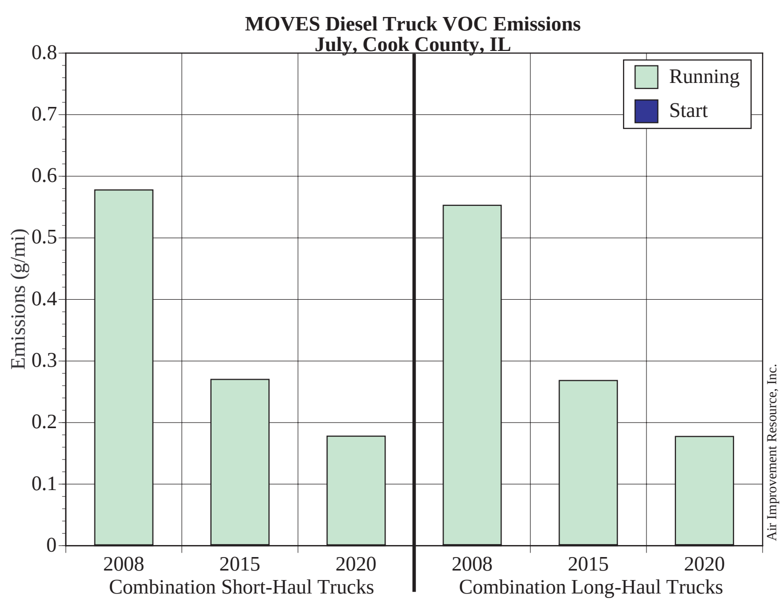


Figure 10-23. Diesel Truck CO in Winter

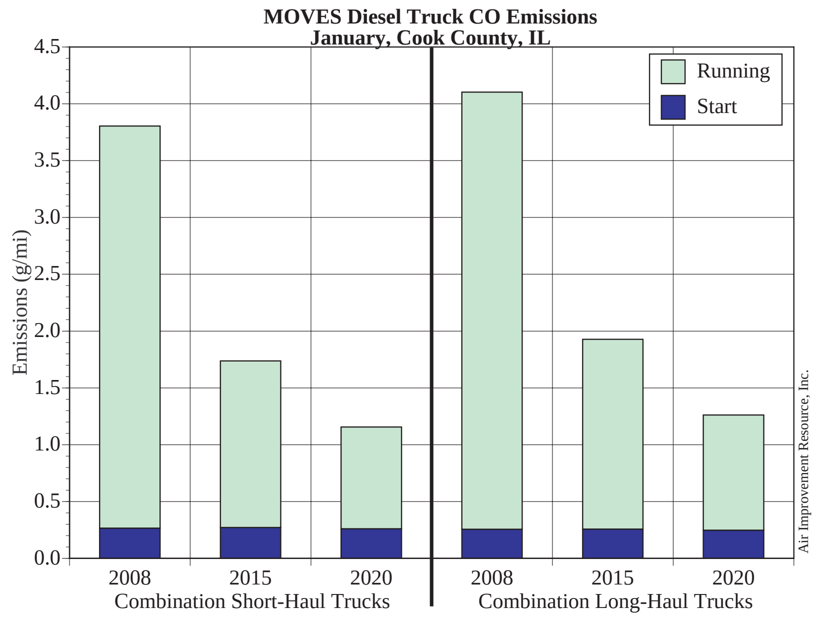


Figure 10-24. Diesel Truck CO in Summer

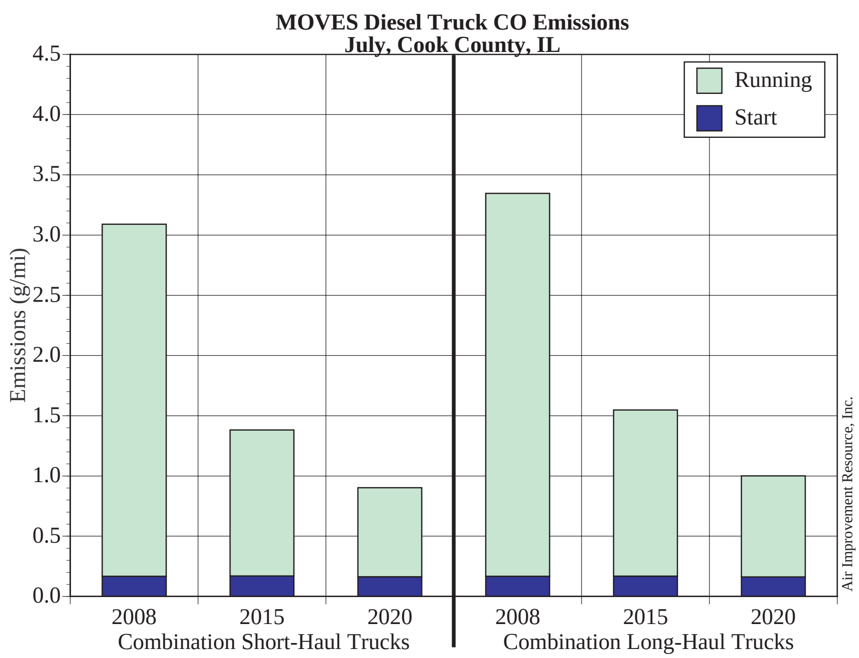


Figure 10-25. Diesel Truck NOx in Winter

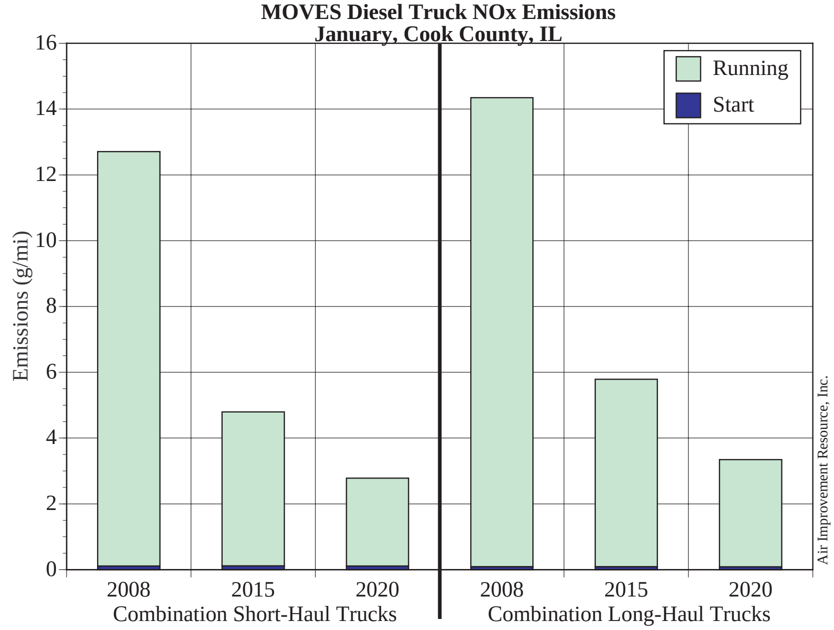


Figure 10-26. Diesel Truck NOx in Summer

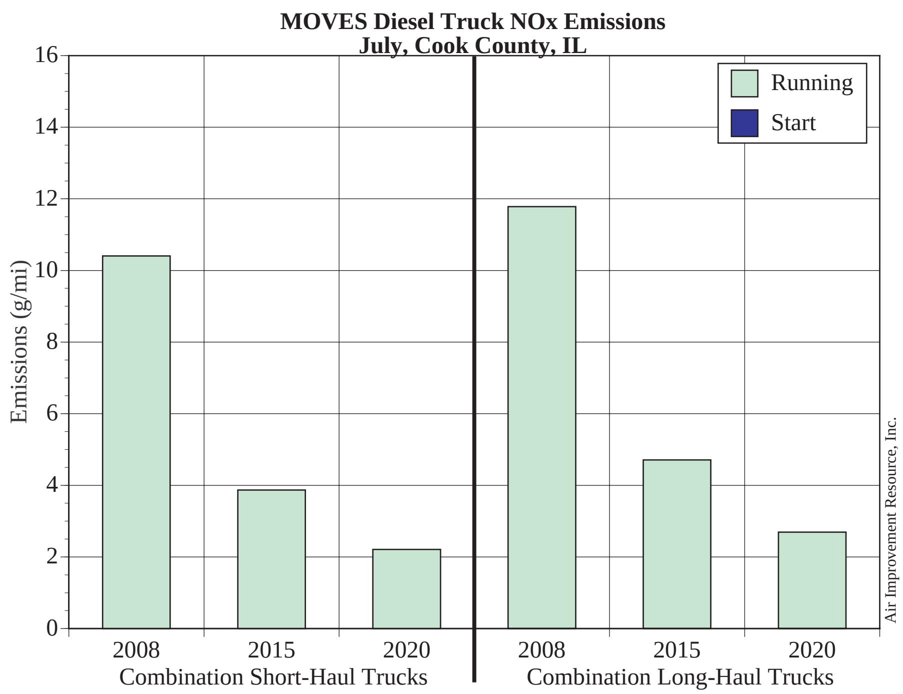


Figure 10-27. Diesel Truck PM in Winter

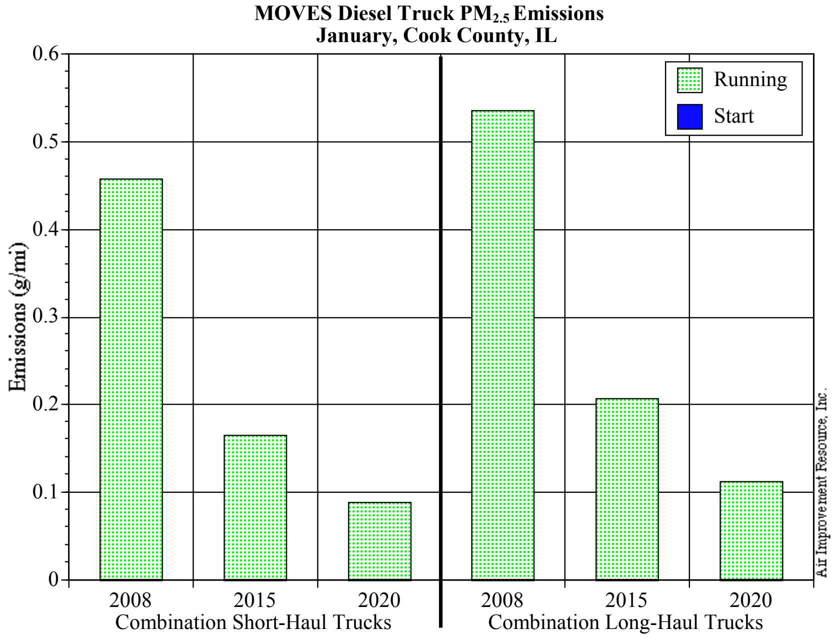


Figure 10-28. Diesel Truck PM in Summer

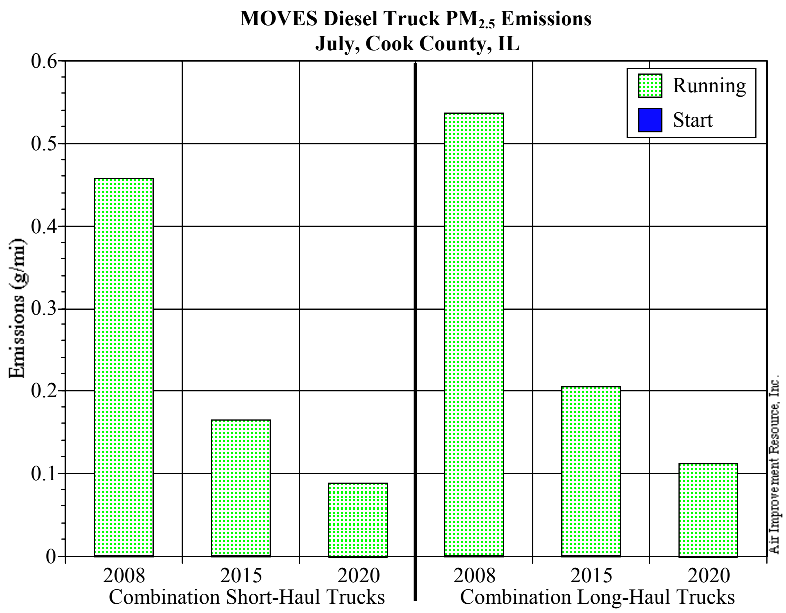


Figure 10-29 through 10-33 show a further breakdown of long-haul diesel emissions by the emission components in the summertime – crankcase, start, refueling (HC only) running, and extended idle. Emissions in tons are for a one month period. Results include any activity changes predicted by MOVES. Figure 10-29 shows that extended idle emissions are estimated to be the majority of THC emissions, followed by running

THC emissions. The running THC emissions are reduced in the future, but extended idle emissions are not.

Figure 10-30 shows that running emissions are the major CO component in 2008, but extended idle emissions become the major component by 2020. Crankcase and start CO emissions are small. Figure 10-31 for NOx also shows the reduction in running NOx emissions due to new emission standards, but idle NOx increases. Figure 10-32 shows large reductions in running PM due to lower emission standards. The idle PM emissions are reduced with time because EPA has not yet updated these in the draft model to be consistent with the idle speeds assumed for HC, CO, and NOx (currently they are based only on the slow idle speed). EPA plans to revise the idle emissions for PM for the final model, so it is likely that with the final model idle PM will increase with time. As indicated earlier in the document, the idle emissions should be based on the driver survey information of time at low idle and high idle, instead of the EPA assumption that all extended idles are at the high idle emission rate.

Figure 10-29. Diesel Long Haul THC Components

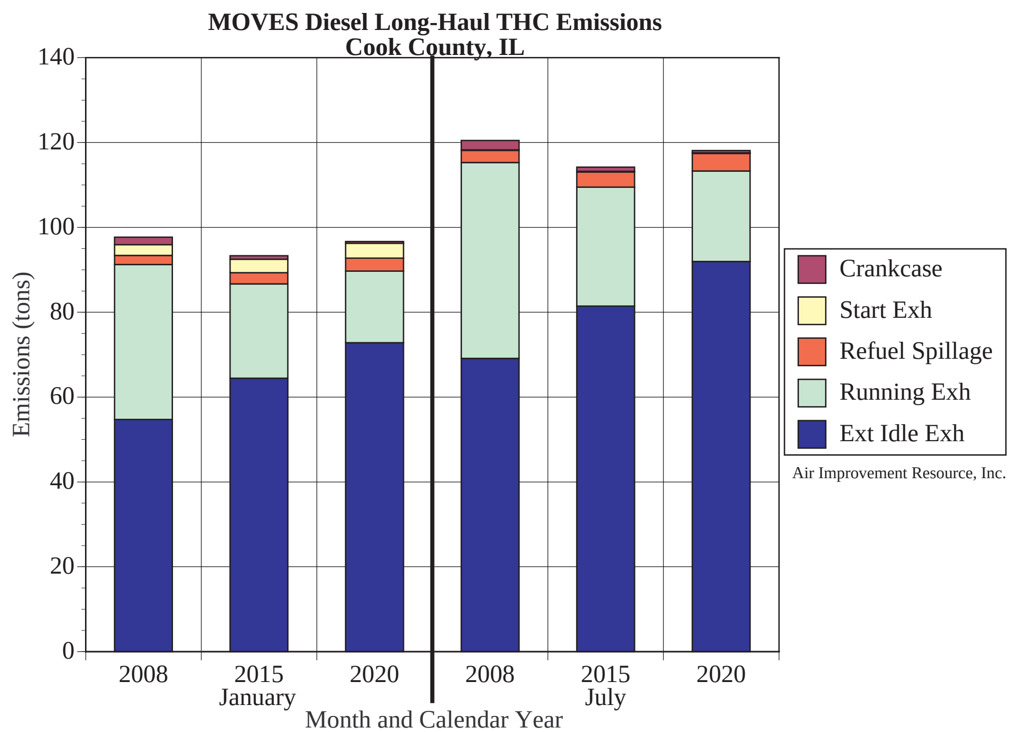


Figure 10-30. Diesel Long Haul CO Components

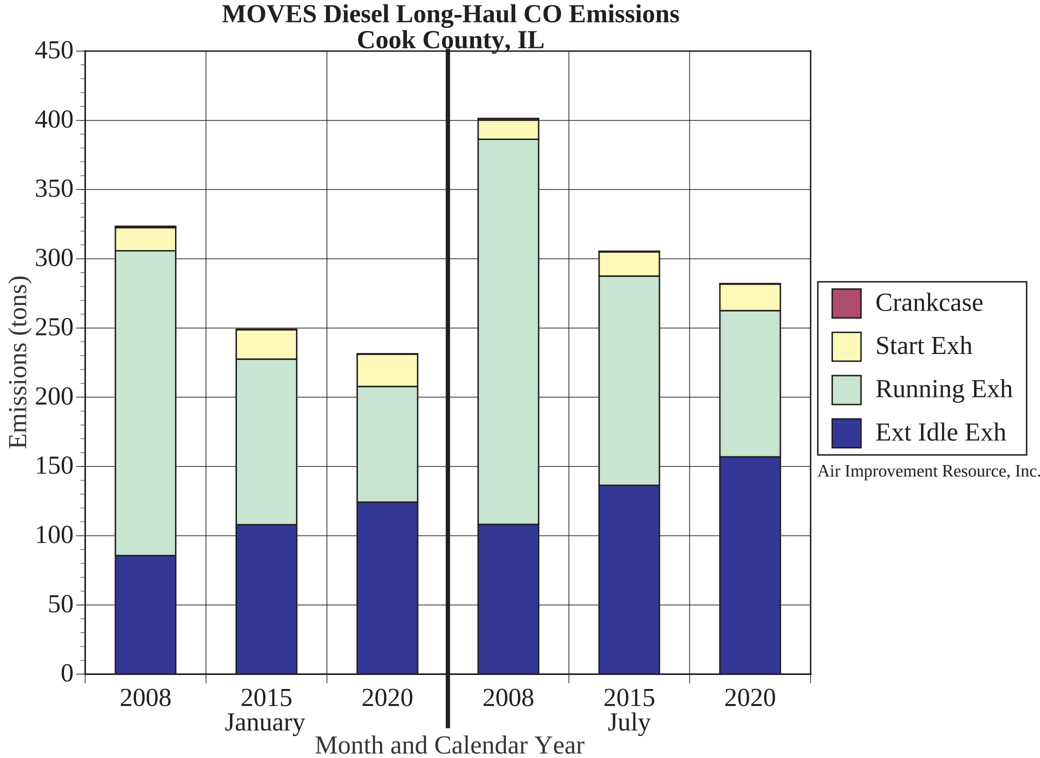


Figure 10-31. Diesel Long Haul NOx Components

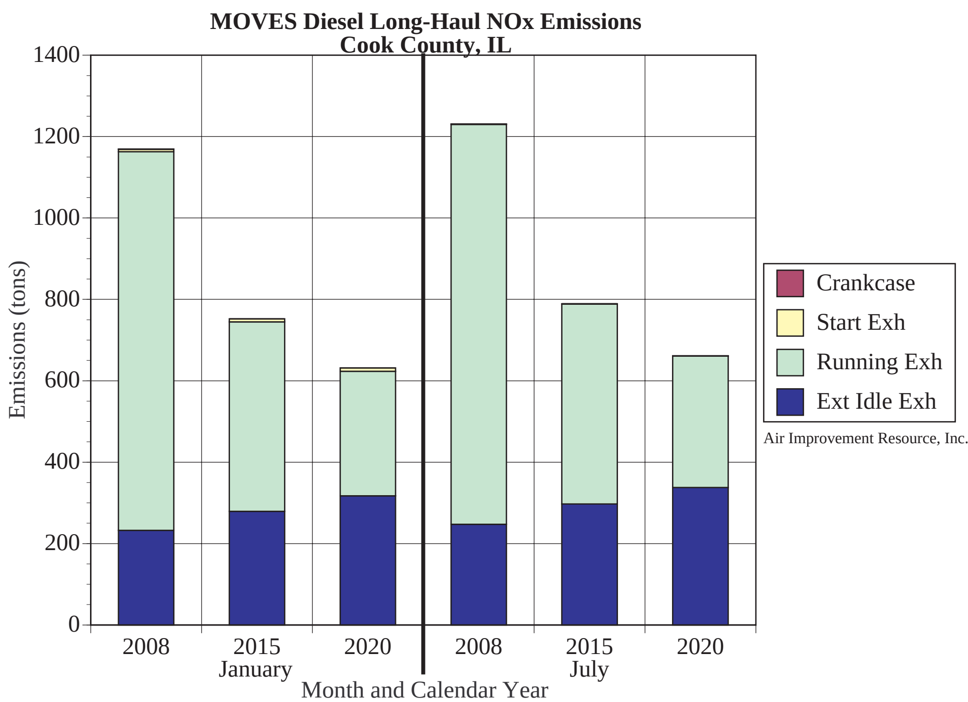
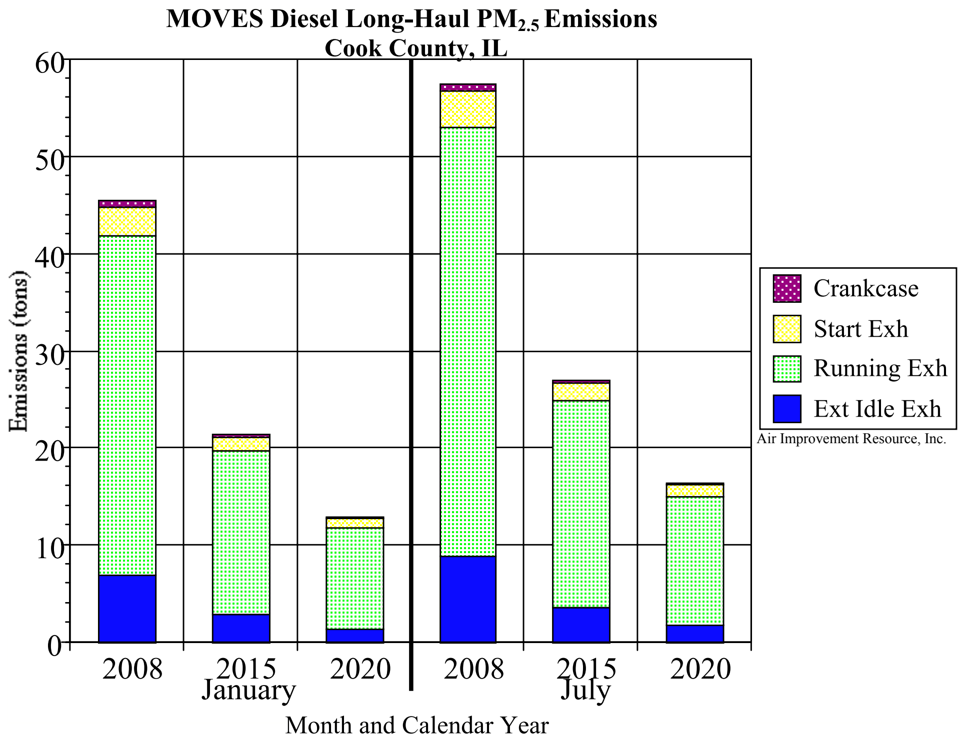


Figure 10-32. Diesel Long Haul $PM_{2.5}$ Components



10.4 1995 and 2008 Model Year Passenger Cars

This section shows emissions for 1995 and 2008 model year passenger cars as they age, from calendar year 2008 to calendar year 2020. The 1995 vehicles are Tier 1 vehicles. The 2008 model year vehicles are Tier 2 vehicles with near-zero evaporative emission controls. In calendar year 2008, the 1995 vehicles would be 13 years old and have over 100,000 miles, and in 2020 they would be 25 years old and have quite high miles. The 2008 model year vehicles would be new in calendar year 2008, and 12 years old in 2020.

Winter and summer evaporative emissions for 1995 model year cars are shown in Figures 10-33 and 10-34. The primary driver for increased evaporative emissions are increased leaks, but in the summer, there are also increases in venting emissions. For example, in 2008, leak emissions are about 0.05 g/mi. But in 2020 the leak emissions are over 0.4 g/mi. It is not clear why winter leaking emissions are higher than summer leak emissions, unless the vehicle miles traveled per vehicle has a seasonality pattern where the VMT is lower in the winter than in the summer.

For the summer plot, it appears that permeation emissions are also projected to increase with vehicle age. However, as indicated in Section 6 permeation emissions are not assumed to increase with age. It is possible that the increase in permeation emissions on a g/mi basis could be due to the same per vehicle permeation emissions, but lower vehicle miles traveled per vehicle. This should be investigated further.

Figure 10-33. 1995 MY Evaporative VOC in Winter

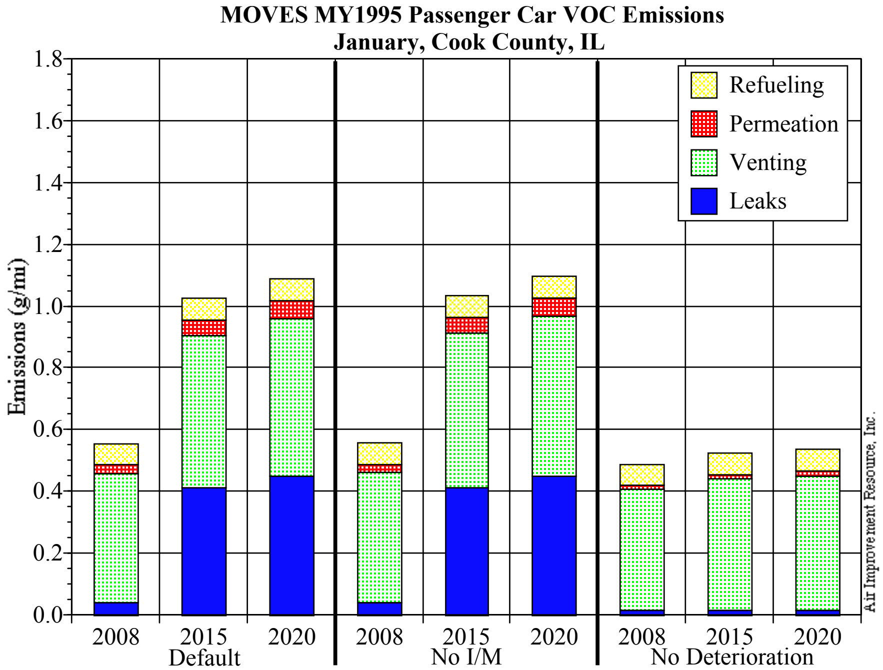
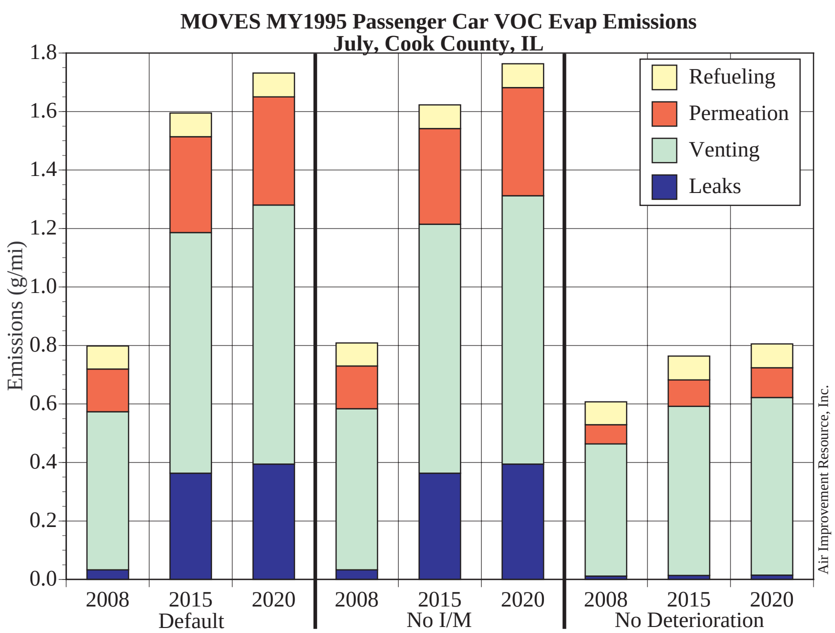


Figure 10-34. MY 1995 Evaporative VOC in Summer



VOC exhaust emissions for 1995 vehicles are shown in Figures 10-35 and 10-36. These show increases in both start and running emissions with vehicle age. The reason that no deterioration case emissions increase with age is unclear; perhaps it is because the starts stay about the same, but the vehicle miles traveled decrease. The same basic patterns are shown for CO emissions in Figures 10-37 and 10-38.

Figure 10-35. MY 1995 Exhaust VOC in Winter

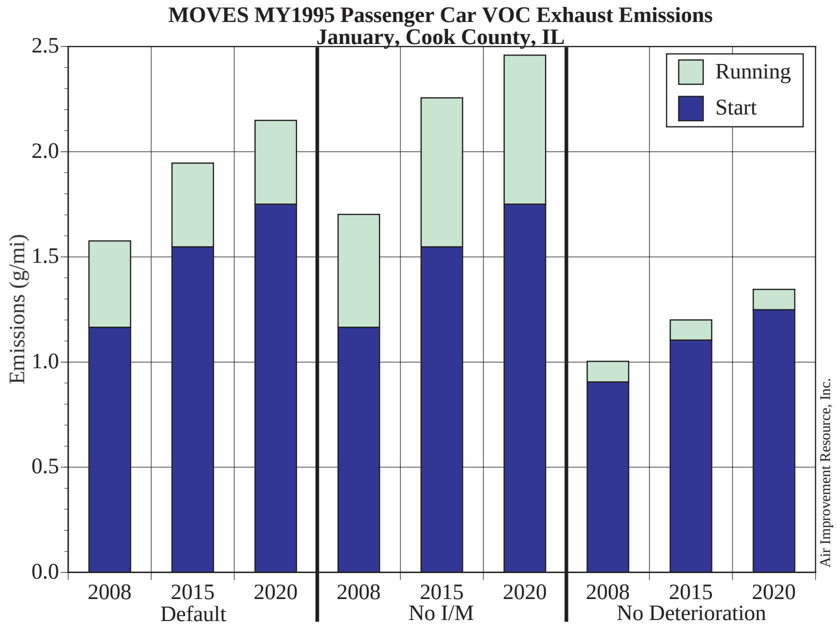


Figure 10-36.MY 1995 VOC in Summer

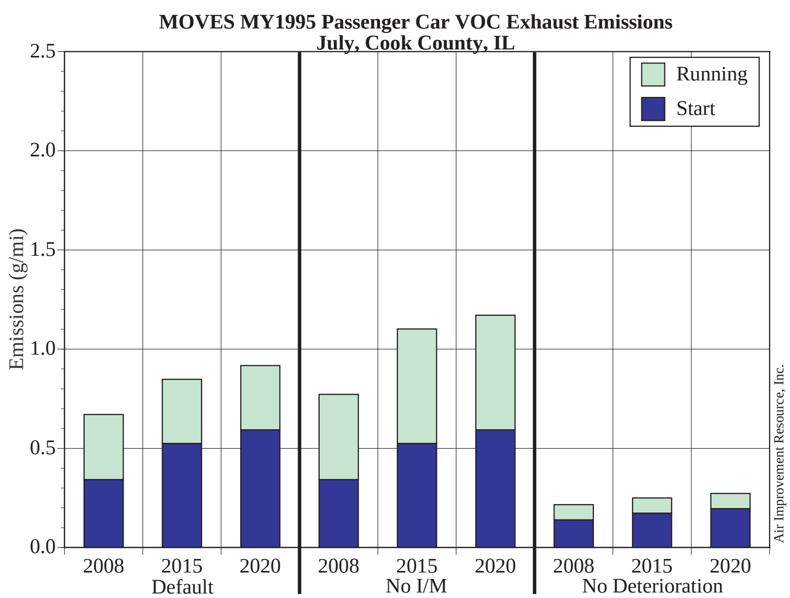


Figure 10-37. MY 1995 CO in Winter

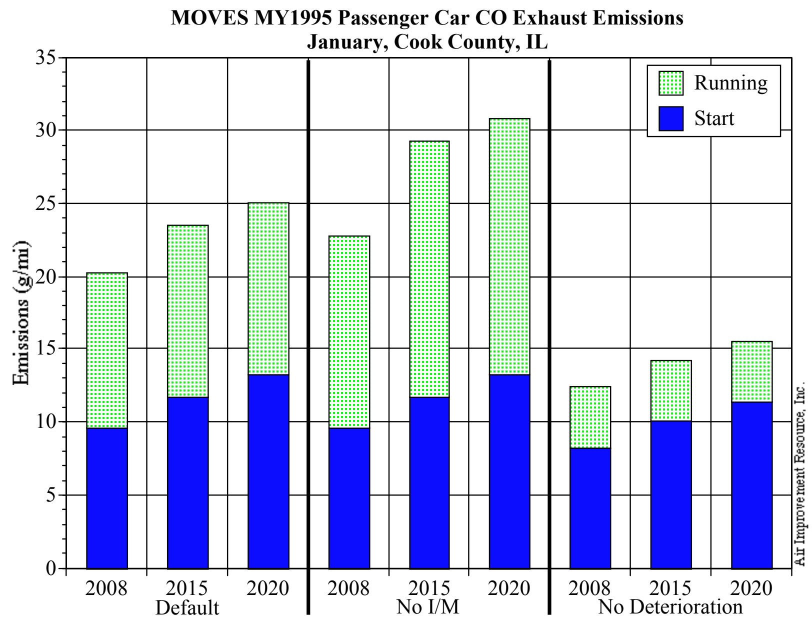
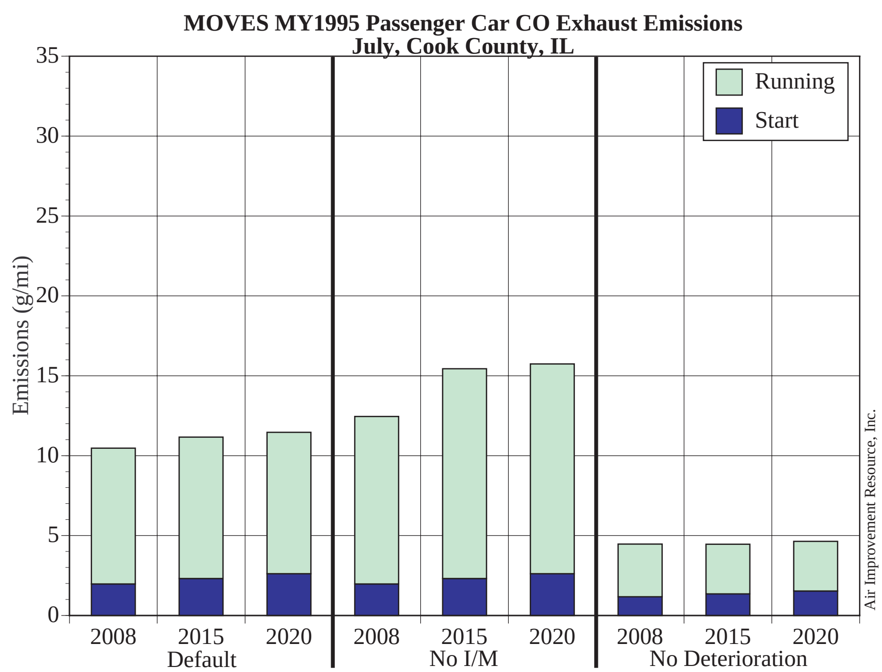


Figure 10-38. MY 1995 CO in Summer



NOx emissions are shown in Figures 10-39 and 10-40. The NOx emissions for the default and no-I/M cases are already quite high in 2008, owing to the fact that the 1995 model year vehicles already have in excess of 100,000 miles (on average) in 2008. Winter emissions are somewhat higher than summer emissions. For the default case, NOx emissions in the summer are almost 1.5 g/mi, which is quite high considering these vehicles are certified to NOx emission standards of 0.4 g/mi (on the FTP) at 100,000 miles. The no deterioration case is about 0.7 g/mi. This level could be considered the level of emissions that reflects real-world driving versus the FTP. The increase from 0.7 to 1.5 is an estimate of EPA's estimated deterioration for NOx. The reason for this increase in NOx is the log-linear increase in emissions versus age as estimated from the I/M data.

Figure 10-39. MY 1995 NOx in Winter

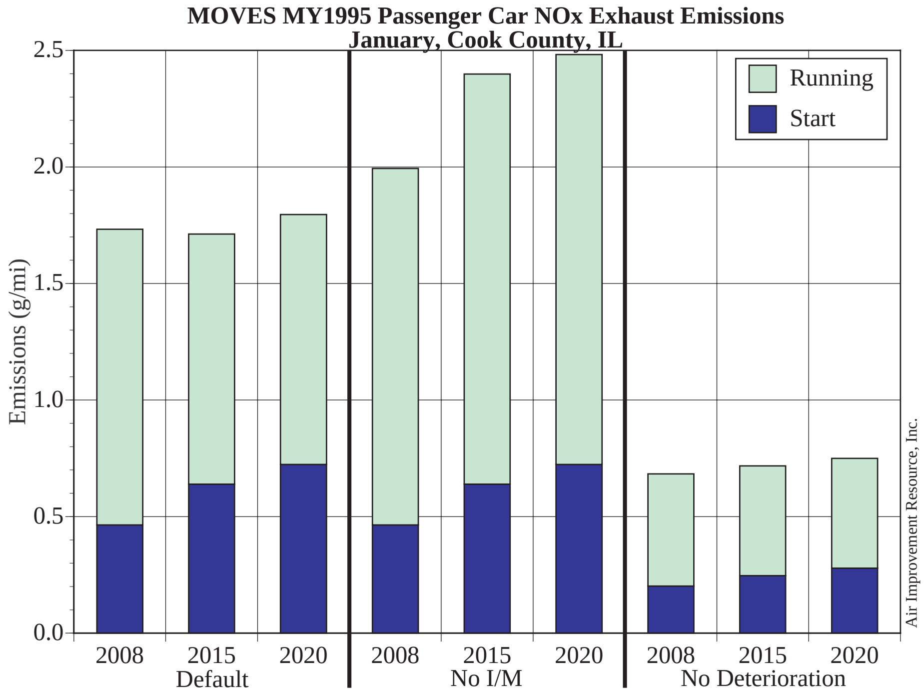
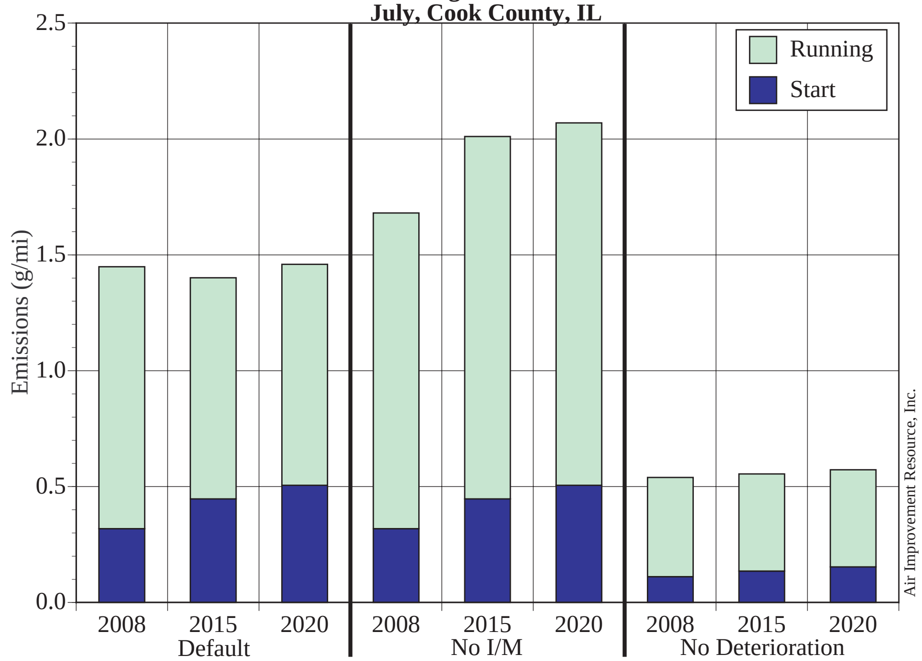


Figure 10-40. MY 1995 NOx in Summer

MOVES MY1995 Passenger Car NOx Exhaust Emissions
July, Cook County, IL



$PM_{2.5}$ emissions for the 1995 model year vehicles are shown in Figures 10-41 and 10-42. The default and no I/M cases are equivalent, and are much higher than the no deterioration cases for the January plot. There are large increases in both start and running emissions. By the time these vehicles are 20 years old (2015), the model is assuming average winter emissions of around 145 mg/mi. While the ages are younger, the range of cold temperature emissions of 79 Tier 1 vehicles in the Kansas City study are 1 to 189 mg/mi, with an average of 16 mg/mi. So clearly, EPA is projecting PM emissions from these vehicles become much higher than they are in Kansas City (almost 10x higher). This may not be reasonable, and is partly the result of EPA's temperature correction factors being applied to running as well as cold start emissions.

Figure 10-41. MY 1995 PM in Winter

MOVES MY1995 Passenger Car $PM_{2.5}$ Exhaust Emissions

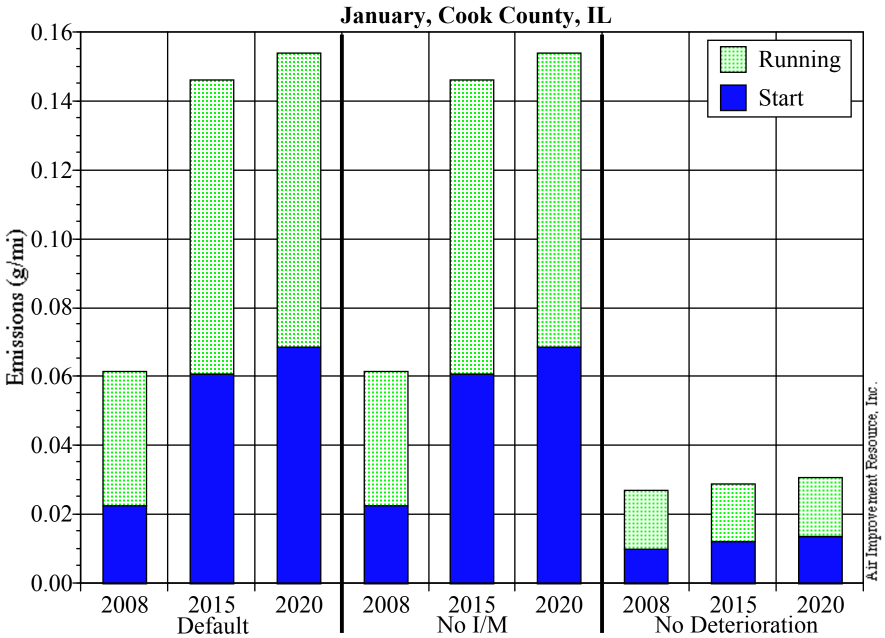
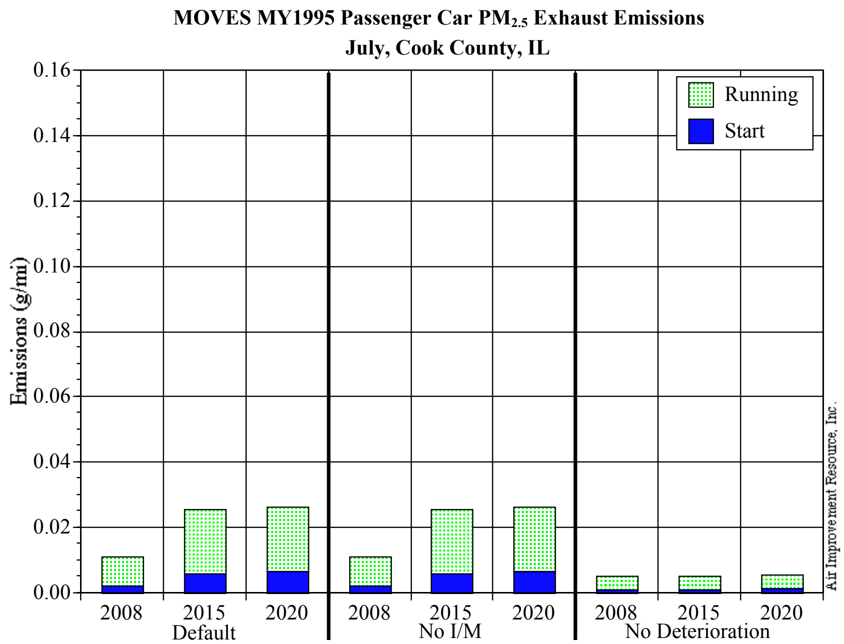

January, Cook County, IL
Running
Start
Emissions (g/mi)
0.16
0.14
0.12
0.10
0.08
0.06
0.04
0.02
0.00
2008
2015
2020
Default
No I/M
No Deterioration
Air Improvement Resource, Inc.

Figure 10-42. MY 1995 PM in Summer



Passenger car evaporative emissions for the 2008 (Tier 2) model year are shown in Figures 10-43 and 10-44. Emissions in 2008 and 2015 are about the same, but an increase is seen in 2020. The major reason for the increase is leaks, but in the summer there are also increases in permeation and venting emissions.

Figure 10-43. MY 2008 Evaporative VOC in Winter

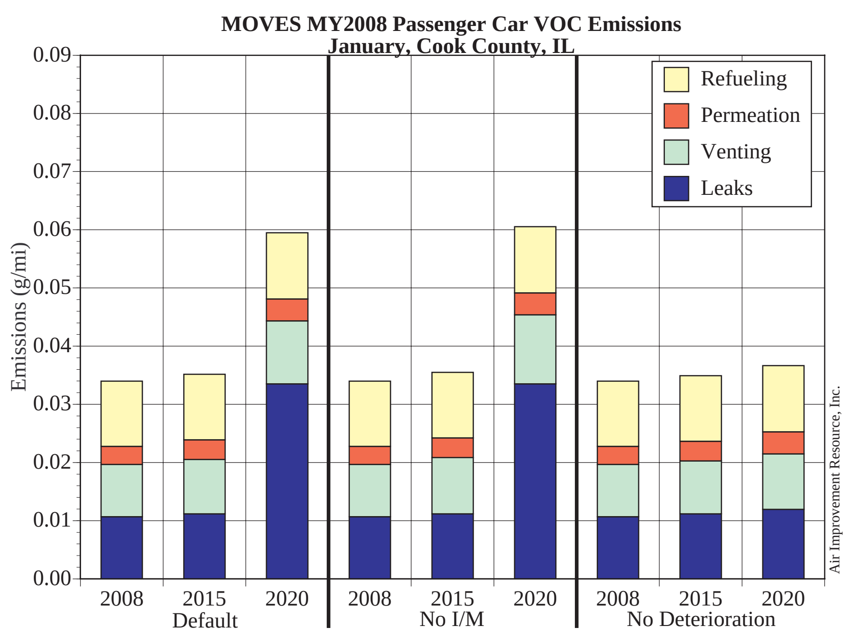
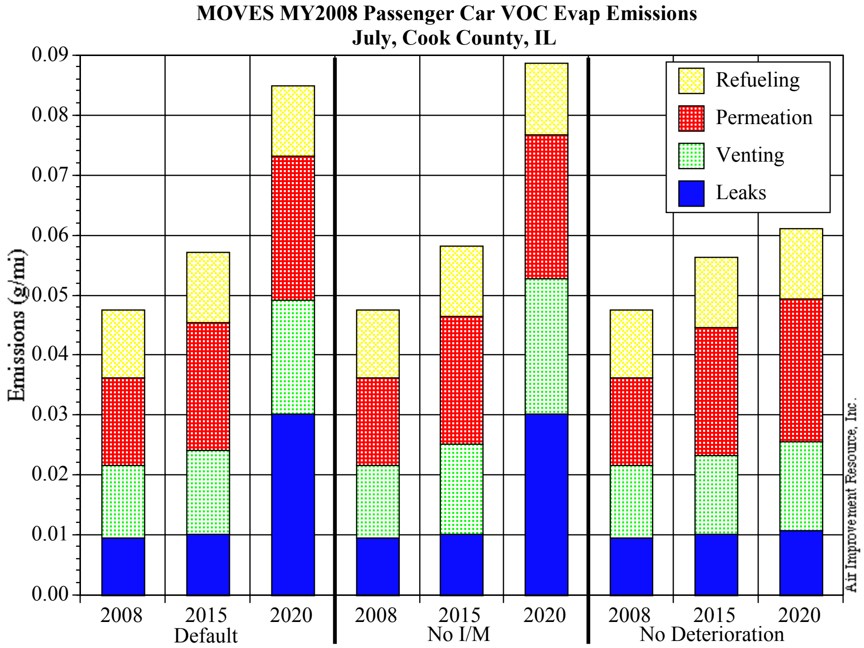


Figure 10-44. MY 2008 Evaporative VOC in Summer



VOC exhaust emissions for the 2008 model year are shown in Figures 10-45 and 10-46. Winter emissions increase with age, and the increase is due to higher start emissions, with a very small contribution due to increased running emissions. Winter emissions for 2008 model year vehicles in calendar year 2008 are 0.26 g/mi, and only 0.03 g/mi in the summer, so the winter emissions are estimated to be over 8 times higher than they are in the summer.

Figure 10-45. MY 2008 Exhaust VOC in Winter

MOVES MY2008 Passenger Car VOC Exhaust Emissions

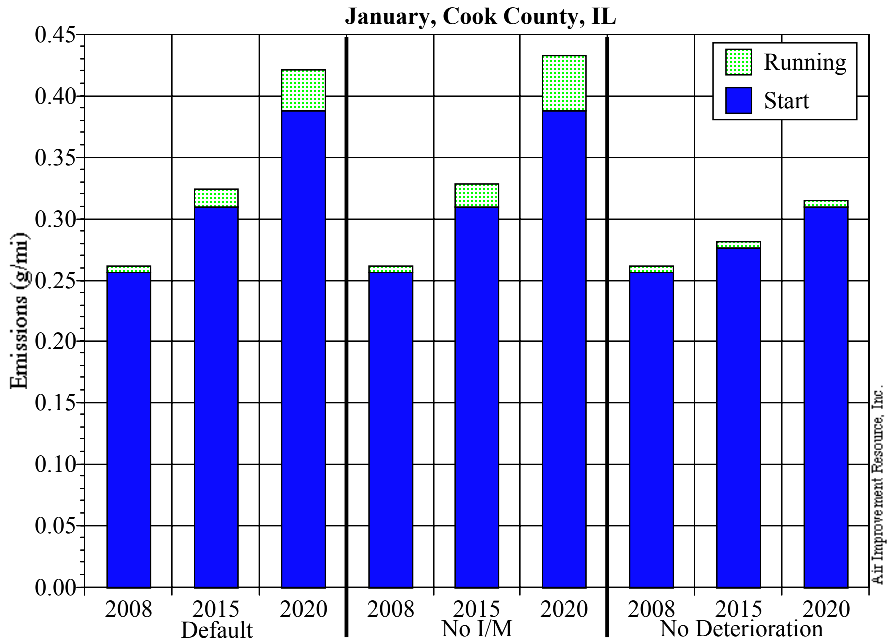
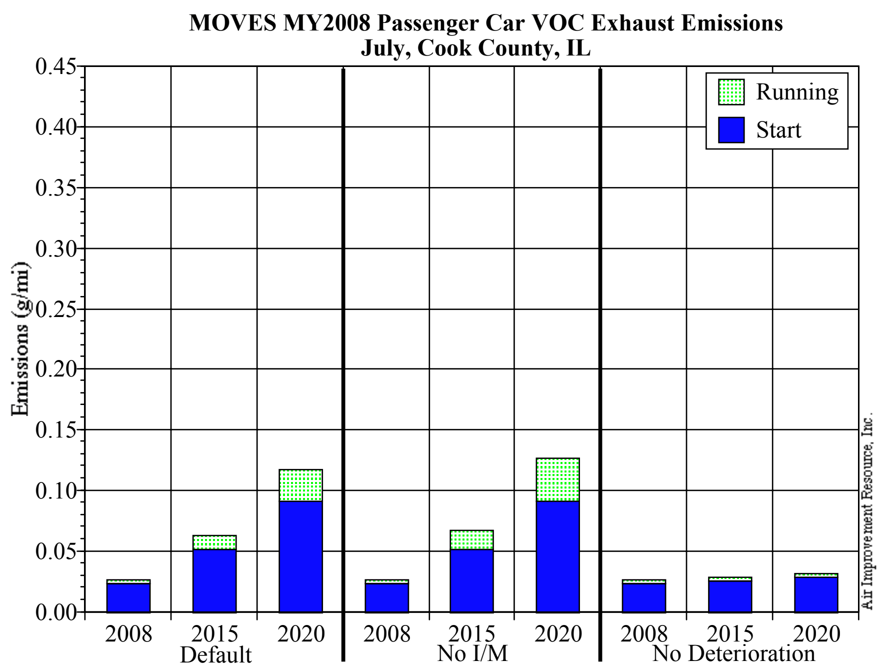


Figure 10-46. MY 2008 Exhaust VOC in Summer



CO emissions are shown in Figures 10-47 and 10-48. Emissions increase with age, but unlike HC, both the start and running emissions increase.

Figure 10-47. MY 2008 CO in Winter

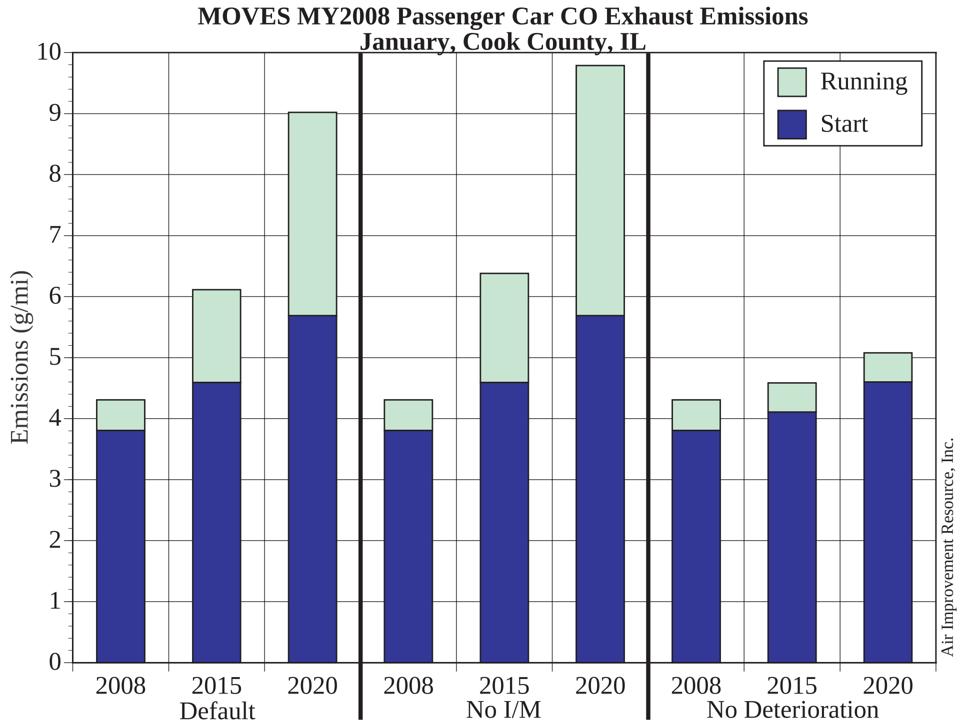
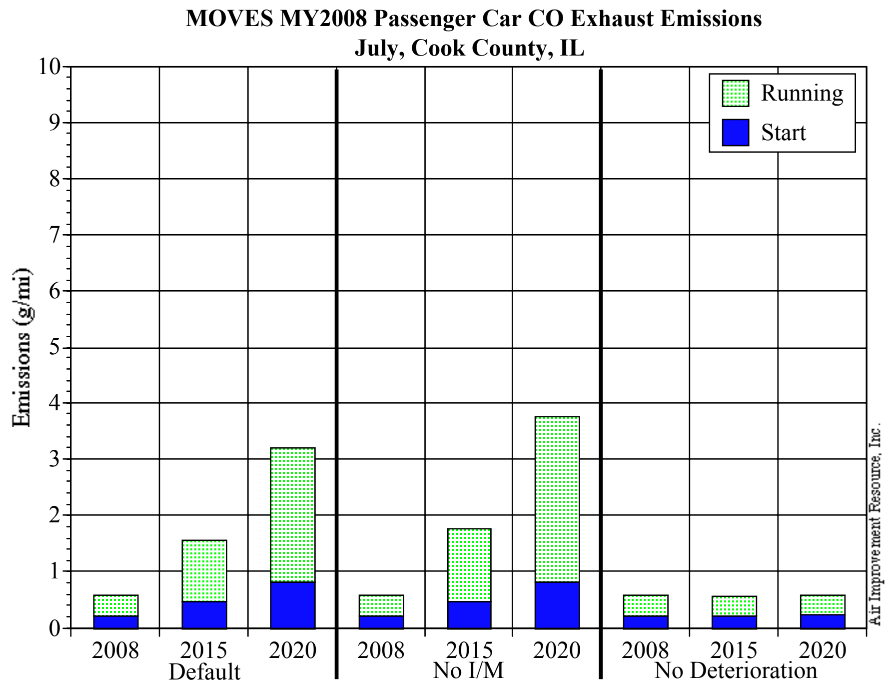

MOVES MY2008 Passenger Car CO Exhaust Emissions
January, Cook County, IL
Emissions (g/mi)
0
1
2
3
4
5
6
7
8
9
10
Running
Start
2008
2015
2020
Default
2008
2015
2020
No I/M
2008
2015
2020
No Deterioration
Air Improvement Resource, Inc.

Figure 10-48. MY 2008 CO in Summer



NOx emissions for the 2008 model year are shown in Figures 10-49 and 10-50. There are very significant increases in NOx emissions versus age. For the no-I/M case for summer, NOx emissions are estimated to increase from about 0.04 g/mi to over 0.2 g/mi, a five-fold increase. Both start and running emissions increase. Even with I/M, the NOx emissions increase to about 0.18 g/mi, a four-fold increase. This seems excessive, and is probably driven by the log/linear deterioration increase at moderately high VSP levels.

Figure 10-49. MY 2008 NOx in Winter

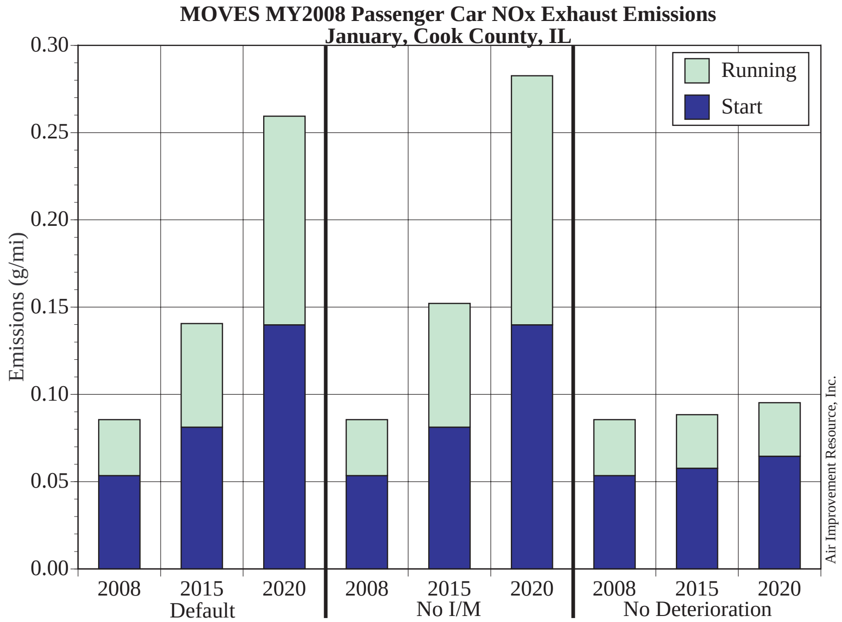
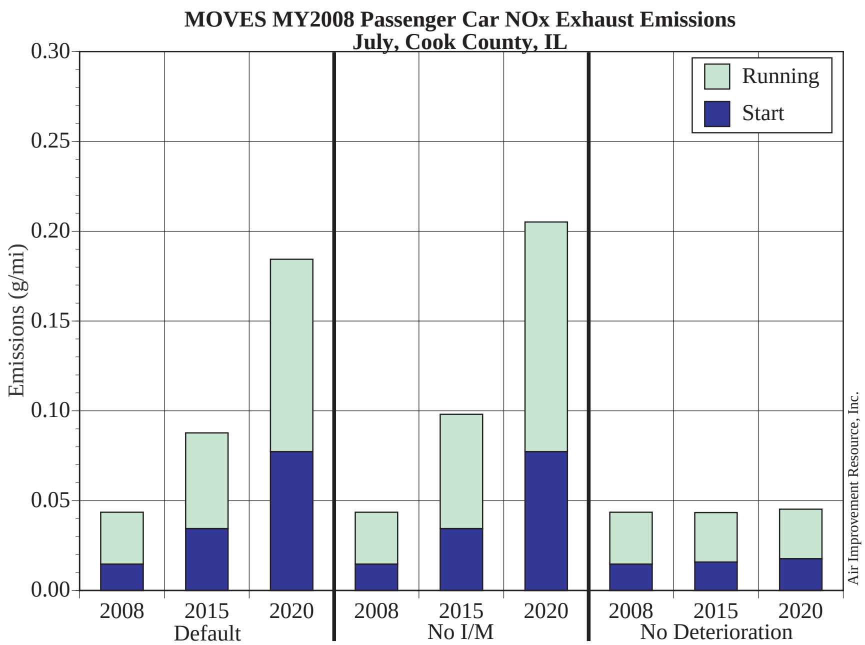


Figure 10-50. MY 2008 NOx in Summer



$PM_{2.5}$ emissions for the 2008 model year are shown in Figures 10-51 and 10-52. Winter emissions for the default case increase from 13 mg/mi when new in 2008 to over 30 mg/mi in 2020. There are increases in both start and running PM emissions.

Figure 10-51. MY 2008 PM in Winter

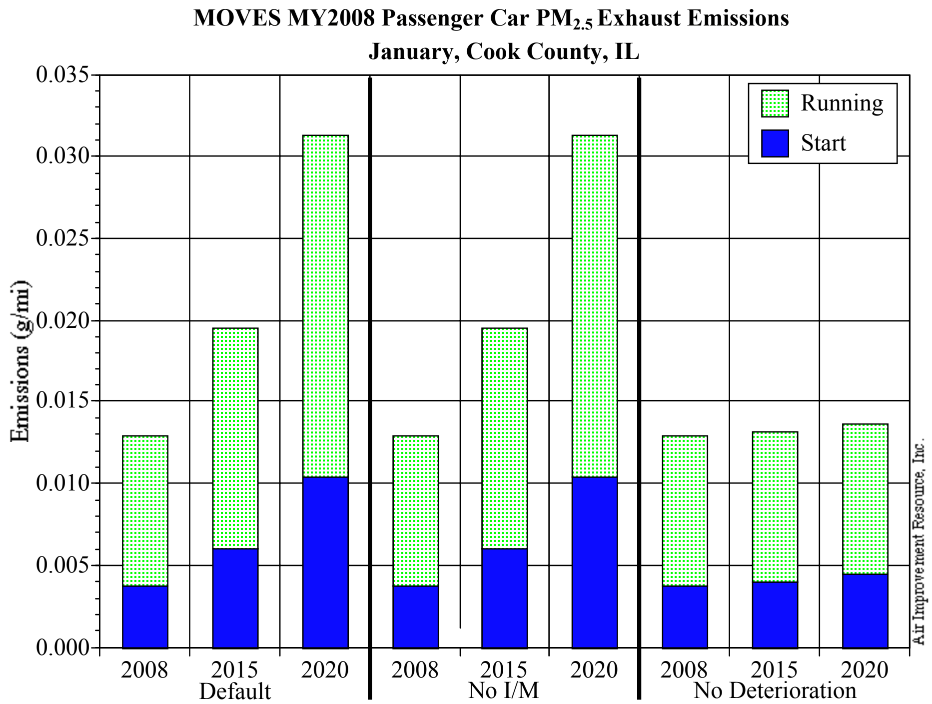
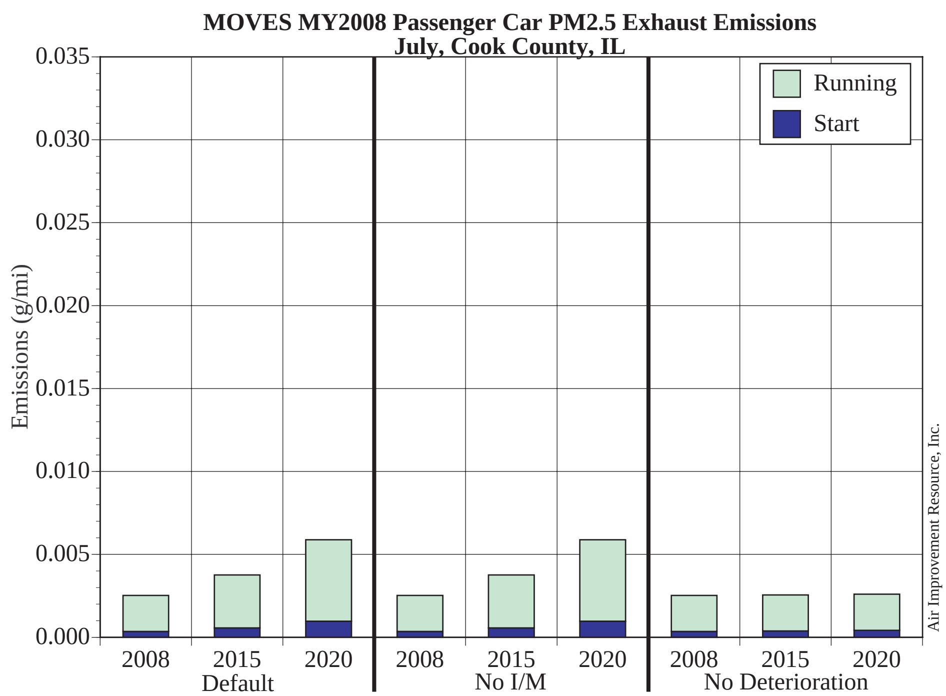


Figure 10-52. MY 2008 PM in Summer



10.5 Comparison of Car and Truck Emissions

Figures 10-53 through 10-60 compare VOC, CO, NOx, and $PM_{2.5}$ emissions for 2008, 2015, and 2020 for cars and trucks. Several vehicle classes are included in this comparison: passenger cars, passenger trucks, light commercial trucks, short-haul trucks, and long-haul trucks. The passenger vehicles are almost entirely gasoline. The light commercial trucks are mostly gasoline. The short-haul trucks are a mixture of gasoline and diesel, and the long-haul trucks would be almost entirely diesel.

For VOC emissions in January, cars and light trucks have higher emissions than heavy trucks on a g/mi basis. In 2015 and 2020, the emissions of cars and light trucks, and heavy trucks are very similar. In summer, car VOC emissions are lower than heavy trucks, but passenger trucks and commercial trucks are somewhat higher than diesel trucks. In 2015 and 2020 for the summer, the emissions of cars and trucks are very similar on a g/mi basis.

For CO, winter or summer, the emissions of gasoline-fueled cars, light passenger trucks and light commercial trucks are higher than diesel-fueled heavy-duty trucks, but these differences narrow with future calendar years.

For NOx, winter or summer, the emissions of heavy-duty trucks are higher than for the other vehicle classes.

Finally, for $PM_{2.5}$, the PM from heavy trucks is high relative to cars in 2008, but the differences are narrowed considerably by 2020 due to the 90% reduction in PM required on diesels starting in model year 2007.

Figure 10-53. Total VOC in Winter

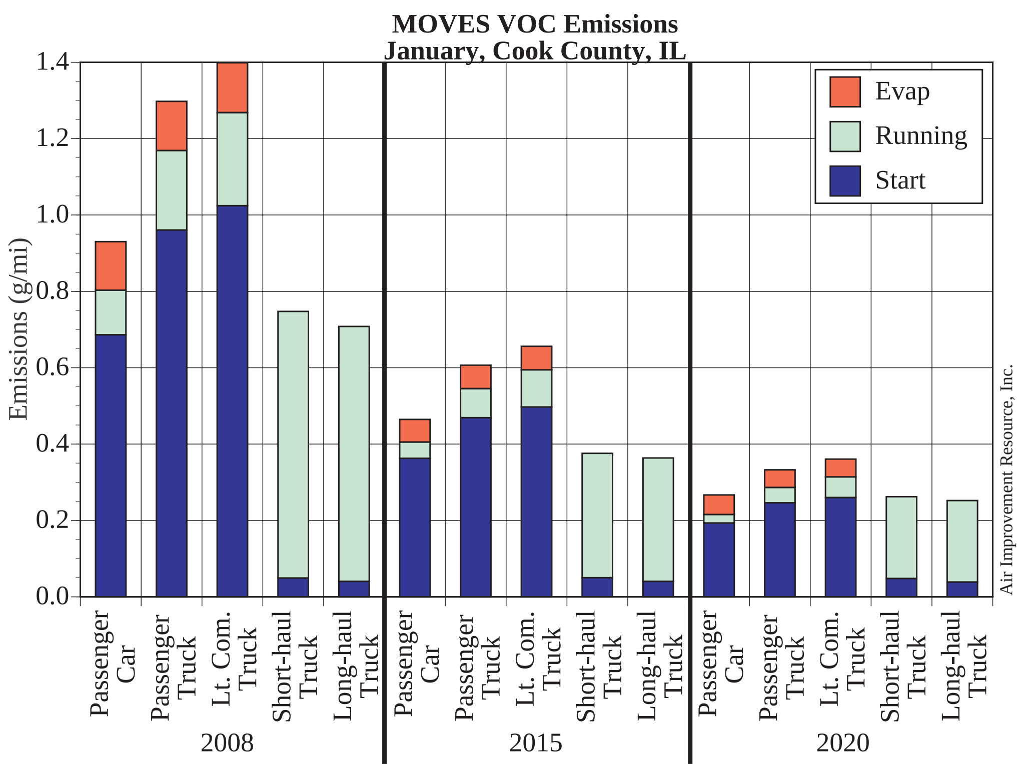


Figure 10-54. Total VOC in Summer

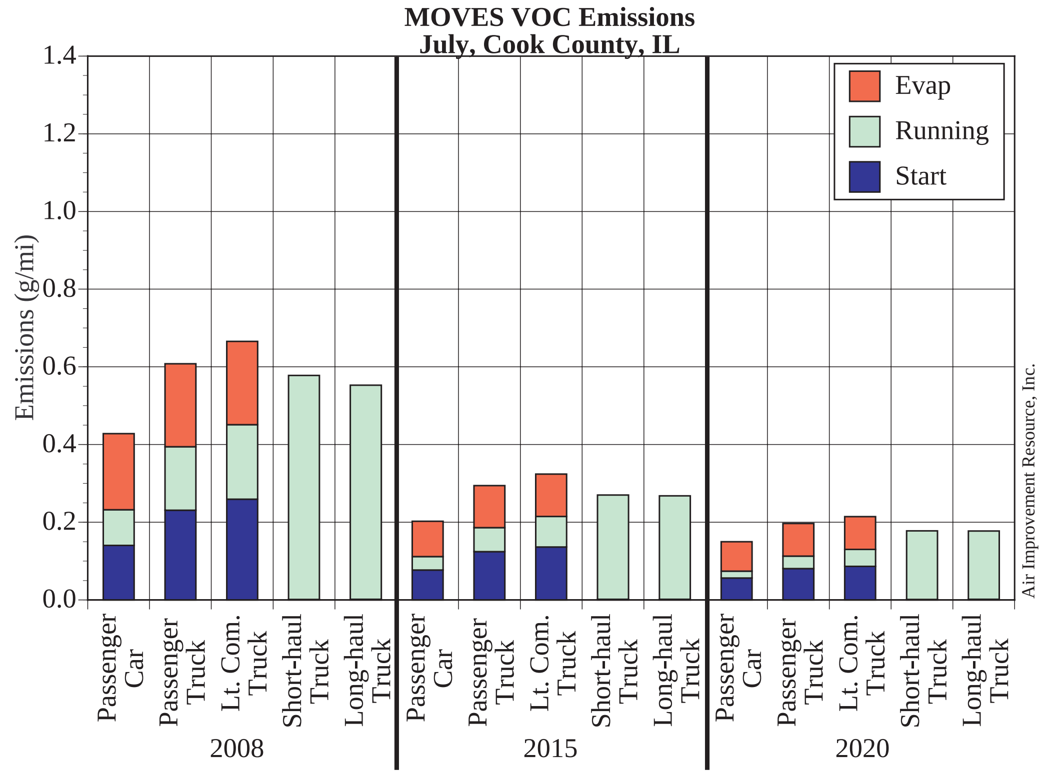


Figure 10-55. CO in Winter

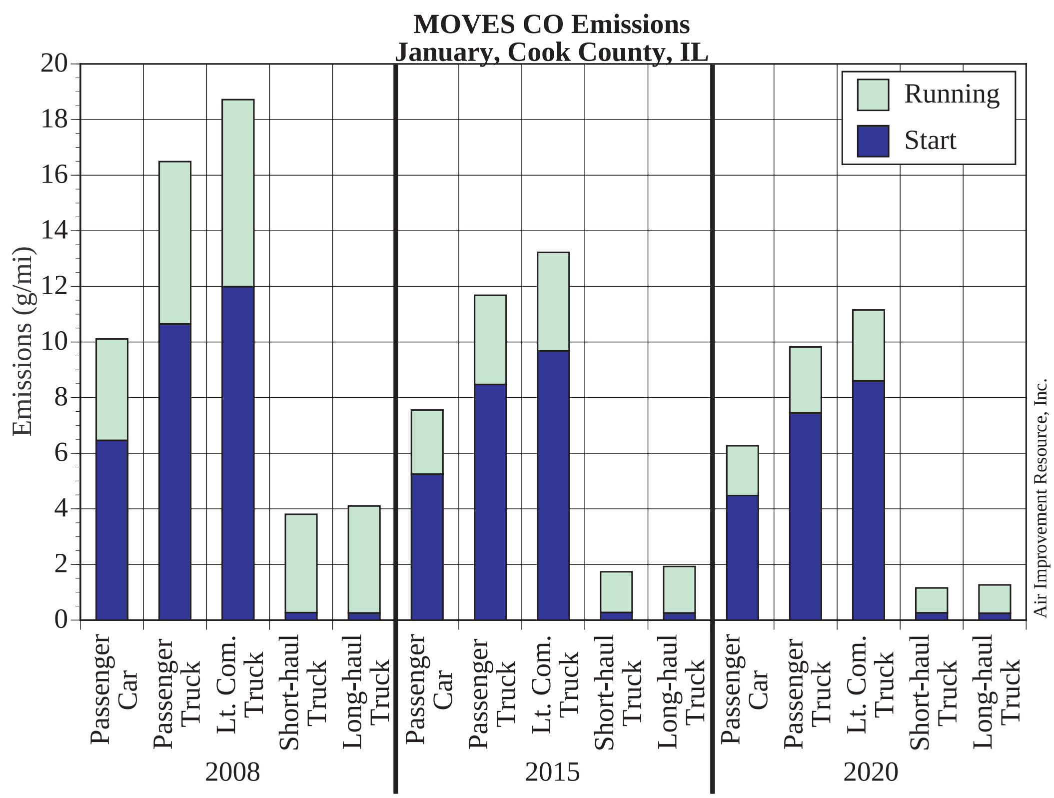


Figure 10-56. CO in Summer

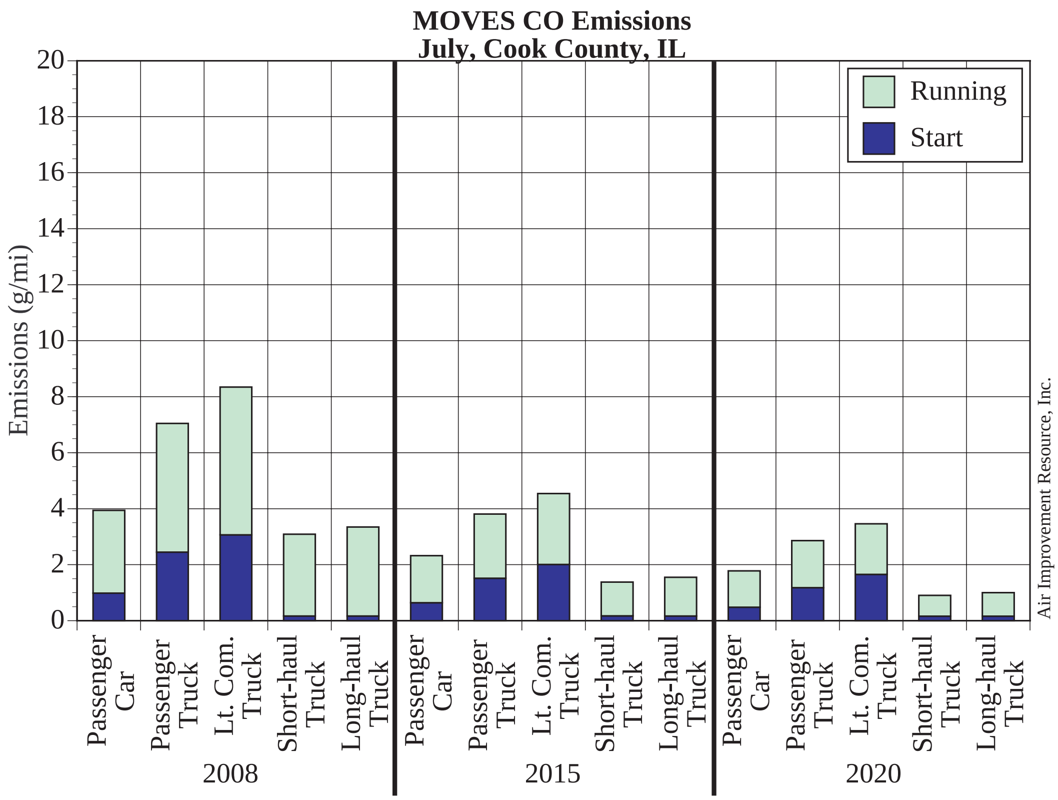


Figure 10-57. NOx in Winter

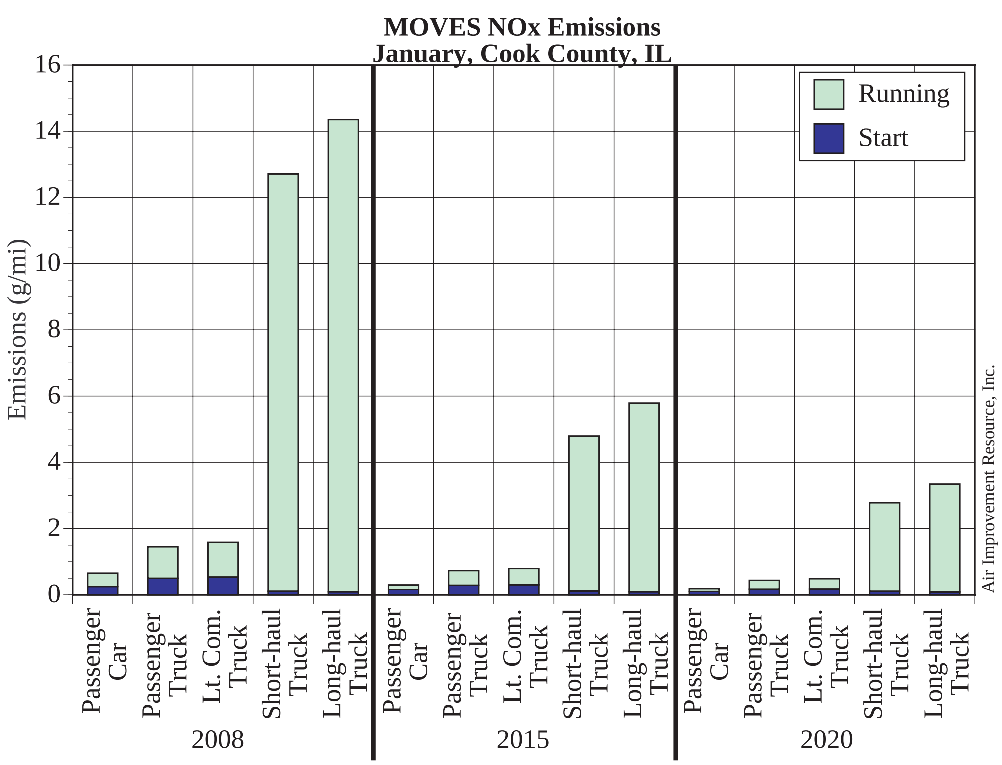


Figure 10-58. NOx in Summer

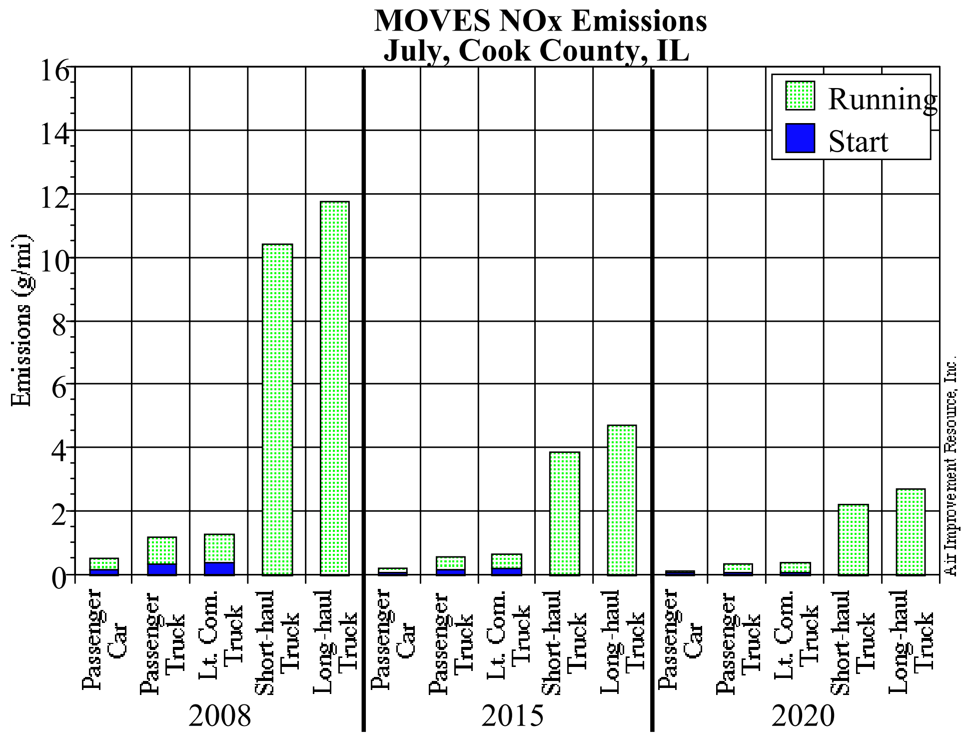


Figure 10-59. PM in Winter

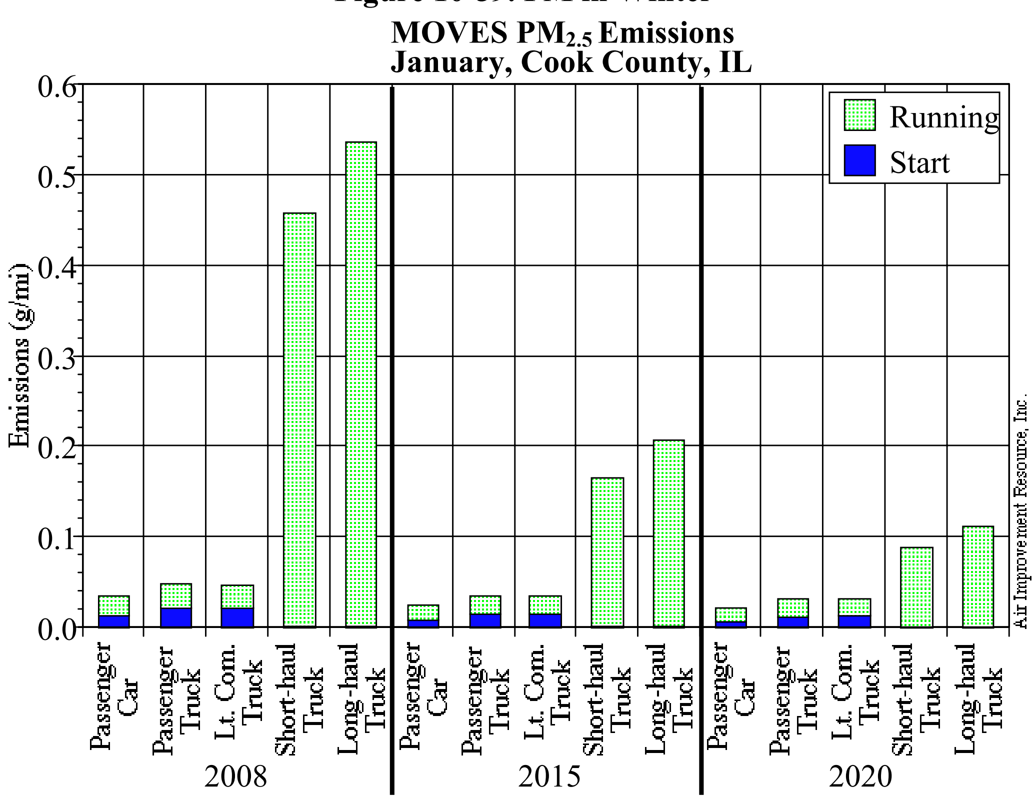
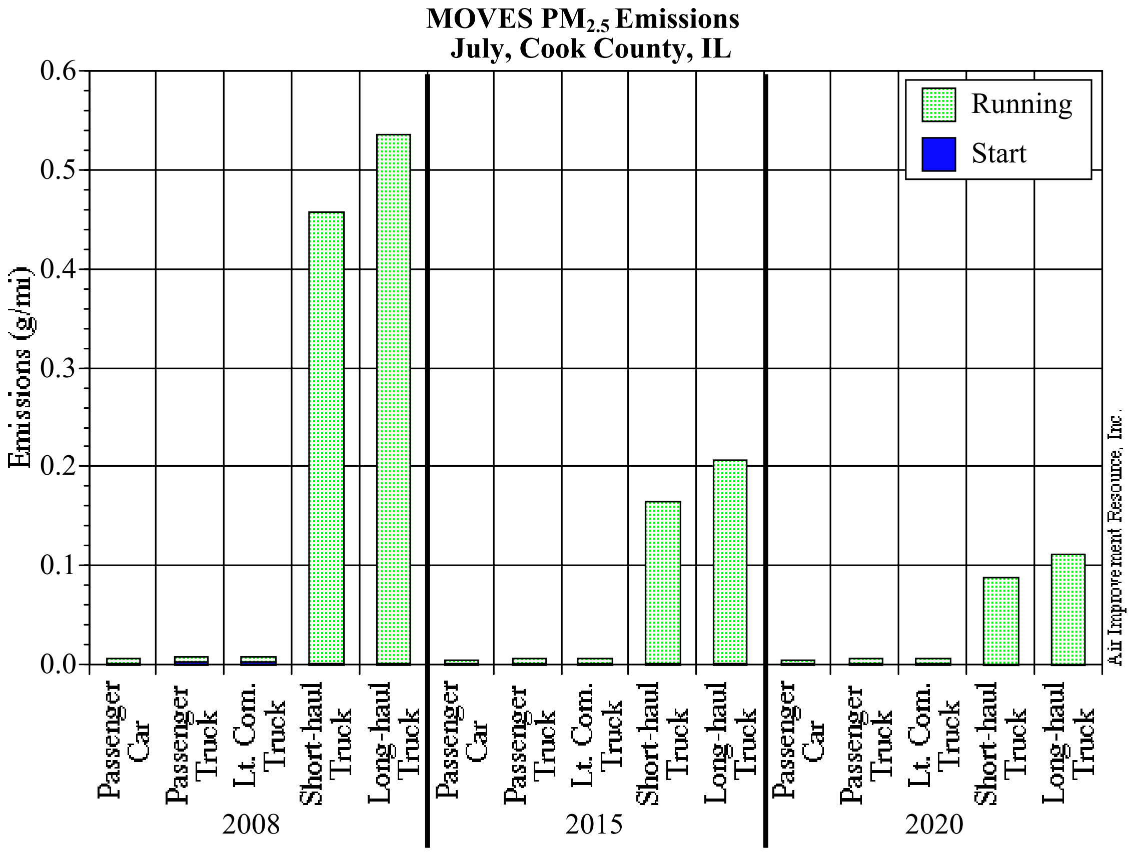


Figure 10-60. PM in Summer



10.6 Impact of the High VSP Correction Factors

As indicated in Section 5.2.4, the running emission rates at higher VSP bins are adjusted using data other than the Arizona I/M data. To evaluate the impact of these adjustments on running VOC, CO, and NOx emissions, an emission rate file for MOVES without the VSP adjustments was created and the model for Cook County was rerun for just model year 2000 passenger cars and passenger trucks. Start emissions are not in these plots. Results are shown in Figures 10-61 to 10-63. The "modified" case is without the high VSP corrections. These plots show that the high VSP correction factors appear to have little effect on overall emissions from model year 2000 cars and light trucks.

Figure 10-61. Exhaust VOC High VSP Adjustment Effects

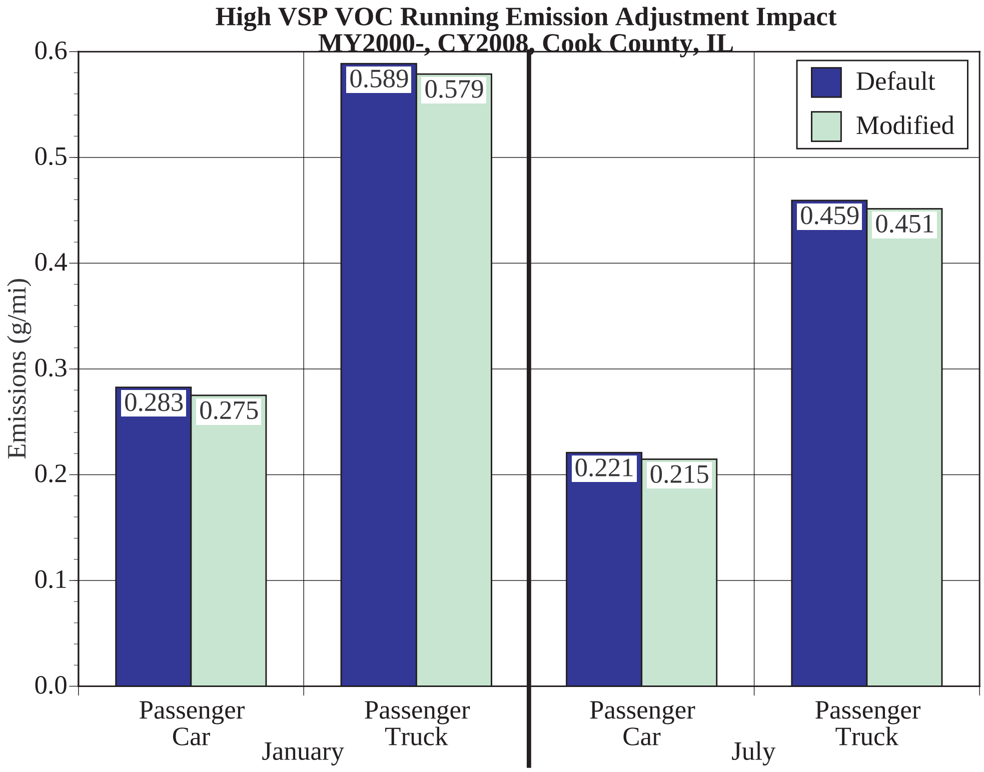


Figure 10-62. CO High VSP Adjustment Effects

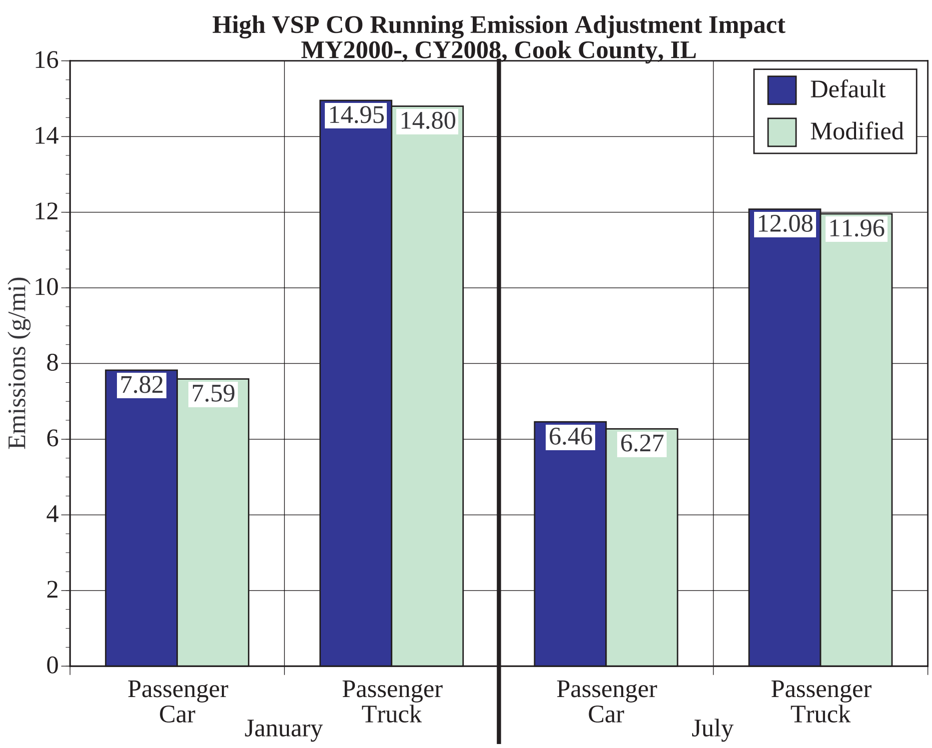
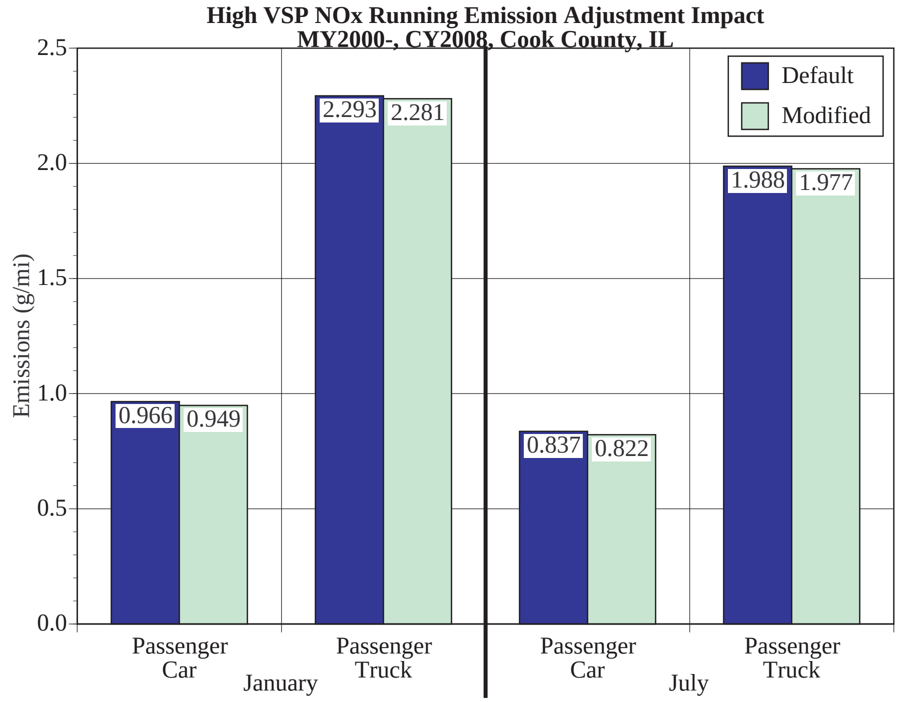


Figure 10-63. NOx High VSP Adjustment Effects



10.7 Tier 2 Emissions Compared to Current Vehicles

This analysis examines the impact of the EPA Tier 2 vehicle emission standards on fleet emissions. The Tier 2 standards started with the 2004 model year, and were fully phased-in in 2008. Thus, the 2004-2007 model years are a mixture of NLEVs and Tier 2 vehicles.

To illustrate the impacts of Tier 2 vehicles, MOVES emissions for light-duty gasoline vehicles in Cook County in summer for calendar years 2008, 2015 and 2020 were examined. Also examined were emissions for three model year groups, 2003 and earlier, 2004-2007, and 2008+. Finally, emissions for vehicles with default deterioration, and for all vehicles with no deterioration were examined. For the no deterioration case, it was assumed that emissions in ages above the 0-3 years are the same as for the 0-3 year age group. The results are shown in Figures 10-64 (Evaporative emissions), 10-65 (Exhaust VOC emissions), 10-66 (CO) and 10-67 (NOx). The model 2003- values for the 2020 without bar (right hand bar) are 0.5, 8.0, and 1.0 tons/day for Figures 10-65, 10-66, and 10-67, respectively.

Figure 10-64. Comparison of Model Year Group Contributions, Evaporative VOC

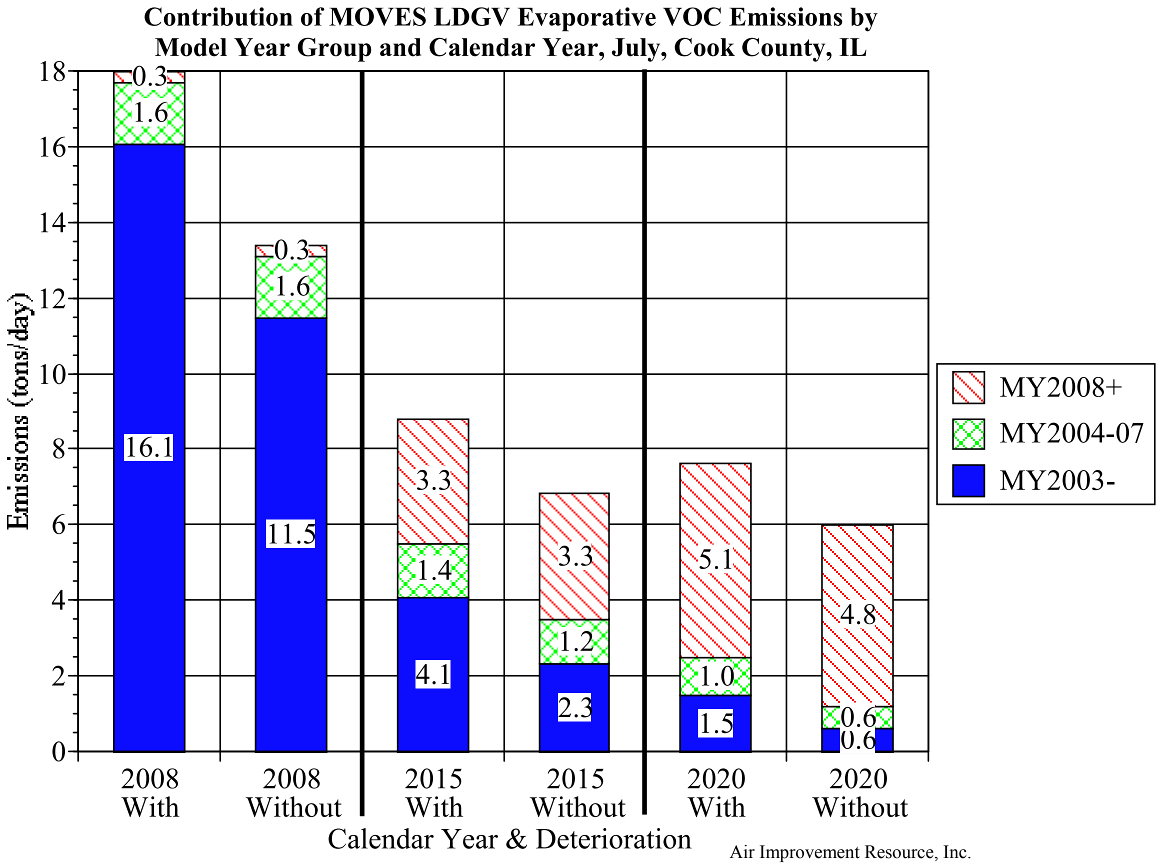


Figure 10-64 shows that the 2008+ model year Tier 2 group reduces emissions significantly between 2008 and 2020, as it becomes a greater contributor to overall LDGV emissions. Deterioration in Tier 2 evaporative systems accounts for only 0.3 tpd of emissions in 2020. The emissions without deterioration in 2020 are 66% below the 2008 emissions with deterioration, and the emissions with deterioration in 2020 are 59% lower for the with deterioration case than in 2008.

Figure 10-65 shows that exhaust VOC in 2020 with deterioration is 64% lower in 2020 than in 2008. The exhaust VOC in 2020 without deterioration is 85% lower than emissions in 2020 with deterioration. Emissions due to deterioration have dropped considerably between 2008 and 2020. In 2008, the deterioration effect is about 14 tons per day. By 2020, the deterioration effect is estimated at 4.5 tpd.

Figure 10-65. Comparison of Model Year Group Contributions, Exhaust VOC

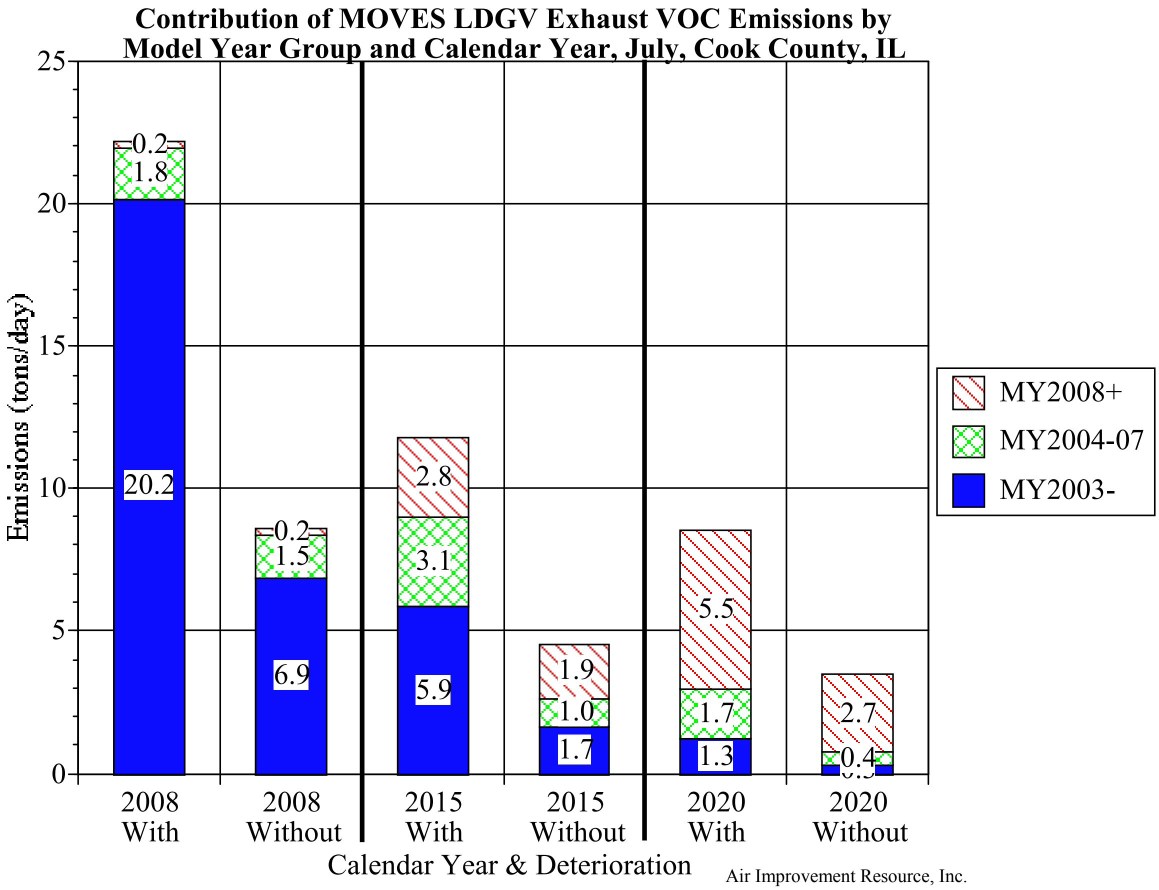


Figure 10-66 shows CO emissions. CO drops from about 450 tpd to 250 tpd in 2020 (with deterioration). EPA predicts a significant amount of deterioration in CO for Tier 2 vehicles. Without deterioration, Tier 2 emissions are only 66 tpd, but with deterioration, Tier 2 emissions are estimated at 173 tpd, almost 3 times higher.

Figure 10-67 shows NOx emissions, which are reduced from 55 tpd in 2008 to about 16 tpd in 2020. EPA predicts that deterioration will double Tier 2 emissions, from about 5.3 tpd in 2020 to 10.2 tpd.

Figure 10-66. Comparison of Model Year Groups Contributions, CO

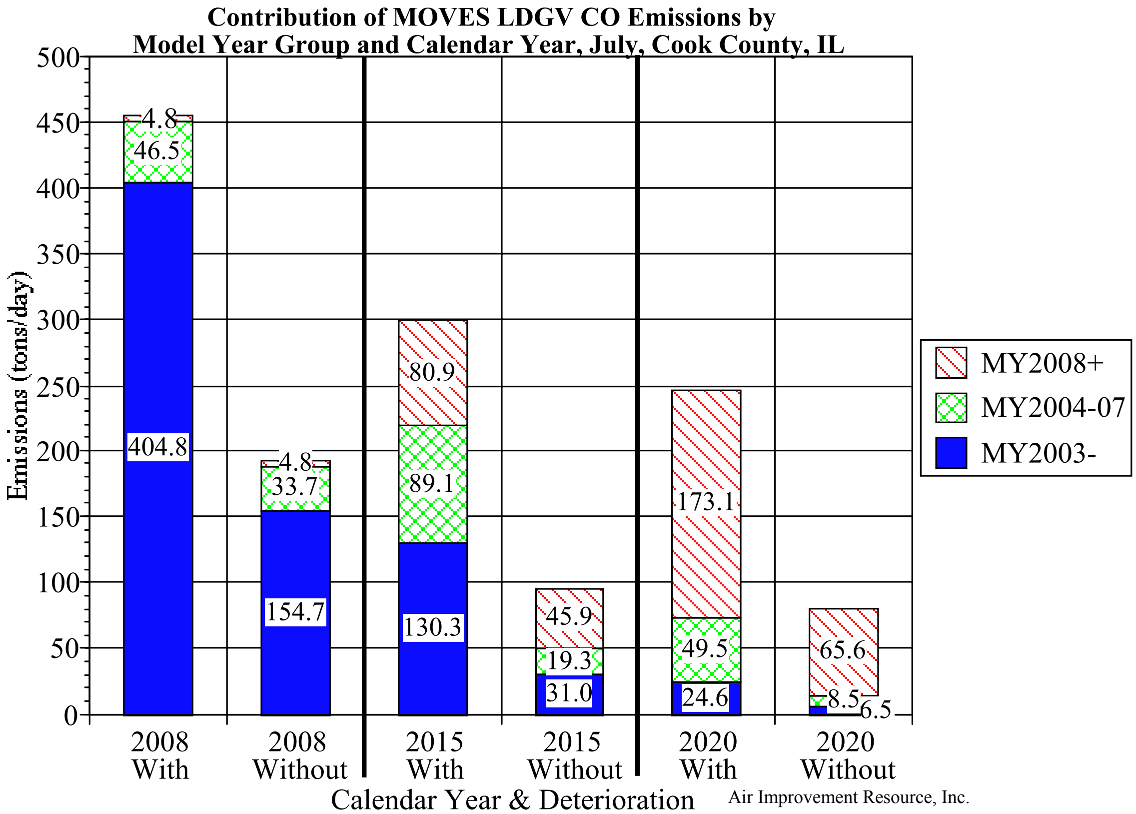
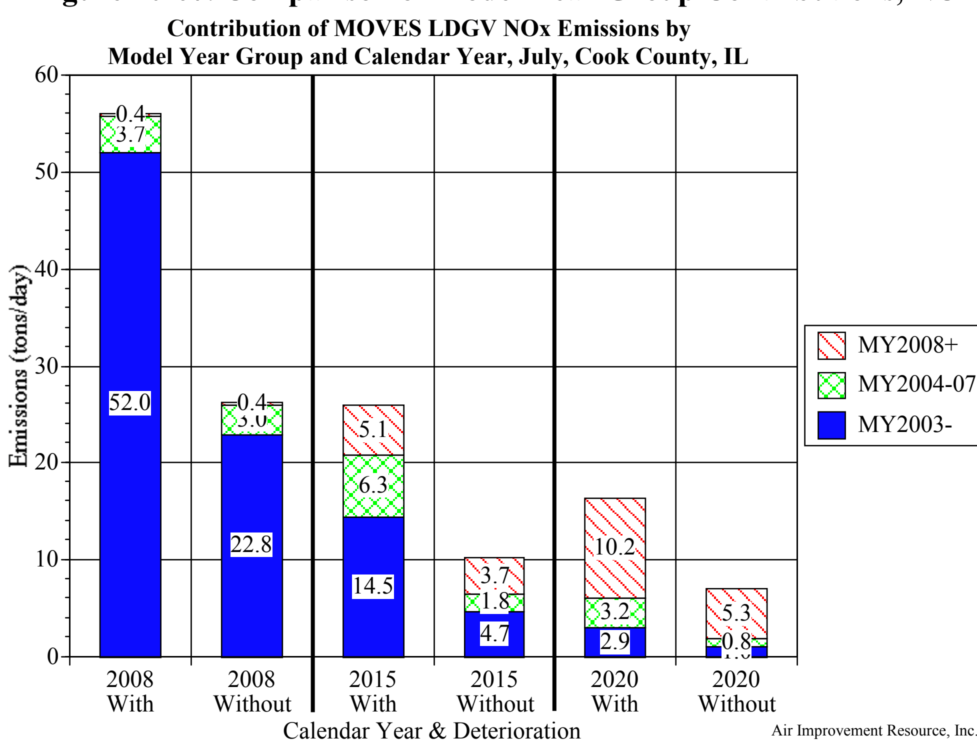


Figure 10-67. Comparison of Model Year Group Contributions, NOx



Overall the Tier 2 vehicle standards significantly reduce emissions from the light-duty fleet. MOVES predicts that deterioration will double exhaust VOC and NOx emissions, and almost triple CO emissions of these vehicles. It is doubtful that this would occur, with or without an I/M program, because the average age of these Tier 2 vehicles is only about 6 years in 2020. Emissions would be more accurately predicted by modifying the deterioration rates by using a linear deterioration function rather than log –linear and the off-cycle emissions projections by re-evaluating emissions in the high-vsp bins.

A second way to view the different model years' contributions to emissions in 2020 is to evaluate the emissions of every model year in 2020. This is shown in Figures 10-68 through 10-72. These figures show the tremendous reduction in emissions from the 1990 model year through the 2020 model year. The emission standards of all vehicles after 2010 are the same in this plot, so any increase in emissions between model year 2020 and 2010 is due to deterioration. Vehicles prior to 2010 are a mixture of Tier 2 vehicles and earlier vehicles. Clearly, when the fleet turns over to all Tier 2 vehicles, emissions from light-duty vehicles will be much lower.

Figure 10-68. Emissions by Model Year, Evaporative VOC

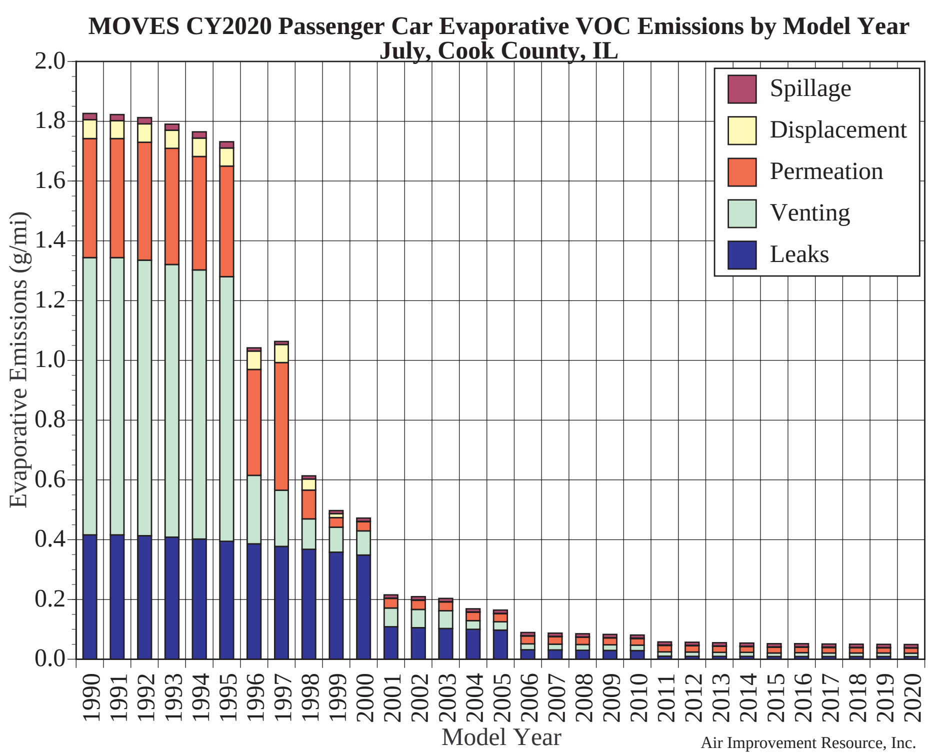


Figure 10-69. **Emissions by Model Year, Exhaust VOC**

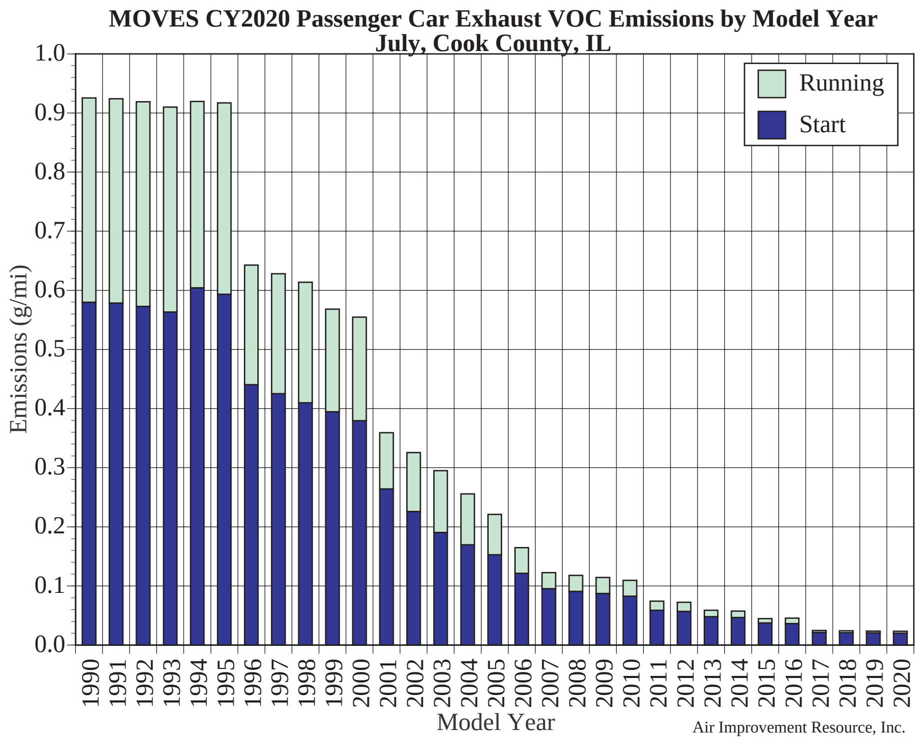


Figure 10-70. Emissions by Model Year, CO

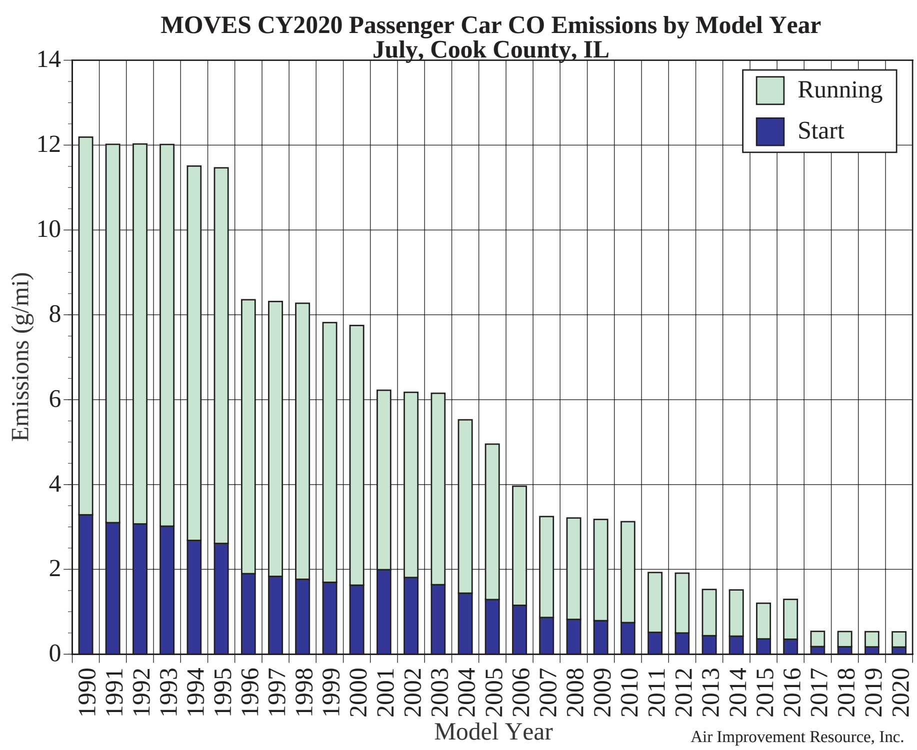


Figure 10-71. Emissions by Model Year, NOx

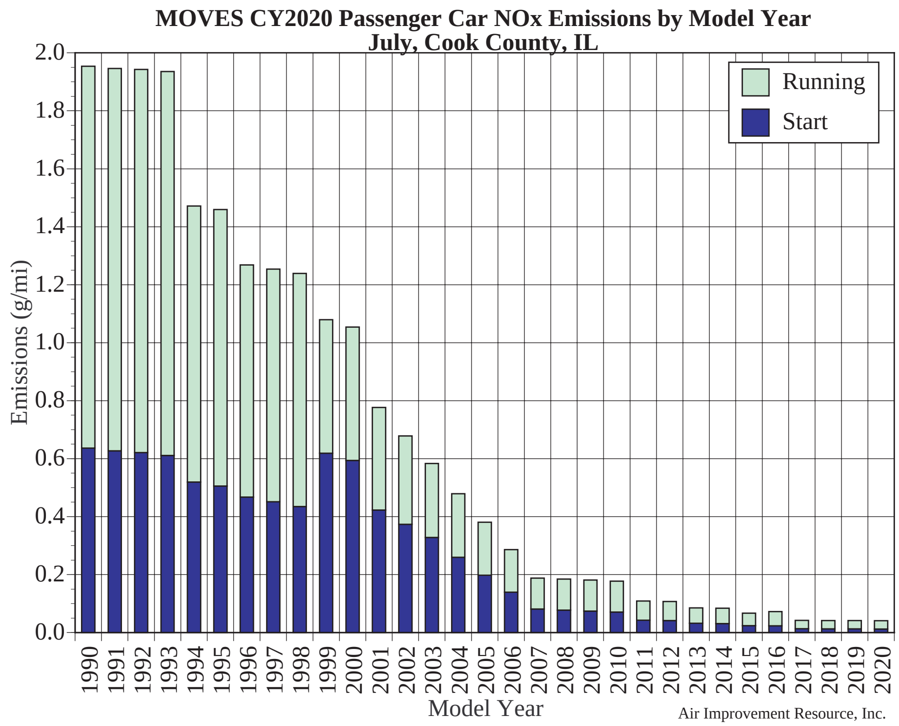
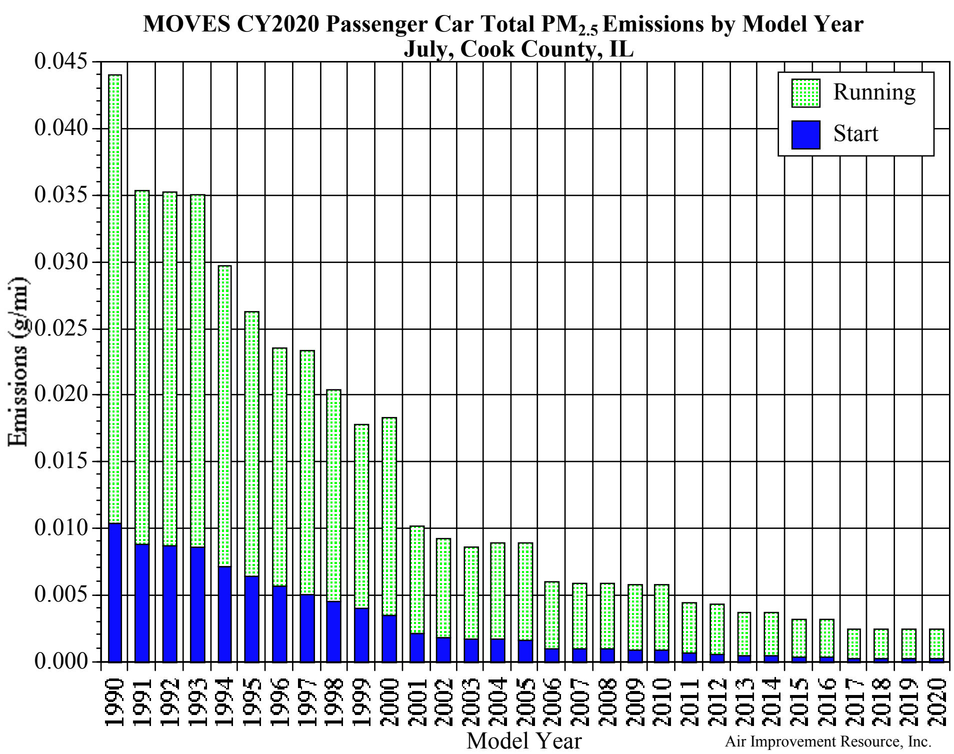


Figure 10-72. Emissions by Model Year, $PM_{2.5}$



10.8 Fuel Parameter Sensitivity Runs

This section contains plots of VOC, CO, NOx and PM emissions with varying fuel properties. To create these plots, MOVES is run with different fuel properties under a single set of conditions. The location is Chicago, during the summer, for the 2008 calendar year. The plots are for light-duty gasoline vehicles (passenger cars). In some cases, the investigators also examined just the 2008 model year in calendar year 2008.

The version of MOVES that was evaluated is the August 2009 version that did not have fuel binning, and used a fuel correction model as described in FACA documents and EPA's draft report.[59] [60] Comments on the technical basis for the fuel model are described in Section 8.

The base values are 2008 RFG fuel assigned to Cook County, IL by MOVES, and the tested values are chosen to be reasonable extremes for the individual parameters. Intermediate values are chosen to be the "Average Value" defined by EPA for the various bins in the original MOVES program. The bin averages are chosen so that the functional relationships would be the same whether the binned or non-binned version of MOVES is used. The fuel property values tested are summarized in Table 10-1.

Table 10-1. Fuel Property Values Tested

Fuel Property	Base Value	Tested Values
Sulfur	30	5, 15, 30, 50, 90 ppm
Ethanol	0	0, 5, 8, 10 vol.%
RVP	6.9	6.9, 7.5, 8.7, 9.2, 10 psi
Aromatics	26.1	17.5, 26.1, 32 vol.%
Olefins	5.6	5.6, 9.2, 11.9 vol.%
E200	41.1	41, 50 vol.%
E300	83.1	78.6, 83.0, 89.1 vol.%

The accompanying charts show the components of exhaust: VOC, CO, NOx, PM10 and $PM_{2.5}$. For exhaust emissions, the program predicts start and running emissions, and the total, which is the sum of the two. For evaporative emissions, MOVES predicts the following components: refueling spillage, refueling displacement, fuel leaks, permeation, and vapor venting. In some cases, the responses are different in direction or degree from the EPA Complex Model, and these could be the subject of further investigation. There are two general conclusions from these plots. One is that PM emissions are affected by sulfur only, and evaporative emissions are affected by RVP and ethanol content only.

In the accompanying figures, PM and evaporative emissions will not be shown for fuel properties that do no impact them.

[59] Ed Glover and Megan Beardsley, "MOVES 2009 Fuel Effects Update", MOVES Review Workgroup, 9/14/09.

[60] "Development of Gasoline Fuel Effects in the Motor Vehicles Emissions Simulator (MOVES2009), Draft Report", EPA-420-P-09-004, August 2009.

Sulfur – Sulfur responses are shown in Figures 10-73 through 10-77 for the entire light-duty gasoline fleet in calendar year 2008, and in Figures 10-78 through 10-82 for 2008 model year only. There is non-linearity evident in exhaust emissions of VOC, CO and NOx, consistent with the various exhaust emission models. PM emissions are affected by sulfur concentrations because of the contribution of sulfates to exhaust particulate.

Figure 10-73

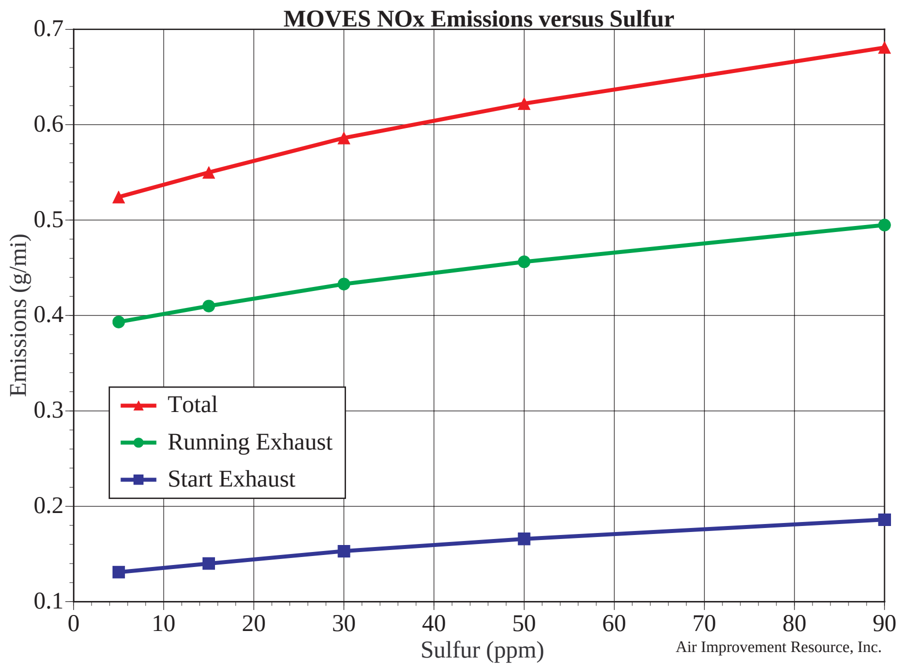


Figure 10-74

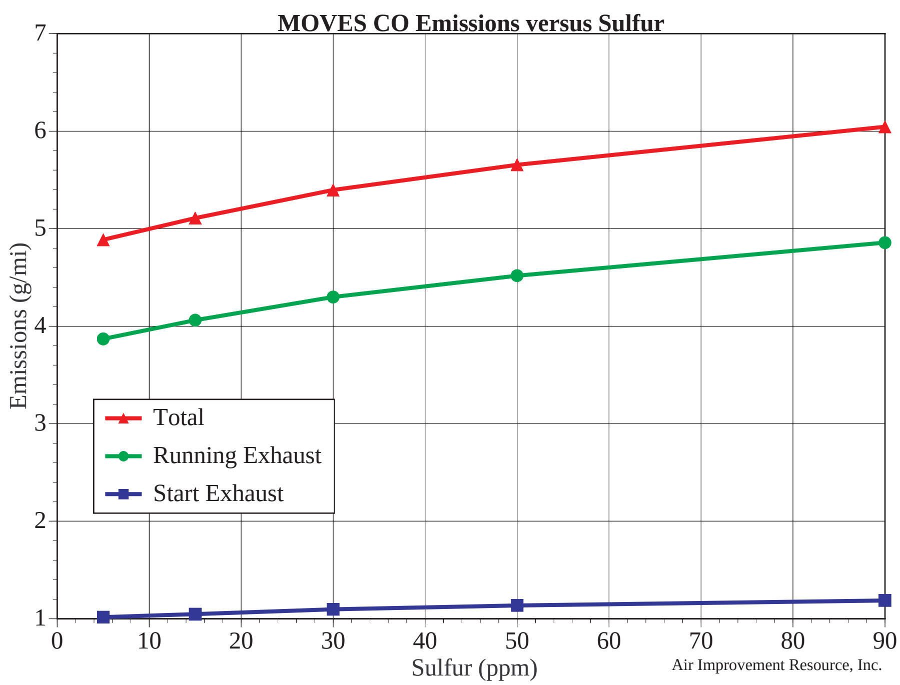


Figure 10-75

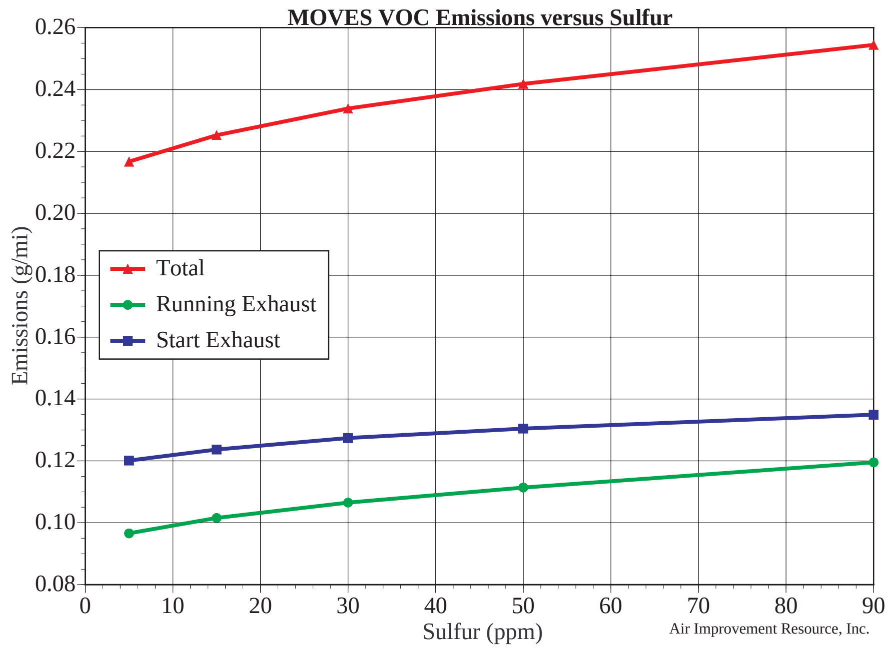


Figure 10-76

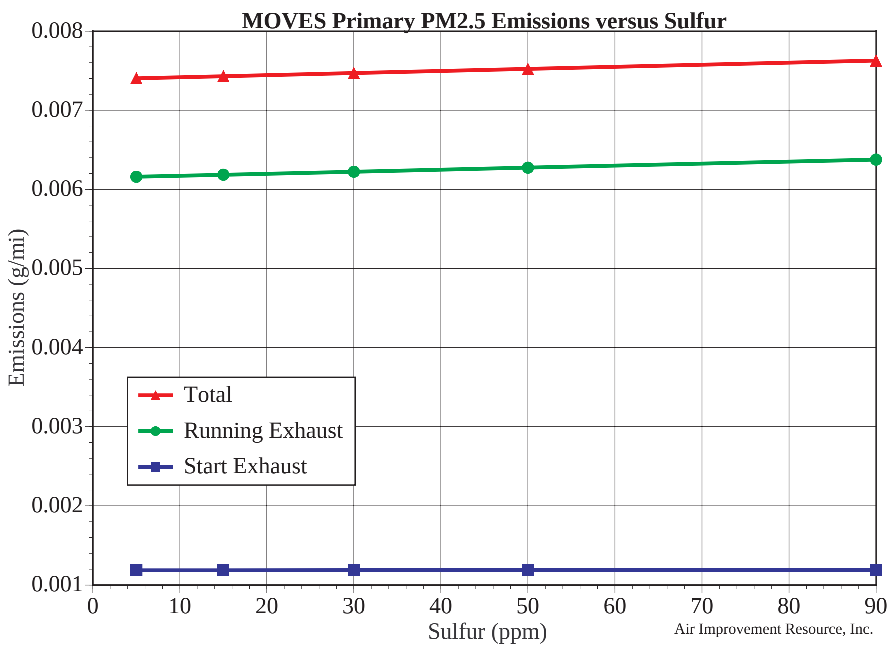
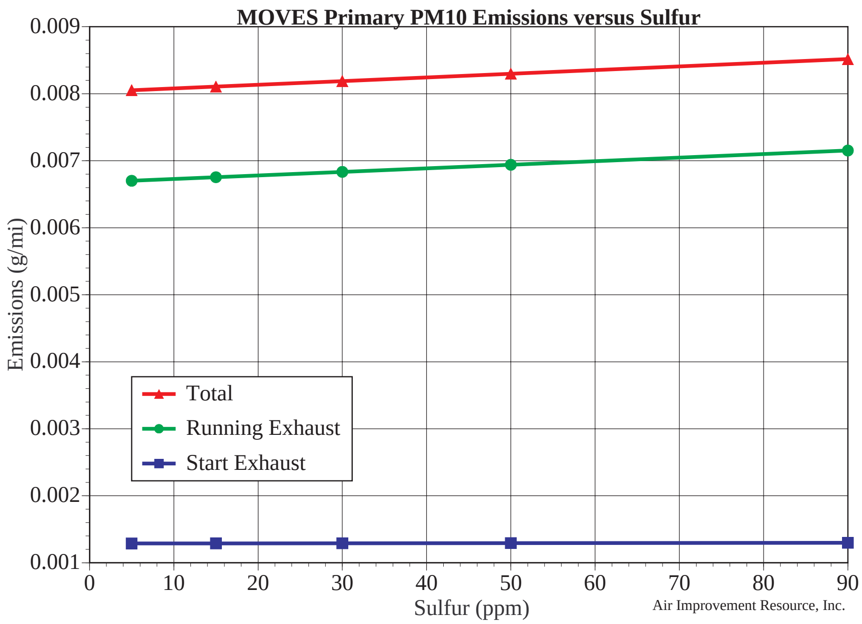


Figure 10-77



For the entire fleet of passenger cars, decreasing sulfur from 90 ppm to 5 ppm reduced total NOx emissions by 23%. On a percentage basis, start emissions are reduced by a larger amount than running emissions (30% versus 21%), but since start emissions are generally low for NOx, total exhaust emissions are weighted more towards running emissions.

CO emissions decreased 19% when sulfur is reduced from 90 ppm to 5 ppm. Running emissions are affected more than start emissions (20% versus 14%) and this is to be expected since most sulfur affects catalytic converter function, and CO start emissions occur when the catalyst is not fully warmed-up.

VOC exhaust emissions decreased 15% when sulfur decreased from 90 ppm to 5 ppm. For VOC, the impact on start emissions is higher than for running emissions (-19% versus -11%); the reason for this is not clear.

$PM_{2.5}$ and PM_{10} emissions are reduced by 3% and 5% respectively when sulfur is decreased from 90 ppm to 5 ppm.

All emissions exhibit non-linear responses with larger effects at the lower end of the sulfur range. This is consistent with emissions data and with the Complex Model.

For the 2008 model year fleet, decreasing sulfur from 90 ppm to 5 ppm decreased NOx emissions by 54%, VOC emissions by 31% and CO emissions by 38%. These results, especially the large reduction for NOx, show the major non-linearity for exhaust emissions in the latest technology. These results are consistent with the EPA Predictive Model, but may overstate the benefits of reducing sulfur at the lowest levels as discussed in Section 8. $PM_{2.5}$ and PM_{10} emissions are reduced 7% and 13% respectively for the

2008 MY fleet. See the discussion in Section 8 for more details of the non-linearity discussion.

Figure 10-78

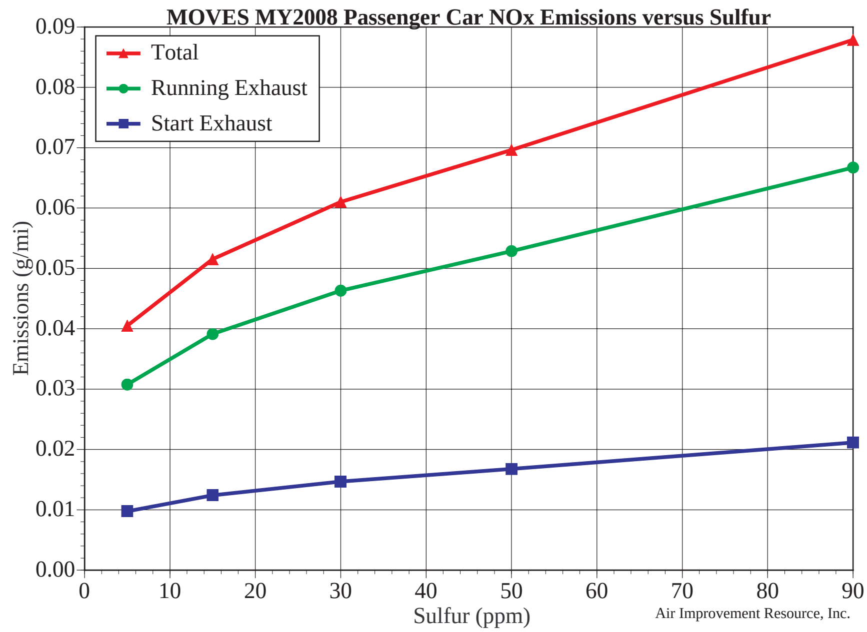


Figure 10-79

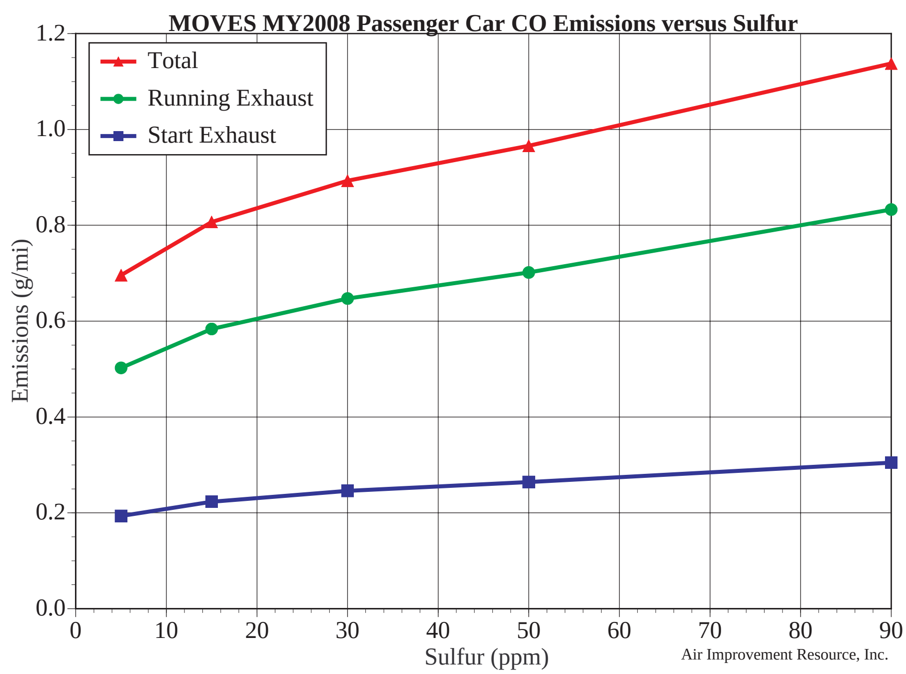


Figure 10-80

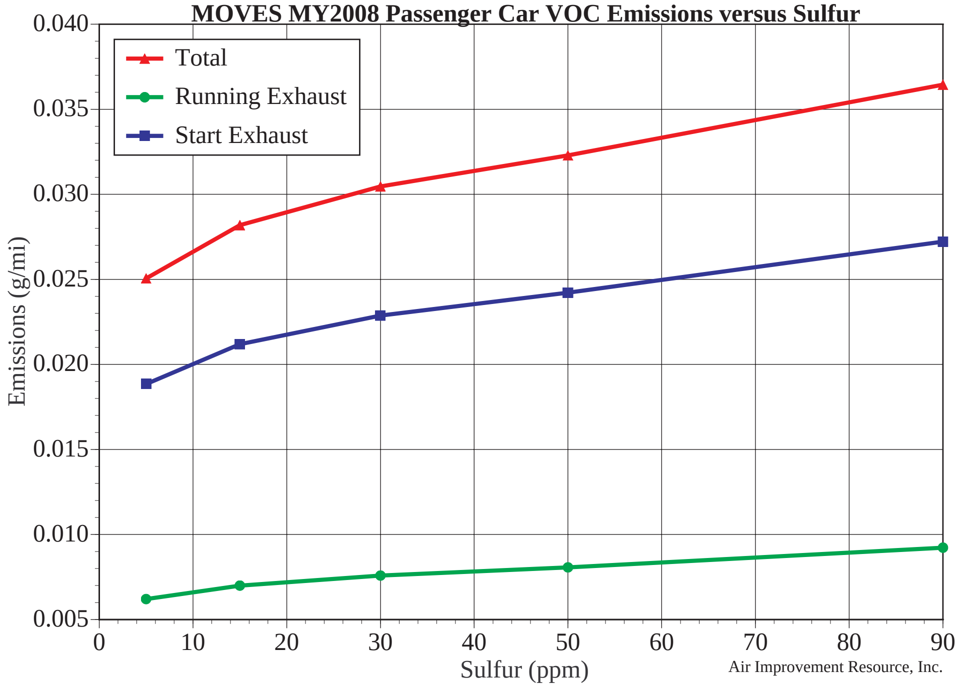


Figure 10-81

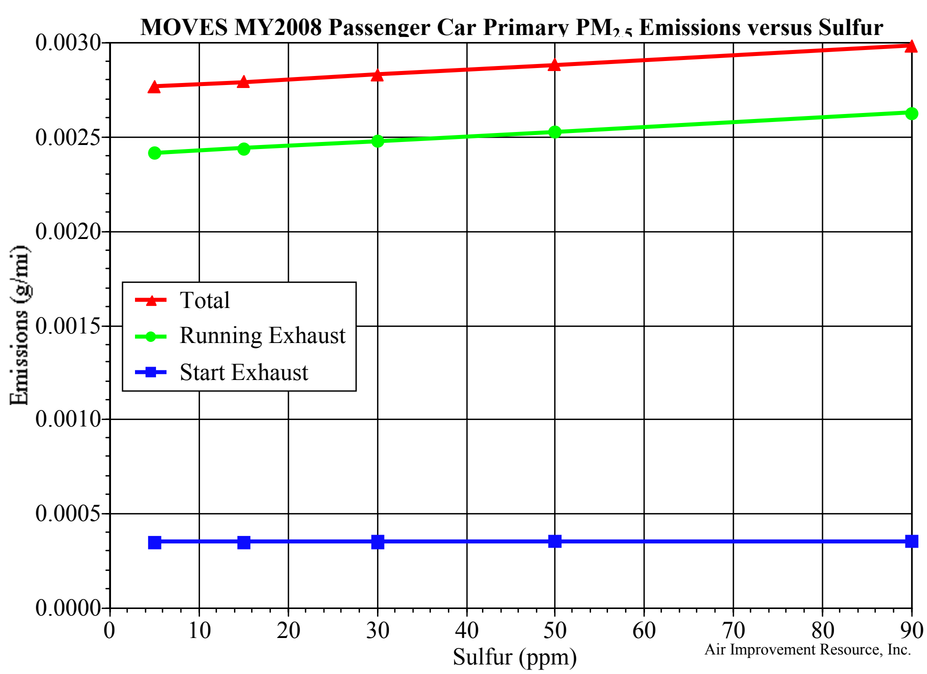
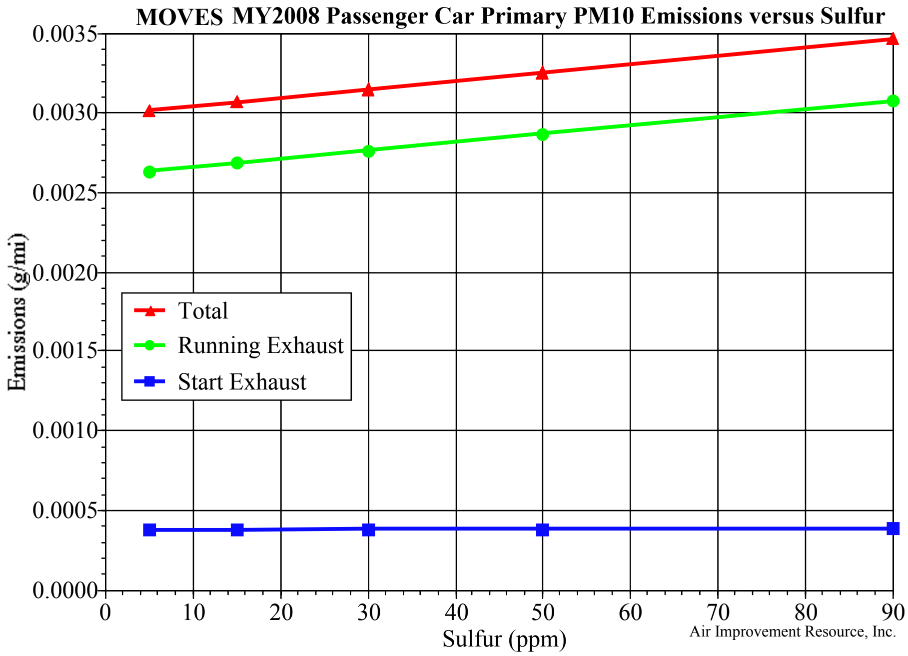


Figure 10-82



Ethanol – Exhaust NOx, VOC, and CO emissions are run for four levels of ethanol – 0%, 5%, 8%, and 10%, and are shown in Figures 10-83 through 10-85. VOC emissions could only be run for the extremes – 0% and 10%. EPA is fixing a flaw in the MOVES software that will allow the intermediate values to be run. It is expected that there will be significant non-linearity for ethanol in the VOC emissions. When ethanol increases from 0% to 10%, NOx emissions increase by 7.8% and CO emissions decrease by 11.7%. Both responses are linear. These results are consistent with available emissions data. In the Complex Model, NOx emission decrease by 0.3% over this range, and EPA has used a different relationship as discussed in their documentation of the RFS rule. For VOC, permeation emissions increase with increasing ethanol content, but start emissions and running VOC emissions also appear to increase (exhaust emissions increase by 6.3% from 0% ethanol to 10% ethanol). This is unexpected and inconsistent with the direction of CO emissions, and is in the opposite direction of the California Predictive Model for increasing ethanol content. The reasons for this should be investigated after the MOVES fuel module has been revised with respect to exhaust VOC emissions and ethanol.

Figure 10-83

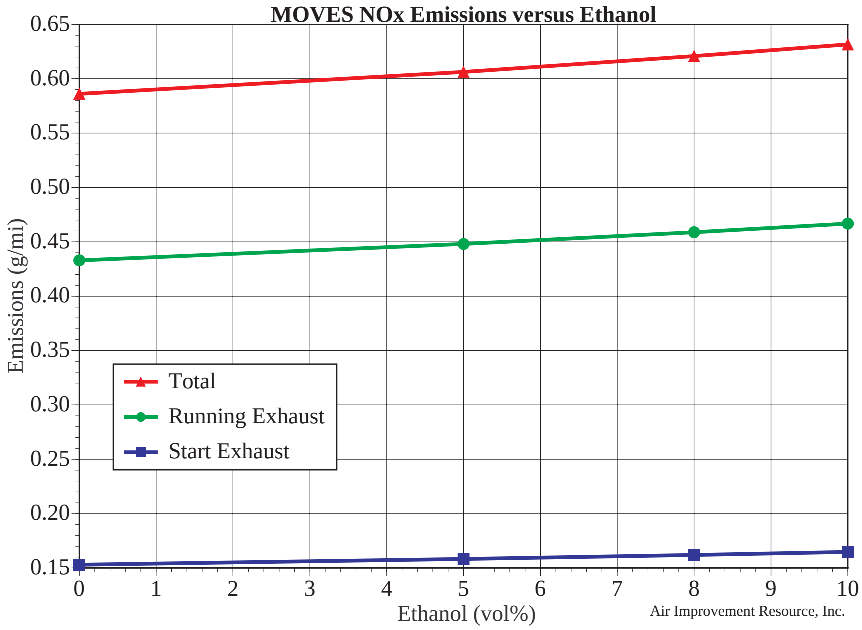


Figure 10-84

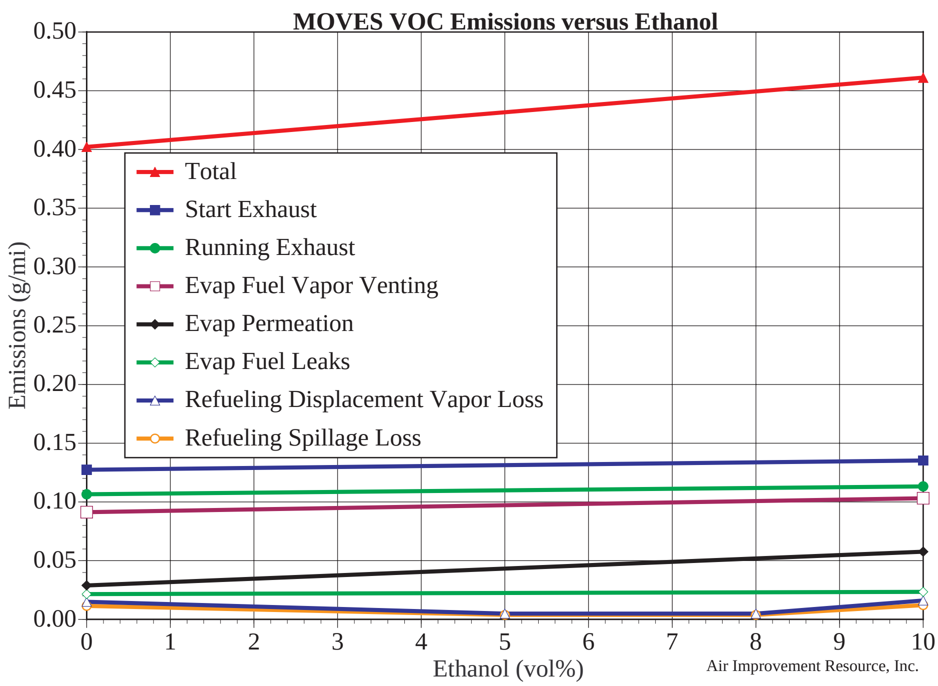
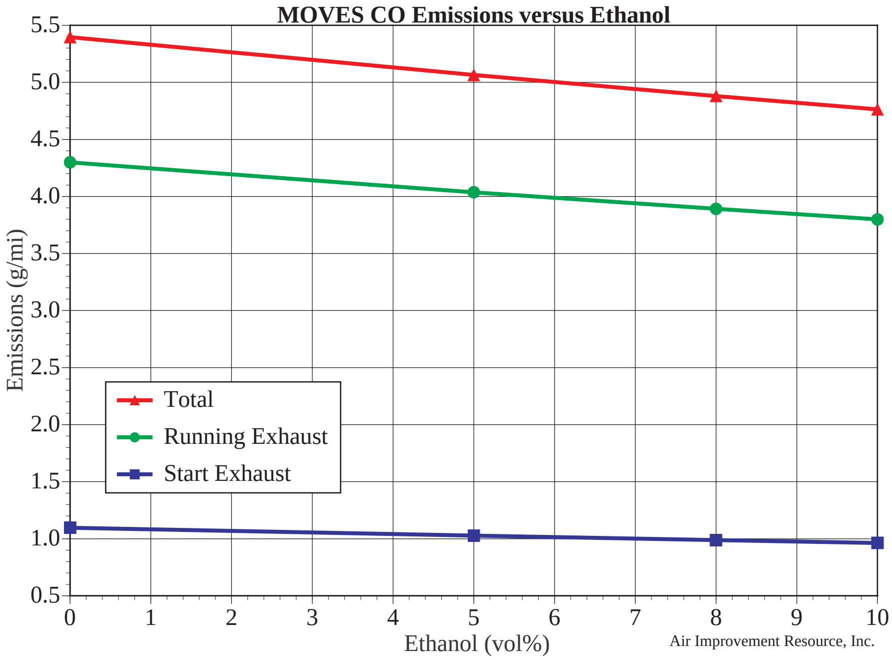


Figure 10-85



<u>RVP</u> – The major effect of RVP is on evaporative VOC emissions as shown in Figure 10-86. Increasing RVP from 6.9 psi to 10.0 psi increased total evaporative emissions by 16.5%. As expected, refueling spillage, fuel leaks and permeation are not affected by RVP. Over this same range, refueling displacement vapor loss increased 52% and vapor venting increased 23%. The effect of RVP on exhaust VOC (Figure 10-87) over the same range is relatively small, an increase of 3%. The response of CO (Figure 10-88) to changes in RVP is mildly non-linear with a minimum around 7.5 psi, which is evident in both start and running exhaust emissions. It is not clear why this is occurring or what technology group is driving the non-linearity. NOx emissions (Figure 10-89) increase as RVP increases, with a change of 3% between 6.9 psi and 10.0 psi.

Figure 10-86

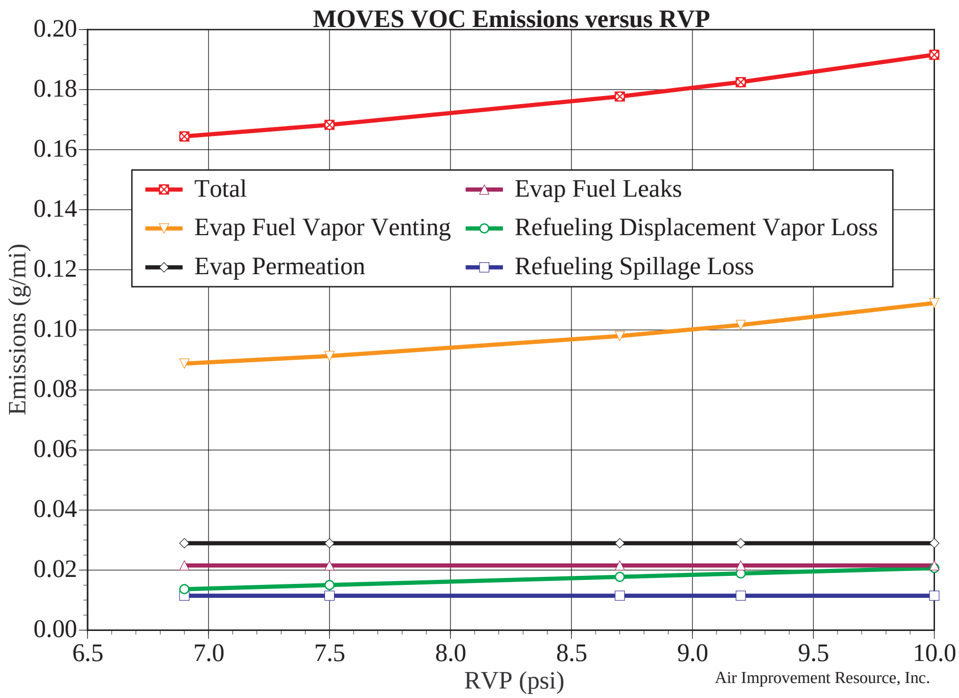


Figure 10-87

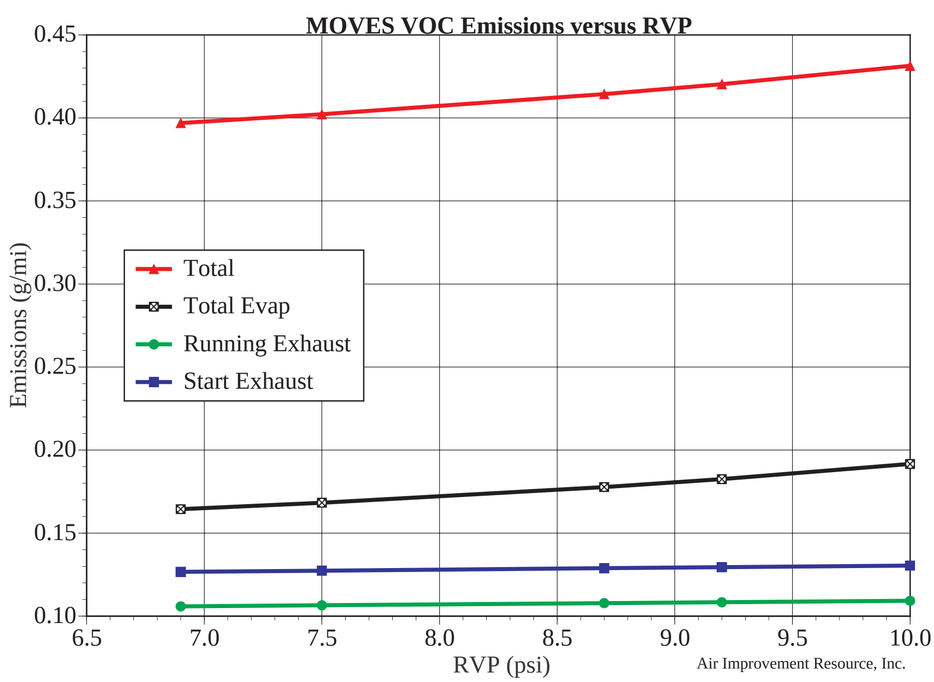


Figure 10-88

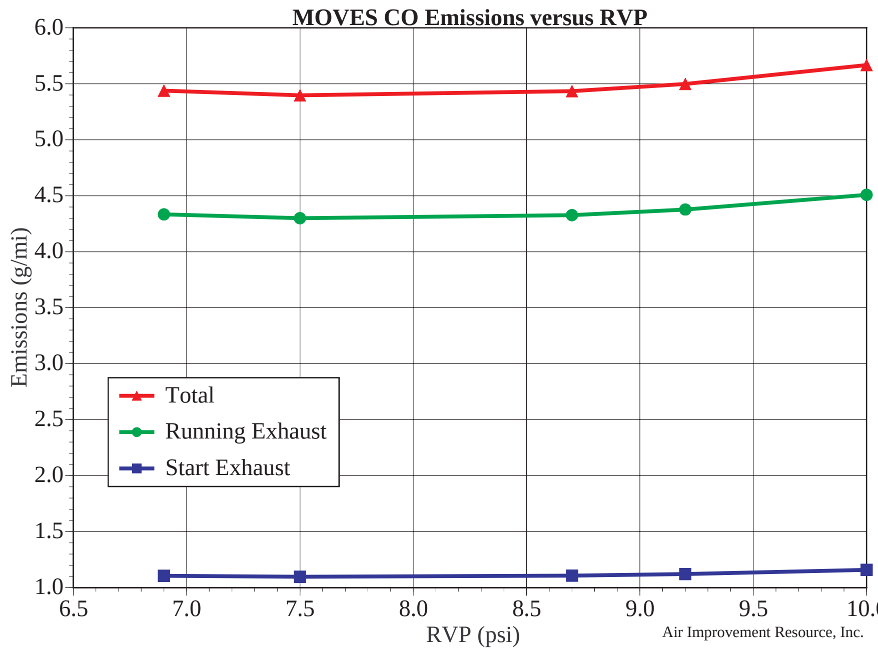
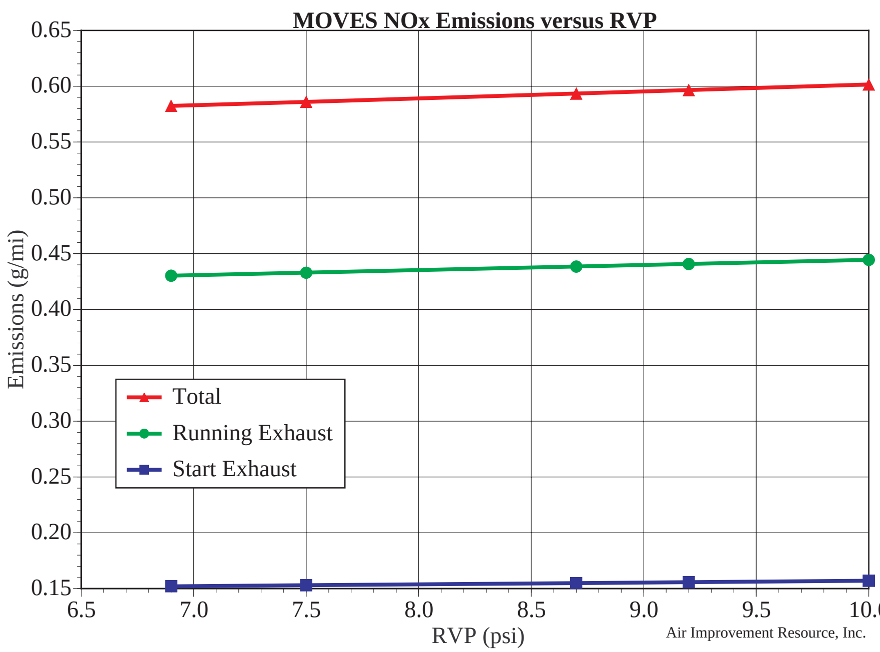


Figure 10-89



Olefins – Emissions effects of olefins are shown in Figures 10-90 through 10-92. Olefins have small effects on exhaust emissions. Increasing olefin concentrations from 5.6% to 11.9 % increases NOx emissions by 2% and decreases VOC emissions by 1.6%. CO emissions are essentially flat. These results are consistent with emissions data and with the Complex Model.

Figure 10-90

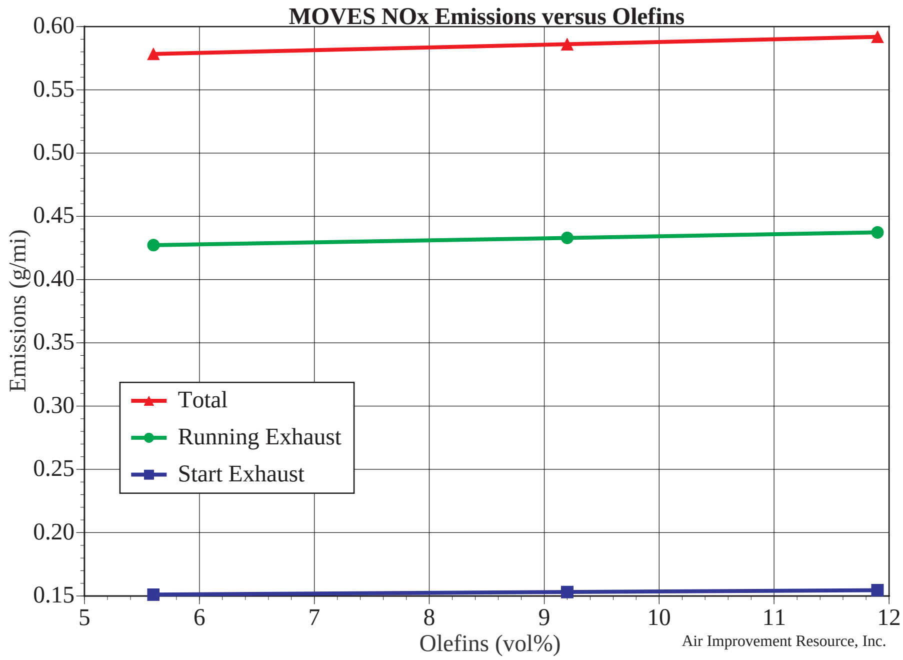


Figure 10-91

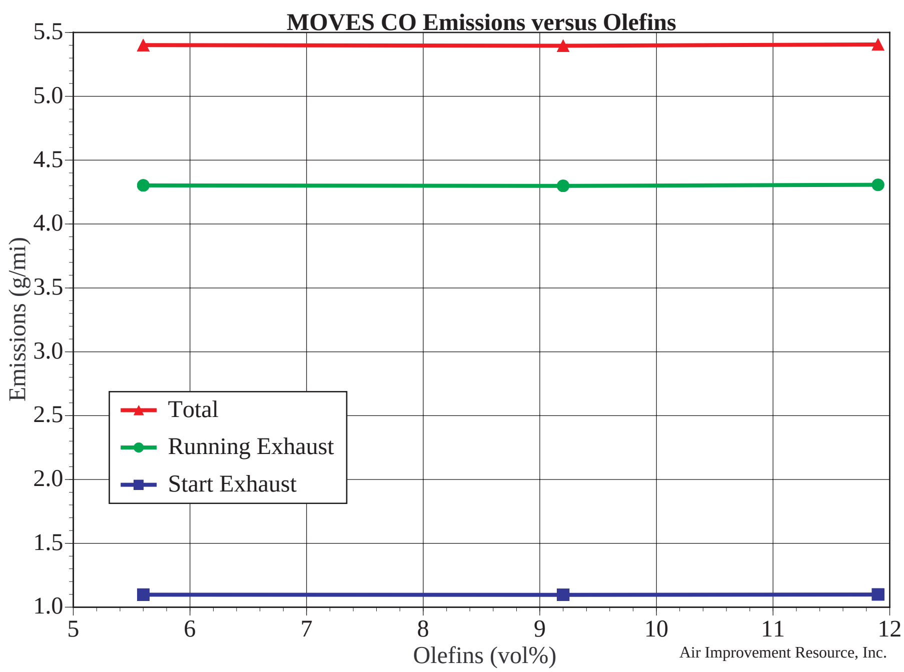
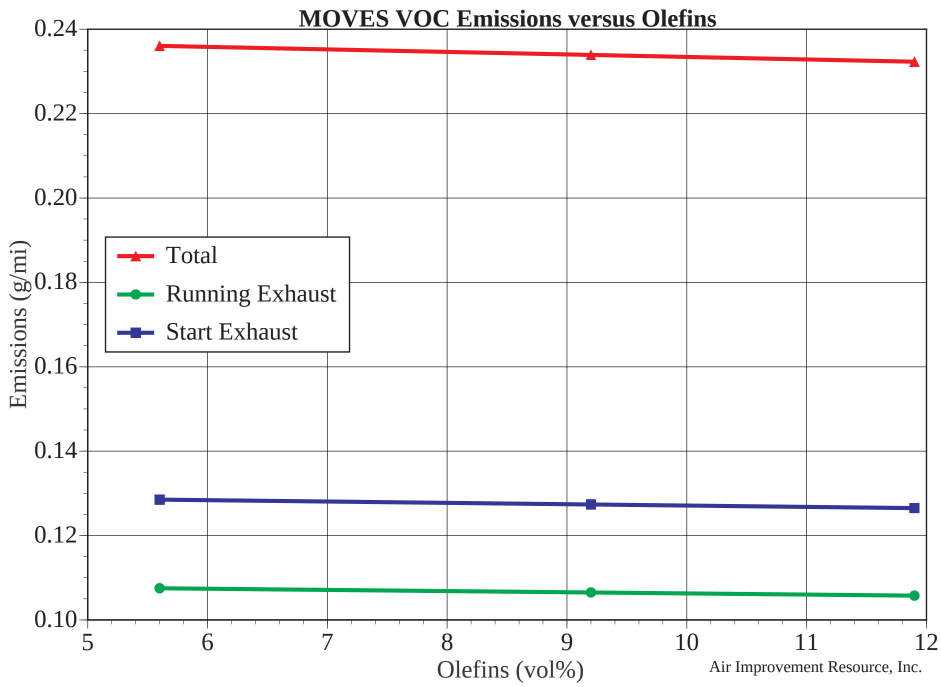


Figure 10-92



Aromatics – Effects of aromatics on emissions are shown in Figures 10-93 through 10-95. Increasing aromatic concentrations from 17.5% to 32% raised NOx and VOC emissions by 3% and raised exhaust CO emissions by 9%. The starting and running changes are about the same for all three constituents.

Figure 10-93

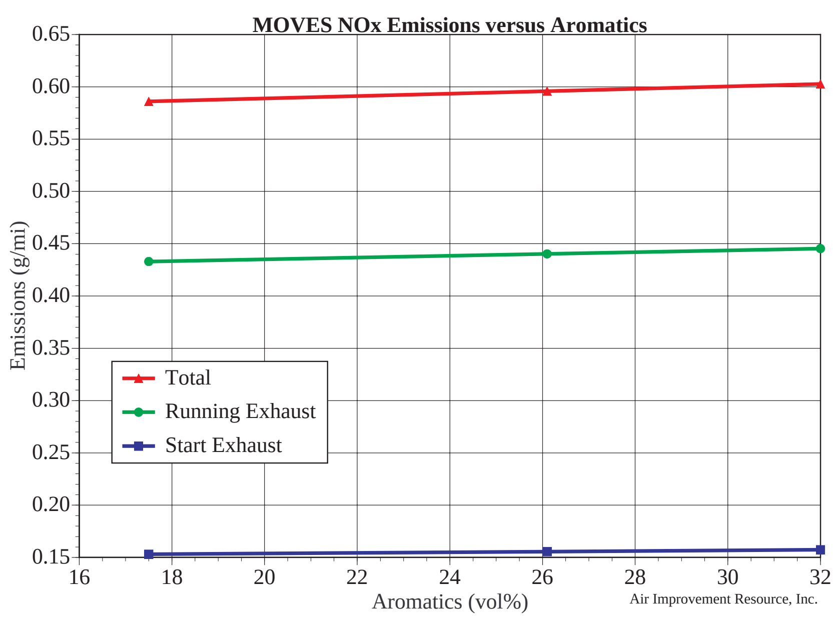


Figure 10-94

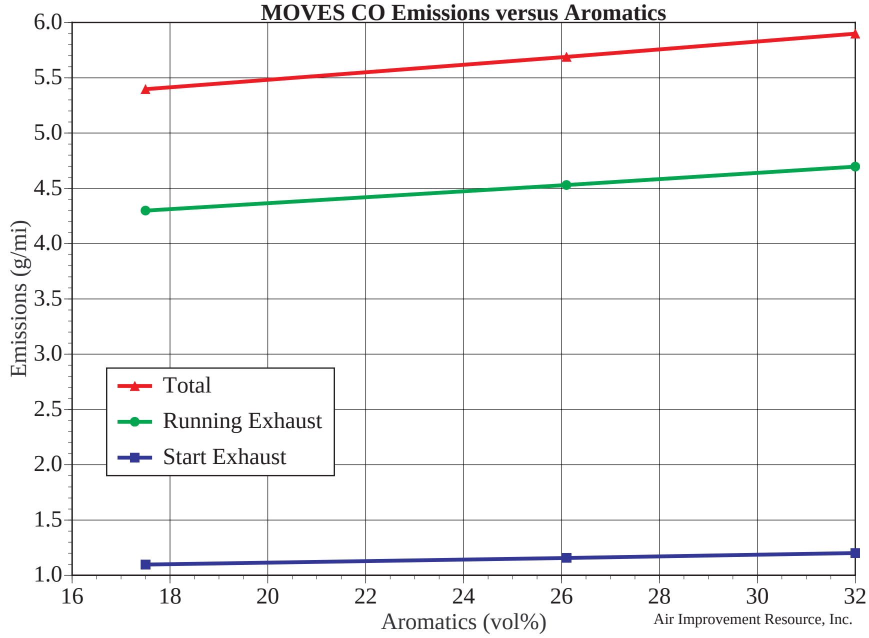
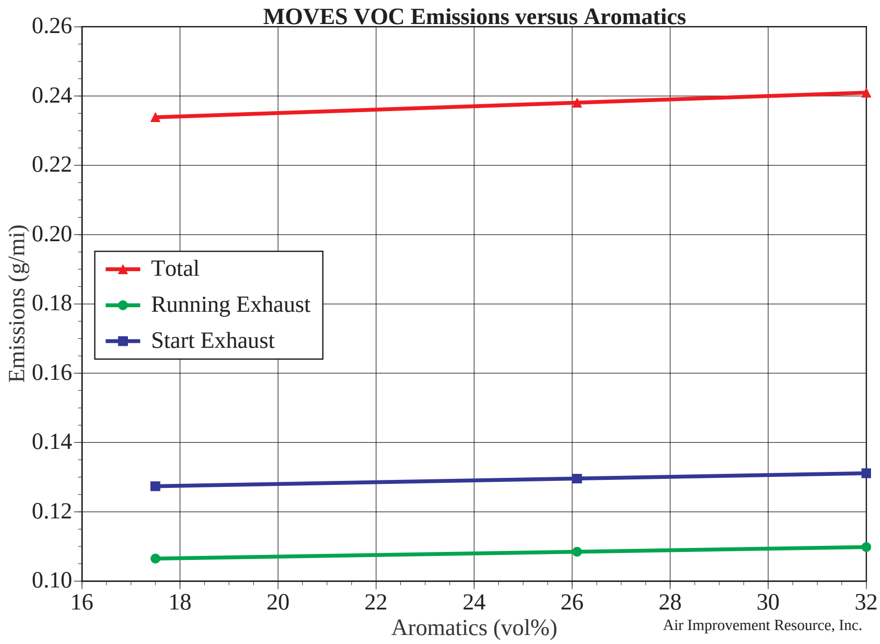


Figure 10-95



E200 – E200 effects on emissions are shown in Figures 10-96 through 10-98. Increasing E200 from 41% to 50% had little or no effect on NOx (-0.1%) and CO (-1.0%). The NOx results are consistent with the Complex Model, which predicted a NOx decrease of 0.5% over the same range. VOC exhaust emissions are non-linear over this range. Over the entire range, exhaust VOC emissions decrease by 5.8%, with the majority of this decrease (3.5%) occurring between 41% and 45.5%. The direction and non-linearity is consistent with emissions data and the Complex Model.

Figure 10-96

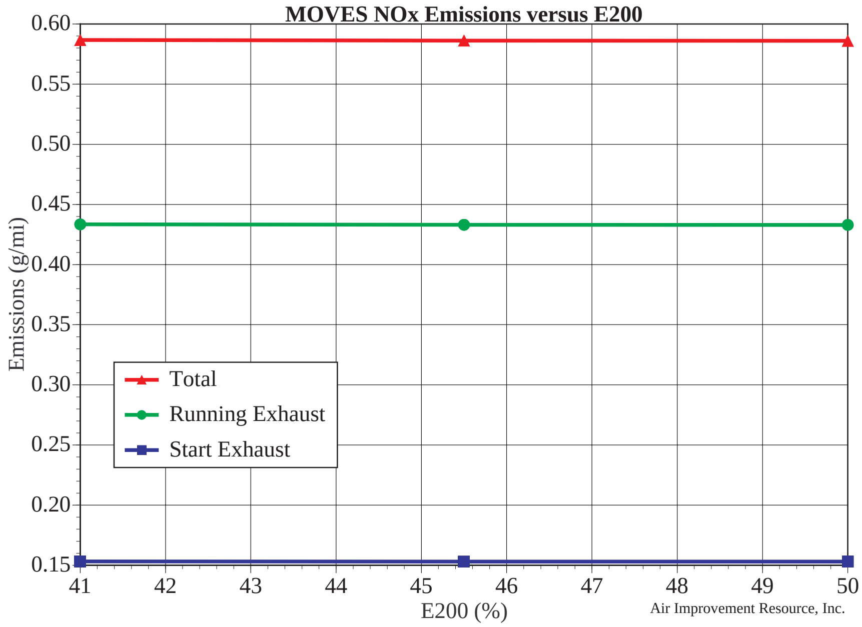


Figure 10-97

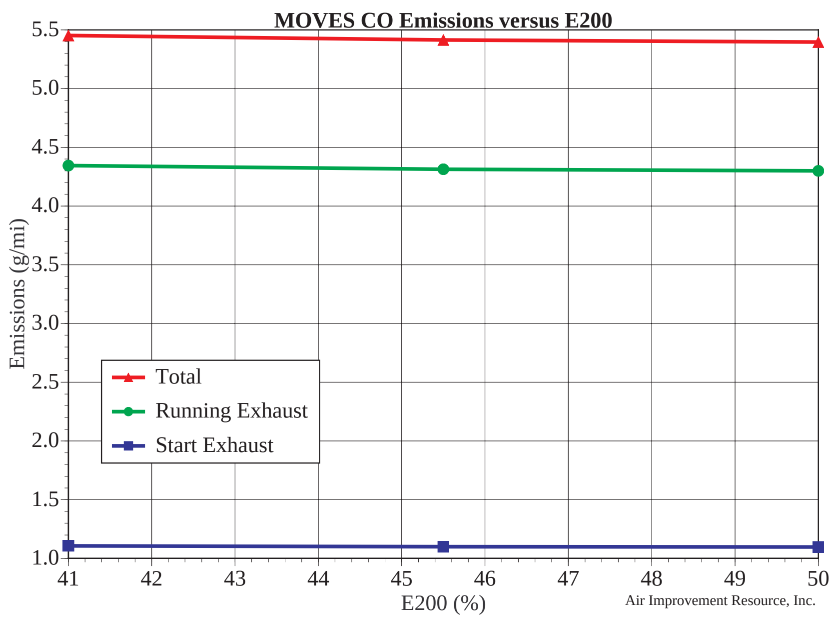
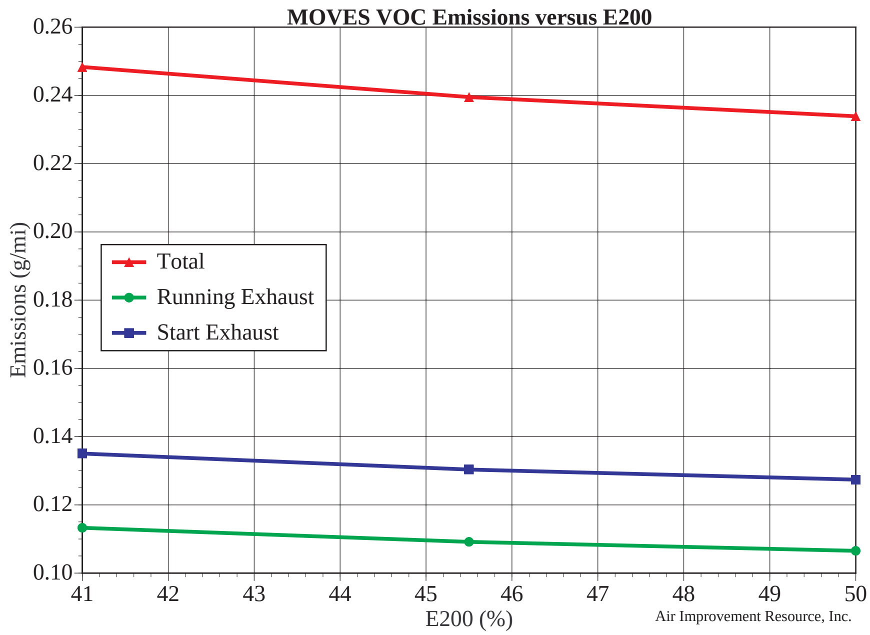


Figure 10-98



E300 – E300 effects on emissions are shown in Figures 10-99 through 10-101. The response of VOC and CO exhaust emissions to E300 is non-linear. NOx emissions appear to be linear, with NOx decreasing by 1.5% when E300 is increased from 78.6% to 89.1%. When E300 is increased from 78.6% to 83%. VOC exhaust emissions decrease 3.1% and CO emissions decrease 1.6%. When E300 increases from 83% to 89.1%, VOC exhaust emissions increase 0.1% and CO emissions increase 1.1%. The decrease at the lower range is consistent with previous emissions data and models. The non-linearity is also consistent with data and models. However, the response should be flat at the higher end of E300 values. The increase shown by the MOVES model could be the result of using a quadratic form of regression equation and may not be supported by data. These results are directionally consistent with emissions data and the Complex Model.

Figure 10-99

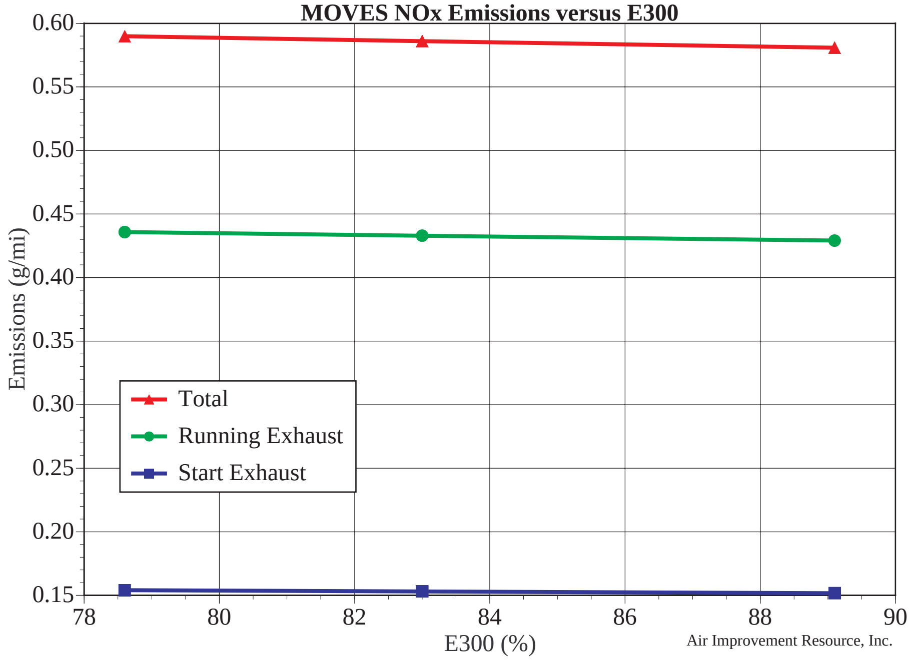


Figure 10-100

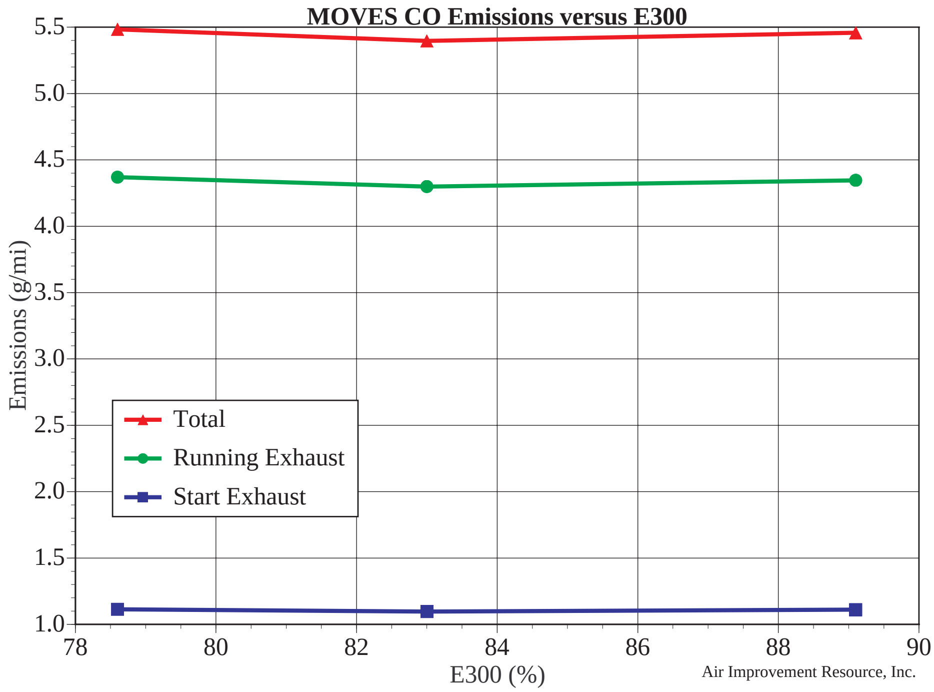


Figure 10-101

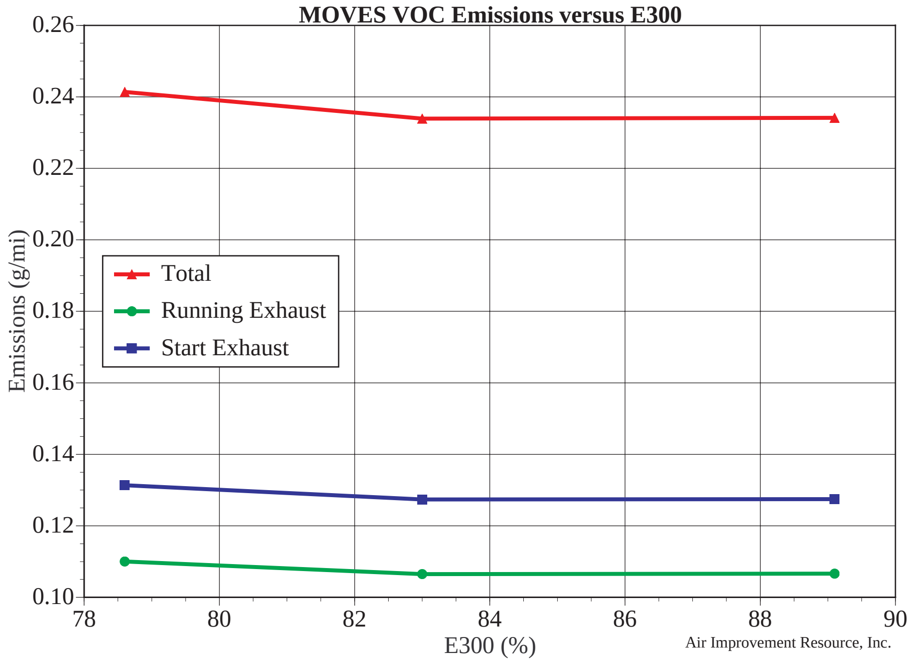


Table 10-2 summarizes the results shown above and shows the results as percent change in emissions when each fuel property in varied between the low and high values.

Table 10-2. Percent Changes in Emissions with Changes in Fuel Properties

		Emissions Change, %					
Parameter	Property Change	Exh. VOC	CO	NOx	Evap. VOC	PM10	$PM_{2.5}$
Sulfur (Fleet)	90→5	-14.8%	-19.2%	-23.0%	No change	-5.5%	-2.9%
Sulfur (2008 MY)	90→5	-31.3%	-38.8%	-53.9%	No change	-13.0%	-7.2%
Ethanol	0→10	+6.3%	-11.7%	+7.8%	+26.3%	No change	
RVP	6.9→10	+3.1%	+4.2%	+3.3%	+16.5%	No change	
Olefins	5.6→11.9	-1.6%	-0.1%	+2.3%	No change		
Aromatics	17.5→32	+3.0%	+9.3%	+2.8%	No change		
E200	41→45.5	-3.5%	-0.7%	-0.1%	No change		
	45.5→50	-2.3%	-0.3%	0.0%	No change		
E300	78.6→83	-3.1%	-1.6%	-0.6%	No change		
	83→89.1	+0.1%	+1.1%	-0.9%	No change		

10.9 Summary of Observations

The following observations arose from evaluating these sensitivity runs:

1. EPA is assuming that some vehicles are liquid leakers in the 0-3 age-year group. This does produce significant evaporative emissions.

2. VOC emissions are mostly start emissions, but CO emissions are mixture of start and running. The reasons for this need to be more clearly understood.

3. NOx exhaust emissions, and to a lesser extent VOC and CO emissions, are significantly higher for light-duty trucks than for Tier 2 passenger cars, although their emission standards are extremely similar. These higher emissions could be due to higher emissions by VSP bin for LDTs than for cars with similar standards as discussed in Section 5.

4. Permeation evaporative HC emissions increase on a g/mi basis, even though the permeation emissions on a vehicle basis are not estimated to increase. The increase in g/mi emissions could be due to a reduction in vehicle miles traveled with vehicle age, but this has not yet been verified.

5. Winter PM emissions for Tier 1 vehicles in Chicago appear to reach an average level that is 10 times higher at high mileages than observed at somewhat lower mileages in the Kansas City at similar temperatures. This is probably due to EPA's assumption of log-linear deterioration in PM with age, but should be investigated further.

6. The NOx deterioration rates for 2008 and later vehicles (Tier 2) passenger cars appear to be excessive, and the reasons for this should be investigated further.

7. Fuel sulfur impacts at low sulfur levels cause large reductions in emissions that are probably not realistic based on more recent test data at lower sulfur levels.

8. The MOVES fuel model causes exhaust VOC to increase at higher ethanol levels (up to E10). This is contrary to the direction for CO (which is reduced), and contrary also to the impacts as predicted by the recent California Predictive Model.

11.0 Summary of Recommendations

This section summarizes all of the recommendations from the various sections of this report. At the outset, it is recognized that building the MOVES model was an ambitious undertaking, and the model has important functions that MOBILE never did, for example, the ability to input different cycles, and estimate emissions at widely varying scales (micro, meso, and macro). While the model is relatively slow compared to MOBILE, it is much more flexible for customizing emission outputs. It is also more flexible in terms of inputting new emissions data sets as they become available. Most of the model seems to run as intended, i.e., the outputs seem to match the methods described by EPA.

Exhaust Start and Running Emissions

1. Start emissions are very significant, and the EPA methods rely on start emissions data developed by CARB on Tier 1 vehicles. Additional testing of LEV2 and Tier 2 vehicles should be conducted soon to check the ratio of start to cold start emissions at various soak times.

2. The start deterioration method borrows heavily from the methodology used in MOBILE6. The frequency and emissions characteristics of higher emitters is explicitly modeled in MOBILE6, but is only implicit in MOVES, and this could affect the outcome. EPA should determine the fraction of higher emitters from MOVES in the data underlying MOVES, adjust MOBILE6 to this fraction, and then determine the start deterioration factor from MOBILE6 to use for MOVES.

3. EPA uses a log-linear deterioration model for determining running emissions as a function of vehicle age and VSP bin. However, the evidence seems to indicate that for Tier 1 and later vehicles, emissions increase with age in a linear, rather than a log-linear fashion.

4. For newer vehicles, 70% of the emissions are found in the four highest VSP bins. This may be a indication of problems with the high VSP correction factors. At the very least, this points to where future data collection efforts should be targeted.

5. The I/M 147 data should not be used to estimate emission reductions from NLEV and Tier 2 vehicles in the higher VSP bins, because there is little operation in the higher VSP bins for this test.

6. The newer vehicle (i.e., Tier 2) method currently used for estimating emissions for the higher VSP bins does not take into account the full implementation of the SFTP rules. The emissions should be modified accordingly.

Evaporative Emissions

7. EPA liquid leaker frequencies estimate that relatively new (0-9 years) vehicles experience leaks. This is questionable.

8. Permeation emissions are not reduced for Tier 2 and MSAT evaporative emission standards. This should be evaluated as soon as possible with testing.

9. There do not appear to be evaporative emissions estimates for PZEVs, which are sold in both California and non-California states.

10. Leak frequencies with age should be updated with new test data as soon as possible.

Heavy-Duty Emissions

11. The impacts of heavy-duty reflash programs should be included.

12. The impacts of OBD regulations should be included as soon as possible.

13. Idle emissions from diesels should incorporate survey results of the fraction of time spent at lower idle speeds, state regulations controlling extended idles, and also the use of "hoteling" facilities.

14. EPA should investigate the amount of tampering and malmaintenance in the base emissions data being used for emission rates, because it could be double-counting many aspects of tampering and malmaintenance by also applying separate factors for these.

Correction Factors

15. The sulfur response below 30 ppm used in MOBILE6 should not be used for MOVES. A new response should be developed from the available data below 50 ppm.

16. MOVES estimates that exhaust VOC emissions increase with increasing ethanol content. This is contrary to the direction of CO emissions, and contrary to the latest information from the California Predictive Model.

17. The Tier 2 cold temperature response should be lower than for Tier 1 vehicles. In addition, the MSAT rules should reduce CO as well as HC.

18. The method used to develop HC temperature correction factors for the MSAT rule should be revised to include a compliance margin at 20° F to be consistent with the margin currently being utilized at 75° F.

19. Since MOVES does not provide a means for assessing the impact of certain metal- or non-metallic additives used in fuel, it may be necessary for EPA to provide guidance on how to model their impacts if their use is of concern to states or other entities.

PM From Gasoline Vehicles

20. The combined MSAT and Kansas City data on matched pairs tested in summer and winter does not support a cold temperature adjustment for running emissions.

21. EPA should check adjustments to low mileage emission rates for older vehicles to account for current fuels and measurement methods.

Sensitivity Runs

22. It is not yet clear why NOx emissions for light trucks are so much higher than passenger cars, when the emission standards are about the same in model year 2010.

23. Winter PM emissions for Tier 1 vehicles appear to reach levels at high mileages that have not been observed for these vehicles in the Kansas City data. This should be investigated further.

24. The NOx deterioration rates for 2008 and later vehicles (Tier 2) appear to be excessive, and the reasons for this should be investigated further.

Appendix A
Method Used to Compare NMIM and MOVES Emissions Based on NMIM VMT

The following method is used to estimate MOVES and MOBILE6 emissions for Atlanta, Chicago, and Salt Lake City.

1) Run NMIM for Fulton County, GA (Atlanta), Cook County, IL (Chicago), and Salt Lake County, UT (Salt Lake City) for the months of January and July, for 2008, 2015 and 2020. By selecting the “county mode”, NMIM will use the county-specific conditions and distributions for each location, month and year. Include all vehicle classes except Motorcycles (MOVES does not yet support that vehicle class.)
2) Obtain the NMIM MySQL pollutant and VMT output files for these runs.
3) Run MOVES for the same three counties, months and years, importing the county-specific data files (created by EPA) via the County Data Manager. These runs must also use the “SCC” aggregation mode so that the output files contain the same level of detail as those produced by NMIM.
4) Obtain the MOVES MySQL activity and emission output files for these runs.
5) Create a database with lookup Keys that cross-reference the 3-digit NMIM SCC codes to the 10-character SCC codes used by MOVES.
6) Create a database with lookup Keys that cross-reference the SCC codes into light-duty and heavy-duty weight groups. .
7) Create a database with lookup Keys that cross-reference the NMIM and MOVES pollutant codes to their pollutant names.
8) Using the SCC cross-reference information, create a lookup Key in the NMIM VMT file based on year, month and 10-digit MOVES SCC code
9) Create a lookup Key in the MOVES emission file based on year, month and 10-digit MOVE SCC code
10) Create a lookup Key in the MOVES activity file based on year, month and 10-digit MOVES SCC code
11) Using the Keys set up in Steps 8, 9 and 10, merge the MOVES activity and NMIM VMT into the MOVES emission records. This merge should work for all records except the MOVES “off network” SCC codes (which end with “00”). These records will be addressed in Step 14 below.
12) Create a new field, MOVES_EF, in the MOVES emissions file by dividing the MOVES emissions by the MOVES VMT. The units for this field are grams/mile.
13) Create another new field, MOVES_NMIM_EMIS, in the MOVES emissions file by multiplying the MOVES_EF by the NMIM VMT, and then multiplying these results by 10^6. (This factor arises because NMIM VMT is in millions of miles/month.) The units for this field are grams/month.
14) Transfer the MOVES “emissionQuant” field into the MOVES_NMIM_EMIS field for all MOVES “off network” SCC codes. None of the calculations in Step 13 should be overwritten.
15) Link the weight class groups into the NMIM pollutant and MOVES emission files via their respective SCC codes

16) Link the pollutant names into the NMIM pollutant and the MOVES emission files via their respective pollutant codes
17) Summate the NMIM pollutant field by pollutant name, year, month, and weight class. The results are in short tons/month. (A short ton is 2000 pounds.)
18) Summate the MOVES_NMIM_EMISS field by pollutant name, month, and weight class. The results are in grams/month.
19) Convert the grams/month results from Step 18 into short tons/month.
20) Add the January and July emissions for each year for NMIM, and for MOVES, and then multiply these results by 6 to convert to short tons/year.
21) Compare the annual results for NMIM and the MOVES_NMIM_EMIS by each category.

Appendix B

Sample Sizes for Cold Start Emissions

Passenger Cars		
Model Year Group	FTP	LA92 (Kansas City)
1969-1980	1,488	4
1981-1982	2,735	0
1983-1985	2,958	5
1986-1989	6.837	17
1990-1993	3,778	11
1994-2000	333	49
2001-2002	18	5
Tier 2 Bin 5	1,900	2 (2003-2004)

Light-Duty Trucks		
Model Year Group	FTP	LA92 (Kansas City)
1969-1980	111	1
1981-1985	910	0
1986-1989	1192	6
1990-2000	1755	26
2001-2002	9	4
Tier 2 Bin 5	Included in Passenger Cars	2 (2003-2004)

Appendix C
Additional Information from EPA on Log-Linear Deterioration in Exhaust Emissions

The following are a series of plots that were provided to AIR by EPA in support of its development of a log-linear model of deterioration to represent exhaust emissions in MOVES. The data are drawn from I/M 147 tests of different model year groups in both Arizona and Illinois. The 10/27 in each title refers to questions AIR asked on 10/27/08, and the Q1 stands for "Question 1", which had to do with the form of the equation for deterioration.

There are six plots, as follows:

- NOx versus age
- Log NOx versus age
- THC versus age
- Log THC versus age
- CO versus age

The 1999-2000 model year group contains too little data to evaluate the form of the deterioration. The 1996-1998 model year group has four years of data in the Arizona I/M program, and is useful to examine. The 1994-1995 model year group has five years, but these two model years were phase-in years for the Tier 1 standards.

The following are observations on these plots:

NOx versus age: The emissions appear to increase linearly for the 1994-2000 vehicles. The 1994-95 vehicles show an acceleration in deterioration between ages 5 and 6, and then this slows.

Log NOx versus age: The log plots for 1994-2000 vehicles appear to "bend over" with age, indicating linear deterioration with age. If the log emissions are regressed with age, they will show an accelerating upward trend in emissions with age in real space, which is not really reflected in the previous plot.

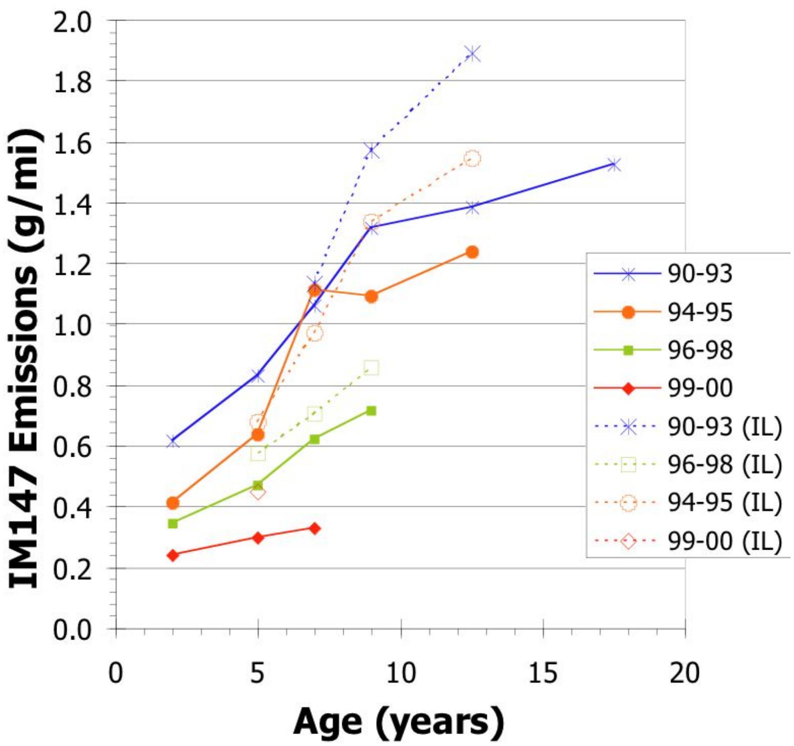
THC versus age: The emissions of 1994-2000 vehicles do appear to accelerate with age.

Log THC versus age: The emissions of 1994-2000 vehicles do not bend over, except for he 1994-95 vehicles. There is not enough data for the later model year groups to determine if they will bend over.

CO versus age: CO emissions of 1994-2000 appear to increase linearly with age.

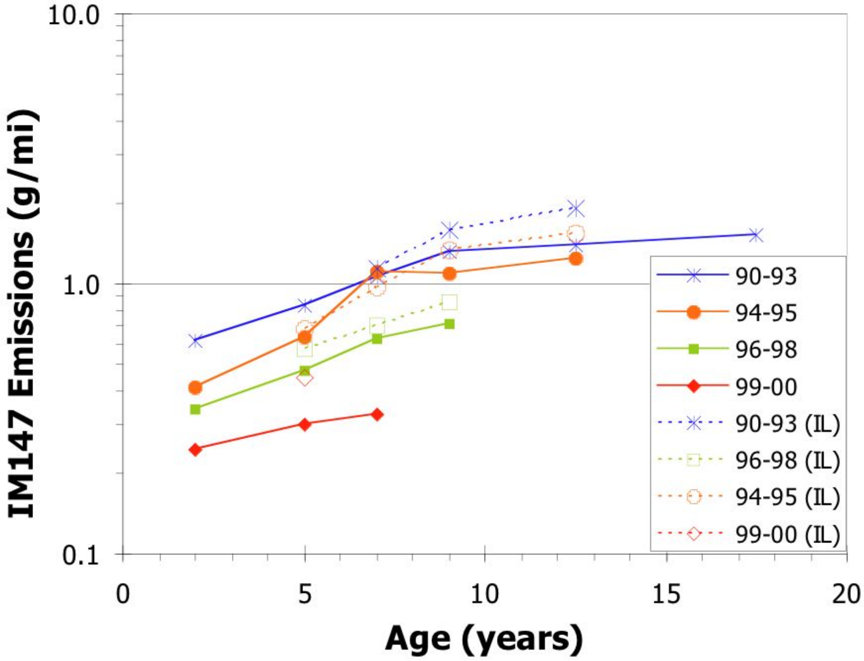
Log CO versus age: CO emissions appear to bend over with age, except for the 1996-98 data, of which there are only 3 points.

10/27, Q1, NOx

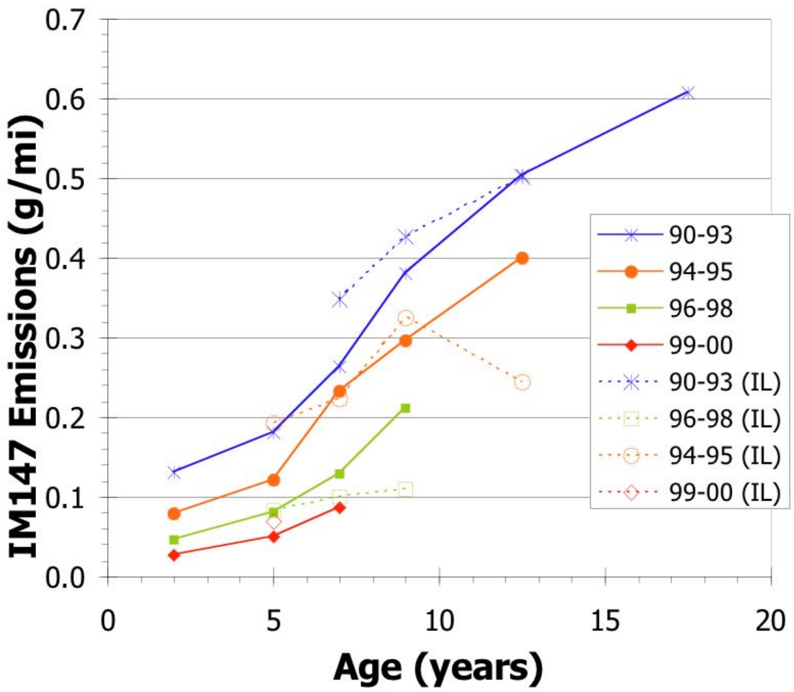


IM147 Emissions (g/mi)
2.0
1.8
1.6
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0.0
0
5
10
15
20
Age (years)
90-93
94-95
96-98
99-00
90-93 (IL)
96-98 (IL)
94-95 (IL)
99-00 (IL)

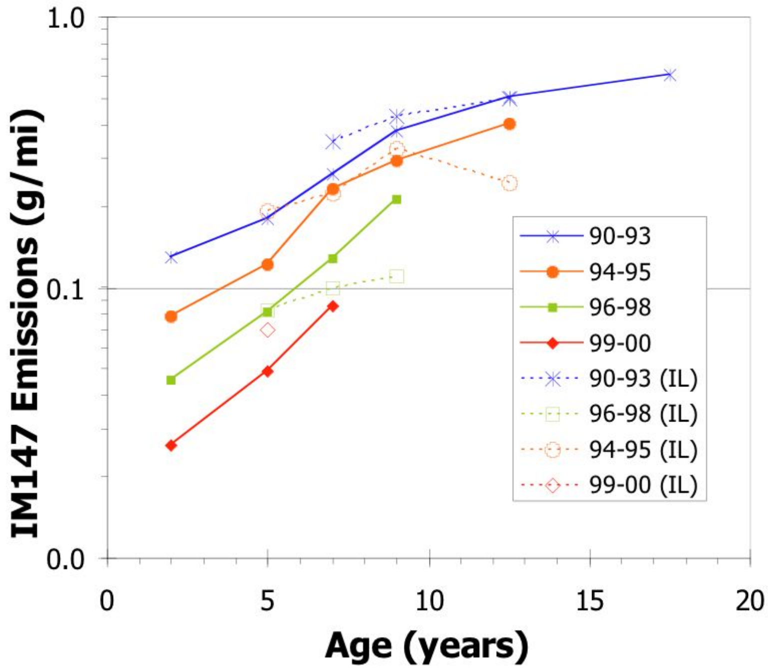
10/27, Q1, logNOx



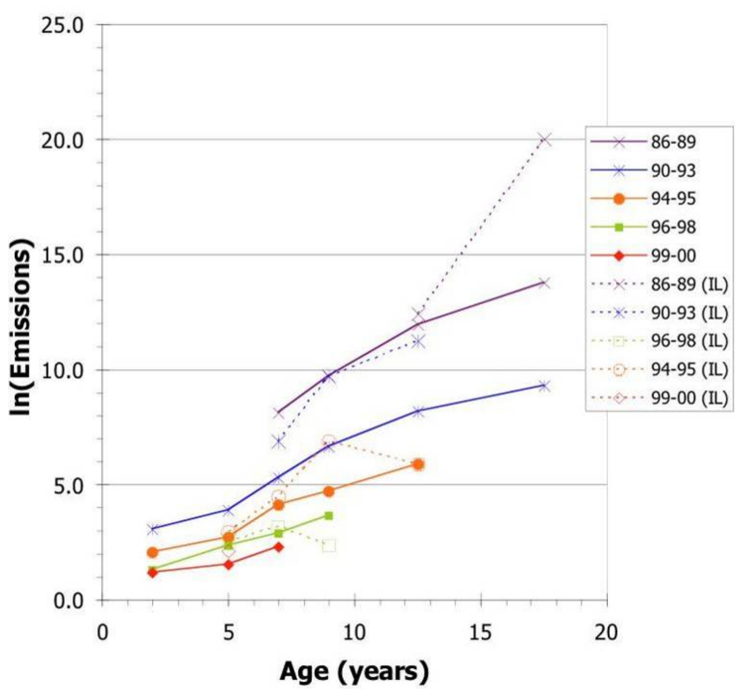
10/27, Q1, THC



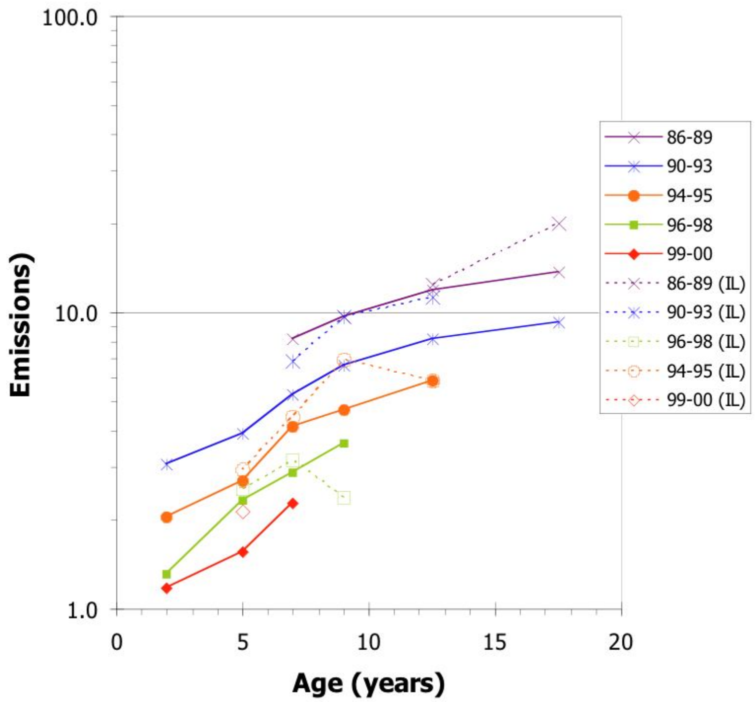
10/27, Q1, logTHC



10/27, Q1, CO

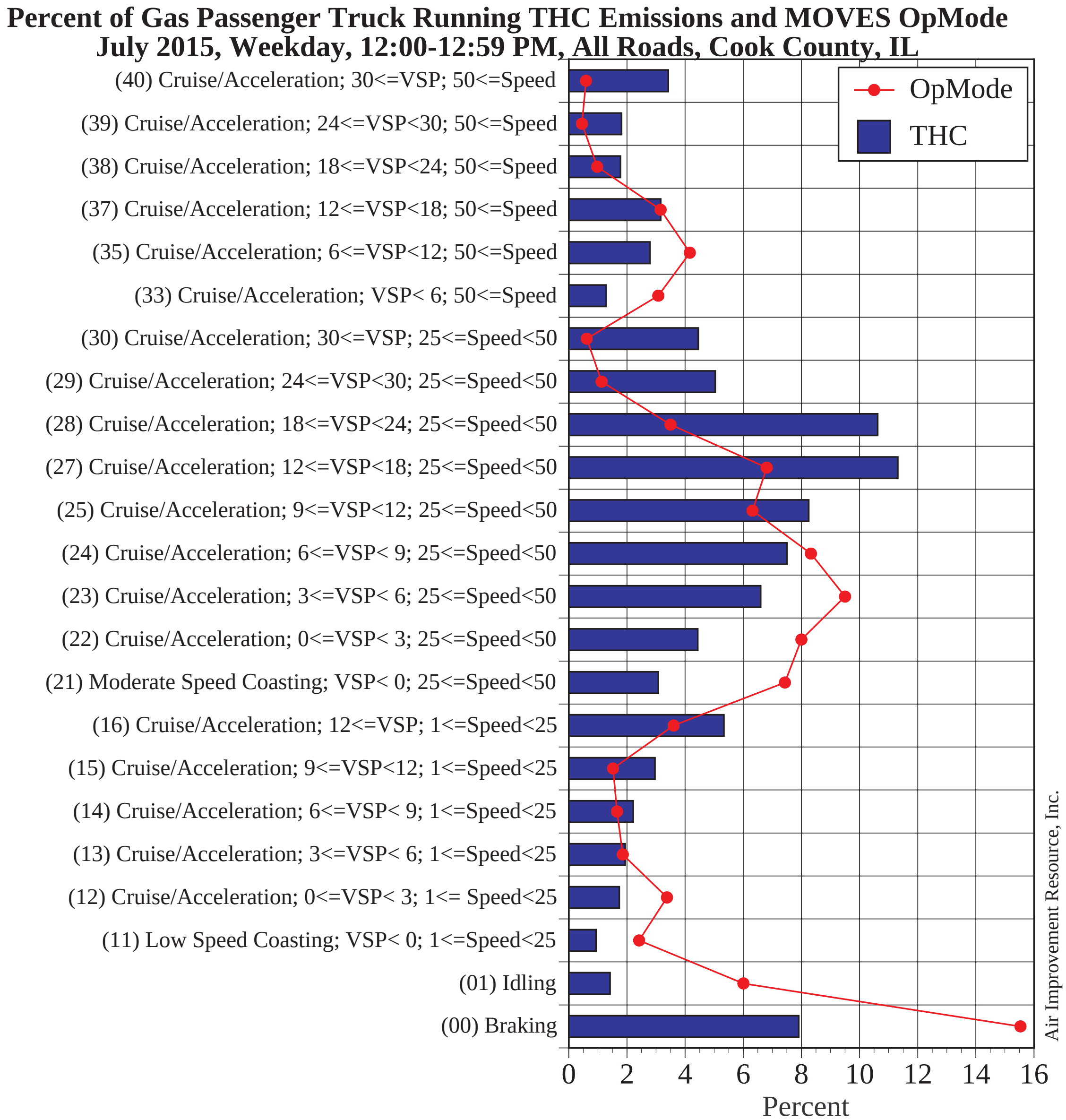


10/27, Q1, logCO

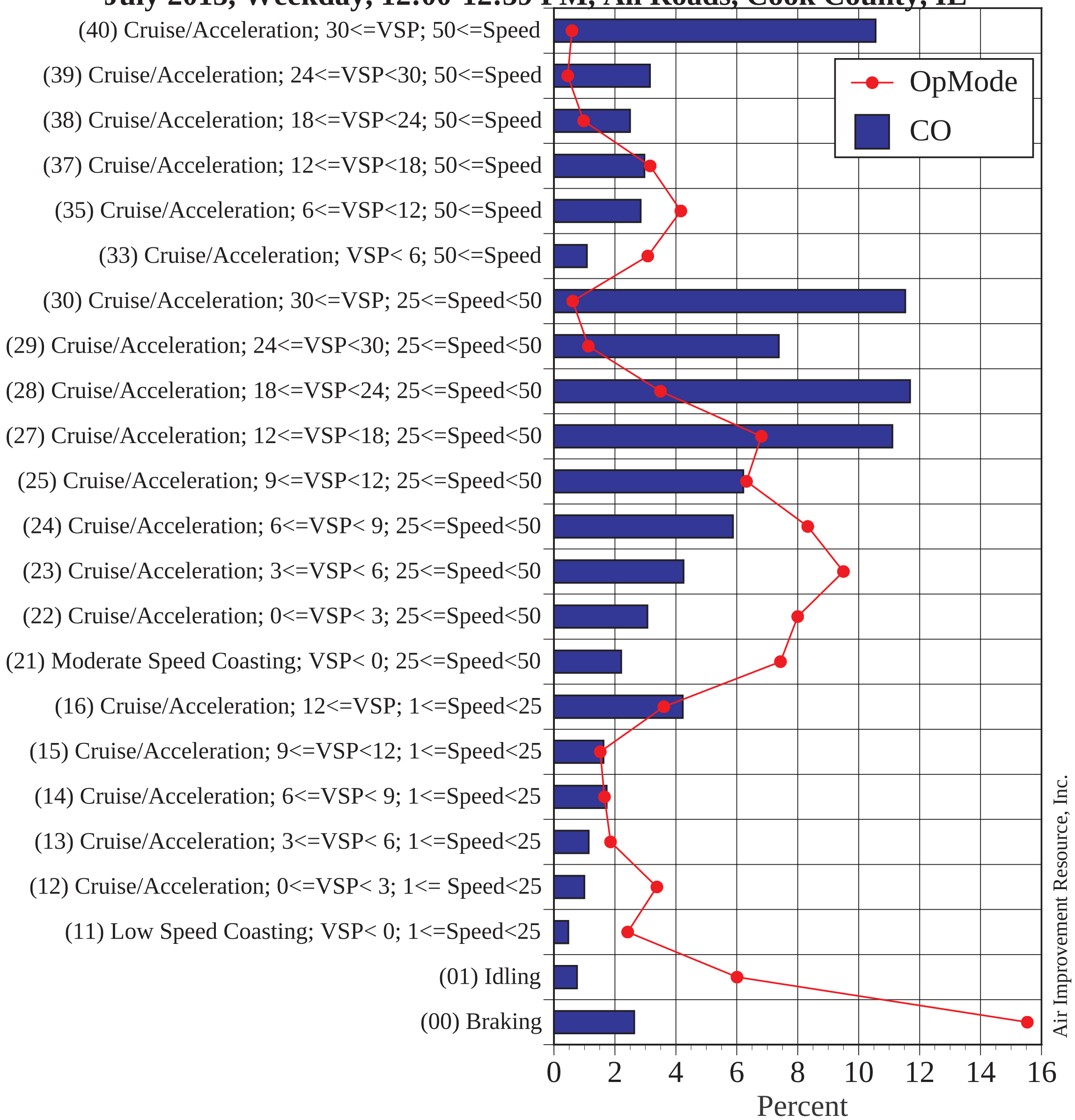


Appendix D
Distribution of Emissions by VSP Bin for LDTs

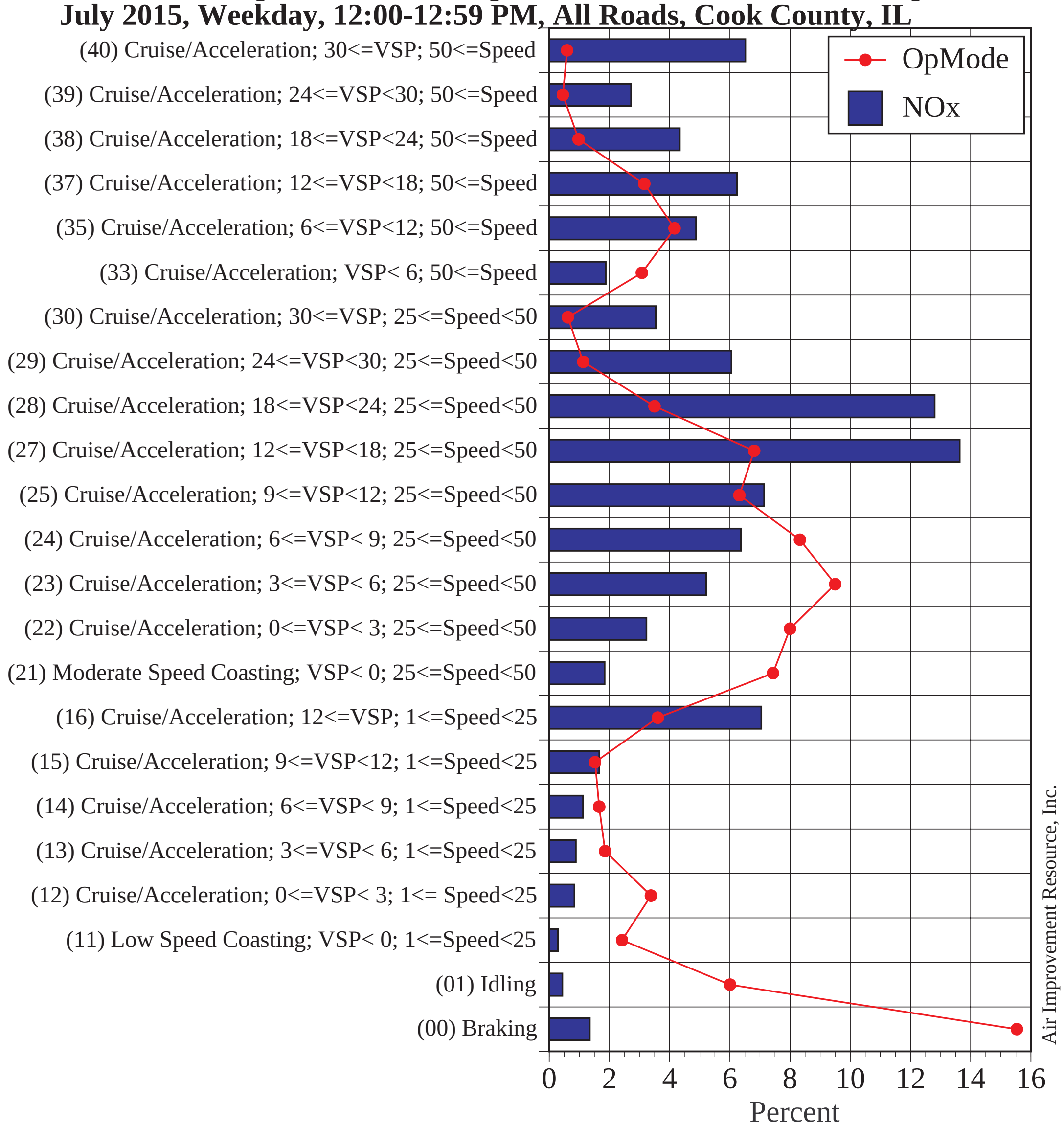
The following charts show light-duty truck emissions by VSP bin from MOVES for Cook County, and also the operating mode distribution by VSP bin. These charts were developed using the same process as discussed for passenger cars in Section 5.3 of the report. The first four charts show emissions of the 2015 fleet, and the next four charts show emissions for the 2009 model year (Tier 2) in 2015. The concentration of emissions in the higher VSP bins is similar for LDTs as it is for passenger cars.



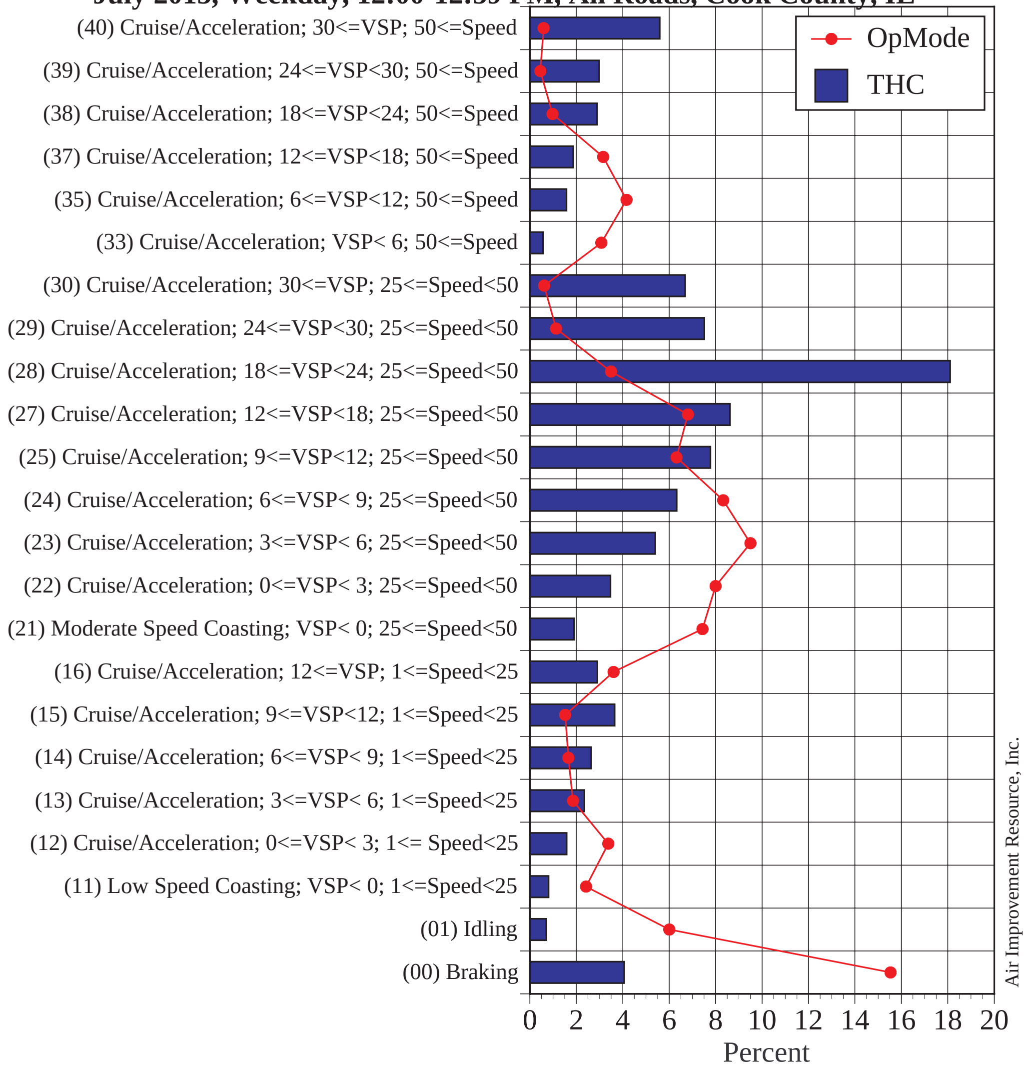
Percent of Gas Passenger Truck Running CO Emissions and MOVES OpMode
July 2015, Weekday, 12:00-12:59 PM, All Roads, Cook County, IL



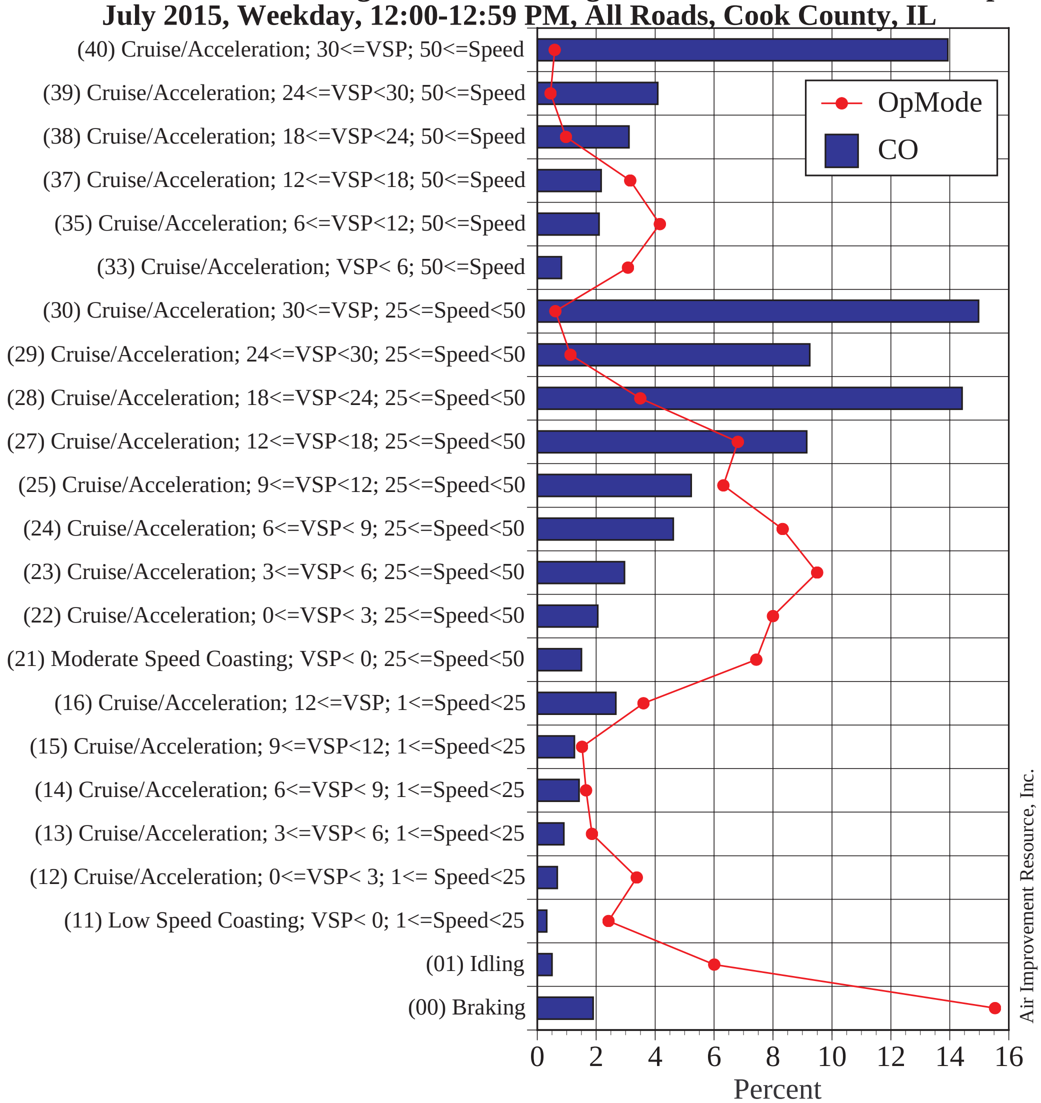
Percent of Gas Passenger Truck Running NOx Emissions and MOVES OpMode
July 2015, Weekday, 12:00-12:59 PM, All Roads, Cook County, IL



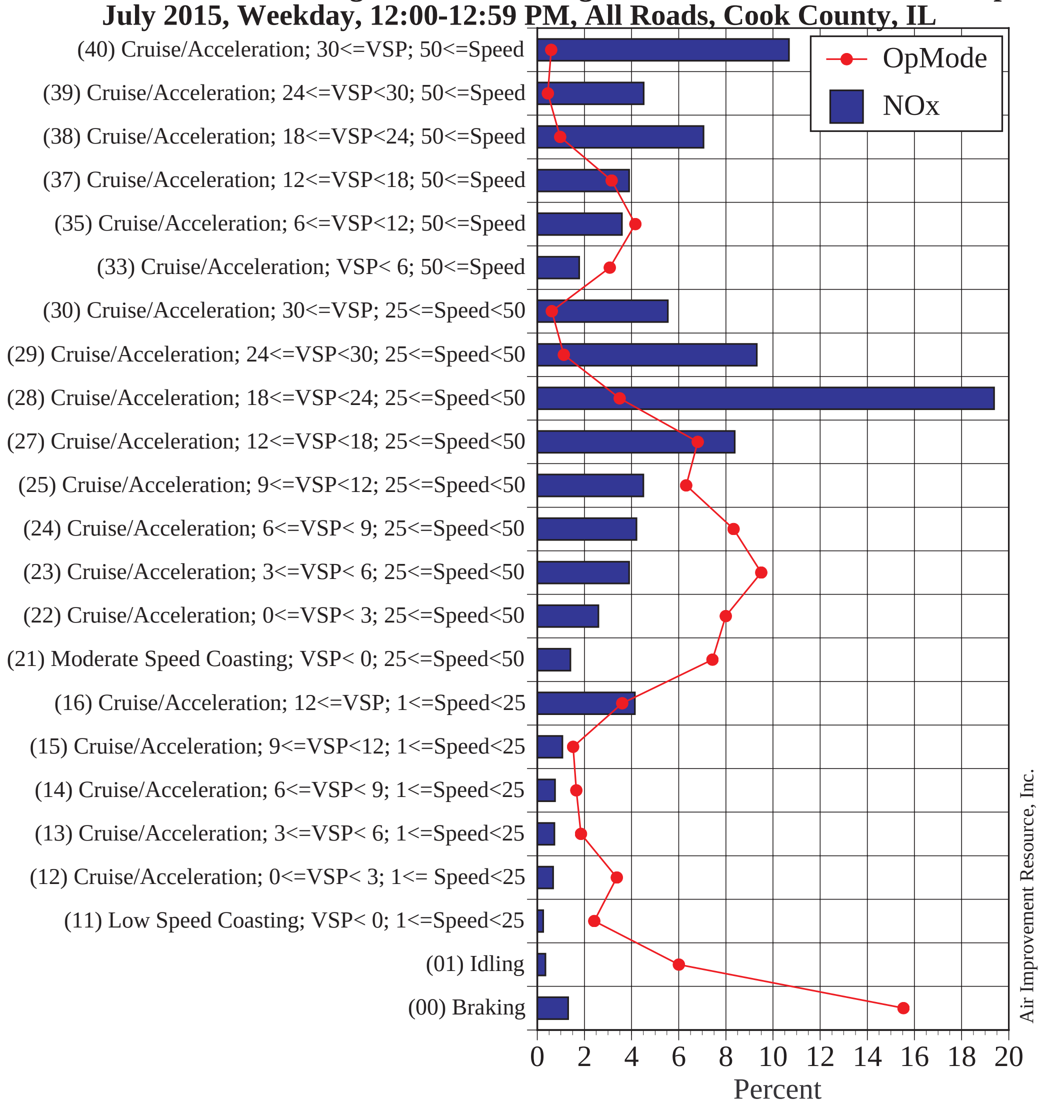
Percent of MY2009 Gas Passenger Truck Running THC Emissions and MOVES OpMode
July 2015, Weekday, 12:00-12:59 PM, All Roads, Cook County, IL



Percent of MY2009 Gas Passenger Truck Running CO Emissions and MOVES OpMode
July 2015, Weekday, 12:00-12:59 PM, All Roads, Cook County, IL



Percent of MY2009 Gas Passenger Truck Running NOx Emissions and MOVES OpMode
July 2015, Weekday, 12:00-12:59 PM, All Roads, Cook County, IL



Appendix E

Additional Information from EPA on Cold Temperature Correction Factors

DRAFT

May 16, 2008

Special Cold Weather Effects

- Larry Landman

There are two sets of regulations that can affect our estimates of emissions at low temperature (i.e., at 20 degrees Fahrenheit), namely the cold weather CO requirement and the cold weather HC requirement.

1. Cold Weather CO Requirement:

The cold weather CO requirement for the 1994 and newer model year LDVs and LDTs limits the composite FTP CO emissions to 10.0 grams per mile at a temperature of 20 degrees Fahrenheit. However, the FTP test results used for our earlier analysis (for those model years) were from vehicles that were certified as meeting that cold weather composite CO requirement. Thus, the temperature adjustments (based on regressions of those FTP results) already incorporated that cold weather CO requirement into MOVES.

2. Cold Weather HC Requirement:

The recently signed MSAT-2 rule included a limit on low temperature (i.e., at 20 degrees Fahrenheit) NMHC emissions for light-duty and some medium-duty gasoline-fueled vehicles. Specifically:

- For passenger cars (LDVs) and for the light light-duty trucks (LLDTs) (i.e., those with GVWR up to 6,000 pounds), the composite FTP NMHC emissions should not exceed 0.3 grams per mile.

- For heavy light-duty trucks (HLDTs) (those with GVWR from 6,001 up to 8,500 pounds) and for medium-duty passenger vehicles (MDPVs), the composite FTP NMHC emissions should not exceed 0.5 grams per mile.

These cold weather standards are to be phased-in beginning with the 2010 model year, specifically:

Phase-In of Vehicles Meeting Cold Weather HC Standard

Model Year	LDVs / LLDTs	HLDTs / MDPVs
2010	25%	0%
2011	50%	0%
2012	75%	25%
2013	100%	50%
2014	100%	75%
2015	100%	100%

2.1 Incorporating into MOVES:

To incorporate this set of HC requirements into MOVES, we must first determine their impact on the start emissions (both cold-start and hot-start) as well as on the running emissions for each class of vehicles.

We already observed that changes in the ambient temperature do not have a significant effect on the running THC emissions. Therefore, we will assume that the full impact of this requirement will be on the start emissions.

Our earlier analysis of temperature effects on the emissions of Tier-2 vehicles was based on a sample of a single gasoline-fueled passenger car and three light-duty trucks that were each FTP tested at zero, 20, and 75 degrees Fahrenheit. The average nonmethane HC (NMHC) composite FTP emissions at 75º F were:

- 0.02 (0.0180) g/mile for the passenger car and
- 0.04 (0.0353) g/mile for the heavy light-duty trucks.

Considering the MSAT-2 standards (0.30 and 0.50, respectively), this would mean the NMHC composite FTP emissions increasing by no more than 0.28 grams per mile (i.e., 0.30 minus 0.02) for LDVs/LLDTs and by no more than 0.46 grams per mile for HLDTs/MDPVs as the ambient temperature drops from 75º F down to 20º F.

Since the composite FTP simulates a trip 7.45 miles in length, those rates convert to total NMHC increases of 2.086 grams (for LDVs/LLDTs) and 3.427 grams (for HLDTs/MDPVs). Those increases represent the increases in the generic start emissions (57 percent hot-start and 43 percent cold-start). Using the ratio of hot-start to cold-start from our earlier analysis, this results in increases in NMHC cold-start emissions (as the ambient temperature drops from 75º F down to 20º F) of:

- 0.5611592 grams for the LDVs/LLDTs and
- 0.9219045 grams for the HLDTs/MDPVs.

Since the MSAT-2 rule assumes that increase in NMHC is linear with temperature (decreasing 55 degrees), then those rates convert to decreases in total NMHC per cold-start of:

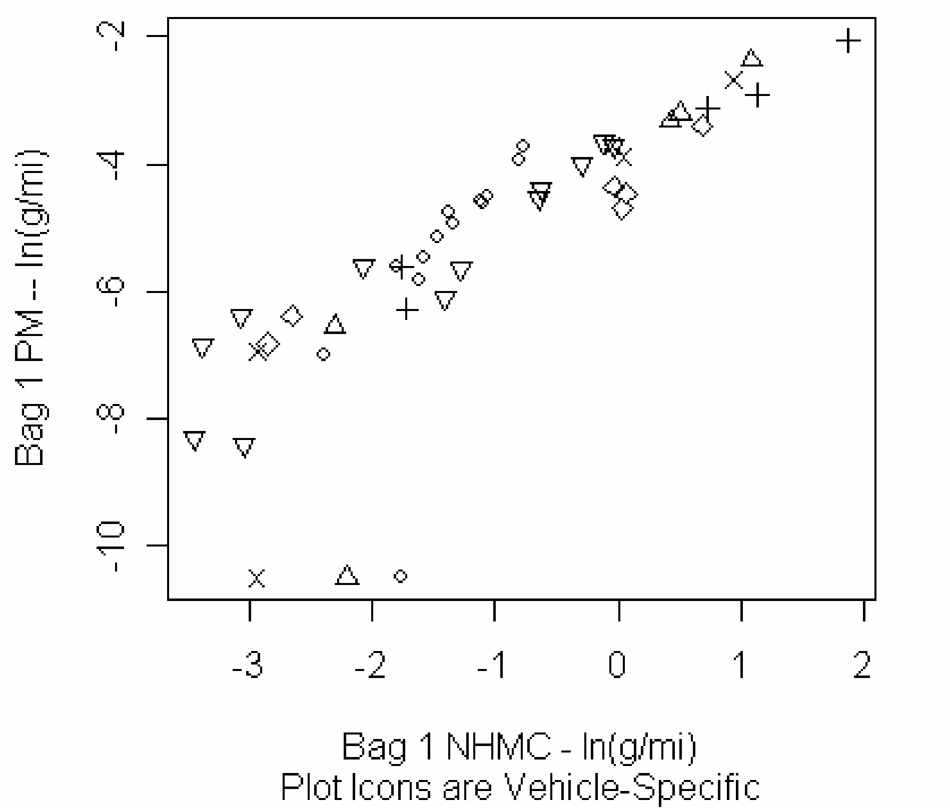
- −0.0102029 grams per degree F for the LDVs/LLDTs and
- −0.0167619 grams per degree F for the HLDTs/MDPVs.

These are the rates (slopes) that we will use in MOVES for cold-starts (i.e., starts that follow a 12 hour engine soak). For the seven shorter soak periods (that MOVES uses as opModes), we will continue to use the ARB soak adjustments for HC emissions for catalyst equipped vehicles to estimate those HC emissions (following the seven shorter soak periods).

3. Cold Weather PM Requirement:

The MSAT-2 rule (signed February 9, 2007) does not explicitly limit cold weather emissions of particulate matter (PM). However, the Regulatory Impact Analysis (RIA) document* that accompanied that rule noted there is a strong linear correlation between NMHC and $PM_{2.5}$ emissions. That correlation is illustrated in the following graph (reproduced from that RIA) of the logarithm of the Bag-1 $PM_{2.5}$ versus the logarithm of the Bag-1 NMHC (for various Tier-2 vehicles).

FTP Bag 1 PM and FTP Bag 1 NMHC for Various Tier 2 Vehicles



* "Regulatory Impact Analysis for Final Rule: Control of Hazardous Air Pollutants from Mobile Sources" EPA Report Number EPA420-R-07-002, February 2007, Chapter 2, pages 2-15 to 2-17.
Available at: http://www.epa.gov/otaq/regs/toxics/fr-ria-sections.htm

Therefore, the limitation on cold weather HC (or NMHC) emissions is expected to result in an ancillary reduction in cold weather $PM_{2.5}$ emissions. In that RIA (Table 2.1.-9), EPA estimated that this requirement would result in a 30 percent reduction of VOC emissions (at 20º F). Also, in that RIA, the ratio of PM to NMHC equaling 0.022 was used to estimate that $PM_{2.5}$ reduction. (The 95 percent confidence interval for that ratio was 0.020 to 0.024.) Applying the same analytical approach that was used in that RIA means that a 30 percent reduction in VOC emissions would correspond to a 30 percent reduction in PM emissions at 20º F (for Tier-2 cars and trucks).

EPA's earlier analysis (for MOVES)* indicated that ambient temperature does affect the rate of running PM emissions as well as start PM emissions, and that effect (for Tier-2 vehicles) is best modeled by (exponential) multiplicative adjustment factors of the form:

Multiplicative factor = $e^{\mathbf{A}*(72-t)}$, where "t" is the ambient temperature

and where $\mathbf{A}$ = 0.0463 for cold-starts and
0.0318 for hot running
(See Table 12 in that EPA document, page 46.)

Therefore, for Tier-2 vehicles <u>not</u> affected by the MSAT-2 requirements, EPA expects (as the temperature decreases from 72º down to 20º F) the PM emissions to increase by factors of:

- 11.10727 fold for cold-starts and
- 5.22576 fold for hot running.

Thus, applying that 30 percent reduction for vehicles that are affected by the MSAT-2 requirements produces estimates (as the temperature decreases from 72º down to 20º F) of PM emissions increasing by factors of:

- 7.77509 fold for cold-starts and
- 3.65803 fold for hot running.

Since the vehicles affected by the MSAT-2 requirements begin to be phased-in starting with the 2010 model year, EPA expects the following (multiplicative) increases (as the temperature decreases from 72º down to 20º F):

---

* "Analysis of Particulate Matter Emissions from Light-Duty Gasoline Vehicles in Kansas City" EPA Report No. EPA420-R-08-010, April 2008, Chapters 7 and 8. Available at: http://www.epa.gov/OMS/emission-factors-research/420r08010.pdf

Multiplicative Increases of PM at 20° Fahrenheit

	LDVs / LLDTs		HLDTs / MDPVs	
Model Year	**Start**	**Running**	**Start**	**Running**
2008	11.10727	5.22576	11.10727	5.22576
2009	11.10727	5.22576	11.10727	5.22576
2010	10.27423	4.83383	11.10727	5.22576
2011	9.44118	4.44189	11.10727	5.22576
2012	8.60814	4.04996	10.27423	4.83383
2013	7.77509	3.65803	9.44118	4.44189
2014	7.77509	3.65803	8.60814	4.04996
2015	7.77509	3.65803	7.77509	3.65803

Solving for the corresponding constant terms so that the preceding exponential equation will yield these increases, gives these "A" values:

Constant Terms

	LDVs / LLDTs		HLDTs / MDPVs	
Model Year	**Cold-Start**	**Running**	**Cold-Start**	**Running**
2008	0.046300	0.031800	0.046300	0.031800
2009	0.046300	0.031800	0.046300	0.031800
2010	0.044801	0.030301	0.046300	0.031800
2011	0.043175	0.028675	0.046300	0.031800
2012	0.041398	0.026898	0.044801	0.030301
2013	0.039441	0.024941	0.043175	0.028675
2014	0.039441	0.024941	0.041398	0.026898
2015	0.039441	0.024941	0.039441	0.024941

We will assume that these same magnitude increases in the $PM_{2.5}$ emissions also apply to the EC and OC emissions.

Although the ARB factors that adjust the start emissions based on soak time were not developed for PM emissions from gasoline-fuel vehicles, the fact that the ratio of PM emissions to the HC emissions are almost constant suggests that we can apply the HC soak adjustment factors to the start PM emissions.

Summary of Temperature Effects in MOVES

1. MOVES now includes a temperature effect for particulate matter (PM) emissions, as well as for the other criteria pollutants. MOBILE6 did not have a temperature adjustment for PM.

2. Temperature primarily affects vehicle emissions that occur immediately after an engine start. MOVES has no temperature adjustment for engine running emissions that occur after the engine warms to operating temperature, except for PM. Exhaust running PM emissions are affected by ambient temperature even after the engine is fully warm for temperatures below 72 degrees Fahrenheit using a multiplicative adjustment.

3. Engine start emissions are reported separately from engine running emissions. Engine start emissions are adjusted for the effects of temperature using an additive adjustment, except for PM. PM temperature adjustments to engine start emissions are multiplicative.

4. Engine start emissions are only affected at temperatures below 75 degrees Fahrenheit (72° for PM). Higher temperatures do not affect engine start emissions.

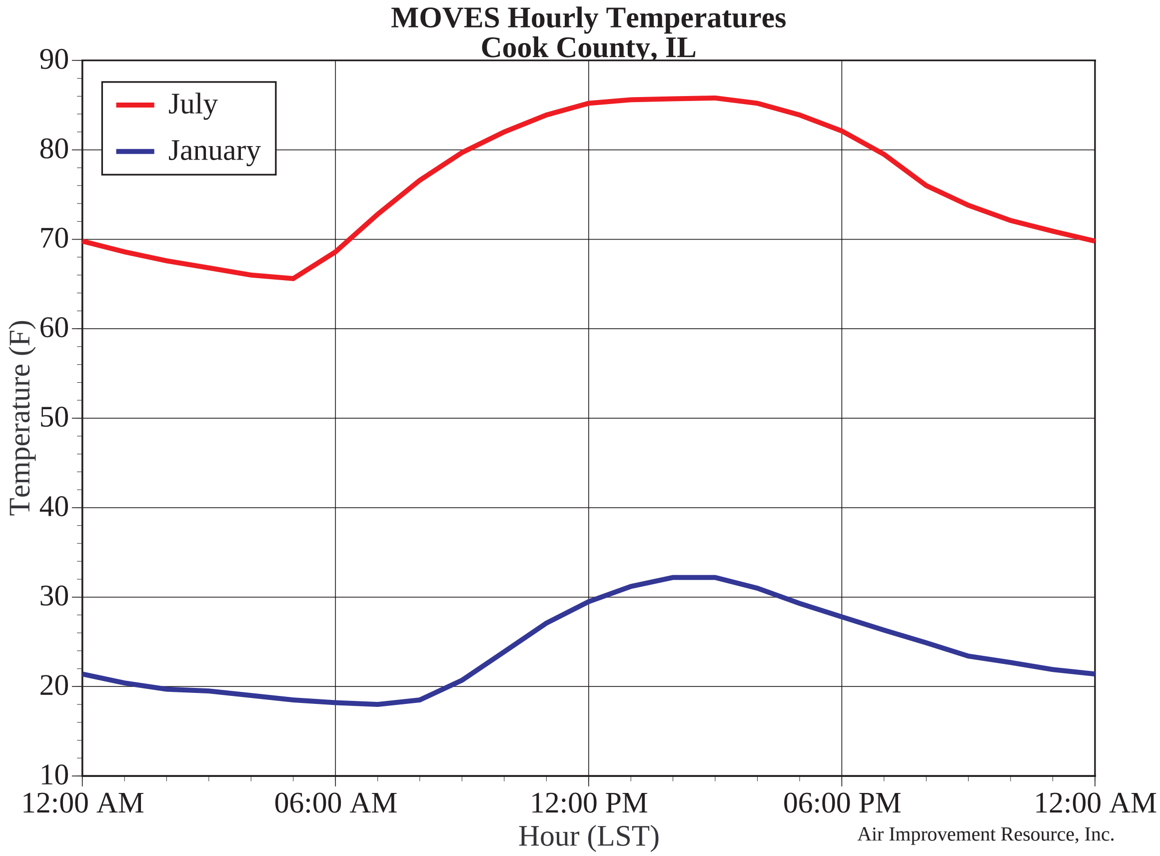
5. CO engine start emissions for 1994 and newer model year vehicles are affected by the cold temperature CO regulations. HC engine start emissions for 2010 and newer model year vehicles are affected by the mobile source air toxic (MSAT) regulations.

6. MOVES uses data that includes measurements down to -20 degrees Fahrenheit.

There is a presentation that describes the temperature adjustments used by MOVES.

Appendix F
Temperatures Modeled in MOVES for Chicago (Cook County)

The temperatures modeled in MOVES for Cook County for January and July are shown in the figure below. Summer temperatures range from 67°F to 86°F. Winter temperatures range from 19°F to 32°F.



Appendix G

E-68 Task 3 Report
Investigation of Validation Methods

1.0 Introduction

The CRC Emissions Committee approved AIR to evaluate different methods of validating the MOVES model. Prior to evaluating these different methods, AIR reviewed the CRC-E64 study, where MOBILE6 was checked using certain methods. The five methods reviewed for MOVES are as follows:

- Near roadway studies and ambient ratios
- I/M data
- Remote sensing data
- Other cycles
- Tunnel studies

The remainder of this appendix is divided into the following sections:

- CRC E-64 Study of MOBILE6
- Roadside Ambient Measurements and Ambient Ratios
- I/M Data
- Remote Sensing Data
- Other Cycles
- Tunnel Studies
- Recommendations

2.0 CRC E-64 Study of MOBILE6

The CRC E-64 study utilized several methods for checking MOBILE6 emission rates. [1] Methods used were:

- Tunnel study comparisons
- HC/NOx and CO/NOx ambient versus model ratios
- Emission ratios of remote sensing data
- Heavy-duty chassis data versus MOBILE6
- Diesel fuel consumption versus sales

The heavy-duty chassis data has been incorporated into MOVES, and EPA has already compared predicted versus actual diesel fuel consumption. Therefore, this discussion will concentrate on the first 3 methods.

Tunnel Studies - The study evaluated four different tunnels and a range of test years from 1982 to 1999. MOBILE6 modeling included the use of local data to match tunnel conditions for speed, temperature, age distribution and fleet mix. Light and heavy-duty emissions were compared between tunnel and MOBILE6 separately. Results varied by tunnel and pollutant. The only consistent trend noted between all four tunnels for light duty vehicles was that MOBILE6 CO emissions were overpredicted relative to the tunnel results. For NOx and NMHC, sometimes MOBILE6 predicted higher emissions than the tunnels, and sometimes lower. For heavy-duty vehicles, MOBILE6 NMHC and CO were higher than the observed results. For NOx, sometimes MOBILE6 was higher and sometimes lower than the tunnel results.

HC/NOx and CO/NOx ambient versus model results – in this method, ratios of species in emission inventories prepared using MOBILE were compared with corresponding ratios in ambient monitoring data during morning commute hours at urban locations with significant mobile source impacts. This allows the evaluation of the degree to which MOBILE6 based emission inventories reproduce the observed pollutant mixture. HC/NOx ambient ratios agreed reasonably well with the model at most sites, but CO/NOx ambient ratios exceeded model estimates at all sites. In general, this contradicts the tunnel results. The report concluded that " Results of he ambient-inventory reconciliation analyses presented….cannot be used to directly infer the accuracy of MOBILE6 emission estimates since the mobile source contributions to ambient concentrations cannot be separated from those of other source categories."

Emission ratios of remote sensing data - This study also evaluated emission ratios from remote sensing data in Denver and Chicago. The study focused on emission ratios rather than absolute emissions, to reduce uncertainty associated with converting emissions from a fuel basis to a g/mile basis. Rough vehicle classifications were used to develop the average age of the RSD fleet for comparison with MOBILE6. Comparisons were made with and without a correction for vehicle specific power. Results indicated that, relative to NO, MOBILE6 overestimates CO emissions from newer vehicles. RSD HC/NO ratios were found to more closely resemble MOBILE6 HC/NO ratios than for CO/NO. When the RSD data were corrected for the difference between the MOBILE6 ramp cycles and the actual ramp speeds, the CO/NO ratios were more similar.

Overall, this study indicates that it is very difficult to check an inventory model with actual results such as ambient, RSD, and tunnel data because of the differences in fleets, operating conditions, and a variety of other factors. A number of the methods, however, seemed to indicate that CO emissions from MOBILE6, and estimated CO deterioration from the fleet, was too high in MOBILE relative to the different data sources. Definite conclusions about NOx and HC were more difficult to reach. The study emphasizes the importance in utilizing at least 2 methods of checking MOVES. If there is agreement between at least two methods with respect to the trend of a certain pollutant, then greater weight can be placed on that conclusion.

Implications for MOVES Validation - For validating MOVES, we are evaluating roadside ambient measurements, I/M data, remote sensing data, data on other testing cycles, and tunnel studies. Some of these methods have been used by the EPA in checking MOVES.

3.0 Roadside Ambient Measurements and Ambient Ratios

One of the options for evaluating MOVES is to examine near roadway monitoring studies that have monitors sited in a way to clearly identify the motor vehicle related concentrations. These could be used to see if the magnitude of the MOVES model emission factors and the emission estimates are consistent with observed ambient concentrations. This is somewhat difficult to accomplish because MOVES estimates emissions at different scales in tons, while the ambient measurements are in concentrations such as ug/m^3. However, it would be possible to examine pollutant ratios near roadways, for example, $PM_{2.5}$/CO ratios. Also, near roadway emission measurements are not purely measuring emissions from the fleet of on-road vehicles, as background emissions from other sources complicate these measurements. It may be possible, however, to check the trends in emission rates from MOVES by evaluating trends in ambient concentrations near a roadway. If near-roadway ambient measurements are available for two different years that are a reasonably far apart (say, for example, 5 years), and If changes in traffic flow are accounted for (traffic flow can increase or decrease, also causing differences in ambient measurements), it may be possible to compare the trends in MOVES predicted emissions over the time period to the trends in ambient measurements over the same period. However, in our view, it is probably not possible to convert MOVES emissions into an ambient concentration at a roadway to compare with the ambient measurements, unless a local dispersion model is used, and this introduces additional uncertainty.

There is limited information in the literature about roadway traffic generated contributions to ambient pollutant levels. This information is limited in part because of the recent emphasis in EPA's $PM_{2.5}$ monitoring programs to establish monitors at locations that satisfy the primary objectives of determining each area's compliance with established ambient air quality standards, and of providing data that are compatible with health effects research needs. The most extensive near roadway study information in the published literature is from monitoring conducted near major freeways in the South Coast of California. [2] EPA and the Federal Highway Administration (FHWA) have a three-year monitoring plan in progress that is expected to yield new data for three areas of the country that might be useful for MOVES evaluations, however, the data are not yet publicly available.

The South Coast of California studies were conducted by Zhu et al. [2] These studies measured PM concentrations near the Interstate 405 freeway and the Interstate 710 freeway. The Interstate 405 study showed that the particle number concentration near the freeway was about 25 times greater than at background locations, and that the concentration of ultrafine particles drops to background levels within 300 meters downwind of the freeway. For the conditions of these measurements, relative concentrations of CO and black carbon (BC) near the freeway tracked each other well as distance from the freeway increased. Average traffic flow during the sampling periods was 13,900 vehicles per hour. Ninety-three percent of the vehicles were gasoline powered cars or light trucks.

The Interstate 405 study included measurements of concentrations of CO, BC, PM and particle number at increasing distances from the freeway. CO and BC were intentionally selected because their ambient concentrations are closely related to vehicle emissions. The Interstate 710 study was conducted in part because the freeway has a much higher percentage of heavy-duty diesel truck travel than the Interstate 405 freeway.[3] On the 710 freeway, more than 25 percent of the vehicles are heavy-duty diesel trucks. Measurements were taken at 17, 20, 30, 90, 150 and 300 meters downwind and 200 meters upwind from the freeway center. Average traffic flow during sampling periods was 12,180 vehicles per hour, or about 13% less than the 405 freeway study. The distribution of particle number versus distance from the road was similar to the 405 freeway. CO levels near the 710 freeway were lower than for the 405 freeway, owing to less traffic flow and lower CO levels from diesel vehicles than gasoline vehicles. Black carbon concentrations near the 710 freeway were significantly higher than near the 405 freeway, due to higher PM emissions from the trucks.

FHWA and EPA have initiated the first of three near road studies. The proposed study locations are Las Vegas, Detroit, and Raleigh. The Las Vegas study is underway and near completion. Some of the Las Vegas data are being used currently by EPA's Office of Research and Development (RTP North Carolina). The FHWA/EPA near- road studies focus on evaluating near-road concentration gradients. Because the impetus for these studies was the settlement of the lawsuit with the Sierra Club regarding a freeway expansion in Las Vegas near an elementary school, the data collection focus is on air toxics as well as criteria air pollutants. The measured mobile source air toxics are benzene, 1,3-butadiene, formaldehyde, acetaldehyde, and acrolein. Criteria pollutant measurements include CO, NO/NO2, and $PM_{2.5}$. Each study is intended to collect ambient data for approximately one year. Non-air quality measurements at each of these sites include wind speed and direction, traffic volume, vehicle speeds and the fleet mix.

Another near roadway study in the recent literature was performed in Seattle, WA. [4] Curtis, Gilroy and Harper (2004) studied the relationship between BC levels at an urban near roadway monitoring site, and a heavily traveled freeway. The analysis was possible because a routine air-monitoring site was close to a major downtown freeway (I-5). The monitoring site is about 20 meters west of the southbound lane of I-5. Daily volumes along this section of I-5 average 284,700 vehicles per day (in 2003). Light-duty traffic has peak weekday flows above 10,000 vehicles per hour, with diesel traffic of about 1,000 vehicles per hour. The study showed higher levels of BC on weekdays and Saturdays than on Sundays. The lower emissions on Sundays were attributed to lower heavy-duty vehicle traffic on those days.

We are not recommending performing comparisons of MOVES to roadside ambient data studies at this time, since EPA may do this in the near future, and the results from the CRC-64 data for ambient data in general were inconclusive. However, we do think there is potential for evaluating $PM_{2.5}$/CO ratios from ambient studies to MOVES emission ratios, as long as there is adequate roadside information on gasoline and diesel vehicles.

4.0 I/M Data

EPA developed the MOVES exhaust emission rates using the Arizona I/M data. The Arizona I/M program currently uses the I/M147 test, a test that is a portion of the I/M240 test. Many vehicles are fast-passed if their results are low enough in the first part of the test, but Arizona performs full I/M147 tests on a random sample of vehicles.

Since EPA developed the exhaust emission rates on the Arizona I/M data, the Arizona I/M data cannot be used to evaluate these emission rates. However, there are other areas of the country with I/M tests, and it is possible that I/M data from other areas could be used to check the MOVES emission rates. There are several limitations, however. One is that many other I/M programs do not perform random I/M147 or I/M240 tests on the fleet. Full tests are only performed on vehicles that fail or may fail the fast-pass test. A second limitation is that since most I/M programs only test cars and light trucks, this method cannot be used to evaluate MOVES emissions for heavy-duty vehicles. A third limitation is that many I/M programs do not perform tests on 1996 and later vehicles equipped with onboard diagnostic systems, or OBD. Thus, comparisons are limited to older vehicles.

EPA did use this method to check the MOVES emissions. EPA used the calendar year 2000 Chicago I/M data to perform some evaluations of the MOVES emission rates. [5] In this program, some 1995 and earlier vehicles are given full I/M240 tests. There is only a single test; no replication is performed. In addition, vehicles that are waiting in the I/M queue can cool down, and thus have higher start emissions than fully warmed up vehicles. For this reason, EPA eliminated the emissions data for the first 120 seconds of the I/M240. There were 74,248 vehicles with valid I/M tests that were included in this analysis. The MOVES model operating mode distribution was altered by inputting the driving cycle of the second half of the I/M240 cycle to match the I/M tests. EPA considers this test to be a “lightly-loaded” test.

Results of these comparisons, which are shown for LDVs in Attachment G-1, showed generally good agreement between MOVES and the Chicago I/M data for the pre-1996 vehicles. There is less agreement for the older vehicles, probably due to much smaller I/M sample sizes at the higher ages.

Other I/M program data comparisons are extremely limited because of the lack of random tests performed on vehicles. Primarily for this reason, we are not recommending further evaluation of I/M data in this validation effort.

5.0 Remote Sensing Data

EPA also compared MOVES emission rates to Remote Sensing Data (RSD) in Chicago and Georgia. [5] The Chicago site was a single on-ramp site from Algonquin Rd to I290 E in northwest Chicago. The on-ramp was an uphill cloverleaf, so speeds were moderate because of the curve. No adjustment was made to MOVES emissions by VSP bin for the grade. Measurements were taken in calendar years 2000, 2002, 2004, and 2006, in the month of September, between 9 am and 7 pm. There were 9,133 total observations. CO and NOx were measured.

The Chicago RSD results appeared to be lower than MOVES, especially at ages above about 6-7 years. The differences were more pronounced for LDTs than for passenger cars. The NOx results for cars and light duty trucks are shown in Attachment G-2. Not too much can be drawn from this comparison, however, because it is only one site in Chicago.

The second RSD comparison that EPA used was in Georgia. The RSD data is part of the Continuous Atlanta Fleet Evaluation, and included calendar years 2001-2008. Measurements are taken year round, between 8 am and 6 pm. A total of 58,000+ measurements were used in the comparison. The operating mode distribution of the Atlanta measurements has most of the operation in VSP bins 24-29. These are higher speed/load bins, and a very different distribution overall than the Chicago location.

The Georgia comparison shows much lower emission rates for RSD than the MOVES model for both CO and NOx. These results for LDTs are shown in Attachment G-3.

Other comparisons of RSD data and MOVES could be made, because of the wide variation in results. However, because RSD data only tests on-road vehicles under limited operating conditions (narrow set of speeds and no cold start emissions), and because EPA has already examined two locations of RSD data, we are not recommending further evaluations of RSD data in this study.

6.0 Other Cycles

LA-92 (also called the Unified Cycle)

The Kansas City PM program evaluated emissions on a number of vehicles utilizing the LA-92 test, which is a self-weighting cycle that is representative of driving in California (i.e., more representative of all-around driving than the FTP or I/M240 cycles). CARB uses this test in all in-use testing. THC, CO, and NOx were evaluated.

This database has an advantage over the I/M and RSD data in that it represents a much wider range of vehicle operation than the either the I/M or RSD data. The disadvantage is that the database can suffer from possible “recruitment bias.”

EPA inputted the LA-92 cycle into MOVES and compared MOVES versus the actual KC testing results. [5] The results of this comparison showed much higher THC, CO, and NOx for MOVES than the KC data at ages 7 years and higher. Under 7 years, the results were close for THC and CO, but even under 7 years, NOx was significantly higher for MOVES than the KC data.

California’s in-use testing utilizing the LA-92 represents another important database that could be compared to MOVES. The California in-use testing database is much larger than the KC database. Of course, like the KC data, this database can also suffer from possible recruitment bias, but we think this comparison should be made.

US06

EPA also compared MOVES emissions to US06 emissions from the In-Use Vehicle Program (IUVP) data. [5] The US06 is an extreme driving cycle, with a high fraction of high speed and high acceleration operation. CRC has repeatedly requested these data from the EPA as a part of this project but the data have not been provided. This comparison shows generally good agreement between the final MOVES emissions rates and the US06 data for late model vehicles for both THC and NOx. However, the IUVP data for CO appear to be higher than MOVES, although this is only on vehicles in the 0-3 year age group. It is not possible to use the IUVP data to make comparisons at higher ages.

Tunnel Studies

AIR reviewed three recent tunnel studies that were conducted by the University of California at Berkeley and others. [6, 7, 8] In the first study, light duty vehicle emissions of CO, NOx and NMHC were quantified as functions of vehicle speed and engine load in the Caldecott tunnel for downhill and uphill travel on a ~4% grade. Emissions were measured on weekdays in July and August of 2001. Downhill driving conditions were studied on 5 mornings from 5 to 11 am, and uphill driving emissions were studied in the afternoon and evening over 9 sampling days. Vehicle average speed inside the tunnel was determined using video cameras with synchronized clocks that were located at both ends of the tunnel. Emissions were measured in grams per liter of fuel consumed, and also converted to grams per mile (these are values for the 2001 fleet of vehicles going through the tunnel). The study found that CO emissions were 16-34 g/L during downhill driving, and range from 27-75 g/L during uphill driving. NOx emissions were 1.1-3.3 g/L for downhill driving and 3.8-5.3 g/L for uphill driving. NMHC emissions were over 3 times greater for downhill driving than uphill driving. For NOx and CO, distance-based emissions factors show greater dependence on load that fuel-based emission factors. Comparisons were also made between the tunnel emissions data versus average speed and CARB's EMFAC model. The EMFAC model seemed to overpredict NOx at all speeds, and predicted flat CO emissions versus speed, where the tunnel data showed increasing emissions versus vehicle speed. Researchers indicated that the raw data is still available for comparison with MOVES predictions.

The second study repeated the Caldecott tunnel measurements in 2004. The focus of this published work was on trends in benzene in the tunnel over the 1991-2004 time period. The study found that the reformulated gasoline regulations (Phase 2 RFS) implemented in 1996 significantly reduced benzene levels. In addition, the study found that fleet turnover also reduced benzene and non-methane organic carbon levels.

The third study was published in 2008 and was based on NOx and PM measurements made at the same tunnel in 2006. Light duty and medium/heavy duty NOx and PM were compared from 1997 and 2006, and were also compared to EMFAC rates. The EMFAC NOx emission rates were somewhat close to those measured in the tunnel for light duty PM and NOx, but EMFAC emission rates were significantly lower than the tunnel for medium/heavy-duty vehicles.

We think the Caldecott tunnel data could provide another important benchmark comparison for MOVES. There are two drawbacks with the data – one is that the most recent data is 2006, so it does not reflect many of the latest Tier 2/LEV2 technologies. The second is that tunnel data does not evaluate cold start emissions, since all vehicles are in the warmed-up condition. However, all validation data has some drawbacks, and we do not think the comparison should be eliminated from consideration based on these two drawbacks.

Caldecott emissions have been compared to EMFAC, but not to MOBILE or MOVES emissions (at least not in these 3 papers; these 3 papers predate MOVES but they do not predate MOBILE). We would propose to evaluate HC, CO, NOx, and PM, for both light and heavy-duty vehicles. We would obtain the emissions data from UCB, the vehicle registration data, and humidity, and speeds. We would then input the registration and humidity data into MOVES (speed and grade will be converted to a VSP bin or bins), evaluate MOVES emissions for these years and this location (California – San Francisco), and compare the results to the tunnel emissions (MOVES also estimates emission for states with California emission standards). Acquisition of ambient temperature data is considered unnecessary because MOVES does not correct running emissions for ambient temperature. Most comparisons would probably be in grams of emissions per liter of fuel. MOVES can provide this output with some effort. In addition to comparing fleet emissions in several different calendar years, we would also compare emissions between MOVES and Caldecott at different speeds (and VSP levels).

7.0 Recommendations

We are recommending two possible studies for further validation work. The first would be to compare MOVES emission rates to CARB's in-use surveillance LA-92 emissions rates by model year. This would be similar to EPA's comparison of MOVES to the KC data tested on LA92s. These data may have somewhat of a recruitment bias associated with them. We would make the comparisons between MOVES and the LA92 database with the LS92 database uncorrected for bias, and then evaluate the possibility of correcting the LA92 database for recruitment bias in the same manner that the KC data were corrected (lower and higher emitters were recruited separately and then re-weighted).

The second effort would be to compare MOVES to Caldecott tunnel emissions for a couple of recent calendar years (2004 and 2006, or 2001 and 2006).

We believe these two efforts would augment EPA's efforts to check MOVES exhaust emission rates.

References

1. "Evaluation of the U.S. EPA MOBILE6 Highway Vehicle Emission Factor Model", CRC Project E-64, March 2004.

2. Zhu, Hinds, Kim, and Sioutas, "Concentration and Size Distribution of Ultrafine Particles Near a Major Highway", JAWMA, 2002.

3. Zhu, Hinds, Kim, Shen, Sioutas, "Study of ultrafine particles near a major highway with heavy-duty diesel traffic", Atmospheric Environment 36 (2002) 4323-4335.

4. Gilroy, Harper, and Donaldson, "Urban Air Monitoring Strategy – Preliminary Results Using Aethalometer™ Carbon Measurements for the Seattle Metropolitan Area", Puget Sound Clear Air Agency.

5. Choi, Warila, and Koupal, "MOVES Validation", FACA MOVES Review Workgroup, April 26, 2010, U.S. EPA.

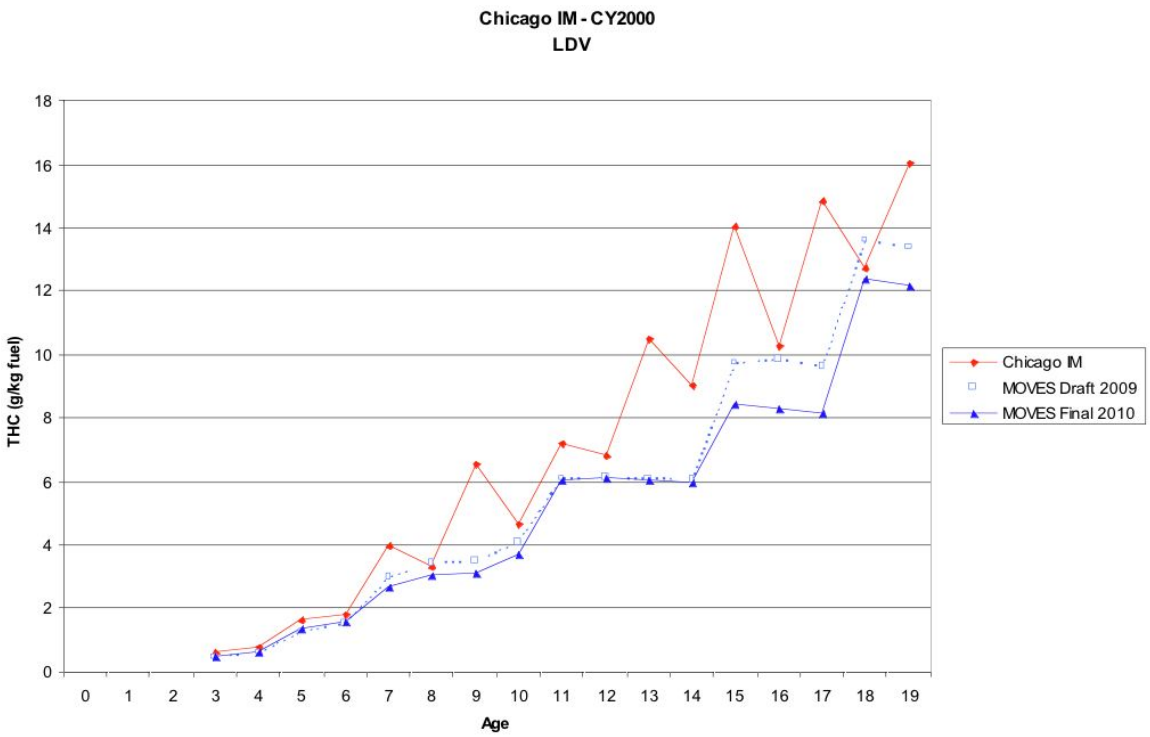
6. Kean, Harley and Kendall, "Effects of Vehicle Speed and Engine Load on Motor Vehicle Emissions", American Chemical Society, published 7/23/2003.

7. Harley, Hooper, and Kean, "Effects of Reformulated Gasoline and Motor Vehicle Fleet Turnover on Emissions and Ambient Concentrations of Benzene", Environmental Science and Technology, Vol. 40, No. 16, 2006.

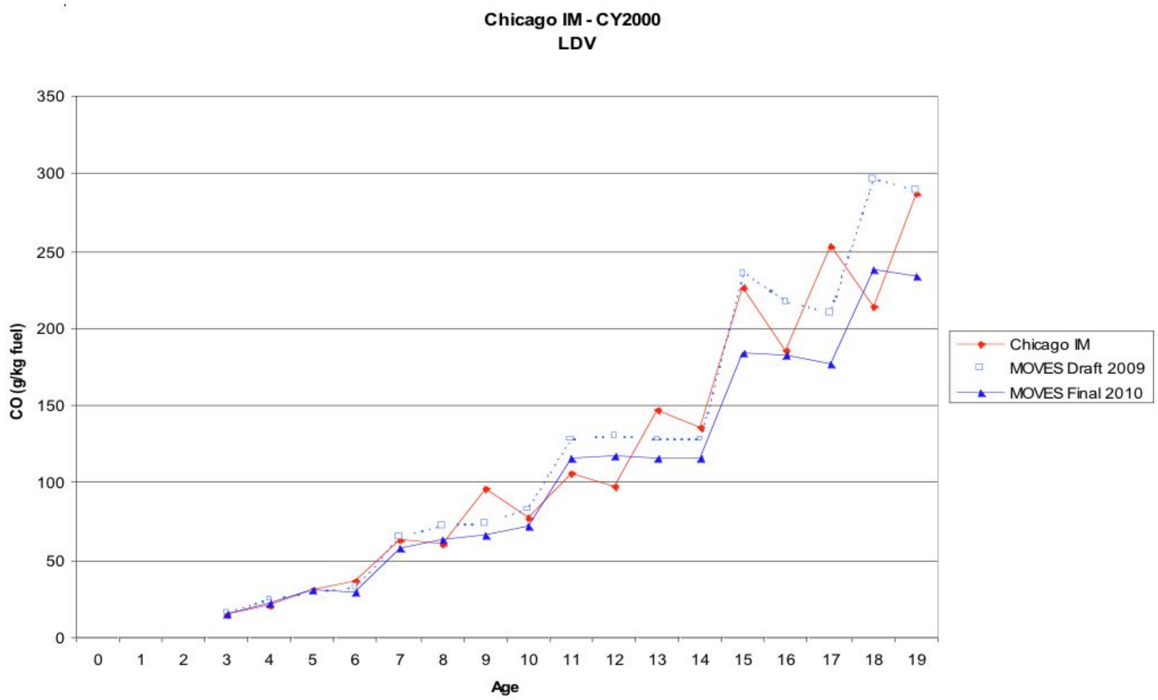
8. Weiss, McLaughlin, Harley, Lunden, Kirchstetter, Kean, Strawa, Stevenson, and Kendall, "Long-term changes in emissions of nitrogen oxides and particulate matter from on-road gasoline and diesel vehicles", Atmospheric Environment, 42 (2008), 220-232.

Attachment G-1
EPA Chicago I/M Comparison

Total Hydrocarbons (THC)

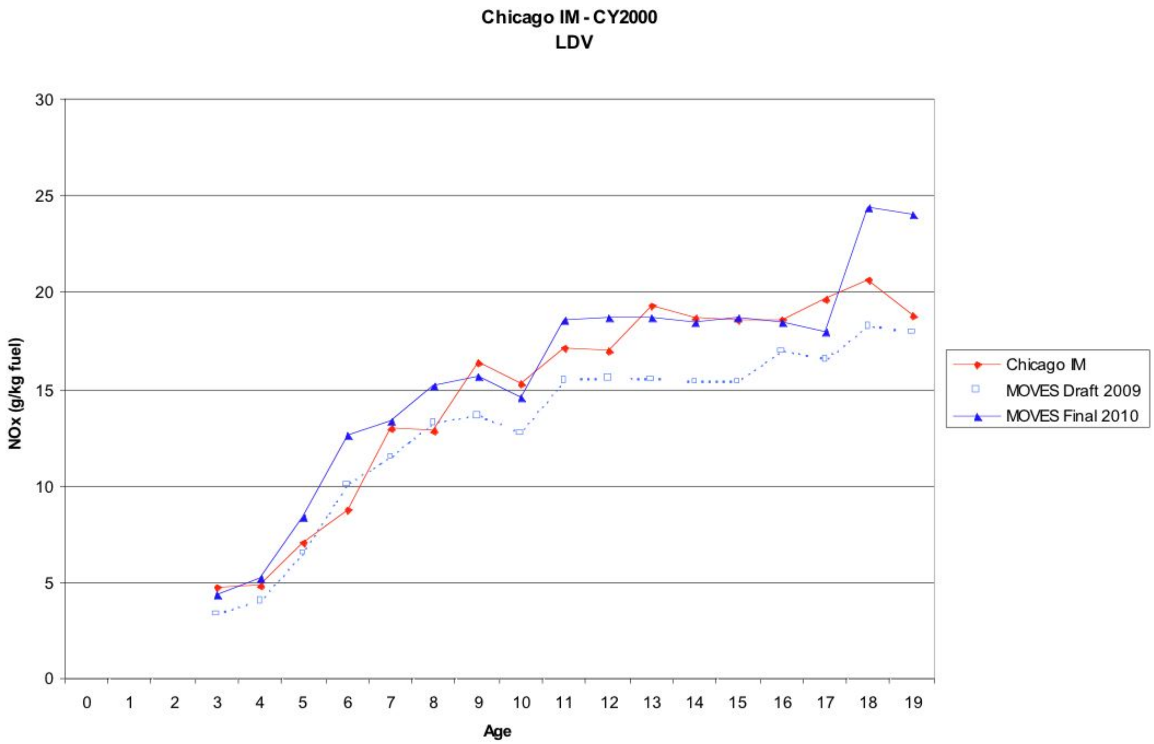


CO



Chicago IM - CY2000
LDV
CO (g/kg fuel)
Age
Chicago IM
MOVES Draft 2009
MOVES Final 2010

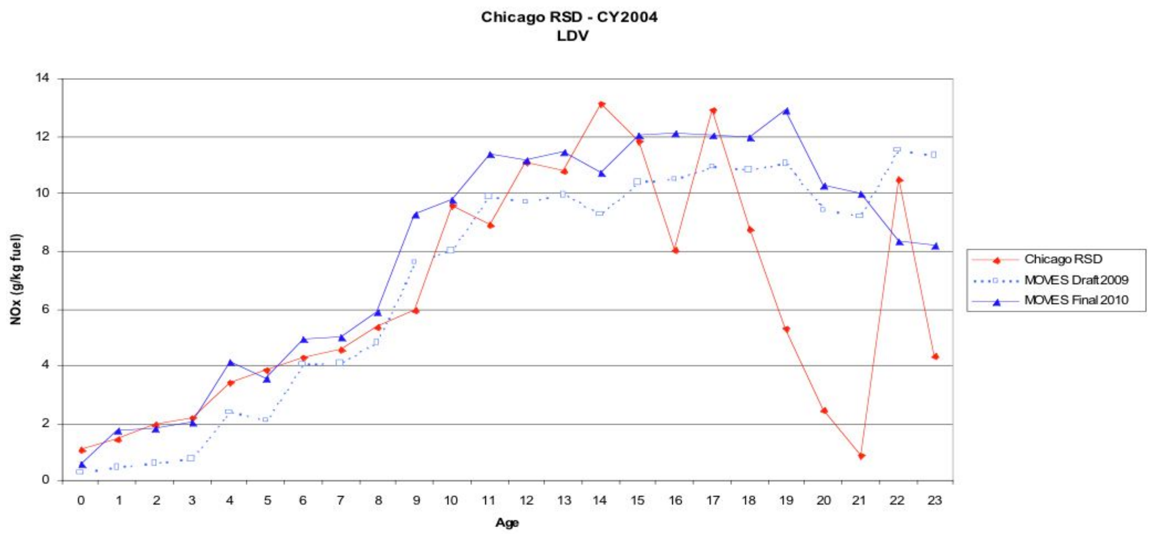
NOx



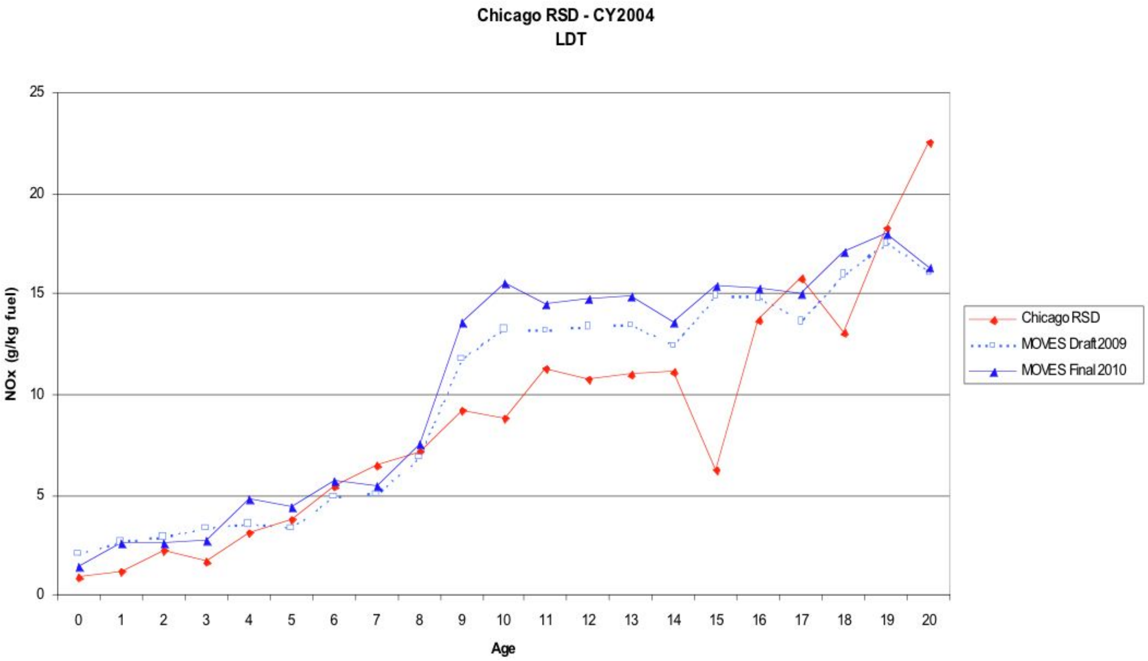
Chicago IM - CY2000
LDV
NOx (g/kg fuel)
30
25
20
15
10
5
0
0 1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19
Age
Chicago IM
MOVES Draft 2009
MOVES Final 2010

Attachment G-2
EPA's Comparison of Chicago NOx RSD Results to MOVES Emission Rates

NOx – Passenger Cars

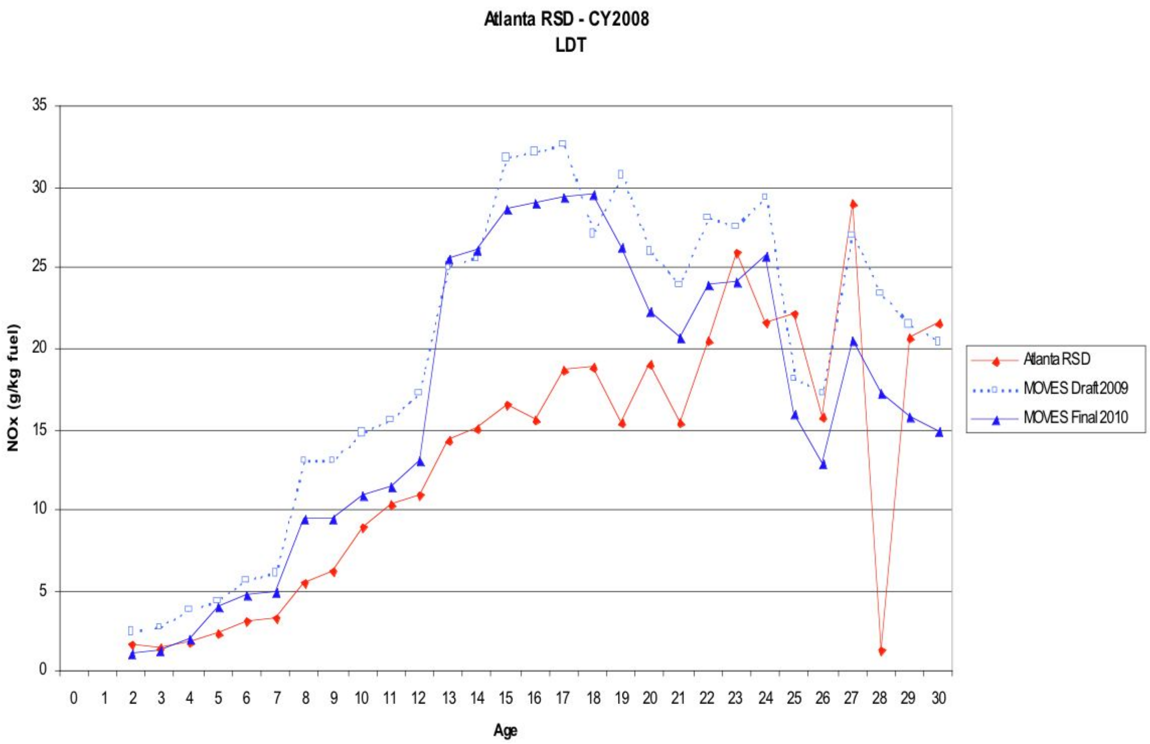


NOx- LDTs

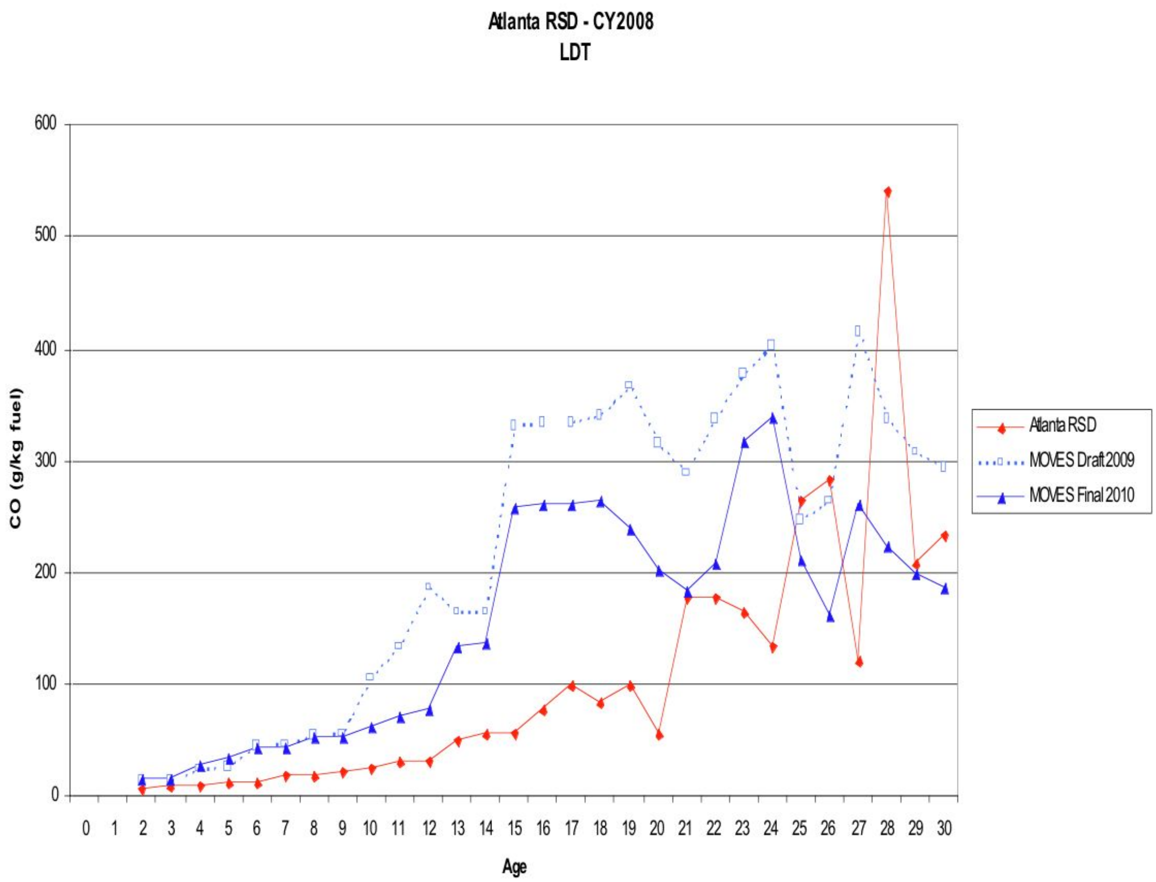

Chicago RSD - CY2004
LDT
25
20
15
10
5
0
NOx (g/kg fuel)
0
1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
20
Age
Chicago RSD
MOVES Draft 2009
MOVES Final 2010

Attachment G-3
EPA's Comparison of Georgia's RSD Measurements to MOVES Emissions

NOx - LDTs



CO-LDTs


Atlanta RSD - CY2008
LDT
600
500
400
300
200
100
0
CO (g/kg fuel)
0 1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 26 27 28 29 30
Age
Atlanta RSD
MOVES Draft 2009
MOVES Final 2010