Analysis of Particulate Matter Emissions from Light-Duty Gasoline Vehicles in Kansas City

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1 Executive Summary

This report summarizes and analyzes the particulate matter data from the Kansas City PM Characterization Study conducted in 2004 and 2005. It supplements the Eastern Research Group contractor report to EPA (2008), which describes the details of the study and provides some preliminary results. The Kansas City PM Characterization Study had many different goals and measurements thus producing a large array of data. This report analyzes only certain cross sections of this large data set within the scope of the subsequent modeling that is required to implement emission rates into an inventory model.

It should be first noted 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.

The first part of this report contains an evaluation of whether the data collected in the Kansas City Study is properly representative of the light duty vehicle fleet. Based on our review of the random sample of vehicle used, we conclude that the program was largely successful in properly capturing vehicles by household size, age, residence type, and household income that is demographically representative of the Kansas City metropolitan area.

We also evaluated whether high emitters are properly represented in the vehicle sample. Due to lack of data from other sources about fractions of PM high emitters in the vehicle fleet, we looked at high emitter rates for carbon monoxide and hydrocarbons among the vehicles in the Kansas City study compared to other studies based on remote sensing and I/M data. We concluded that CO and HC high emitter rates for older vehicles from the Kansas City Study were comparable with other sources; however there is less certainty whether the program captured the dirtier HC emissions from newer vehicles. We believe that the evidence implies that a proper representation of high emitters were captured in this study, however further analysis is required in order to prove this conclusively.

The remaining parts of the report describe in detail how the Kansas City data were analyzed and the results of that analysis. Key results are summarized in the following paragraphs.

For PM, 50% of the emissions came from 13% of the vehicles. It was also found that light trucks had slightly higher PM emissions than cars. These results are generally consistent with past studies.

The emission trends show a clear drop in emissions levels (for PM as well as HC, CO, NOx) with later model year vehicles. However, we have not yet determined whether the drop is due to technology changes (compliance with tighter standards) or whether it reflects varying levels of vehicle deterioration. More than likely, it is a combination of both of these factors, but quantifying this distinction will be reserved for a future publication.

Elemental or black carbon accounts for roughly 20% of the PM emissions, with the organic carbon accounting for the rest. It was found that elemental carbon roughly doubles during starts compared to hot running operation. However the fractions were not found to depend heavily on model year or temperature. These results may be important to studies that attempt to relate PM emission inventories to ambient PM concentrations.

The testing program measured a number of vehicles under different temperature conditions (summer and winter). The study results indicate that PM increases exponentially as temperature decreases so that for every 20˚F drop, PM doubles. This effect is more pronounced for cold starts.

Applying the KC data in a draft version of EPA's MOVES model results in an estimated average nationwide increase of light duty gasoline PM emissions of about 1.6 times compared to MOBILE6.2. Emissions are generally higher than MOBILE6.2 in winter months and lower than MOBILE6.2 in summer months. Overall annual emissions are expected to be significantly higher in areas with colder winters, even while summer emissions in those areas may be lower compared to MOBILE6.2. However, these comparisons do not fully account for all local conditions which may have an impact on a local inventory analysis. Because we have not yet completed our analysis of the relative impacts of deterioration and technology, we cannot yet predict how PM emissions in MOVES and MOBILE6.2 will compare for future years. In addition, PM from light-duty gasoline sources only form a fraction of the overall PM inventory, where stationary, non-road, diesel, road dust, wood-burning, and many other sources (natural and man-made) also play a significant role. However, even for light-duty gasoline PM, there is much work to be done before a final estimate of inventory impacts can be determined.

In the future, EPA will continue to investigate factors that contribute to or reduce the formation of PM. EPA has also observed the variability of measurements (even for back-to-back tests) and will continue to explore testing methodologies and procedures that may contribute to the non-repeatability of some measurements. It is also important to resolve the differences between Kansas City and the more numerous inspection and maintenance data. In the future, it would be important to examine trends in the speciated hydrocarbons and organic PM from the standpoint of toxic emissions and also to quantify the PM emissions attributable to oil consumption. This is likely to expand the scientific understanding of PM formation. For modeling purposes, it is important to understand the modal or load-based behavior of PM as well as determine the relative impacts of technology vs deterioration. Resolution of these topics will help us to update EPA's inventory model, MOVES, in order to better generate inventories from the past and into the future.

2 Introduction and Background

In 1998, the Coordinating Research Council conducted major studies on particulate emissions from in-use vehicles in its Project E-24. This work was done in San Antonio, TX (by Southwest Research Institute), Denver, CO (by a variety of groups including General Motors, the Colorado Department of Public Health and Environment, Colorado State University, EPA, and the Clean Air Vehicle Technology Center), and in California (by the Center for Environmental Research and Technology of the University of California at Riverside). This work, discussed elsewhere in this paper, involved testing of several hundred vehicles and was designed to obtain PM emission data on vehicles of different model years in different locations. This work, as had other projects, showed the presence of high-emitting vehicles which have a substantial impact on overall PM emissions from gasoline-fueled vehicles. This work was extremely valuable in providing initial data for emission factors on gasoline vehicles. Also, numerous source apportionment studies (discussed later in this paper) showed the large contribution to overall ambient PM from gasoline-fuel vehicles. However, none of these studies were designed to determine the frequency of high-emitting vehicles in the overall fleet or to accurately predict the emissions from the overall fleet.

A major limitation in previous emissions testing studies has been the way vehicles have been recruited. Most studies have not incorporated random sampling in the study design due to the high non-participation rate and the high costs associated with generating and implementing a random sampling plan. Therefore, few studies, and no studies evaluating light-duty PM emissions, can be used to represent the actual distribution of vehicle emissions in a large population. Most test programs select the first vehicle that meets the test program vehicle specifications usually based on model year, manufacturer, make, engine family or odometer. Gathering emission data from vehicles this way will provide what that particular vehicle is emitting in the laboratory but will not tell you whether that vehicle is representative of an entire group of similar vehicles. Therefore, modelers are always trying to determine how to apply these emission data to represent the vehicle fleet over various geographic scales. The National Research Council's report on modeling mobile-source emissions released in 2000 stated as part of their recommendations that EPA should:

- "Develop a program to enable more accurate determination of in-use emissions";
- "Begin a substantial research effort to characterize high exhaust … emitting vehicles";
- "Update their models with the best available data on PM emissions," and
- "Incorporate estimates of mobile-source toxic emissions into our models".

EPA's staff started developing and proposing a test plan in 2001 to foster interest from potential parties for this type of a test program. Through this effort, EPA was able to develop a consortium of sponsors by early 2003 that included: the Coordinating Research Council (CRC), the U.S. Department of Energy's (DOE) National Renewable Energy Laboratory (NREL), the U.S. Department of Transportation (DOT) Federal Highway Administration (FHWA), and the State and Territorial Air Pollution Program Administrators/Association of Local Air Pollution Control Officials (STAPPA/ALAPCO) through EPA's Emission Inventory Improvement Program (EIIP). EPA also established a cooperative research and development agreement

(CRADA) with the CRC that allowed for their sponsorship of this test program and provided EPA with technical expertise. An advisory committee, consisting of most of the sponsors, was established as an oversight committee for the test program. This committee advised EPA's staff on the proper testing methodologies, procedures and assisted in resolving any important issues that arose before, during and after the test program. EPA was responsible for managing the testing contract, the contractor and making the final technical decisions on how the test program would be conducted. After a competitive solicitation process and evaluation, EPA awarded the testing contract to Eastern Research Group (ERG).

Some earlier test programs conducted by EPA's Office of Research and Development and others indicated that temperature might influence the amount of PM emitted from light-duty vehicles, especially at colder temperatures (Stump et al., 2002; Cadle et al., 1999). To help address these temperature concerns, EPA split the study into two equal rounds of vehicle testing. One round of vehicles was tested during the summer months and a second round of vehicles in the winter months. This allowed for the widest temperature profile. To evaluate trends between the rounds, 41 vehicles from the summer test program also underwent testing in the winter phase. In addition, a reference vehicle with well-characterized emissions was tested weekly throughout the program. The study also conducted detailed gaseous and PM speciation, including toxics, in exhaust emissions in approximately 25 vehicles each round.

Finally, all the data gathered under this program is undergoing data validation and is being analyzed by EPA to help meet some of the National Research Council's recommendations in developing better modeling tools for mobile-source emissions. All this data will be stored in the EPA OTAQ's Mobile Source Observation Database (MSOD) and made available to the general public for their scientific review. Information pertaining to the specific study design and quality management plan can be found in the supporting documentation (EPA, 2008).

2.1 Emission and Fuel Regulations

The Clean Air Act of 1970 gave EPA the broad authority to regulate motor vehicle pollution, and the Agency has implemented multiple emission control policies to reduce emissions from passenger cars and the light trucks. Efforts by government and industry since 1970 have greatly reduced typical vehicle emissions. EPA has issued many successful control programs, the National Low-Emission Vehicle (NLEV), Reformulated Gasoline (RFG) and Tier 2 vehicle and gasoline sulfur standards are important recent examples that will continue to help reduce car and light-duty truck emissions into the near future. In that same period of time, however, the number of vehicles and the distance driven have steadily increased. This increase in travel by passenger cars and light trucks will continue to make motor vehicles significant contributors to air pollution inventories well in the future.

Exhaust emissions of particulate matter from gasoline powered motor vehicles and diesel powered vehicles have changed significantly over the past 25 years (Sawyer and Johnson, 1995; Cadle et al., 1999). These changes have resulted from reformulation of fuels especially the removal of lead additives, the wide application of exhaust gas treatment in gasoline-powered motor vehicles, and changes in engine design and operation. Particularly, as emission standards

reduced exhaust hydrocarbons with the introduction of catalysts in 1975, the organic component of exhaust PM also decreased. Lead, which was the major PM component in gasoline vehicle exhaust, was virtually eliminated with the introduction of unleaded gasoline mandated for the 1975 model year vehicles and the later phase-out of lead in all motor vehicle gasoline.

For some time, it has been well-known that in-use (gas-phase) emissions of gasoline-powered motor vehicles can be significantly higher than the standards to which they were certified when new. Age or odometer-related deterioration, engine or fuel system malfunction, broken (or removed) catalysts, and poor owner maintenance can result in higher emissions. As a result, prior emission studies have indicated that emissions from similar vehicles can span several orders of magnitude (Hildemann et al., 1991; Cadle et al., 1997; Sagebiel et al., 1997; Yanowitz et al., 2000). Because of evolving tailpipe emission standards, along with the wide variability of emissions between vehicles of the same class, well-defined average emissions profiles for the major classes of motor vehicles have been difficult to establish.

EPA has regulated HC, CO, and NOx exhaust emissions from gasoline vehicles since 1968. These regulations have become increasingly stringent with time including those for model year 1975 which resulted in introduction of catalysts, 1981 when the 3-way catalyst was widely used, and in later years such as for the 1994 model year [check year] when the Tier 1 standards were introduced. In contrast to EPA's strict regulations on diesel smoke (effective in the 1970 model year) and diesel PM (effective with the 1988 model year), EPA did not regulate PM emissions from gasoline vehicles until relatively recently. The first PM standards were part of the Tier 1 regulations, which phased in starting with the 1994 model year, through 1996. These standards were designed to provide more of an upper limit on PM emissions rather than to effect actual reductions. EPA further regulated gasoline PM as part of the Tier 2 regulations [CFR 65, 2000] ranging from 0.00 g/mile for Tier 2 bin 1 (for zero-emission vehicles) or 0.01 g/mile for Tier 2 bins 2 to6) to 0.08 g/mile for Tier 2 bin 10. These regulations were phased in from the 2004 to 2006 model year. Since new model year gasoline vehicles typically have lower PM emissions, emission testing is not generally done on these vehicles in the certification process.

The majority of exhaust PM emitted by catalyst-equipped motor vehicles is in the $PM_{2.5}$ size range (particulate matter mass with aerodynamic size of 2.5um or less, typically collected on a filter). Kleeman et al., (2000) have shown that gasoline and diesel fueled vehicles produce particles that are mostly less than 2.0 μm in diameter. Cadle et al., (1999) found that 91% of PM emitted by in-use gasoline vehicles in the Denver area was in the $PM_{2.5}$ size range, which increased to 97% for "smokers" (i.e., light-duty vehicles with visible smoke emitted from their tailpipes). Durbin et al.,(1999) found that 92% of the PM was smaller than 2.5 μm for smokers. The mass median diameter of the PM emitted by the gasoline vehicles sampled by Cadle et al., (1999) was about 0.12 μm, which increased to 0.18 μm for smokers.

Gasoline PM consists mostly of carbonaceous components including elemental carbon as well as those derived from organic constituents, generally higher molecular weight hydrocarbons. Gasoline PM consists largely of the higher molecular weight hydrocarbon/organic constituents which comprise approximately 60-99% of the total PM mass. Elemental carbon comprises a relatively small fraction of the PM mass. A figure of 26% of gasoline PM being element carbon has been found in the emission testing done for the Northern Front Range Air Quality Study

(Watson et al, 1998) . Vehicles that burn oil (frequently older vehicles) have a higher amount of PM mass and, generally, a higher fraction of high molecular weight organics (certainly those associated with oil combustion) in the PM. Vehicles running rich (e.g. under enrichment or during cold temperature starting) can emit higher levels of elemental carbon (black soot).

Gasoline PM also contains a small amount of sulfate, generally in the form of sulfuric acid, from oxidation of the SO_2 formed in combustion. A fraction of the SO_2 (roughly 10%**)** is oxidized over the catalyst to form sulfuric acid/sulfate. Non-catalyst vehicles have roughly 1% of the SO_2 in vehicle exhaust further oxidized to sulfuric acid/sulfates. The fraction of gasoline PM that is sulfate varies but has been prior to 2006 generally no more than 5% of total PM mass making it a relatively small constituent. About 100 vehicles were tested in an EPA baseline study on sulfate emissions which is a major basis for PM data on sulfates (Somers et al 1977). However, effective in 2006, EPA Tier 2 regulations resulted in the reduction of gasoline sulfur content to 30 ppm from an average of roughly 300 ppm. This reduction in gasoline sulfur content should result in gasoline PM containing much smaller amounts of sulfates.

2.2 Causes of Gasoline PM Emissions

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

Simply put, particulate matter primarily forms during combustion (and afterwards) when carbon-containing molecules condense into solid form. This PM is generally higher molecular weight hydrocarbon compounds, some of which originate in the fuel/oil and some of which are formed in combustion. Unlike diesel engines, elemental (molecular) carbon or soot is not very prevalent with gasoline engines but does form in larger quantities under relatively rich air:fuel ratios. The amount of elemental carbon in PM varies from vehicle to vehicle (and, even for a given vehicle, varies depending on operating conditions and state of repair). There are also other compounds in the fuel or engine oil such as trace levels of sulfur and phosphorus which, in combustion, form sulfates and phosphates, both of which are PM. The sulfur level in gasoline is now very low almost eliminating sulfate formation from gasoline sulfur content but motor oil contains significant sulfur (and phosphorus) compounds. Also, trace metal constituents in gasoline and oil forms PM in the combustion process as metallic oxides, sulfates, nitrates, or other compounds. Attrition products from the catalyst substrate and trace amounts of noble metals can also be emitted as PM. The catalyst attrition products are mechanically generated and are usually coarse particles (>2.5um). Exhaust PM as formed in the engine is generally very small in

size (possibly much of it is nuclei mode PM in the 0.05 micron or smaller size range). In the exhaust system including the muffler, some of the PM agglomerates and grows in size.

There is a wide assortment of technologies in vehicles that can affect PM formation. These technologies were mainly developed to control HC, CO and NOx emissions, but most have the side benefit of also reducing PM since reducing exhaust HC generally also reduces exhaust PM although not to the same extent. Older engines from the 1980s and earlier that deliver fuel through a carburetor typically have poorer fuel droplet quality, as well as poorer control of fuel air stoichiometry. These older vehicles are expected to produce more PM (on average) then their fuel injected engines that followed generally in the late 1980s and early 1990s. Among fuel injected engines, throttle body fuel injection (TBI) used in earlier engines with fuel injection typically has poorer fuel atomization quality and air:fuel ratio control than the port fuel injection (PFI) technology that supplanted it; thus, one might expect older model year fuel injection vehicles to have higher PM emissions (on average) than newer ones. Somewhat before the widespread use of fuel injection, closed loop control systems were developed along with oxygen sensors to improve the stoichiometric chemistry of combustion of catalytic conversion. These closed loop controls improved combustion as well as the effectiveness of the after-treatment system. The after-treatment system on most vehicles consists of a 3-way catalyst. The 3-way catalyst was designed for simultaneous control of hydrocarbons, carbon monoxide, and nitrogen oxides. Vehicles with 3-way catalysts would meet more stringent hydrocarbon and carbon monoxide emission standards while also meeting the first stringent nitrogen oxide standard. In oxidizing hydrocarbons, these systems are resulted in additional PM control. These systems were utilized on almost all gasoline-fueled vehicles beginning in the 1981 model year. On some model-year vehicles in the 1980s and a few more recently, a secondary air injection system was added between the engine and oxidation portion catalyst in order to supplement air to the oxidation reactions on the catalyst. These systems also helped oxidize PM also (though probably not to the extent that it oxidizes CO or HC). The deterioration of these technologies may affect PM and HC quite differently.

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

In addition to fuel, lubricating oil can also get into the combustion chamber via several pathways. Engine components, such as valves, valve seals, piston rings, and turbochargers can wear and deteriorate. During the intake stroke, the negative pressures (engine intake vacuum) can pull oil through the gaps left by these worn parts. In all gasoline automotive engines, the crankcase (where the oil splashes onto the engine components) is vented back into the combustion chamber through the intake manifold. This is known as Positive Crankcase

Ventilation (PCV), and is required in order to remove and burn the excess hydrocarbons in the hot crankcase. Unfortunately, it can also introduce PM precursors and oil into the engine combustion chamber. Because of the relatively small amount of oil consumption compared to the volume of gasoline burned in a vehicle, HC from oil is also small. However, organic PM from oil consumption can be quite significant because oil is a high molecular weight hydrocarbon, and more likely to be in uncombusted droplets. Therefore as vehicles get older, those that consume more oil, will probably have very different HC emissions behavior than PM, compared to when it was new. However, oil consumption can "poison" the catalyst material reducing the effectiveness of the catalyst at oxidizing HC.

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

Finally, the fuel itself may have properties that exacerbate PM formation. These are sensitive to the concentrations of the following: lead, sulfur, aromatics, and impurities. With the lower levels of lead and sulfur in fuels recently, the first two are probably less of a factor in the Kansas City program than aromatics would be.

3 Test Program and Report Goals

3.1 Test Program Goals

The program was designed to estimate average PM emissions for the fleet with attention to characterizing the contribution of high-emitting vehicles. A large body of previous work has demonstrated that a small fraction of motor vehicles emits a disproportionate fraction of particulate emissions. These “high-emitters” can exhibit higher emissions rates under certain specific operating or environmental conditions than do “normal emitters”. During the design of the study, we concluded that no reliable method of screening or identifying high-emitting vehicles short of actually measuring them was available. Thus, to use resources effectively, the goal was to employ a sampling design to provide a context for understanding and interpreting the results, including the frequency and contributions of both “normal” and “high-emitting” vehicles. To achieve this goal, the approach adopted was to over-sample from portions of the fleet where high-emitting vehicles were assumed to be most prevalent. This approach was implemented by sampling older vehicles in higher proportions than those vehicles existed in the fleet. During subsequent analysis, EPA will investigate some of the variables and factors (mileage, maintenance, age, environmental conditions, engine design or other emission device technologies, etc.) that might be an influence on these vehicles.

The study also had secondary objectives of advancing the development of analytical methods. One goal was to evaluate the capabilities of onboard portable emission measurement systems (PEMS) in the collection of mass-based emissions data. A similar goal was to evaluate the use of portable activity measurement system (PAMS) to monitor the activity of a vehicle (e.g. engine on/off, vehicle speed, date, time of day etc.) during normal driving. Usage and testing of these instruments will allow EPA to advance the development of instrumentation and associated protocols, and to demonstrate the utility of portable instrumentation in collection of real-world emissions and activity data. Unfortunately, PM was not measured on-road using the PEMS devices during this study, so discussion of PEMS results is beyond the scope of this paper.

Another goal was to investigate the use and measurement of other continuous particulate matter measurement devices. Three devices, (DustTrak, DataRAM and Quartz Crystal Microbalance (QCM)), employing different measurement techniques were used during this test program. Another area of investigation was to advance the development of devices that can measure PM on a continuous basis versus traditional gravimetric measurements. Different versions of the Quartz Crystal Microbalance (QCM) were evaluated in each round of vehicle testing. Some of the results from these measurements can be found in the EPA (and ERG report 2008).

3.2 EPA Analysis Report Goals

This report will discusses preliminary investigation and and analysis of different aspects of the PM data gathered during this program. In its review, EPA will begin by investigating if the study was able to achieve a random sample from the KCMA and if the non-respondents were significantly different from the positive respondents. This is followed by a discussion on EPA's analysis pertaining to the distribution of PM emissions within the KCMA vehicle fleet. It will include preliminary analysis on "high-emitters" or "smokers" and will compare results with previous programs that have tried to identify such groups of vehicles. Third, the report will investigate the trends in the PM data. Trends include temperature effects on PM emissions as well as correlations to model year, age, odometer, emission standards, other pollutants, etc., that might influence these findings. Finally, EPA will provide some analysis of PM modeling methodologies that could be used to incorporate these findings and present current inventory estimates. Future inventory estimates will be presented in another publication.

4 Site Selection

EPA started investigating medium to large metropolitan areas that would have a vehicle fleet representative of typical PM emission for the light-duty vehicles nationwide. It was determined early in our evaluation that the area should not have an Inspection and Maintenance (I/M) or Reformulated Gasoline (RFG) program. In an I/M program area, vehicles are regularly required to be tested and repaired to prevent their emissions from becoming excessive. Thus, I/M programs affect a vehicle's deterioration rate by requiring repairs and maintenance that otherwise might not normally be performed by the vehicle owner. This program changes the "natural" deterioration rate of vehicles. EPA also did not want metropolitan areas that had an RFG

program, since this program reduces the emissions from vehicles in the summer months. This would increase the complexity in trying to remove this emission effect for vehicles in areas that do not have this program. By not selecting an area with these programs, EPA would be evaluating vehicles under "natural" deterioration rates and with standard fuel properties. EPA also wanted to consider an area that would represent a typical metropolitan area that had a developed transportation structure with moderate driving patterns, socioeconomic demographics, and a broad range of seasonal temperatures. After careful review, EPA selected the centrally located Kansas City (MO/KS) metropolitan area to conduct this landmark PM study. The Kansas City Metropolitan Area (KCMA) for this study consisted of the following counties: Johnson County, KS; Leavenworth County, KS; Wyandotte County, KS; Clay County, MO; Cass County, MO; Jackson County, MO; and Platte County, MO.

There were two secondary factors that influenced EPA in its selection of Kansas City metropolitan area (KCMA). The first factor was a prior transportation study that had been conducted in the area recently that could potentially be used in support of vehicle recruitment. The second factor was the KCMA has had multiple emission test programs conducted using remote sensing devices (RSD) which could be used as an additional tool to possibly help in determining if the vehicles were recruited in a random basis.

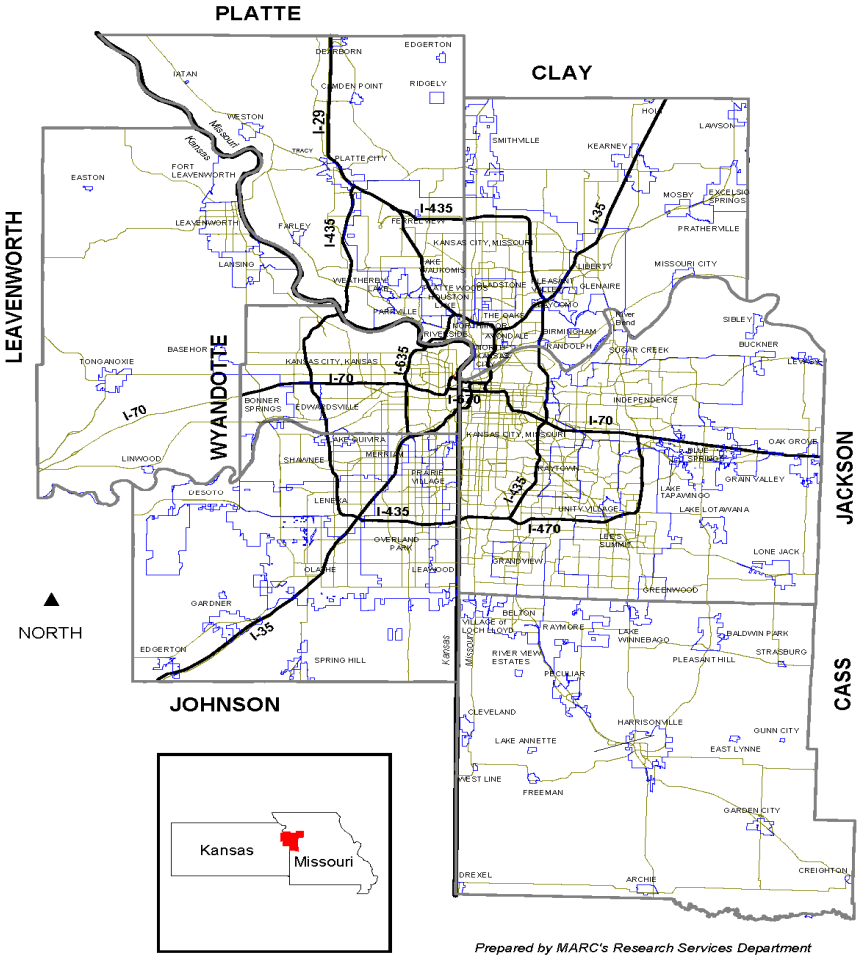


Figure 1, Kansas City Metropolitan Area (KCMA)

5 Vehicle Recruitment Process and Procedures

EPA and its contractor (ERG) designed a vehicle recruitment process to characterize the average PM emission factor from a representative sample of vehicles, with special attention paid to recruitment of initial non-responders, who were thought to be potential drivers of higher-emitting vehicles. Vehicles were divided into eight vehicle type strata: two vehicle types (passenger car and trucks) and four model year age groups (pre-1981, 1981-1990, 1991 – 1995 and 1996 and newer).

To achieve the random stratified vehicle recruitment goals, ERG used vehicles from a transportation study that was conducted by the Mid-America Regional Council (MARC) in 2004 (Kansas City Regional Household Travel Survey Final Report, 2004). The MARC 2004 Household study was a survey of KCMA regional households' travel activities. This study was used as the initial starting point for generating a stratified random sample. The survey procedures for establishing the cohort required the use of a random digit dial (RDD) telephone survey of households (HHs) in the seven county KCMA area from which a sample of vehicles were recruited for testing. The MARC data were compared to the latest U.S. Census data (2000) on key household characteristics including household size, vehicles, household workers, household income, residence type, household ownership and the types of vehicles owned.

Vehicles were randomly selected out of the cohort for testing. It became apparent that the MARC cohort had fewer than expected older vehicles available for recruitment and by the end of round 1 testing, the contractor had exhausted its pool of older vehicles (pre-1981 and 1981-1990) to recruit for testing. In order to address this problem, EPA and our contractor acquired both Kansas and Missouri Vehicle Registration databases which provided a large pool of vehicles that can be sampled and recruited for testing. These databases were used to draw a stratified random sample for recruiting the vehicles necessary to achieve the overall desired sampling strata targets. An incentive survey was also conducted to identify the appropriate levels of incentives necessary to ensure sufficient regional vehicles would be available for the emissions test program. Further analysis conducted on both the MARC and State vehicle registration database and the incentive survey can be found in the supporting documentation to this report (EPA, 2008).

6 Vehicle Testing

The KC study was conducted in three distinct phases: pilot testing, Round 1- summer testing and Round 2 - winter testing. A pilot study was conducted in May 2004. The primary goals of the pilot study were to establish a temporary testing facility in the Kansas City area and to finalize all testing methodologies, testing procedures and data handling procedures. The contractor and EPA staff also tested three EPA-provided "correlation" vehicles to compare EPA's National Vehicle and Fuel Emission Laboratory (NVFEL) dynamometer measurements with those obtained using the EPA's Office of Research and Development (ORD) portable Clayton dynamometer at the KC test facility. The details of the pilot study are discussed elsewhere (EPA, 2008- Appendix BB). The report identified procedural changes that were

incorporated into the Quality Management Plan (QMP) and Quality Assurance Project Plan (QAPP) that were approved before the start of Round 1 vehicle testing (EPA, 2008- Appendices AA & II).

Summer testing (Round 1) started in July 2004. During this round, approximately 261 vehicles were tested under summer conditions at the facility. The contractor had vehicles arrive at the test facility one to two days before actual dynamometer emission testing would occur. Vehicle owners were scheduled at specified times to ensure quick and individual attention was given to each owner. ERG's subcontractor, Nustats, was given the task of making sure that enough vehicles were on at the test facility to meet the test program goals and that vehicles were being recruited on a random basis.

Upon arrival, each vehicle first received a unique identification code for documentation tracking purposes, and was then inspected for test worthiness. Specific vehicle information, in the form of digital photographs, interview questionnaires, checklists, and hard copy data forms, was recorded for later input into the MSOD data table. A study was conducted by the contractor to evaluate what level of compensation was appropriate for the use of their vehicle and owner's time. Some vehicles were picked up and/or delivered to the owner's home or office to help facilitate the use of that vehicle in this study. The owner was also given the option to use one of our loaner vehicles while their vehicle was tested.

The vehicle was inspected with the owner present to document the condition of the vehicle. A more detailed inspection occurred later to ensure that the vehicle could safely be operated on the road and dynamometer. If repairs were required, the vehicle owner was notified and their permission was obtained before repairs were performed. If the repairs could not be performed on-site, the vehicle was taken to a local repair shop. Records of the repair, along with a brief narrative, were maintained. Typical repairs, for example, the replacement of brakes or part of the exhaust system were done for either safe operation of the vehicle or to allow for proper emission testing. Any defects or deficiencies in the vehicle condition that might affect exhaust emissions where not repaired by the contractor. All vehicles were maintained in what is called an "as received" or "as is" condition. During this inspection, a small sample of lubricating oil and, if possible, a fuel sample were taken and stored in a refrigerated unit for later possible analysis.

The next step in the process was to have the vehicle "conditioned" before being tested on the dynamometer. The main purpose for conditioning a vehicle was to make sure each vehicle was operated the same way for a certain period of time before being tested. This "conditioning" process should minimize the impact of the owner's driving pattern and habits on the subsequent emission measurements which might have occurred via the engine's adaptive learning capability. The conditioning route developed by the contractor and approved by EPA was about 45 minutes long and included high speed accelerations, driving at freeway speeds, and driving in stop-and-go traffic patterns through different roadway types: city, arterial and highway. The exact route driven has been documented in the contractor's report. During the conditioning process, a portable emission measurement system (PEMS) manufactured by Sensors Inc. was installed onto the vehicle to monitor emissions. The incorporation of a PEMS device onto the vehicle during its conditioning provided a couple of key quality assurance and control techniques. It first provided "real world" emission and activity data on vehicles that could be compared to

dynamometer data. It also allowed for the gathering of emission data on vehicles that could not be tested on the dynamometer. These excluded vehicles might be too long, too wide, all-time four-wheel drive, or the vehicle's condition (engine) prevented the vehicle from performing the drive cycle on the dynamometer. All of these vehicles would normally be excluded from laboratory testing. This program allowed the PEMS device to be installed on these vehicles and have them driven on the conditioning route, therefore some of these data may be included into the national emission profile analysis.

The PEMS unit used for the conditioning drive underwent a complete warm-up, zero and audit sequence to verify CO, CO_2, NO_x, and THC measurement accuracy. The contractor established procedures including check lists to ensure proper installation and calibrations were performed as necessary to bring the PEMS into proper calibration. The vehicle was driven on the conditioning route described above. The PEMS unit was then uninstalled, data analyzed, and the vehicle was left inside the facility to soak overnight at ambient temperatures. The PEMS data was analyzed to determine the testing order of vehicles for the next day with the cleanest being tested first and the dirtiest last. This was done to reduce any effect that a "high emitting" vehicle might have the on the emission testing equipment.

The following day, the vehicle was pushed onto the dynamometer, and secured. EPA used ORD's transportable Clayton Model CTE-50-0 twin-roll chassis dynamometer. Test inertia and horsepower settings for the dynamometer were determined from EPA I/M lookup tables. A Positive Displacement Pump-Constant Volume Sampling (PDP-CVS) system was used to dilute and transport the vehicle tailpipe exhaust to analyzers during the dynamometer test. Dilution tunnel air was kept constant at 47°C ± 5°C to prevent loss of volatile PM components. Procedures for conditioning the tunnel and analytical equipment to minimize any release of volatile compounds was reviewed during the pilot study and checked daily and weekly during both rounds of tests. The PEMS unit was installed directly onto the vehicle's tailpipe to monitor undiluted emissions, in tandem with the emissions measurements to be performed by the dynamometer bench.

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

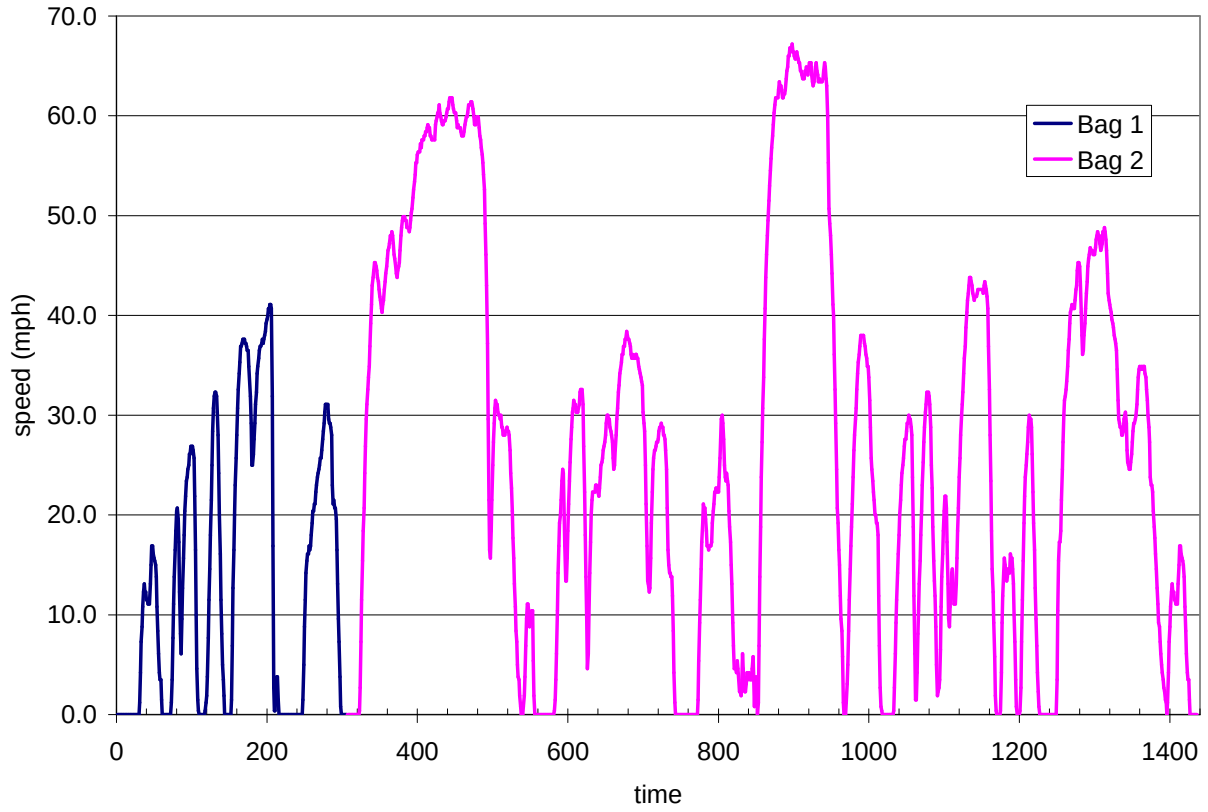


Figure 2. LA92 driving schedule with each of the bags identified.

In addition to the regulated gas pollutants measured via the CVS, continuous measurements of PM mass were taken using an EPA-supplied Booker Systems Model RPM-101 QCM manufactured by Sensor's Inc. and a Thermo-MIE Inc. DataRam 4000 Nephelometer. An estimate of black carbon was measured continuously with a DRI photoacoustic instrument and integrated samples were collected and analyzed by DRI for PM gravimetric mass, elements, elemental and organic carbon, ions, particulate and semi-volatile organic compounds, and volatile organic air toxics. All sampling lines were heated and maintained at 47°C ± 2°C. The samples were extracted from the dilution tunnel through a low particulate loss 2.5 μm cutpoint pre-classifier. Further, details and a schematic of the sampling instrumentation can be found Figure 3 and in EPA, (2008).

At the conclusion of vehicle testing, the vehicle was disconnected from the PEMS and dynamometer sampling systems and removed from the dynamometer. The vehicle was released to the owner after the tests were reviewed by the contractor to confirm a valid emission test had occurred. The CVS tunnel blowers were kept on and monitored by numerous analytical devices to ensure that the tunnel had stabilized (No "off-gasing" of volatile organic compounds). Once, the tunnel had stabilized and proper analytical test procedures completed, the next vehicle was pushed onto the dynamometer and tested.

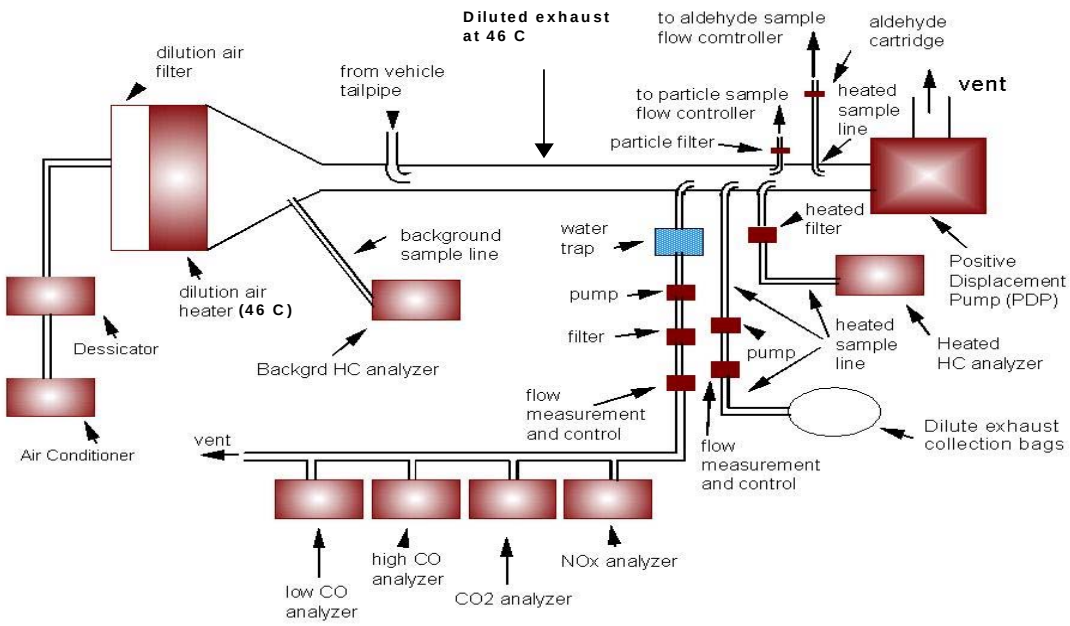


Figure 3. CVS Sampling System Schematic

Winter testing (Round 2) started in January 2005. During this round, approximately 278 vehicles were tested under winter conditions at the facility. Approximately 43 vehicles tested during Round 1 were re-tested in Round 2 to estimate the effect of the ambient temperature on exhaust emissions. The same testing procedures were used except for a couple of small changes as noted in the QAPP documentation in the KC EPA report (2008). The contractor provided to EPA their review of all data verification and validations that were performed throughout this test program. ERG recorded and reported all data and did not remove or eliminated any data. All data exceptions were noted within their report. Further details on these areas are documented in the EPA report on the study (2008).

7 Aggregate PM Results

Many of the general data verification and validation results from the Kansas City program are included in the EPA report (2007). This section will report analysis that was either reviewed or conducted by EPA and is divided into two main areas; analysis on recruiting a random stratified vehicle sample and on the KCMA's vehicle emissions.

7.1 Random Stratified Vehicle Recruitment Results

At the outset, the contractor analyzed the household cohort, to assess its representativeness with respect to households within the KMCA. To achieve this goal, the cohort was compared to

results from Census 2000 on the basis of a series of demographic variables, including household size, income levels, home owner's age and numbers of vehicles owned. Table 1 shows that the cohort distributions were generally similar to those from the Census. The largest difference observed with respect to residence type, in which the prevalence of vehicle owners living in single-family dwellings was higher for the cohort than for Census, with the reverse being true for other types. This difference is explainable in terms of the fact that the cohort was constructed using random-digit dialing (RDD) and therefore related to patterns of telephone ownership. In general, the availability of listed telephone numbers is associated with length of tenure in residence, which is in turn associated with home ownership and single-family dwellings.

It is also of interest to compare the demographic characteristics of vehicle owners participating in the study to those for the cohort and the Census. Table 1 also presents demographic distributions for participants in Summer and Winter Phases. In these cases, some differences can be seen between the study participants and the cohort (and Census). One obvious difference is that only households owning vehicles could be included, explaining the absence of households without vehicles. With respect to numbers of household vehicles, households owning more vehicles participated at higher rates than in the general population. With respect to income levels, households at the low and high ends of the income distribution participated at lower rates than in the general population. In contrast, households in the middle income ranges participated at slightly higher rates than in the population. Overall, the fractions of households in the cohort (round 1+2) is notably lower in the cohort than in the Census for multiple-family residences, residences occupied by a single individual, residences with 0-1 vehicles and households with < $25,000 annual income. In Table 1, each of these categories is tabulated independently of the others. We suspect, however, that there is substantial overlap among the categories, i.e., that single individuals residing in multiple family housing often have lower incomes and thus may not own vehicles. Follow-up analysis would be useful to investigate the extent to which these categories overlap.

The contractor also analyzed the cohort with respect to its geographic distribution within the eight-county Kansas-City metropolitan statistical area (MSA), as shown in Table 2. As with the demographic makeup of the cohort, its geographic distribution tracks that of the Census, with some minor differences. Relative to participants, refusals were somewhat more likely to come from Cass, Clay or Jackson counties, in Missouri. Overall, differences in the geographic distribution of participants and refusals are not striking.

Table 1 Demographic Comparison of MARC Survey Cohort to Census 2000

Characteristic	**Survey Cohort (%) (n=4,001)**	**EPA Round 1**	**EPA Round 2[(1)]**	**EPA Rounds 1 + 2 [(1)]**	**Census 2000 (%)**
Household Size					
1 individual	26.8	16.8	7.06	10.8	27.4
2 individuals	33.3	32.8	36.47	34.9	33.0
3 individuals	16.0	14.4	20.0	18.1	16.2
4 or more individuals	23.9	36.0	36.47	36.1	23.4
Total	100.0				100.0
No. Vehicles Owned					
0 vehicles	5.8	0.0	0.0	0.0	7.4
1 vehicle	32.9	12.8	10.6	12.1	33.9
2 vehicles	42.7	44.8	54.1	49.4	41.7
3 or more vehicles	18.6	42.4	35.3	38.6	17.0
Total	100.0				100.0
Annual Income Level					
< $15,000	9.9	4.8	3.6	4.2	12.2
$15,000 to < $25,000	10.2	10.4	7.1	7.8	11.3
$25,000 to < $50,000	30.2	36.8	31.8	34.3	30.1
$50,000 to < $100,000	35.9	37.6	40.0	40.4	33.6
$100,000 or more	13.8	8.8	12.9	10.8	12.8
No Income Reported		1.6	4.6	2.4	
Total	100.0				100.0
Residence Type					
Single family	76.8	87.2	91.8	88.0	69.0
All other	23.2	12.8	8.2	12.0	31.0
Total	100.0	100.0	100.0	100.0	100.0
Respondent Age					
< 20 years	29.6				29.1
20 – 24 years	4.3				6.1
25 – 54 years	43.3				45.3
55 – 64 years	9.9				8.2
65 or more years	12.8				11.3
Total	100.0				100.0

Source: Kansas City PM Characterization Study, Final Report. Eastern Research Group, ERG 0133.18.007.001. October 27, 2006 (Section 3.2, page 3-6).

(1) This only includes MARC cohort vehicles and does not include vehicle recruited from KS and MO vehicle registration lists

Table 2 Comparison of MARC Survey Cohort to Census 2000 in terms of County of Residence

County	Survey Cohort (%) (n=4,001)	Round 1 Participants	Round 1 Refusers	Census 2000 (%)
Kansas				
Johnson	26.6	25.6	22.2	26.1
Leavenworth	3.5	6.4	2.2	3.3
Wyandotte	9.1	10.4	9.5	8.9
Missouri				
Cass	4.6	9.6	14.0	4.9
Clay	11.1	4.8	6.0	12.3
Jackson	40.6	40.0	43.2	39.9
Platte	4.5	3.2	2.9	4.6
Total	100.0			100.00

Source: Kansas City PM Characterization Study, Final Report. Eastern Research Group, ERG 0133.18.007.001. October 27, 2006 (Section 3.2, page 3-7.

With respect to emissions, it is also of interest to examine the characteristics of vehicles owned by participants in relation to the MARC cohort and the registration database.

Table 3 shows the frequencies of vehicles within the vehicle-type model-year-group strata used for the study. As with households, distributions of vehicle age and type are generally similar between the registration database and the cohort, with the exception of 1996 and later MY cars, for which the cohort has a noticeably higher fraction. At the same time, the frequencies for the MARC survey cohort are slightly lower than the registration database in the remaining seven strata.

Table 3 Distribution of Vehicle Ownership within Vehicle-Type/MY Strata

stratum	MYG	Vehicle Type	KC DMV (%)	MARC Cohort	Round 1 (Planned) n=	Round 1 (Actual)	Round 2 (Planned)	Round 2 (Actual)
1	Pre 1981	Truck	1.2	0.9	6.4	0.8	4.2	3.8
2	1981-90	Truck	3.7	2.3	10.4	8.0	15.7	12.3
3	1991-95	Truck	4.4	3.1	10.4	6.9	12.7	13.2
4	1996 +	Truck	11.6	9.2	15.6	14.9	19.9	21.3
5	Pre 1981	Car	2.2	1.5	6.4	2.3	6.4	6.0
6	1981-90	Car	10.7	8.9	20.4	18.8	14.4	15.3
7	1991-95	Car	18.0	17.4	13.6	14.9	15.3	15.7
8	1996 +	Car	48.2	56.6	16.8	33.3	11.4	12.3
Total			100.0	100.0	100.0	100.0	100.0	100.0

Table 3 also presents sample allocation among strata for Rounds 1 and 2. There are some differences. The planned distribution for Round 1 reflects the over-sampling of older vehicles relative to newer ones, whereas the actual distribution shows the difficulty experienced in locating and recruiting older vehicles. The closer correspondence between the planned and actual differences for Round 2 reflects the modification of the design and emphasis to compensate for low recruitment of old vehicles in Round 1.

Finally, two basic aspects of driver behavior were examined. The first was the mode of transport for travel to work. Options covered included automobile (as driver or passenger) or by other

modes, such as walking or bicycling. In the Census and cohort profiles a substantial majority of respondents get to work by automobile (91-94%), while the remaining respondents travel by other modes or work at home. For the current study more than 99% of respondents traveled to work by automobile. These results are not surprising, given the requirement that participants had to be able to drive their vehicles up to 30 miles to the testing site.

The second characteristic examined was commute time over a range of <5 to >45 minutes. Commute-time distributions were similar between the Census, the cohort and this study with two exceptions. Round 1 participants reported a higher frequency of 10-14 minute trips and a lower frequency of 20-29 minute trips, whereas Round 2 participants reported low frequencies of 10-14 minute trips and higher frequencies of 15-19 minute trips.

The material presented up to this point suggests that sample pools of households and vehicles used for vehicle recruitment were generally representative of the Kansas-City fleet. This examination is necessary but not sufficient to demonstrate that the emissions measurements are themselves representative of those from the study area. An important consideration is the extent to which potential respondents might have avoided participation due to their own perceptions about the emissions of their vehicles. Due to the fact that Kansas City has no I/M program, we might presume at the outset that residents have less motivation to be conscious of their emissions or that they would be more likely to avoid the study than they might in an I/M area, where owners are required to repair their high emitting vehicles. On the other hand, the burden of participation was high for participants, who were required to give up their vehicles and to modify their schedules to make two round trips to the study site. It is thus plausible that much of the reluctance of many respondents to participate can be attributed to perceived or actual inconvenience.

Assuming that the refusal to participate was motivated largely by the respondents' own perception of their vehicles emissions, we would expect that the vehicles owned by refusals would emit more on average than those of regular participants. Fortunately, the results from Round 1 provide an opportunity to examine this assumption. During round 1, the contractor persuaded approximately 50 respondents who had initially refused to change their minds (Table 4). Thus, emissions measurements for the 'converted refusals' can be compared those for respondents who agreed at the outset to participate.

Table 4 Numbers of Participants and Converted Refusals in Round 1

Stratum	Initial participants	Converted refusals	Total	Population Weight
Truck, pre 1981	2	0	2	6.5
Truck, 1981-1990	11	5	16	5.9
Truck, 1991-1995	18	2	20	10.5
Truck, 1996 +	35	5	40	23.0
Car, pre 1981	6	0	6	4.2
Car, 1981-1990	34	12	46	6.7
Car, 1991-1995	29	14	43	12.9
Car, 1996 +	77	17	94	22.7
Total	212	55	267	

Figure 4 and Figure 5 show aggregate PM emissions for both groups plotted against model year on both linear and logarithmic scales. The plots show that emissions for both groups overlap, with no clear differences between them. The visual inspection was confirmed by two quantitative analyses.

First, we performed common log transformations to normalize the distribution of emissions with respect to model year, and then regressed PM against MY for both respondent groups, to test for differences in intercepts and slopes between them. Restricting the analysis to the six strata containing both participants and conversions, a model was fit as a regression on "year," calculated as MY-1980 and treated as a continuous variable, while attempting to fit separate intercepts and slopes for respondent groups and separate intercepts for vehicle type. As expected based on the graphical presentation, results showed no significant differences in either slope or intercept between the respondent groups, as the *p*-values for both parameters "refusal" and "year × refusal" are well above the typical a level of 5%, and even above a less stringent level of 10% (Table 5). However, the model did suggest separate intercepts for cars and trucks. In addition, models were fit with and without population weights for each stratum, which did not substantially affect the results; weighted results are presented in Table 5.

Table 5 Regression Model Parameters for aggregate ln(PM) Emissions vs. Model Year and Participant Group, incorporating Population weights[1]

Parameter	Estimate	Standard error	*t*-value	Pr > | *t* |
Intercept	1.7245	0.1438	11.99	<0.0001
Year[2]	-0.06234	0.007479	-8.34	<0.0001
Refusal[3]	0.1969	0.2633	0.75	0.4554
Year×Refusal	-0.008411	0.01567	-0.54	0.5920
Vehicle Type[4]	-0.3049	0.06239	-4.89	<0.0001

[1]Dependent variable = log(PM), mg/mile, on LA92 cycle.
[2]Calculated as model year – 1980, i.e., the intercept is set in MY 1980.
[3]An indicator variable, =0 for participants , 1 for conversions.
[4] An indicator variable, = 0 for trucks, 1 for cars.

Due to the highly skewed distributions of untransformed data, it is also useful to perform alternative tests that avoid reliance on the assumptions of normality and stable variance underlying classical regression techniques. To address this need, we performed Wilcoxon rank-sum tests to detect differences in the locations of distributions of emissions for the conversion and participant groups. We performed these tests independently in the 1981-1990, 1991-1995 and 1996+ model-year groups for cars. The specific test was the Mann-Whitney normal approximation to the *Z* statistic, which is acceptable because the numbers of participants and conversions were greater than 10 in each respondent group. As with the parametric regression, the non-parametric tests showed no significant differences in aggregate PM emissions between the participant and conversion groups.

Table 6 Summary statistics for Wilcoxon Rank-Sum Tests for Participants and Converted Refusals in Round 1

Stratum	*Z* statistic	Two-sided Pr > | *Z* |
Car, 1981 – 1990	0.8211	0.4116
Car, 1991 – 1995	0.4304	0.6669
Car, 1996 +	0.9012	0.3765

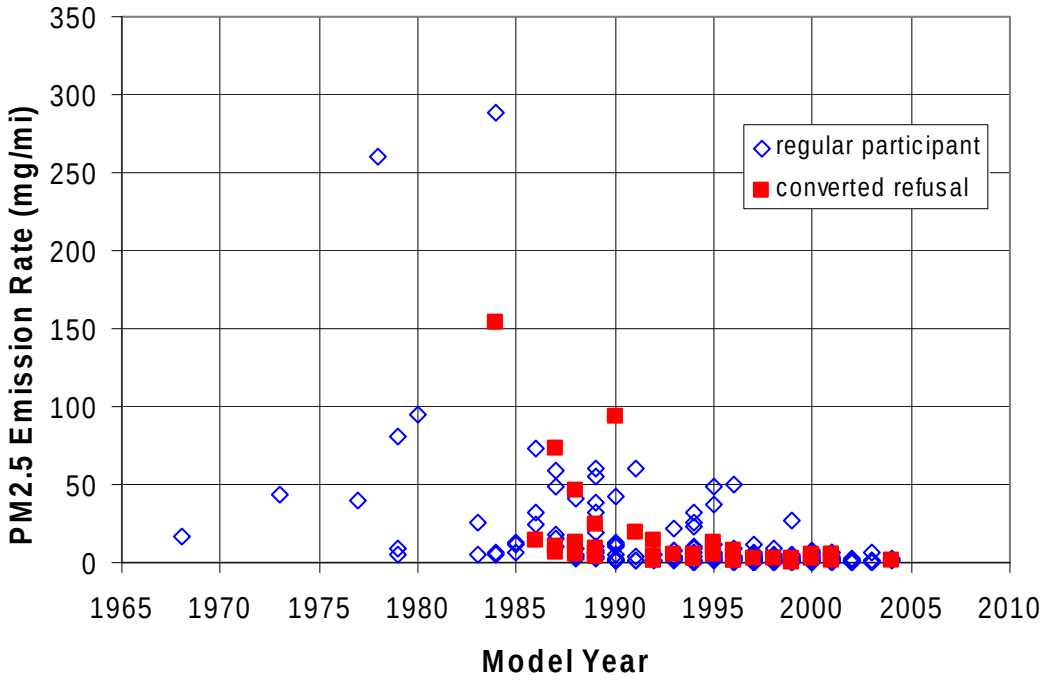


Figure 4. Aggregate PM emissions for Participants and Converted Refusals in Round 1 (Summer).

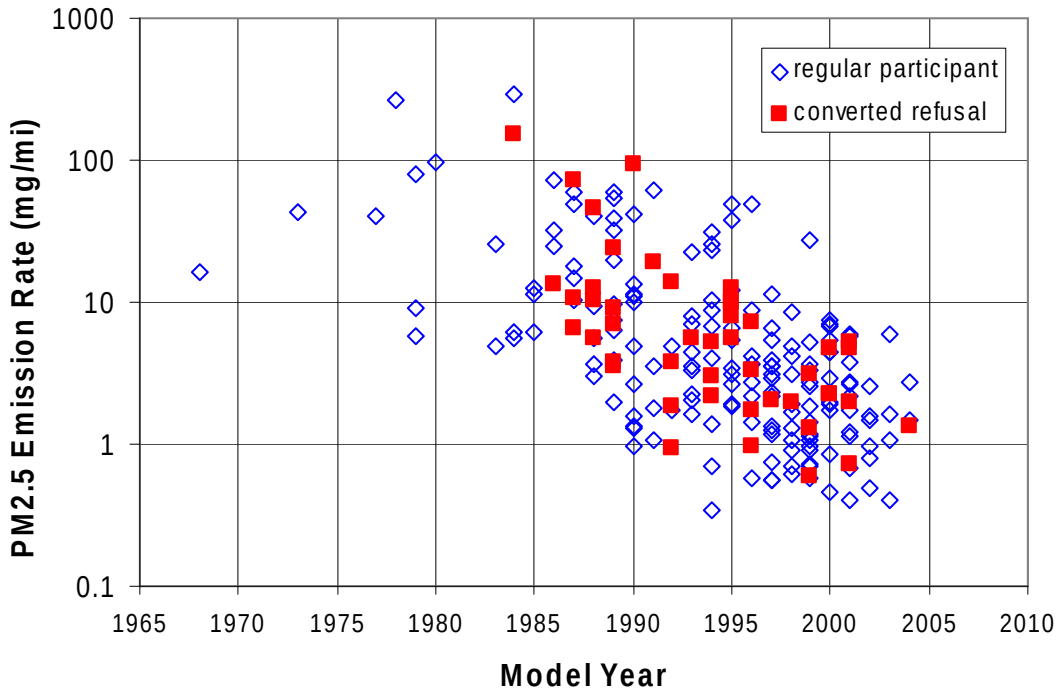


Figure 5. Aggregated PM Emissions in Round 1 (logarithmic scale).

These tests failed to disprove an assumption of no significant differences in PM emissions between the participants and conversions in Round 1. While not definitively resolving the question of the representativeness of the Round 1 measurements, these results allow a tentative conclusion that non-response was motivated largely by reasons other than perceived high emissions. To the extent that this conclusion holds, it allows a further presumption that non-response does not substantially affect the representativeness of the Round 1 results.

7.2 Emission Results

We present the results from the testing in Kansas City in several stages. First we briefly examine the test-to-test variability observed by comparing the back-to-back measurements. Then we show the general trends including: averages broken up by various testing regimes, average emissions of smoking vehicles, and a comparison to past studies. Next we go into further detail on the temperature dependence of PM emissions. These temperature results are compared to past studies and also lay the groundwork for subsequent modeling. Then, we look at trends in the aggregate LA92 emissions by model year and age as well as correlations with other pollutants. In addition to furthering our understanding of PM formation from gasoline engines, this analysis will also help inform the development of new PM models for inventory analyses. Finally, in comparison to the aggregate results, we repeat some of the above analysis, but with the separate bag by bag data from the LA92.

During both summer and winter phases, repeat tests were performed on selected vehicles, with 15 repeats during round 1 and nine during round 2. In this case “back-to-back” means that tests were performed within several days of each other, not on the same day or at the same time. As can be seen in Table 7, the composite PM results show a high degree of variability for some vehicles, particularly during the winter. However, the direction and magnitude of differences do not appear to show consistent patterns with regard to ambient temperature or humidity.

Absolute differences in composite results ranged from -2.9 to 5.6 mg/mi in summer and -183 to 0.8 mg/mi in winter. Corresponding percent differences were -92 to 241% in summer and -97 to 25% in winter. The most striking differences in absolute terms (-183 and -132 mg/mi) occurred in winter (duplicates 5 and 6). The temperature during both tests was between 35 and 45 °F, showing that a large temperature differential does not appear to be responsible. Although not presented here, the repeatability of repeat measurements for gaseous pollutants is substantially higher than for particulate measurement. We present some hypotheses for extent of the test to test variability in a partner paper (Nam et al, in publication).

These results are valuable in that they give insight into the degree of measurement variability for individual vehicles inherent in gravimetric measurement, which may be large enough to merit consideration in analysis and use of the data.

Table 7. Composite Gravimetric Particulate Matter Results for Repeat Back-to-Back Tests during Rounds 1 and 2.

Round	Duplicate	Run #		Temperature (°F)		Relative Humidity (%)		Gravimetric PM (mg/mi)	
		Test 1	*Test 2*	*Test 1*	*Test 2*	*Test 1*	*Test 2*	*Test 1*	*Test 2*
1	5	84060	84062	90.3	83.9	63.8	80.3	0.994	1.236
1	7	84104	84109	78.1	83	55.7	59.1	1.556	1.069
1	4	84110	84115	84.7	84	56.8	57.1	10.006	7.597
1	2	84111	84116	88.4	87.6	51.6	49.5	4.068	1.787
1	12	84120	84123	93.6	85.1	44.4	56.2	5.467	3.428
1	8	84132	84137	80.3	76.9	38.3	34.9	4.334	4.413
1	11	84151	84156	69.4	65.4	46.8	59.4	1.452	2.022
1	3	84166	84169	71.5	70.7	39.2	44	0.604	0.043
1	9	84175	84180	70.1	74.6	47.8	44.4	3.644	9.257
1	6	84198	84200	65.8	65.2	63.2	68.8	2.072	3.687
1	1	84258	84262	70.8	71.8	1.5	50.2	4.798	3.321
1	18	84308	84312	77.4	74.3	42.3	61.1	3.914	0.958
1	16	84321	84328	80.5	82	33.3	40.7	1.366	0.391
1	10	84332	84341	76.8	76	39.7	54.3	2.24	7.647
1	17	84345	84350	74.1	70.7	42.5	49.5	0.327	0.444
2	3	84437	84442	60.1	40.7	47	59.6	2.078	2.535
2	9	84449	84451	25.8	37.6	39	61.3	10.153	4.62
2	5	84465	84468	37.9	36.6	56	46.3	188.706	5.223
2	2	84482	84484	39	40.7	70.1	56.8	14.104	8.658
2	10	84485	84490	38.9	36.9	59.2	55	20.047	3.842
2	1	84537	84543	40.9	49.5	63.8	52.5	3.178	3.982
2	8	84541	84542	49.8	44.6	68.3	61.3	6.332	4.908
2	6	84627	84632	47	45	34.9	30.4	232.116	99.412
2	4	84690	84695	55.2	42.4	38.3	67.6	2.005	2.119

A reporting of (certain) average emissions from a test program is more meaningful if the sample in the test program is corrected, or weighted, to the population of vehicles registered in that region. The fraction of vehicles of a particular model year recruited in this study may not necessarily correspond to the fraction of vehicles of the same model year in the general fleet in the region. Since the fleet figures are available to us, we present "population weighted" statistics where appropriate. These should be distinguished from "VMT weighted" rates, which may differ since older vehicles (while they exist) tend to be driven less, which would affect their relative emissions impact in a region.

The population weighted average emissions are 12 mg/mi, though the distribution has a high degree of skew. The maximum emission rate is 417 mg/mi from a 1973 truck measured at 62°F. The maximum emissions for a car is 260 mg/mi from a 1978 vehicle measured at 79°F. The average test temperature and (population weighted) emissions in the summer is 76°F and 8 mg/mi, respectively, while in the winter it is 45°F and 15 mg/mi.

The population weighted average emissions from cars is 11 mg/mi and for trucks it is 13 mg/mi. These results support the expected trends that trucks tend to have higher PM emissions than cars. There are two likely reasons why light truck emissions could be higher than those of passenger cars. Trucks tend to have larger engine displacements and consume more fuel per mile. Additionally, if PM emissions loosely follow HC emissions, the standards for trucks have lagged

those of cars. Also, emissions in the winter tend to be higher than they are in the summer. We will explore this in greater detail in a later section.

The following figures show the distribution of emissions in the summer and winter. In this dataset, 296 cars and 182 trucks had viable PM data for a total of 478 tests (though some of these vehicles had repeat tests). From a population weighted standpoint, 50% of the total emissions is coming from 13% of the vehicles. Only 1 of the 14 highest emitters (defined here as emitting 50 mg/mi or greater) in the summer was certified to Tier 1 standards (the rest were Tier 0), and of the 14, 6 were trucks while 8 were cars.

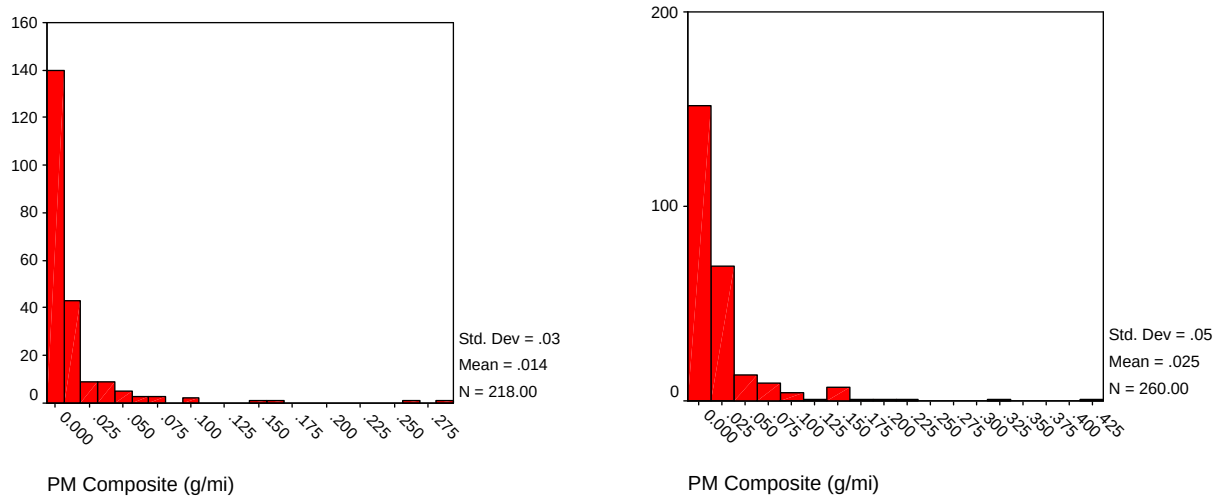


Figure 6. Histogram of Gravimetric PM emissions composited over the LA92 cycle in the summer (left) and winter (right).

Some of the vehicles were observed to have visibly smoking exhaust during idle. Of the 478 vehicles tested, 406 were observed for smoke and categorized into four distinct smoke intensity levels: No smoke (or normal), Low, Medium, and High (N, L, M, H respectively). We further categorize L, M and H as "smokers" and N as "non-smokers". The "smoking" label has been used in the past to indicate "PM high emitters", though it is subjective definition. In the summer phase of testing, there were 10 (L only) smokers observed constituting 5% of the vehicles measured for smoke. The population weighted average emission rate of these 10 vehicles is 43 mg/mi. Of the highest emitting 14 vehicles in the summer, only 2 had observable smoke.

An earlier field study of smoking vehicles in California, found that between 1-2% of the light duty vehicles in the fleet emit visible smoke [Durbin et al, 1999]. The average FTP emissions of a select sample of smoking vehicles from this previous study, was 399 mg/mi. However, one cannot compare the smoking rates of these two studies, since the vehicles in KC were not explicitly recruited by their smoking classification. Moreover, a direct comparison of the frequencies also cannot be compared. This is due to the fact that the observation methods were very different between the studies. In KC the vehicles were observed for smoke while idling (at low loads). In the California field study, smokers were observed when the engines were loaded: at signalized intersections and highway ramps. These vehicles may also have had more exhaust dissipation due to their higher speeds, and the root causes of smoke formation may be quite

different in idling versus the hot running engines. It is important to note that most analyses of smoking vehicles is more qualitative than quantitative.

In the winter phase, 40% of the vehicles were observed to be "smoking" to some degree. Of the highest 26 vehicles tested (which contributed 50% of the winter emissions), 10 were smokers, 8 were not, and 8 were not observed. However, since it may have been difficult for the technicians to distinguish definitively smoke from water condensation due to the cold temperatures, winter smokers should not be grouped together with the rest of the population. Because of the high degree of water condensation in the smoke, the frequency of smoking vehicles seen in this winter study is not representative of high PM smokers in the fleet.

These summer and winter results support the notion that not all smokers emit high PM emissions, and not all high emitters are smokers. However, smokers do tend to have higher emissions as demonstrated in the following table. These levels are considerably lower than the average smoker emission rates from past studies, which were recruited specifically for their visible smoke levels, thus potentially biasing their emission rates higher than this study.

Table 8 shows the population weighted average emission rate for each smoker category during both phases of the study.

Table 8: Population weighted average emissions by smoker classification.

Smoker Category	PM mg/mi	N
No	8.7	338
Low	19.7	68
Medium	49.0	21
High	63.8	8

We have discussed some of the issues with defining a PM high emitter. Choosing a cut-off value is the primary challenge. Basing the cut point on statistics such as standard deviations, or cumulative distributions of emissions is overly arbitrary and dependent on the sample. On the other hand, several previous authors employ "smoker" as a surrogate definition. We have shown that though there is a correlation with higher PM emissions and smoking vehicles, that relationship is subjective and can even be misleading at times, especially in colder temperatures. Finally, a definition based on an emission standard (as with HC, CO and NOx) is also not feasible because PM standards for gasoline fueled vehicles were chosen to mirror those from light-duty diesel. Thus they were not chosen at a stringent (technology forcing) level. The following figure shows the scatter plot of PM emissions compared to the emissions standard. Clearly, the standard is not a good measure of high PM emissions, especially since there were no standards before 1996, and the test program uses a different drive schedule (and temperature) than the certification test.

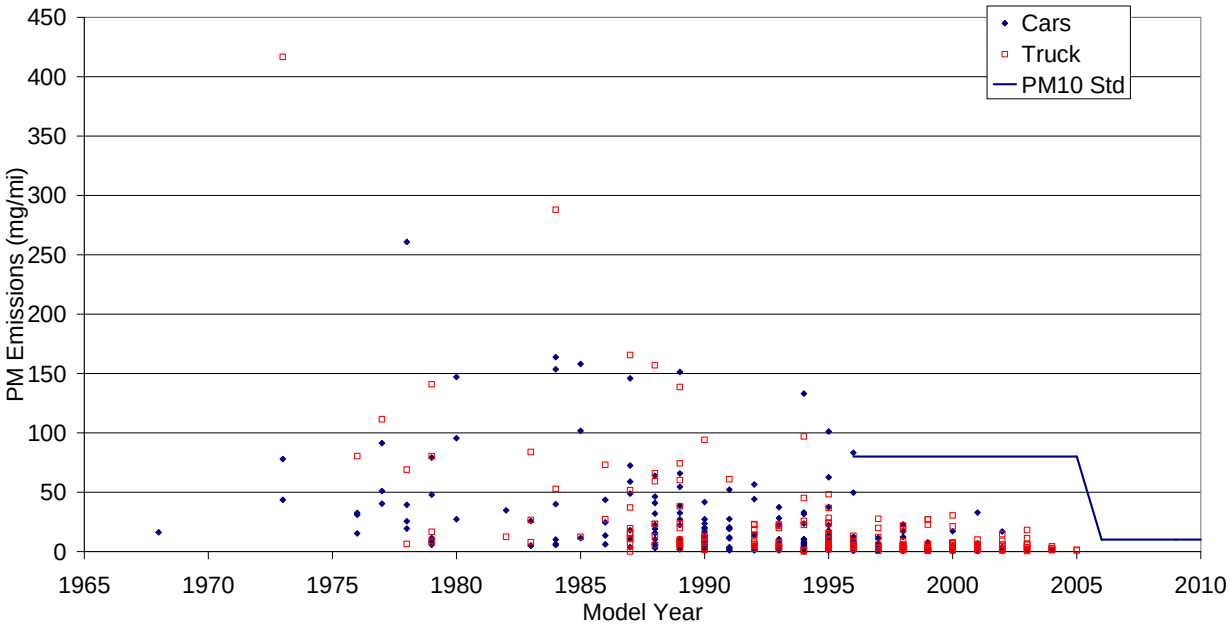


Figure 7. Scatter plot of KC PM emissions as a function of model year, compared to the PM10 emissions standard.

Unfortunately, this study does not answer the question of how to define a PM high emitter. The above analysis describes some characteristics of PM "high emissions" without necessarily adding a high emitter label or cut point. In a later section of the paper, we show cumulative distributions with temperature adjusted data. However, this will have to remain a subject of further study.

7.3 Comparison with past studies

Table 10 compares the results of the present study with two major studies that have preceded it: CRC E24 phases 1, 2, and 3, as well as the Gas Diesel Split (GDS) study. The emissions rates are averaged by the model year strata as used in the Kansas City study to provide a common basis of comparison (though a more recent model year group was added at 2001+). Where possible, smokers that were specifically recruited as part of prior studies are omitted from the comparison, since their inclusion may lead to over-representation of high emission vehicles. The smokers included in the "normal recruitment" (on the basis of model year or mileage) were included in the following analysis. A comparison of these past studies with the present one is informative, but it is important to note that all of the studies are very different, thus making comparisons qualitative in nature. Mainly, a model year comparison is limited since the same model year vehicles in KC would be approximately 8 years older than those in E24, and 3 or 4 years older than those in the GDS, and thus probably have more miles driven. Moreover, the number of vehicles recruited in each of these other studies was smaller, thus making estimates of averages more unstable.

As mentioned earlier, the CRC E-24 project was conducted in 3 phases. Phase 1 is more commonly known as NFRAQS. Table 9 describes the differences between the programs:

Table 9: A comparison of 5 different PM measurement programs.

	CRC E24-1	CRC E24-2	CRC E24-3	Gasoline-Diesel Split	Kansas City
Region	Northern Front Range, CO	SCAQMC	Dallas-Fort Worth, Houston	Los Angeles	Kansas City
Recruitment	Local Merchants, etc.	Registration + phone	SwRI employee vehicles + local school	BAR via smog check + local advertising	Random digit dialing + registration database
Test Year	1996-1997	1996-1997	1996-1997	2001	2004-2005
Drive Schedule	FTP	FTP	FTP	LA92	LA92
Temperatures	23-67F in winter	Lab	Lab	64-99F	20-100F
PM size	PM10	PM10	PM10, PM2.5	PM2.5	PM2.5
Fuel	Local	Local	Local	Local	Local
Fleet Certificatio	National	CA	National	CA	National

Given the time span between the different studies, it is unexpected that the emissions levels are approximately consistent (within 95% confidence bounds). The exception is in E24-2, where the vehicles have significantly lower emissions in two of the model year groups. This unexpected consistency across the studies could be due to a number of factors: the KC fleet within these model year groups may be cleaner than the other studies, the other studies may have oversampled dirtier vehicles, a "survivor effect" may be keeping the oldest emissions rates from exceeding a certain level by a process where owners scrap the dirtiest vehicles in the fleet, or the emission rates may be mostly dependent on model year (or technology), i.e. emissions rates may not deteriorate significantly over time. The latter is an unlikely hypothesis. It is likely, however, that a combination of these factors is in play.

Table 10: A comparison of average emissions by model year groups from 5 different test programs: KC, CRC E24 phases 1, 2, 3, and Gasoline Diesel Split Study. "PM ttl" has no PM size preselection.

Model Yr Group	KC Summer 2004		KC Winter 2005		E24_1_Summer 1996		E24_1_Winter 1997		E24_2 1996 - 1997		E24_3 1996 - 1997		Gas-Diesel Split 2001	
	PM2.5	N	PM2.5	N	PM ttl	N	PM ttl	N	PM ttl	N	PM ttl	N	PM2.5	N
pre-1981	69.0 ± 58.3	8	75.7 ± 40.6	24	95.4 ± 34.7	25	78.3 ± 36.3	15	34.0 ± 12.2	14	148.5 ± 94.4	10	59.0 ± 54.4	6
81to90	29.8 ± 12.7	56	36.4 ± 11.8	68	46.0 ± 24.1	47	37.8 ± 11.4	33	28.1 ± 19.1	54	64.4 ± 29.3	25	34.2 ± 16.2	18
91to95	9.7 ± 3.6	48	20.0 ± 6.9	72	2.5 ± 0.7	17	30.4 ± 32.6	7	2.8 ± 1.2	50	5.8 ± 1.2	12	3.6 ± 1.6	26
96to2000	3.7 ± 1.4	77	8.9 ± 3.1	61	4.5 ± 5.0	3	3.0	1	2.3 ± 2.2	11	6.4 ± 1.7	6	3.9 ± 3.7	3
2001+	2.2 ± 0.6	29	6.8 ± 2.7	35										

In the following sections, we look at trends of PM as a function of a variety of independent variables. Such variables include temperature, model year, and age. We also examine the correlations with other pollutants since the mechanisms of formation of the more traditional criteria pollutants (HC and CO) are better understood. It follows that these trends may shed light on PM formation processes as well. The correlations may also show how controls of HC may or may not influence PM emissions.

7.4 Temperature Effects on Composite Data

In this section, we examine the effects of temperature on the KC data. The vehicles tested in Kansas City were tested under ambient temperature conditions. Before other PM trends can be determined, we must first isolate (or adjust for) the effect of temperature. Some studies in the past (cited below) have shown that PM emissions increase as temperature decreases. However, this effect has never been conclusively quantified and compared across studies. This is what we hope to accomplish.

The first pass at the analysis is conducted on the composite (or aggregate) data only. Composite data consists of the 3 bags of LA92 emissions data weighted by the appropriate weighting factors. Temperature effects are scrutinized in 3 ways: by examining all the data, the correlation vehicle, and the summer-winter paired data. The correlation vehicle (1988 Ford Taurus) was measured 24 times throughout both phases of the program, each test occurring at a distinct ambient temperature. Also, 43 vehicles were tested, both in the summer and winter. This paired test database is valuable in looking at the fraction of the emission rate variability explained by temperature. The paired test data is shown in Figure 8. The figure clearly shows that winter emissions exceed summer emissions in nearly all cases. Some vehicles have 10 times more PM emissions in the winter as in the summer. This leads us to conduct much of our subsequent analysis in log-space, which allows for quantification of effects across a large range of results.

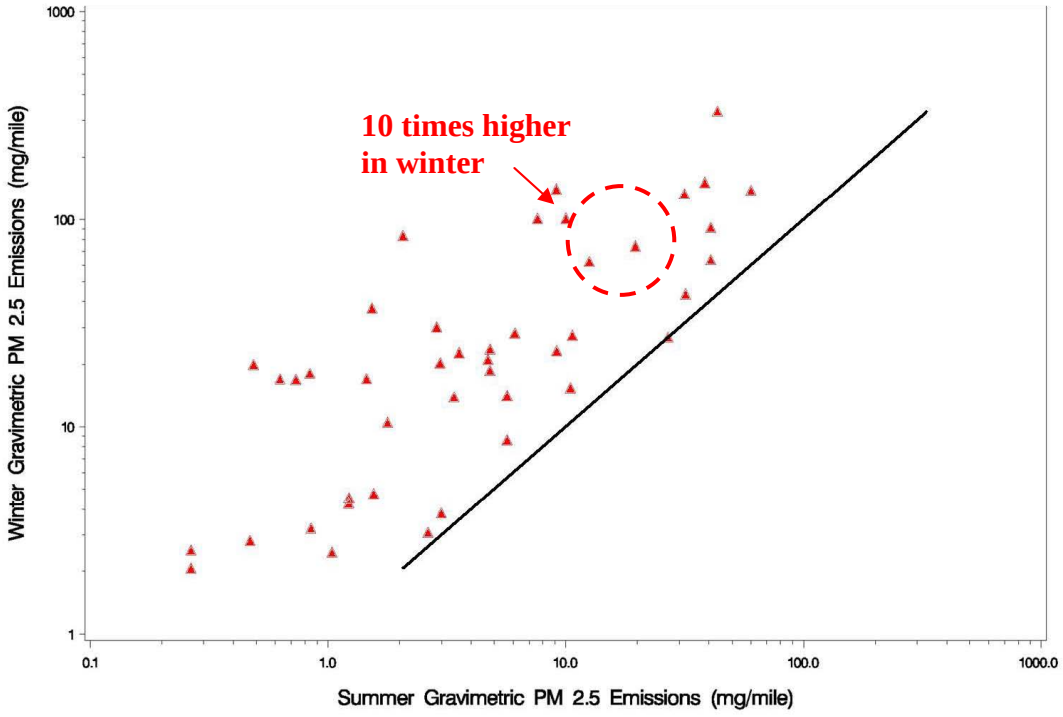


Figure 8. Scatter plot of winter vs summer PM emissions on log scale (EPA, 2008).

The PM trend with temperature is shown in Figure 9. The red (empty) diamonds are all the vehicles (except the correlation vehicle) and a trend line is drawn through this grouping. The solid blue diamonds are the summer-winter paired tests. And the solid blue squares are the correlation vehicle repeat tests. The slope for all the data is -0.0188 (in log space), and for the correlation vehicle is -0.0415.

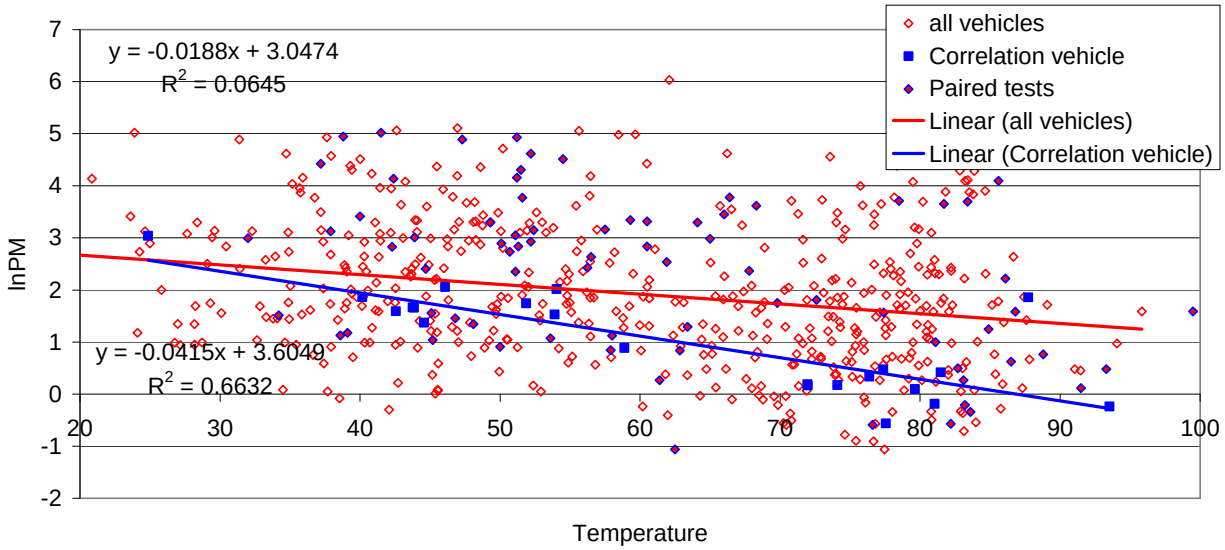


Figure 9. lnPM vs test (ambient) temperature for all vehicles, paired tests, and correlation vehicle.

The matching paired tests each have separate slopes which were calculated using the following linear equation in log-space.

$$m = (\ln PM2 - \ln PM1)/(T2-T1)$$

Pairs were omitted from the analysis if any of the conditions exist: one of the PM values is missing; the temperature difference between two tests was less than 10°F. If the temperature difference is less than 10°F, the test-to-test variability dominates over any temperature effects and the slopes become ill-defined. Unfortunately, these criteria eliminate 10 of the 43 paired tests. The remaining slopes are plotted by model year in Figure 10. The average of the slopes is -0.036 +/- 0.009 (95%CI) (we will explore a more robust method for estimating this slope parameter next). However, the plot seems to indicate that there is no apparent model year trend with temperature. This leads us to believe that temperature effects on PM emissions are independent of vehicle technology. Also, though not shown, the trend does not appear to be dependent on average temperature of the repeat tests.

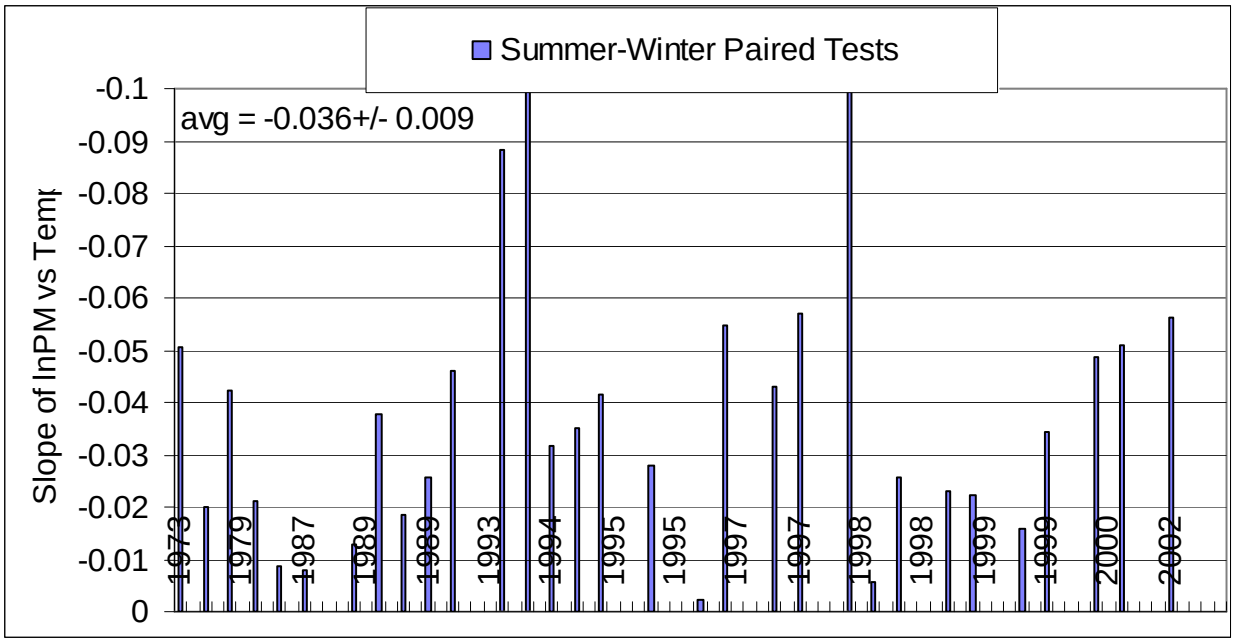


Figure 10. Temperature slope from individual matched pairs as a function of vehicle model year.

In order to estimate the effect of temperature from the summer/winter matched pair vehicles, we rely on a univariate general linear model, run on the SPSS statistical software. The "vehicle" was treated as a fixed factor (categorical variable) and the correlation vehicle was weighted by a factor of 0.09 in order to give it the same weighting as the other 33 matched pairs. Otherwise, the 22 points of the correlation vehicle would dominate the linear model. This is the temperature effect that should be used in order to adjust all of the PM data to a baseline temperature for subsequent comparisons of summer and winter data. The matched vehicle slopes are shown graphically in Figure 11. There are clearly some vehicles with a weaker temperature effect and others with a stronger one, however a statistical mean can be discerned. Using the solution to the previous equation, the correction will be applied in the following manner:

$$\ln PM2 = -0.03356*(72°F - T1) + \ln PM1$$

This is based on the assumption that it is the slope that drives the temperature effect; thus the offset in the slope is defined by each individual test and is ignored for the purposes of this temperature correction model.

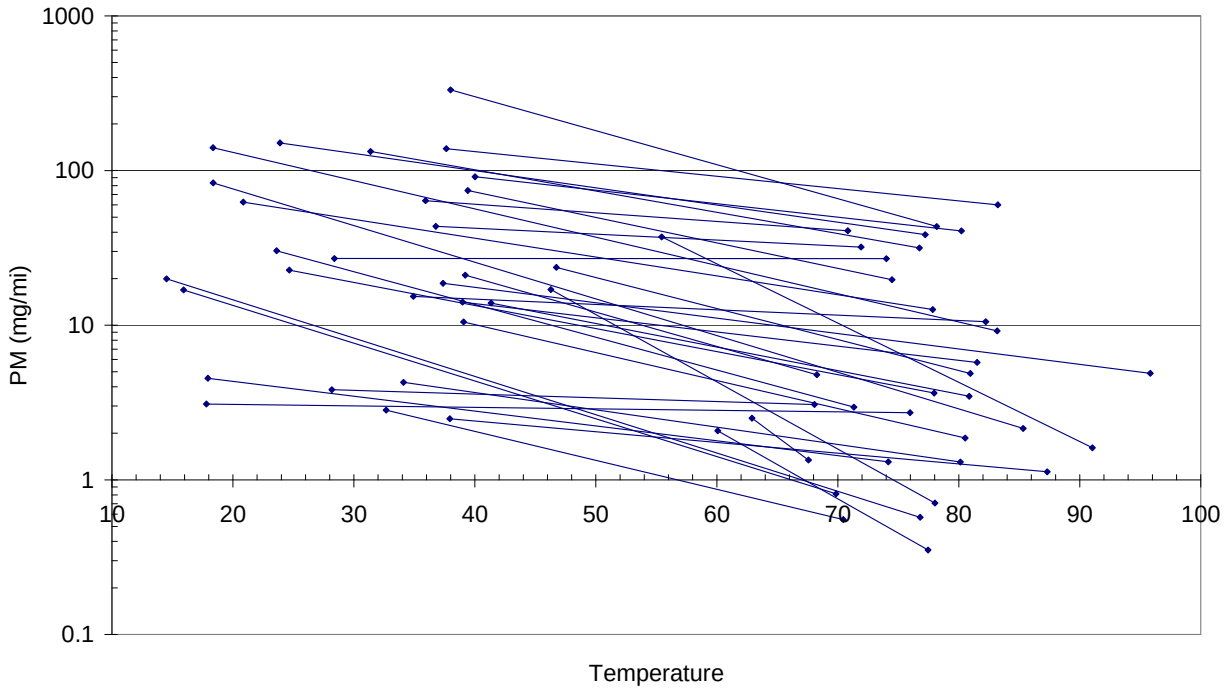


Figure 11. 33 Matched pair vehicles as a function of temperature on log scale.

Temperature effects on PM emissions have been evaluated in some recent testing programs. In 2000, Southwest Research Institute measured the PM from 7 vehicles at 30°F and 75°F [Whitney, 2000]. In 2001, the EPA (at the Office of Research and Development) conducted testing on 9 vehicles with model years ranging from 1987 to 2001 at 75°F, 20°F, 0°F and some at -20°F. This is referred to as the ORD data [Stump, et al, 2005]. In 2005, EPA tested 4 Tier 2 vehicles at 75°F, 20°F and 0°F in support of the Mobile Source Air Toxics (MSAT) rulemaking [Stanard, (2005)]. The results of these three test programs are compared with the Kansas City data on Figure 12. Note that the trends (slopes) are quite similar, and that the slope for the KC data lies in between the studies . The MSAT data is considerably lower, since the program specifically targeted Tier 2 vehicles. It is again, interesting to note that temperature trends (slopes) seem to be independent of vehicle technology, since the slopes are relatively constant from the various test programs.

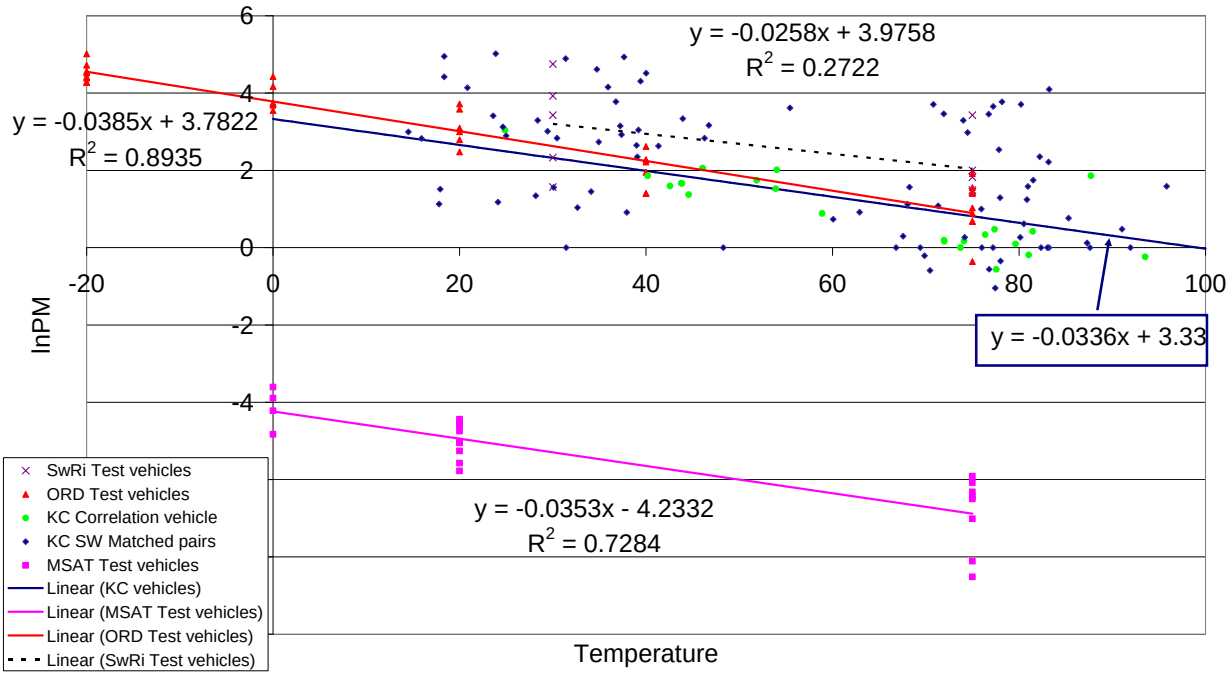


Figure 12. PM temperature trends from 3 different test programs in log space.

Figure 13 shows the some of the same data as the previous figure but in terms of absolute PM emissions. The fit is to the KC data, which is consistent with the ORD data despite the very different fleet mix tested.

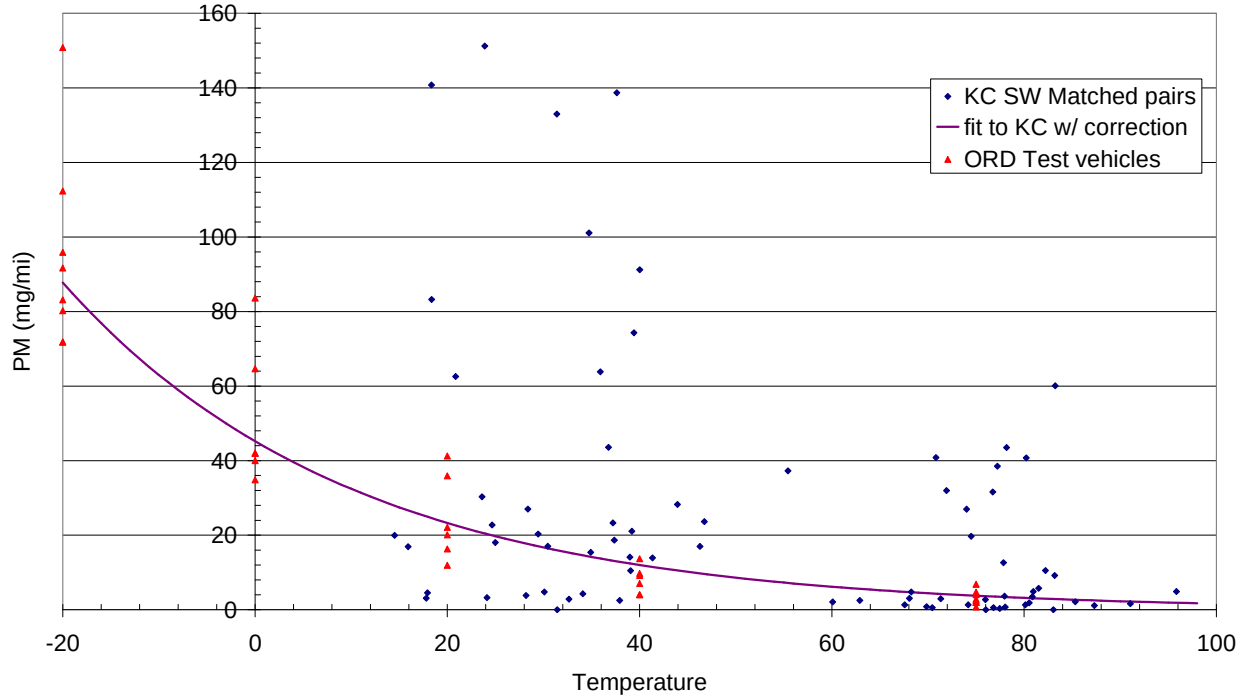


Figure 13. PM temperature trends from 3 different test programs in linear space. The fit the KC data is MSE corrected.

A similar analysis of temperature trends for the other pollutants was conducted and the results are shown in Table 11. All of the slopes are significantly different from zero at the 5% level. PM is the most sensitive to temperature, whereas for some other pollutants the effects are small. For the remainder of this report, the correlation vehicle is usually omitted from all trend reporting since the vehicle is not part of the randomly sampled population, and the emissions from this vehicle is not necessarily representative of what would be expected to be seen in the existing fleet.

Table 11. Exponential emissions dependence on temperature (P<.05). The number of matched tests varies with pollutant since some pollutants had invalid data.

Pollutant	Slope	std error	N
PM	-0.0336	0.0029	34
HC	-0.0124	0.0012	44
CO	-0.0145	0.0014	44
NOx	-0.00234	0.00082	43
CO2	-0.00077	0.00014	44

In this section, we have shown the PM follows a well-behaved pattern with temperature. PM emissions increase exponentially with decreasing temperature. This pattern has been confirmed by comparison with two other independent studies. Later in this report, we will examine these effects further by looking at the cold start and hot running dependence on temperature. With this temperature behavior, it is possible to isolate the effect of this variable in the Kansas City data so that we can further examine trends in PM emissions due to other factors, such as age or model year.

7.5 PM Emissions Trends from Composite Data

We now look at trends in PM emissions with other criteria pollutants as well as trends with model year or age. We tend to examine these trends on a logarithmic scale. There are several reasons for this:

- Emissions trends tend to be log normally distributed (or some similarly skewed distribution) [Frey, et al., 2002].
- Because of the skewed distribution, linear plots tend to show points clustered about the origin with some outliers. This does not typically give the viewer as much information about the pattern of the data as with a log scale plot.
- Emissions trends with time (either model year or age) tend to follow functions that either increase or decrease exponentially or level off smoothly over time.

Figure 14 shows the histograms of the logarithm of the emissions for winter and summer. While the emissions are not perfectly log-normally distributed, the distributions do indicate that they would be a significantly skewed if these were plotted on linear scale (like in Figure 6).

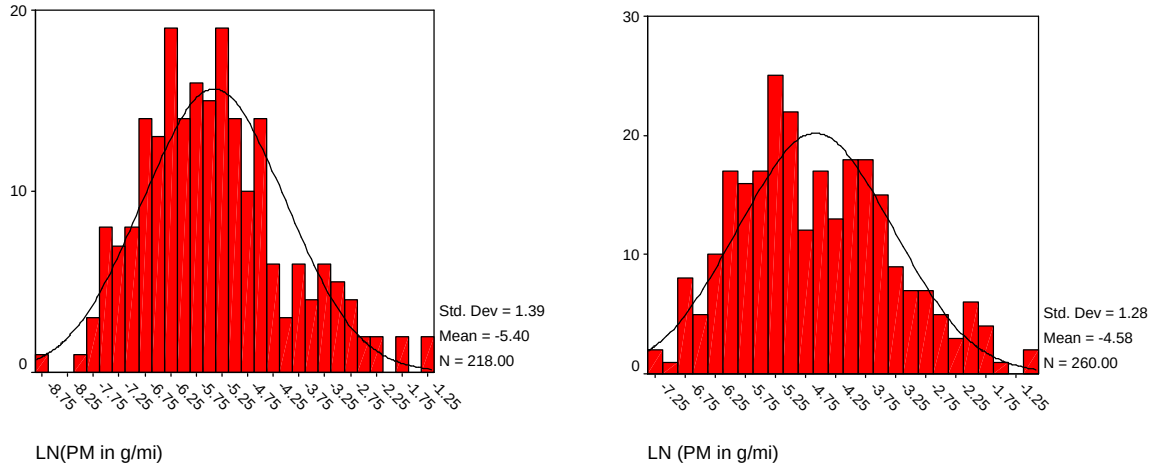


Figure 14. histogram of the logarithm of summer (left) and winter PM emissions (right)

Figure 15 shows cumulative frequency distributions of composite emissions, with rounds 1 and 2 combined by (model year) stratum. Note that the values from round 2 have been adjusted for temperature using the equation in the previous section. However the distributions have not been population weighted, so the distributions are not necessarily indicative of the actual KC fleet.

The distributions show expected patterns, with the oldest vehicles showing highest emissions, the youngest vehicles the lowest. In addition, within each model-year group, values for cars and trucks are distributed more closely than are values for adjacent model-year groups. In absolute terms, the distributions show that the skew in the data is more pronounced for older vs. younger vehicles. For example, the difference between the 90^{th} and 50^{th} percentiles is approximately 4.0, 6.5, 60 and 100 mg/mi for cars in the 96+, 91-95, 81-90 and pre-80 groups, respectively. The skew in the distributions is also shown by the differences between medians and means, which fall roughly between the 70^{th} and 80^{th} percentiles.

Figure 16 shows the same cumulative frequency distribution as Figure 15 but on a logarithm scale. One can note that the distributions look nearly log-normal, though in some of the stratum, the high emissions tail appears to be slightly longer than the low side.

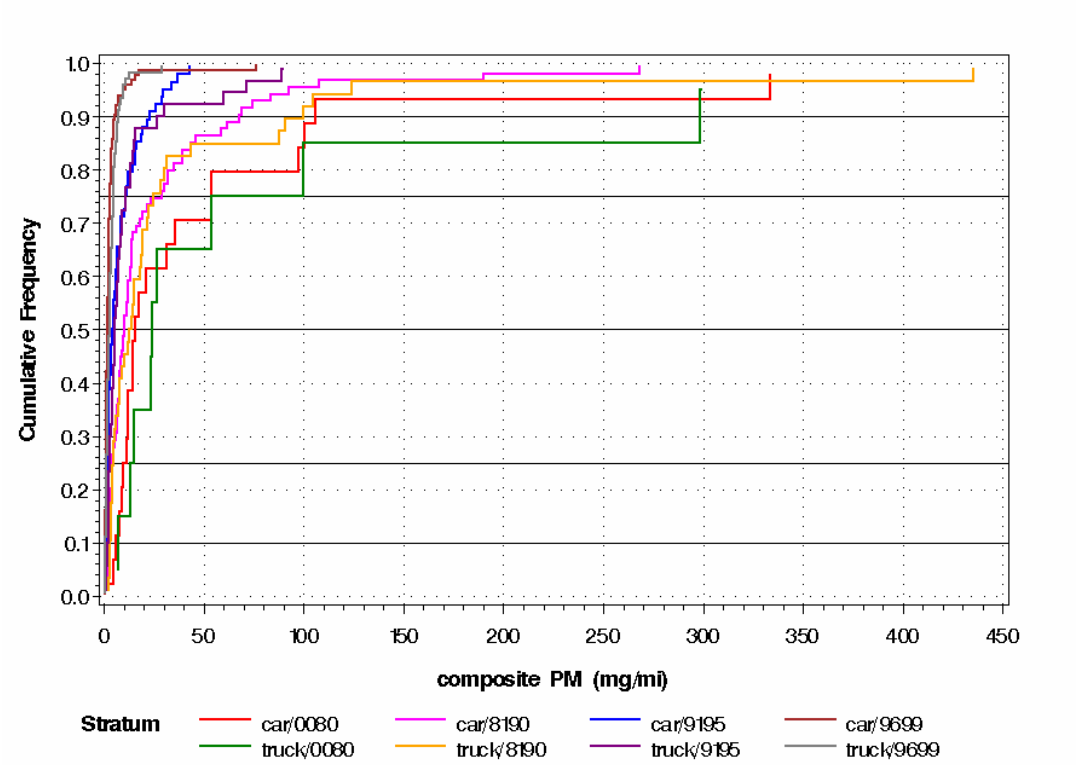


Figure 15. Cumulative distribution of PM emissions (temperature adjusted to 72F) for the model year groups.

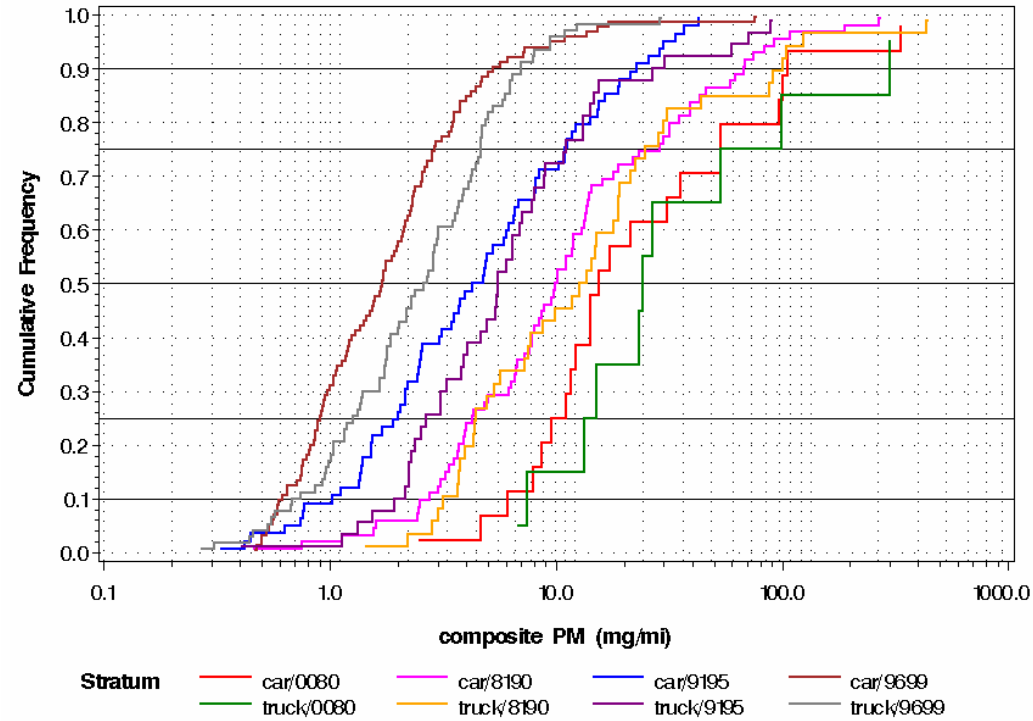


Figure 16. Cumulative distribution of PM emissions (temperature adjusted to 72F) for the model year groups on a logarithm scale.

The trend of PM with model year is shown in Figure 17. It is interesting to note that the slope of the line changes little whether the temperature adjustment is made or not, though the R^2 is slightly improved (not shown). This is due to the fact that there was a reasonable mix of model years in both summer and winter. The following figure which shows temperature adjusted PM as a function of model year is provided only for the purposes of examining trends and does not form the basis for an emission rate model.

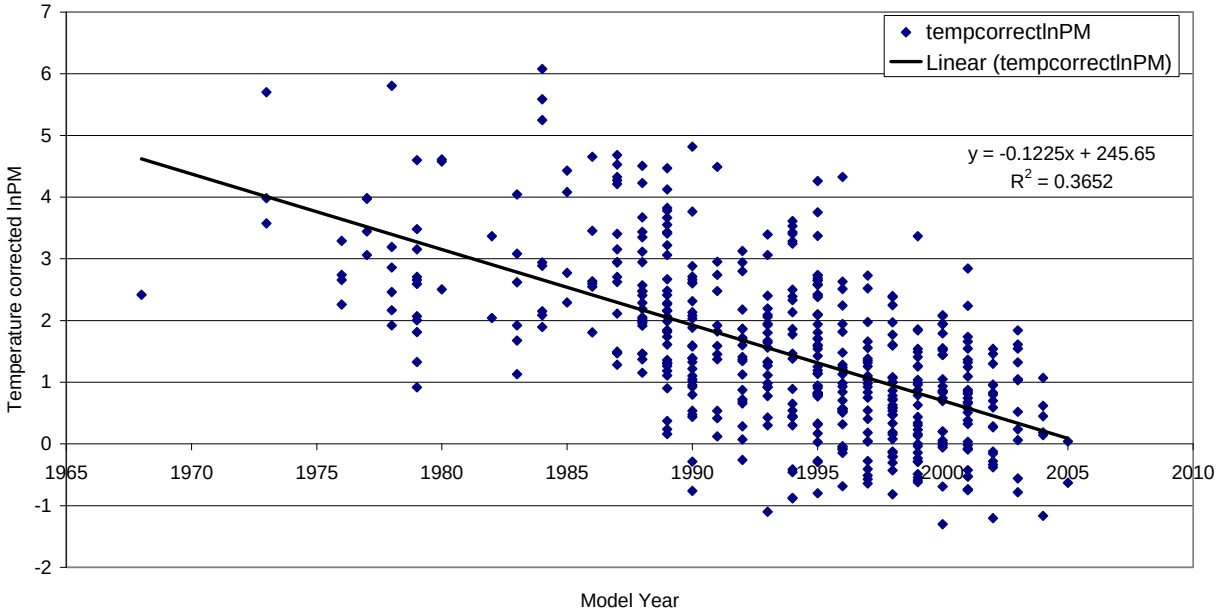


Figure 17. Temperature adjusted PM as a function of model year plotted in log scale.

When quantifying deterioration, it is always interesting to compare emissions trends with age as well as odometer. However, odometer readings are unreliable in older vehicles since a 6th digit was not present in many vehicles. This "rollover" effect tends to bias odometers low for older vehicles. Figure 18 shows odometer as a function of model year. Note that before 1995, the rollover effect is pronounced. Before 1987, very few vehicles had a 6th digit to report.

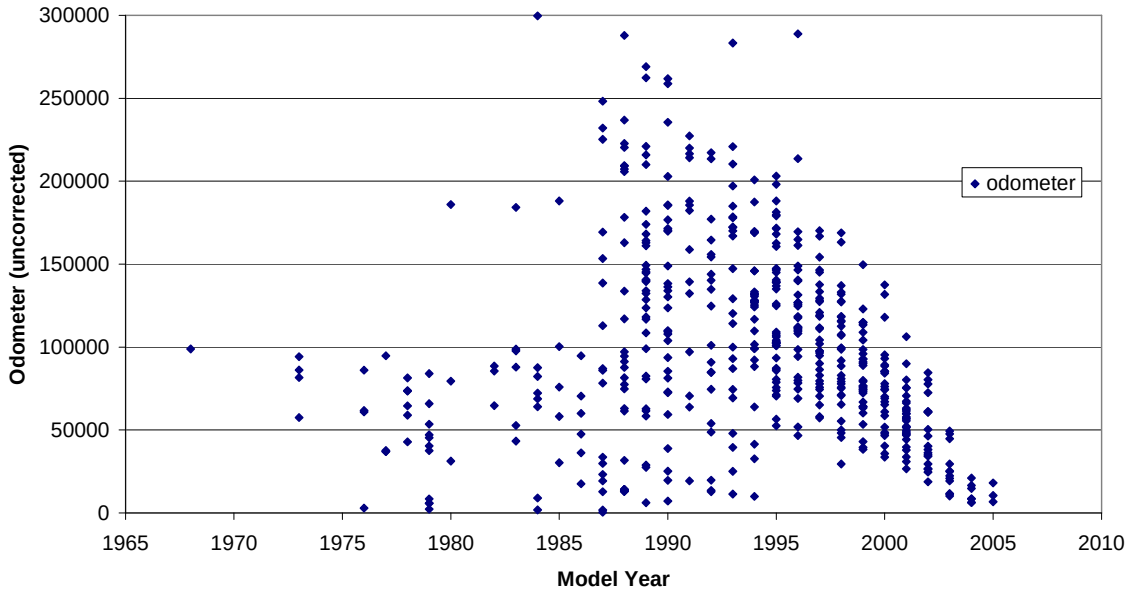


Figure 18. Odometer as a function of model year.

Thus it is clear that a deterioration trend based on odometer would only be reliable for vehicles with model years 1995 and later. For these model years, PM can be plotted as a function of odometer, as shown in the following figure for 1995+ vehicles in 25,000 mile bins. The dip at higher odometers (>125,000 miles) may indicate a "survivor effect", where the malfunctioning (and higher emitter vehicles) take themselves out of the mix; or it could be natural variability. Again, one must be cautious in drawing conclusions about deterioration from this plot, since there are a range of model years, technologies, and ages grouped together.

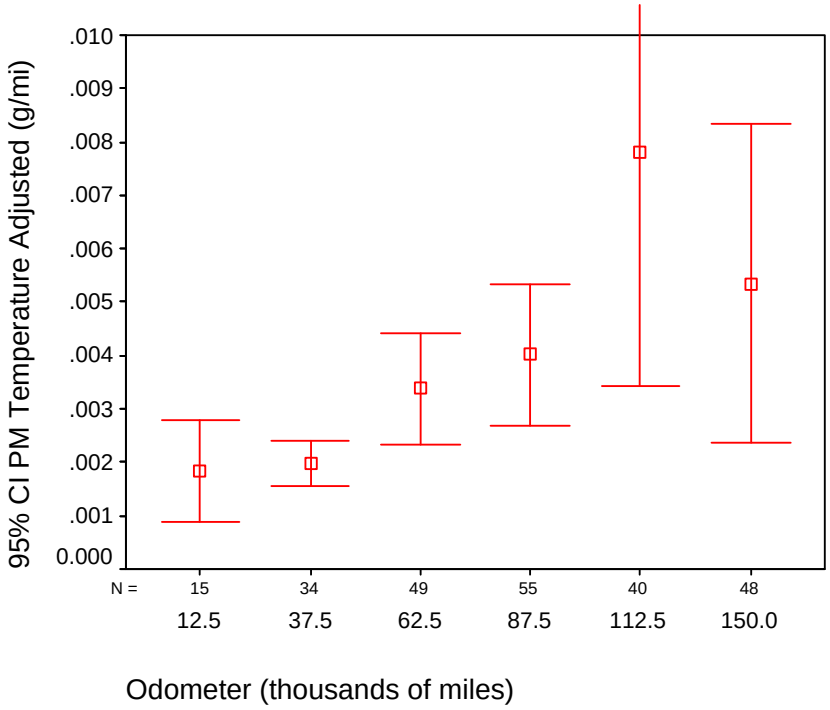


Figure 19. PM emissions as a function of odometer for model year 1995 and later vehicles.

For most of the remaining analysis we will rely on age as a surrogate for odometer. As vehicles age, they typically have higher odometers; and even the vehicles that have low mileage accumulation there are deterioration paths that depend on other factors such as frequency of starts, weather, accidents, and maintenance. Model year and age are roughly equivalent in this analysis since the Kansas City data was only collected over the period of one year. Further, this makes it difficult to separate the effect of model year and technology from age and mileage or deterioration, especially for pre-1995 vehicles.

The following series of charts shows some of the other pollutant trends with model year (not temperature adjusted). Note that both HC and CO show a logarithmic trend, but NOx shows a more rapid decline starting in 1995. The fit lines are shown more for comparison than an accurate measure of trends. At this time, it is impossible to tell whether the decreases in emissions are due to technology (model year) or deterioration (age). These modeling results are the subject of a partner publication (Nam, 2008).

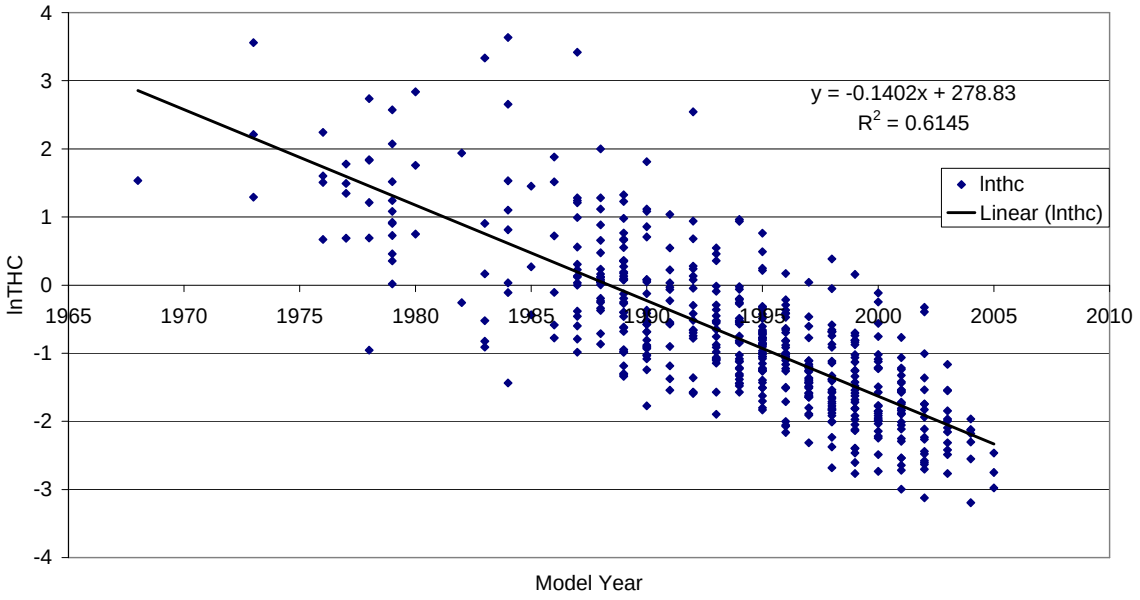


Figure 20. THC emissions as a function of model year.

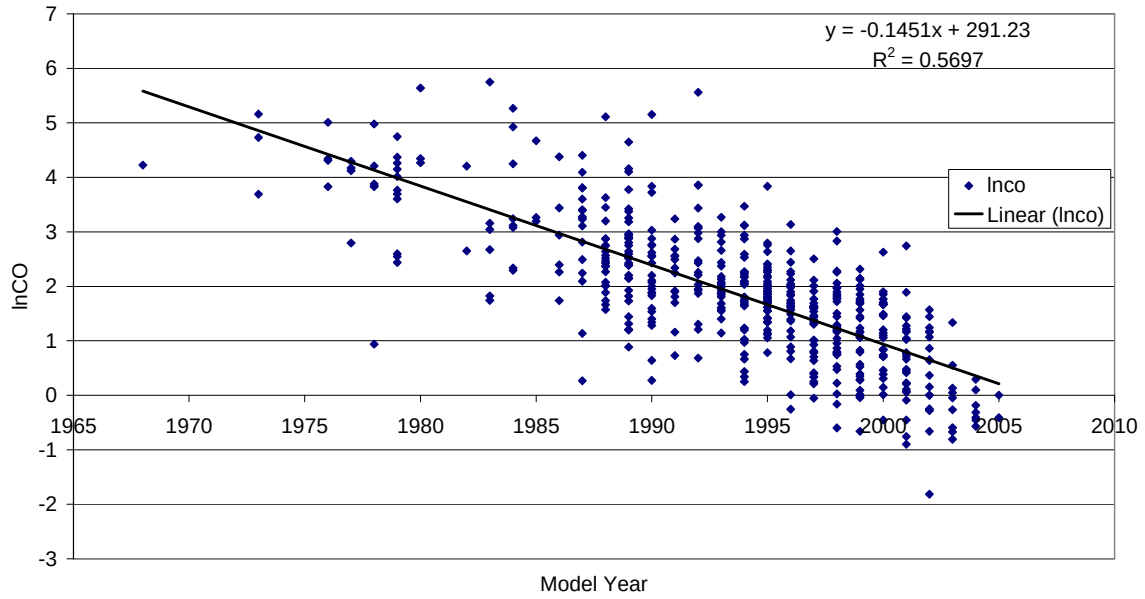


Figure 21. CO emissions as a function of model year.

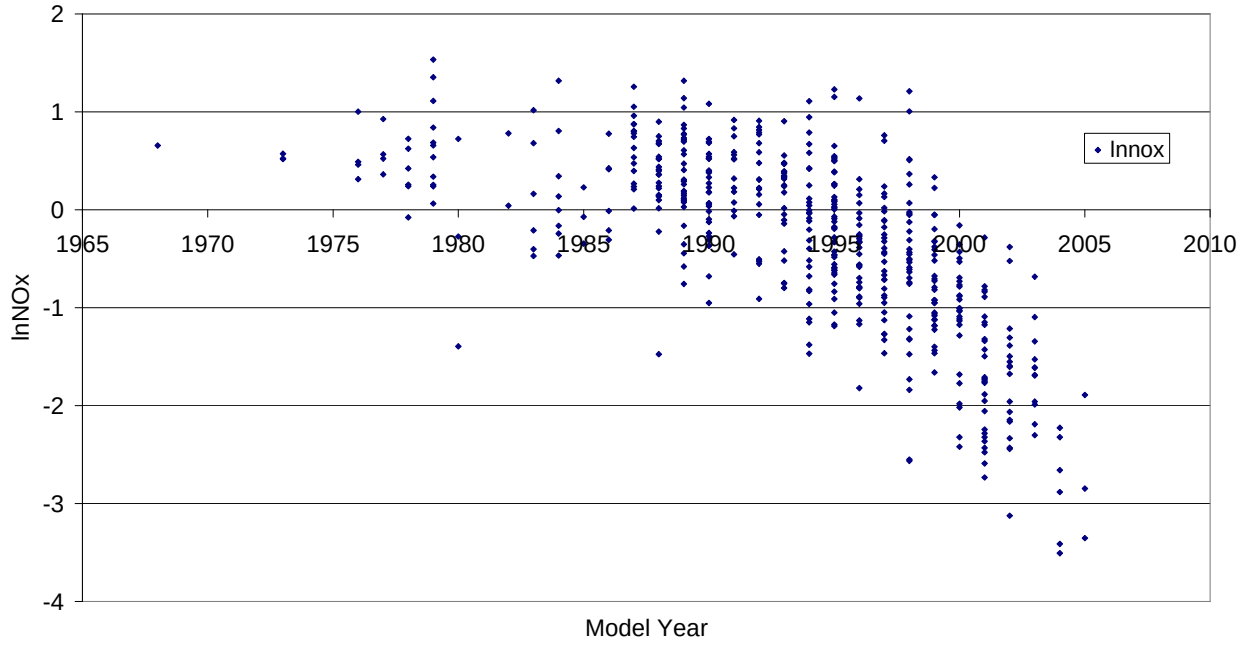


Figure 22. NOx emissions as a function of model year.

Figure 23 shows the correlation of PM with HC. The R^2 is 0.56 and is a better correlation than that of PM with CO, NOx, or CO_2 (not show, though R^2 is 0.46, 0.26, and 0.03 respectively). We will further explore the link between HC and PM when we split out bag results.

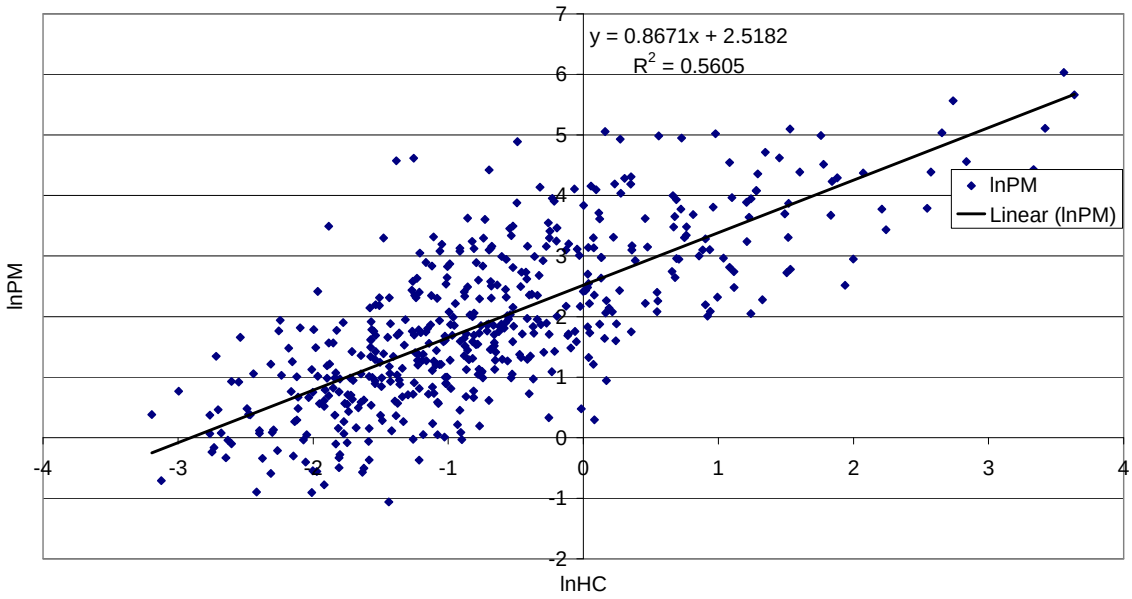


Figure 23. PM correlation with HC (not temperature adjusted).

8 Bag Analysis of Kansas City Data

The analysis in the previous section was based on a single aggregate emission rate from an LA92 test. In order to gain a deeper understanding of trends in PM, it is advantageous to examine the emissions from each bag (or phase) of the test. This is especially true if we want to study the differences between cold start and hot running PM formation. Subtracting bag 3 from bag 1 is a common method of quantifying the excess emissions from the cold start since the bags are identical except for their soak times. There is a slight error in defining this as "cold start" since it is actually a cold start compared to a hot start. A true cold start would subtract a hot running bag 3 from a cold start bag 1. Unfortunately few test programs perform hot running bag 3 equivalent tests. However, emissions from a cold start are usually significantly greater than for a hot start, thus rendering the effect of hot start miniscule by comparison (though it may not be insignificant).

It is widely known that when an engine starts "cold", the emissions are higher. This is due to several reasons: The engine is less efficient when cold, the catalytic converter is too cold to convert pollutants, the engine is usually running fuel rich (excess), and the combustion tends to quench (or freeze out) when the cylinder walls are cold. These same physical effects tend to increase PM emissions as well.

Figure 24 shows a regression of cold start PM with cold start HC. The correlation appears to be similar to the composite results from Figure 23 above. This is partly due to the fact that a large fraction of the PM emissions occur during cold start. On average 30% of the PM emissions from the vehicles tested occurred during this cold start increment, and bag 1 emissions are (based on straight averages) 7.5 times that of bag 3. These statistics are similar to that of HC cold start

emissions, where 33% of HC emissions occurred during this cold start increment, and bag 1 emissions are nearly 4 times that of bag 3. Cold start CO statistics are similar to those of HC.

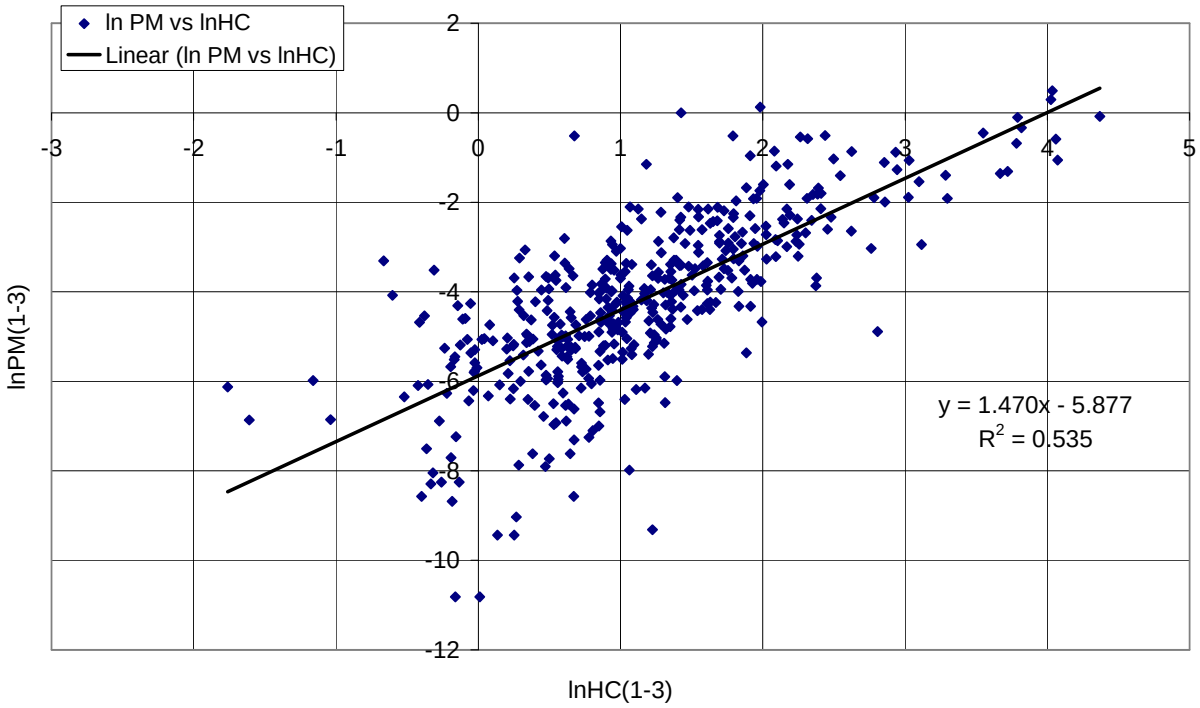


Figure 24. Cold start PM emissions correlation with HC.

A similar but inferior correlation is shown for PM and CO below.

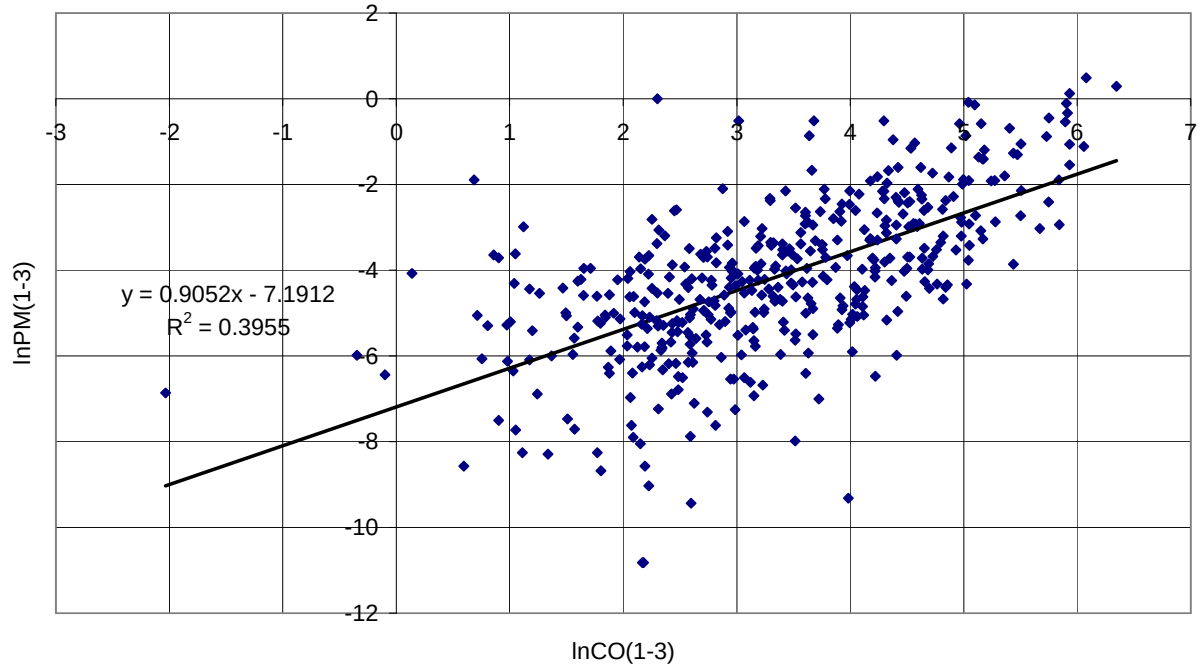


Figure 25. Cold start PM emissions correlation with CO.

For comparison, the following figures show the phase 2 (hot running) PM compared to HC and CO. The correlations are decent, though not as strong for the cold start.

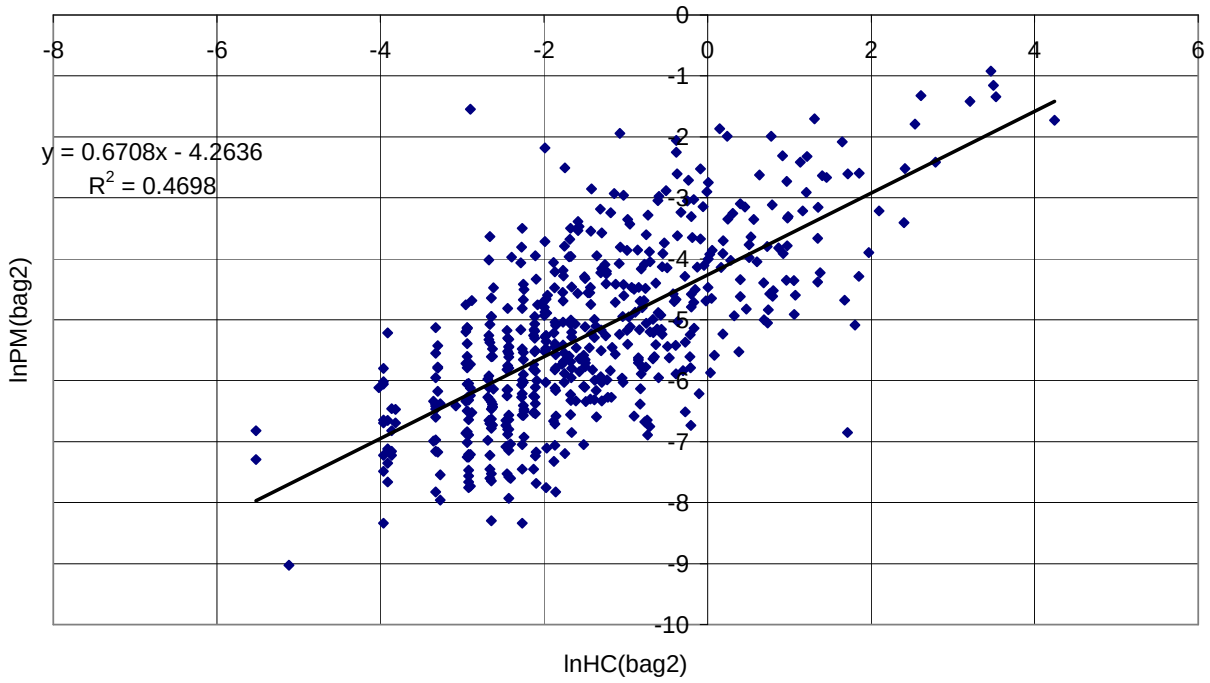


Figure 26. Bag 2 PM emissions correlation with HC.

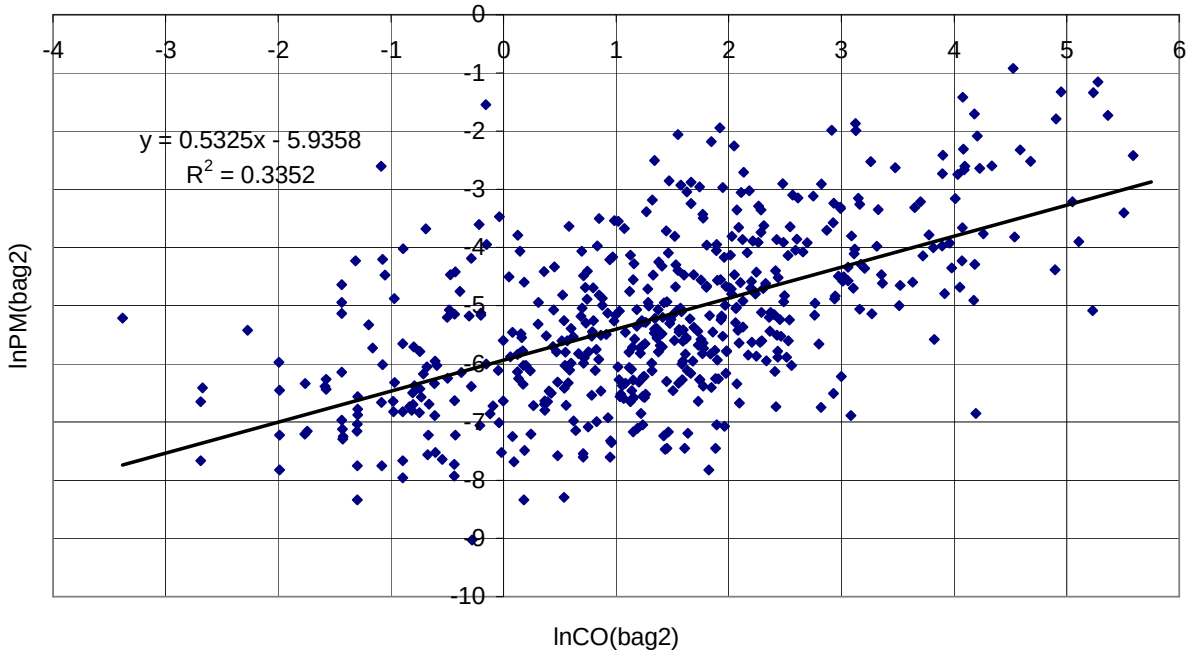


Figure 27. Bag 2 PM emissions correlation with CO.

The LA92 has more aggressive high speed driving during phase 2 (bag 2) of the test than bag 1 or 3. This phase is also twice as long as phases 1 and 3 put together. Due to these factors, on average bag 2 PM emissions (in total grams) are 61% of the total. This is compared to 44% for phase 2 HC emission compared to the total. Phase 2 CO statistics are similar to that of HC. This large phase 2 emissions in the LA92 is in contast to the standard FTP cycle (the certification test), which has a much milder phase 2.

These statistics for cold start and bag 2 PM emissions compared to that of HC and CO provide evidence that PM emissions follow some of the same trends as the other criteria pollutants. However it is important to note the differences, especially the fact that PM seems to be more sensitive to cold start than HC and CO. It also follows that PM may also be more sensitive to temperature than the other pollutants. We explore this question next.

8.1 Temperature Analysis of Bag Data

We repeat the analysis conducted in the previous temperature section, but in more detail. The following two figures show the temperature trend for cold start and hot running PM respectively. We can see from the slope that the temperature behavior previously seen in Figure 9 is largely driven by cold start. The emissions are higher during normal (higher temperature) "cold starts" due to three primary reasons: the engine is colder and not running as efficiently, the catalyst is not yet "lit-off", and the engine often runs a fuel rich mixture in order to combust under cold conditions. Under cold temperature conditions, all of these factors are enhanced and are prolonged. These combinations of factors cause emissions to be an exponential function of temperature. Quantitatively, the cold start slope is more than double that of hot running. The hot running PM shows a small but present temperature effect.

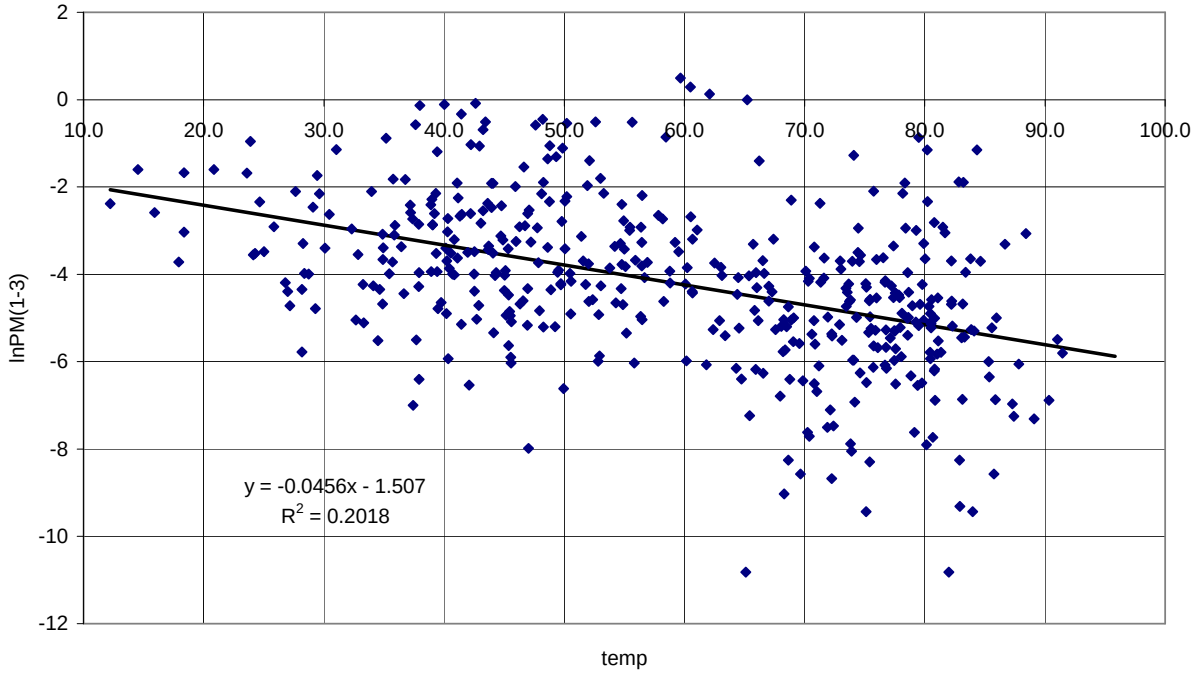


Figure 28. Cold start PM emissions as a function of temperature

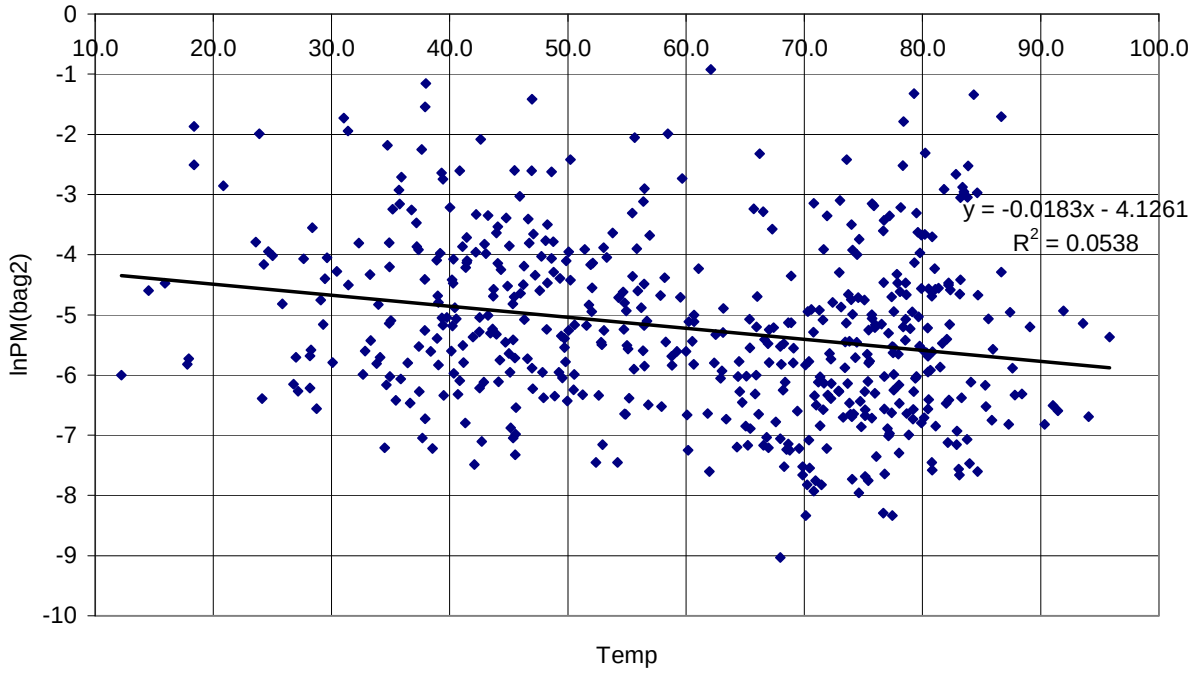


Figure 29. Bag 2 PM emissions as a function of temperature

By comparison, the next two figures show cold start and hot running temperature trends for HC. Note that the temperature seems to affect the cold start, but not the hot running phase 2 HC. CO trends are not shown, but the trends are similar.

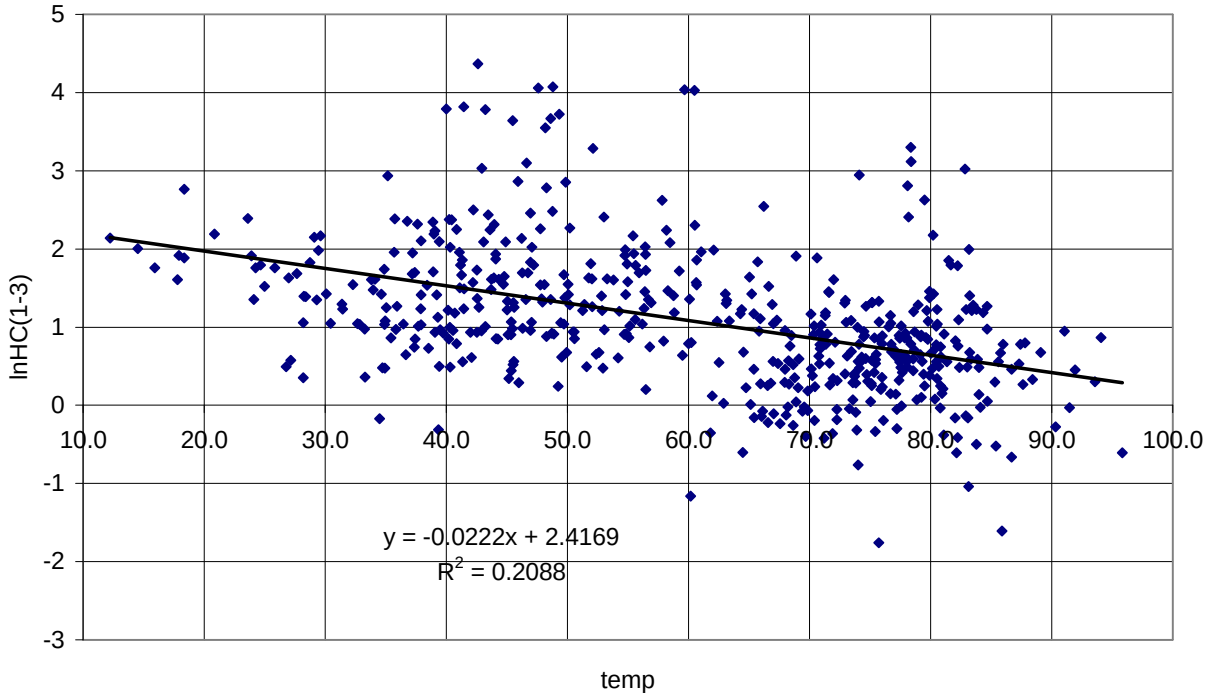


Figure 30. Cold Start HC emissions as a function of temperature

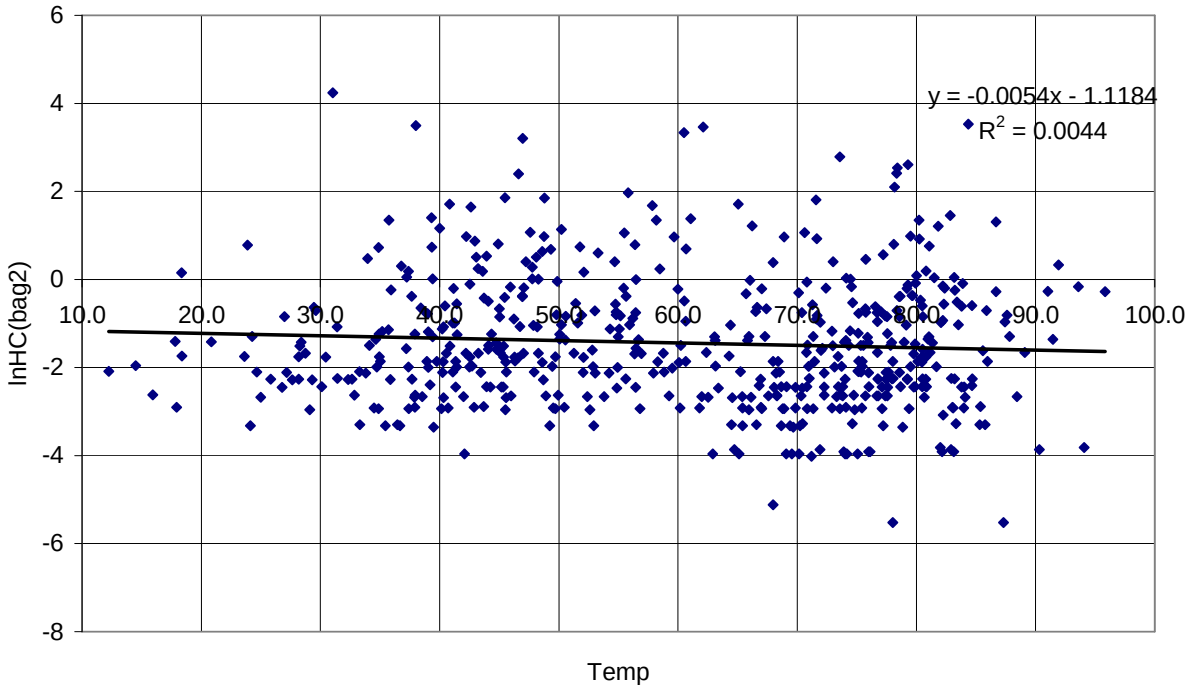


Figure 31. Bag 2 HC emissions as a function of temperature.

In order to gain a more thorough understanding of temperature effects, we look at the matched pair vehicles similar to what we've done above. For this analysis it was necessary to make assumptions for those unusual cases where the cold start emissions (bag 1- bag 3) was less than or equal to zero, since the logarithm is undefined. In these instances, the logarithm was set to the minimum value of the other test. For example, the pm correlation vehicle minimum is 0.5128 and for the rest of the matched vehicles, it is -0.9943. Two values in the correlation vehicle and five values of matched pair tests were substituted. For NOx, half of the correlation vehicle tests required substitution, so they were all omitted from the model; however, 2 of the matched pair vehicles were substituted with -2.142. However, these substitutions for NOx are inconsequential, since it is clear from Table 12 that NOx is the least of the temperature sensitive pollutants. In fact, for cold start, temperature does not have a significant effect, and effect is minimal for hot running as well. The table also indicates that PM is the most sensitive, both in cold start and running, compared to the other pollutants; and that for PM, HC, and CO, the temperature effect is much more pronounced for cold start than hot running. Similar to the trends for composite PM emissions, there is little discernable trend of slope of PM for bag 2 or cold start as a function of model year, with either changes in (delta) temperature or the absolute value of temperature.

Table 12. Cold Start and Hot running slopes with temperature for all pollutants (p<0.05 for all, except NOx cold start).

Process	Pollutant	Slope	std error	N
Cold Start	PM	-0.0463	0.0052	32
	HC	-0.0204	0.0012	44
	CO	-0.0348	0.0021	44
	NOx	0.0017	0.0035	39
	CO2	-0.0065	0.0010	44
Hot Running	PM	-0.0318	0.0028	41
	HC	-0.0073	0.0012	43
	CO	-0.0050	0.0017	44
	NOx	-0.0034	0.0009	43
	CO2	-0.00072	0.0001	44

8.2 Model Year Trends by Bag

Applying these temperature adjustments to convert emissions to their equivalent at 72°F, as we have done above, we now look at model year trends. The following four figures show the model year trends of bag 2 as well as bag 1-3 (cold start) PM separated by car and truck (for the binned plots). Interestingly, the slopes are nearly identical to each other, and also identical to those of the composite PM (Figure 17). Prior to around 1987, the emissions doesn't increase. This could be a "survivor effect" where older vehicles that emit more are scrapped out of the fleet.

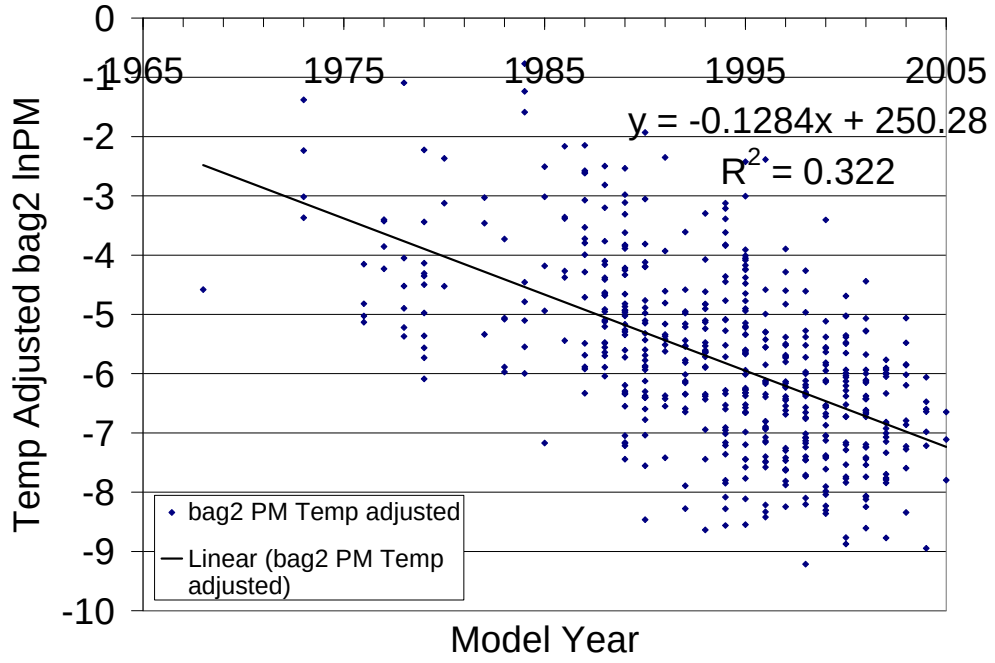


Figure 32. Temperature adjustted natural log of hot running PM as a function of model year.

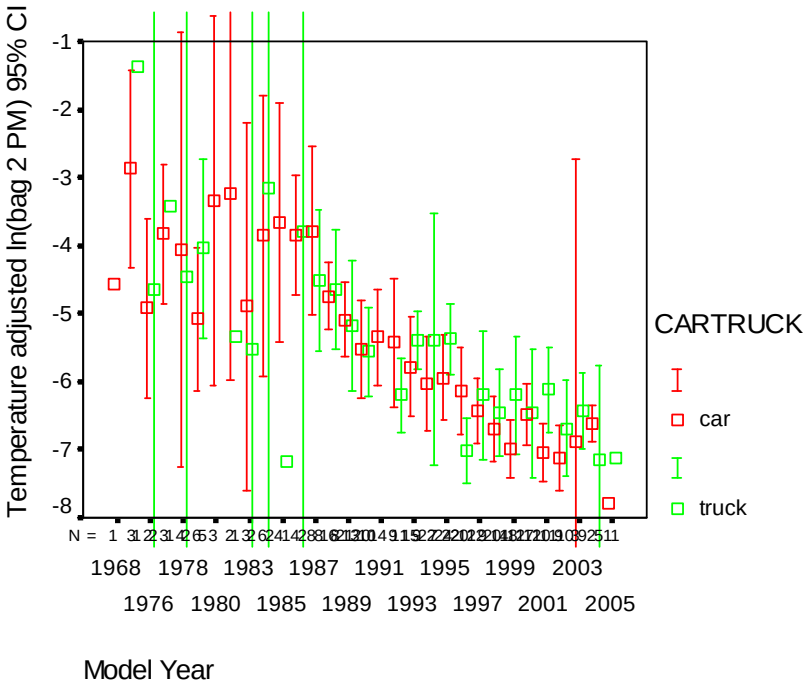


Figure 33. Temperature adjusted natural log of hot running PM as a function of model year bin.

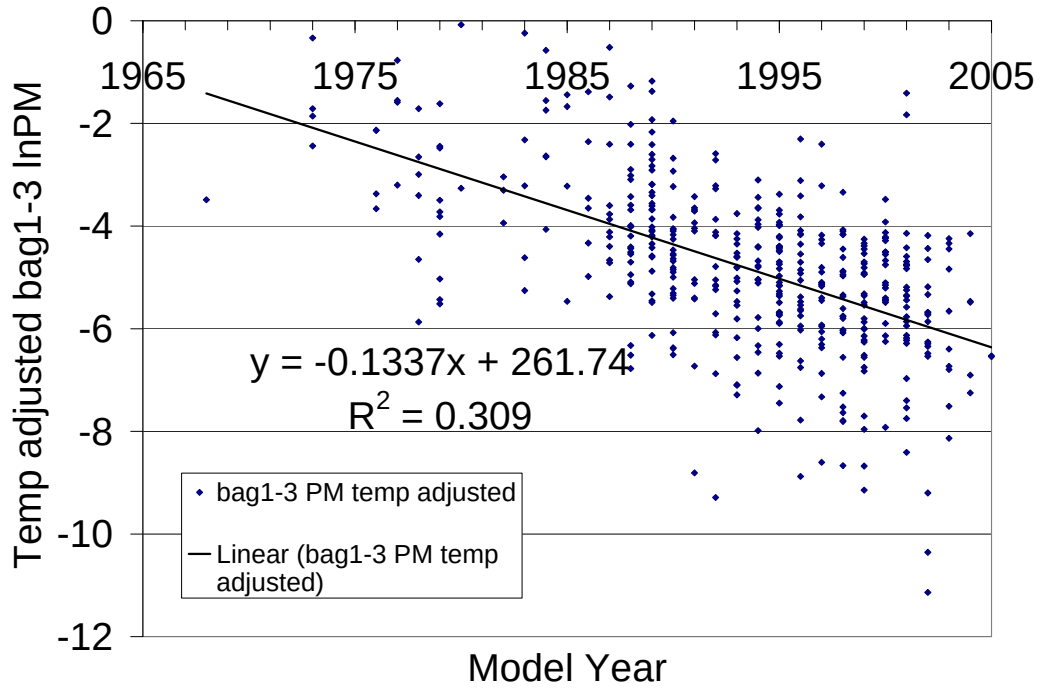


Figure 34. Temperature adjusted natural logarithm of cold start PM as a function of model year

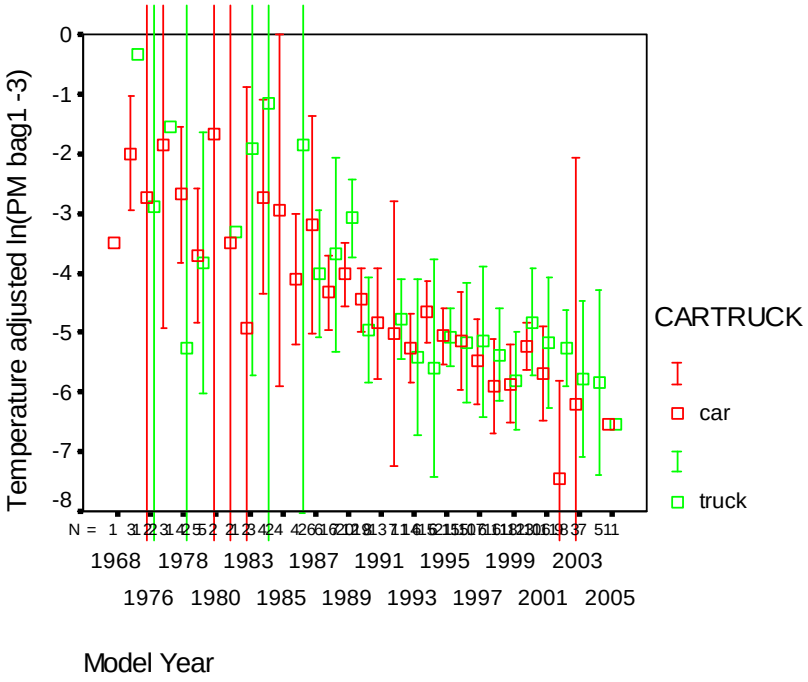


Figure 35. Temperature adjusted natural logarithm of cold start PM as a function of model year bin

8.3 PM to HC Ratios

It is interesting to examine the ratio of PM to HC for the separate bags. This relationship between HC and PM can be used to make decisions about PM in the absence of information, such as projecting future rates, estimating effects on HC control programs on PM, etc. Figure 36 shows this trend as a function of model year for the composite emissions (combined bags). The plot indicates that there is no discernable trend in PM/HC ratio with model year. The mean value is 0.0234 +/- 0.0032 (95%CI). These are not temperature adjusted values.

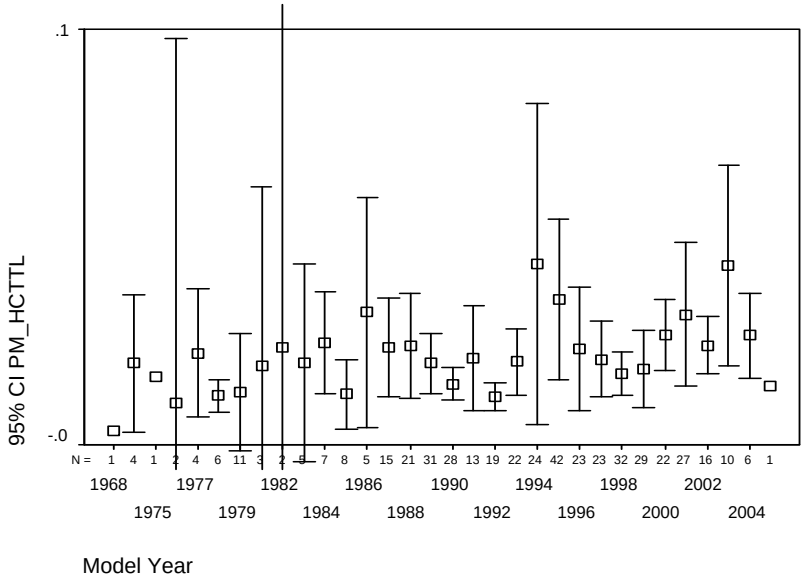


Figure 36. The ratio of PM to HC as a function of model year.

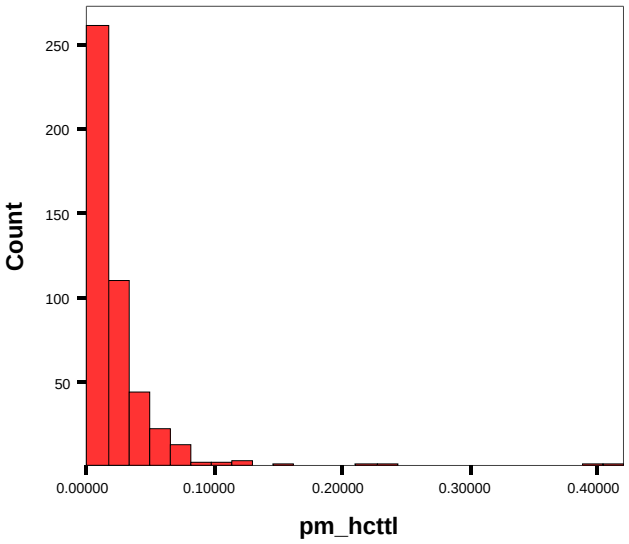


Figure 37. Distribution of PM/HC ratios.

Figure 38 shows the cold start PM:HC ratio as a function of model year. In contrast to aggregate emissions, the ratio for cold starts does seem to be decreasing with model year, i.e. PM is decreasing faster than HC for starts over the years, though it is steady after 1990. Though not shown, this effect is more pronounced in trucks than cars. There are several possible reasons for this:

- The transition from carbureted to throttle body to port fuel injection has allowed for higher fuel and ignition quality during starts, thus potentially reducing PM faster than HC.
- Engines were running richer during starts in older vehicles, thus producing soot.
- Oil consumption during starts may have decreased over the years.

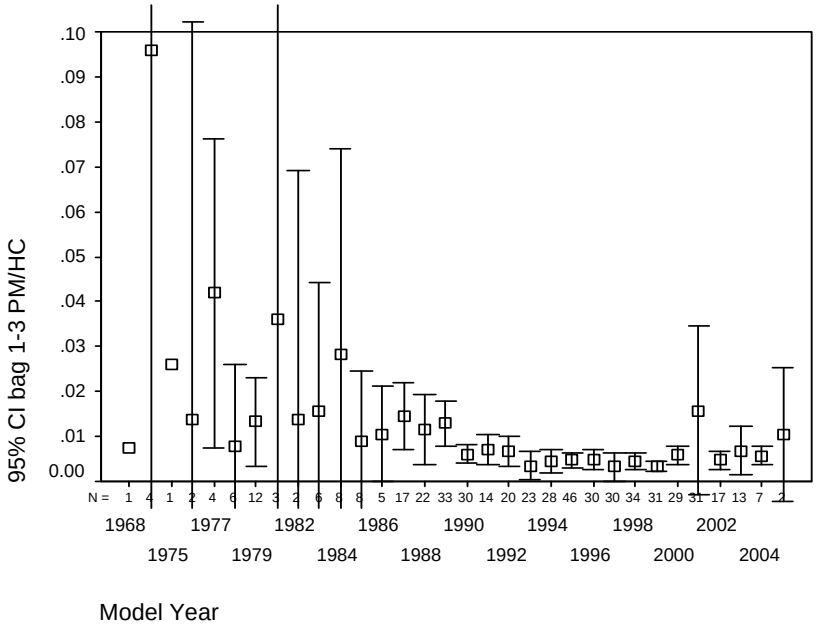


Figure 38. The ratio of cold start PM to HC as a function of model year.

Figure 39 shows bag 2 PM:HC ratio. In contrast to cold start, the ratios for hot running emissions increase with model year. This indicates that while HC is being controlled to an even greater degree over the years, PM may be approaching a limit. This limit may be due to oil consumption, which may not be affected by same control strategies to achieve HC emission standards.

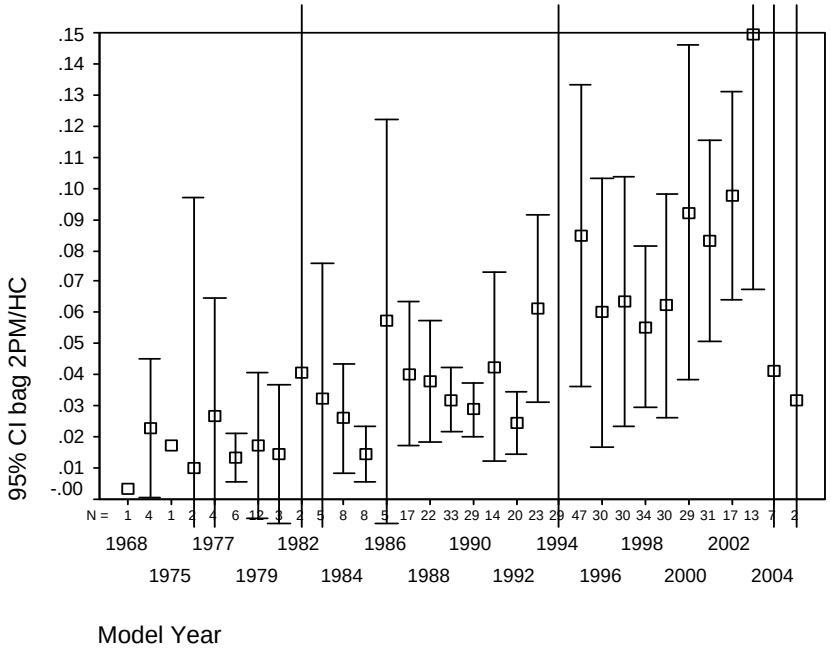


Figure 39. The ratio of bag 2 PM to HC as a function of model year.

To further accentuate the difference between hydrocarbons and PM, we compare the plots of bag 2 vs bag 1 emissions for HC and PM respectively in Figure 40. Clearly the correlation between bag 2 and bag 1 is stronger for HC than it is for PM (correlation coefficient is 0.78 and 0.57 respectively). Thus while it is sometimes advantageous to employ trends in HC to inform and supplement a PM analysis, there are limitations to this approach.

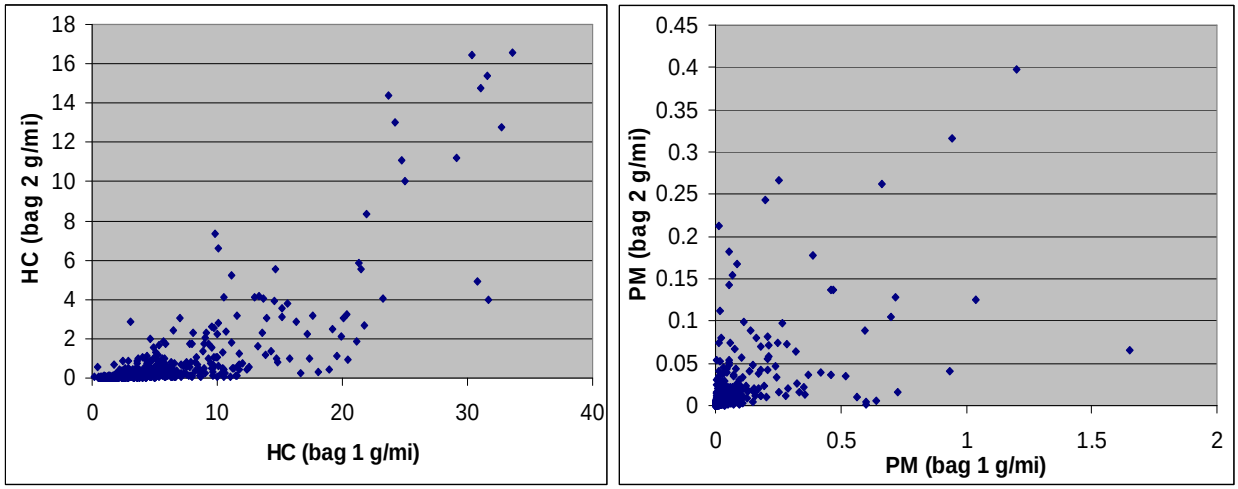


Figure 40. Plots of bag 1 vs bag 2 emissions for HC and PM respectively.

8.4 Elemental (Black) to Organic Carbon (EC/OC) Ratios

Vehicle exhaust particulate matter consists of many different chemical species. Among these are elemental carbon, organic carbon (EC and OC), sulfates, nitrates, trace metals and elements. The vast majority of the PM emissions is in the form of EC or OC. Elemental carbon PM (or soot or black carbon) is produced during combustion when fuel or fuel droplets are pyrolyzed (or carbonized) under low oxygen levels. In this process hydrogen is stripped from the carbon atoms in the hydrocarbon, and carbon soot residue remains. EC is formed in gasoline engines primarily when the fuel air mixture is rich (even in localized portions of the air/fuel mixture of the engine). The hot oxygen-starved and fuel rich environment favors pyrolysis reactions. We might expect to see higher EC fractions in gasoline engines when the engine starts, or when the vehicle goes into an enrichment mode such as under heavy engine load. These fine soot particles are generally non reactive in the atmosphere, though they may act as agglomerization centers for particle growth both in the exhaust stream and in the atmosphere. In other words, other compounds including organic carbon adsorbs onto the surface of the elemental carbon. In turn, these adsorbed organic carbon compounds can react in the atmosphere, generally in an oxidation type reaction. Organic carbon PM are clusters of organic molecules that agglomerate and grow throughout combustion, as the exhaust cools, and finally as it disperses into the atmosphere. In gasoline engines OC can be formed normally during combustion from the fuel or the lubricating oil. Sulfate emissions have largely been controlled through fuel sulfur controls, and, previously, by the closer control of air:fuel ratio necessary for the three-way catalyst to effectively function. We expect the sulfate emissions to be much lower than past studies. Likewise, we also assume that nitrates and trace metals and elements are small on a mass basis by comparison. Therefore, we spend the remainder of this section discussing EC and OC only.

It is important to separate out EC and OC since photochemical models require these separate inputs which our emission models provide. Also, the ratios are helpful for validating emissions (and air quality) models to source apportionment studies. Finally, EC is easier to measure and more stable in the atmosphere than OC, therefore it is useful to track for a variety of purposes.

In the Kansas City study, EC was measured using two different methodologies. The first was through the use of the Thermal Optical Reflectance (TOR) methodology. This procedure also measured OC and total PM, but unfortunately, not all the vehicles in the study were measured using this technique. The other instrument used to measure EC in this program was the photoacoustic analyzer, which measures real-time EC. More information can be found on these techniques and their calibration and comparison results in EPA (2008) and Fujita et al. (2006). The former reference indicates that the photacoustic analyzer has good correlation with TOR EC measurement especially at higher PM levels, however, at lower levels (in bag 3 for example), the correlation is poorer. This is not surprising since all instruments have limited ability to measure small signals. To accentuate the full range of operation, Figure 41 shows a plot of a comparison of the two instruments in log space. The plot reinforces the excellent agreement between the two instruments in bag 1 of the test, when emissions levels are at their highest. The correlation (and slope) is also good for the high values in bag 2, however, as the measurements get smaller, the photoacoustic analyzer seems to be shifted by about 2.4 mg/mi (near the origin of the plot). An

adjustment equation may be appropriate if the TOR is the accepted standard, but since this offset mainly affects small measurements only, it will probably have little impact on emissions inventory models.

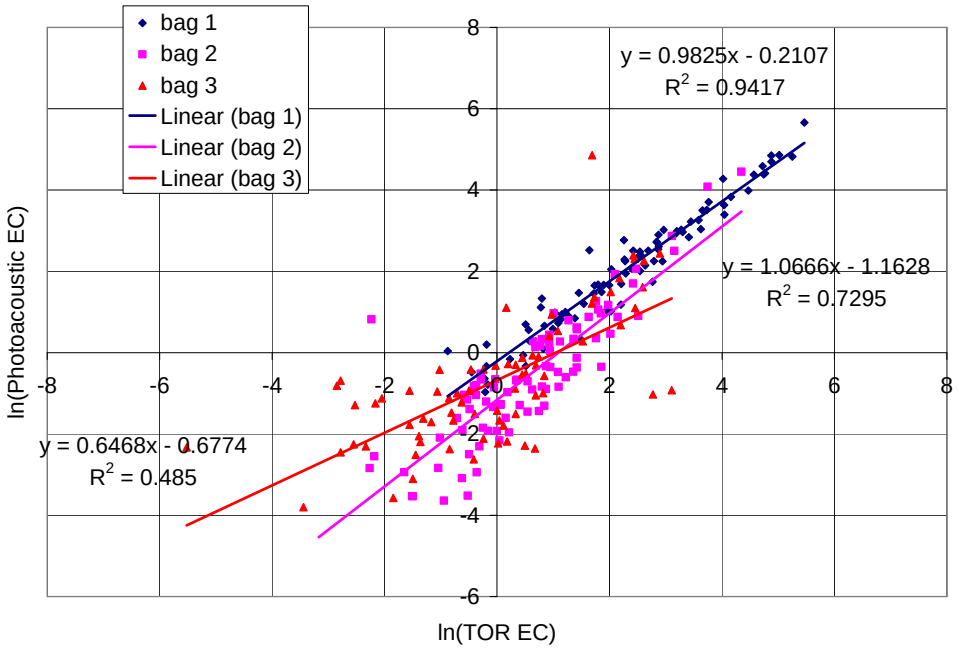


Figure 41. Comparison of Photoacoustic to TOR EC measurements on a logarithmic scale.

We present trends of the ratio of EC to total PM (EC/PM) only. Since in most cases the sum of EC+OC = PM, generalizations can be extended to OC/PM as well accounting, of course, for the inverse relationship between EC and OC. There may be a small amount of non-carbon emission in the PM, but we are assuming that it is negligible for now.

We explore the EC/PM ratio for the four measurement techniques employed in this study: photoacoustic analyzer (PM, measured real-time EC), Dustrak analyzer (DT, measured real-time PM optically), gravimetric filter (PM), and thermal optical reflectance (TOR, which measured both EC and total carbon, TC).

Table 13 shows the comparison of the 3 different ratio methods using these instruments. The values were determined from the ratio of the average values in the numerator and denominator. The TOR ratios have two major limitations: the ratios are unexpectedly high and, after eliminating bad data points, there are only 75 valid measurements. Due to the latter condition (primarily), the TOR ratios will not be used in subsequent analysis. The photoacoustic to dustrak ratios present a reasonable approach, however, since the Dustrak and PM are not perfectly correlated (EPA, 2008), we will use the photacoustic to gravimetric filter ratios for the remainder of this paper.

Table 13. Elemental to total PM ratio for 4 different measurement techniques.

	all	start	running
PA/DT	0.128	0.188	0.105
PA/PM	0.197	0.340	0.164
EC/TC TOR	0.382	0.540	0.339

In the next 3 plots, we look for other factors that may affect the EC/PM variability. Temperature model year and vehicle weight are all examined. Figure 42 shows the relationship between EC/PM to test temperature. These values are averaged for all test values within a 10ºF bin and then ratioed. We conclude from this plot that there is very little temperature dependence to this ratio (though there may be a very small effect for hot running bag 2). Any temperature dependence is miniscule compared to the temperature effects presented earlier for total PM. One might have expected cold start EC ratios to be higher in colder temperatures due to the potential for extended rich starts, however the data does not seem to support this hypothesis.

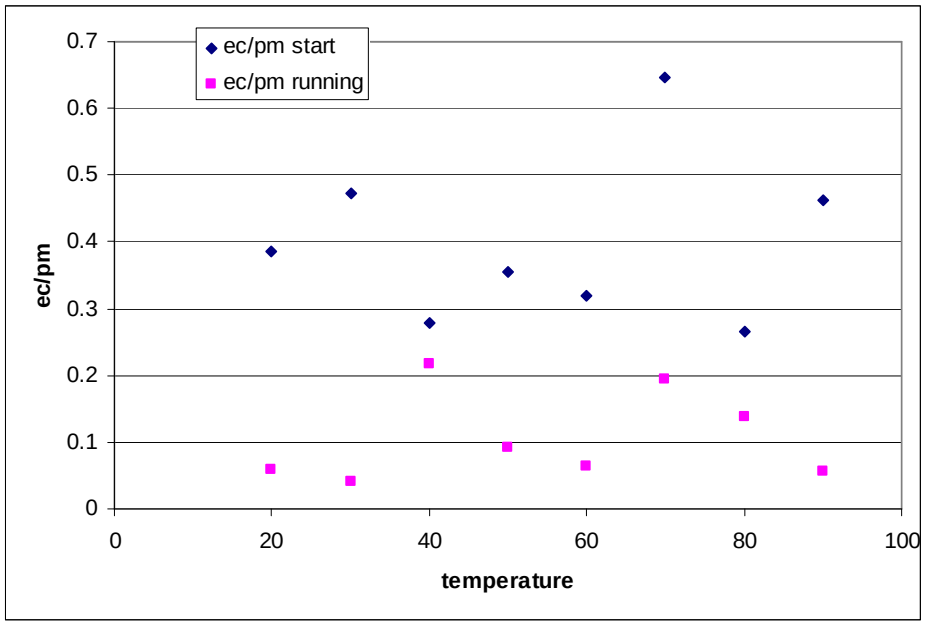


Figure 42. Elemental Carbon to Total PM ratio as a function of test temperature.

Figure 43 shows the EC/PM ratio within model year bins. We conclude from this plot that there seems to be very little model year or age dependence on the EC/PM ratio.

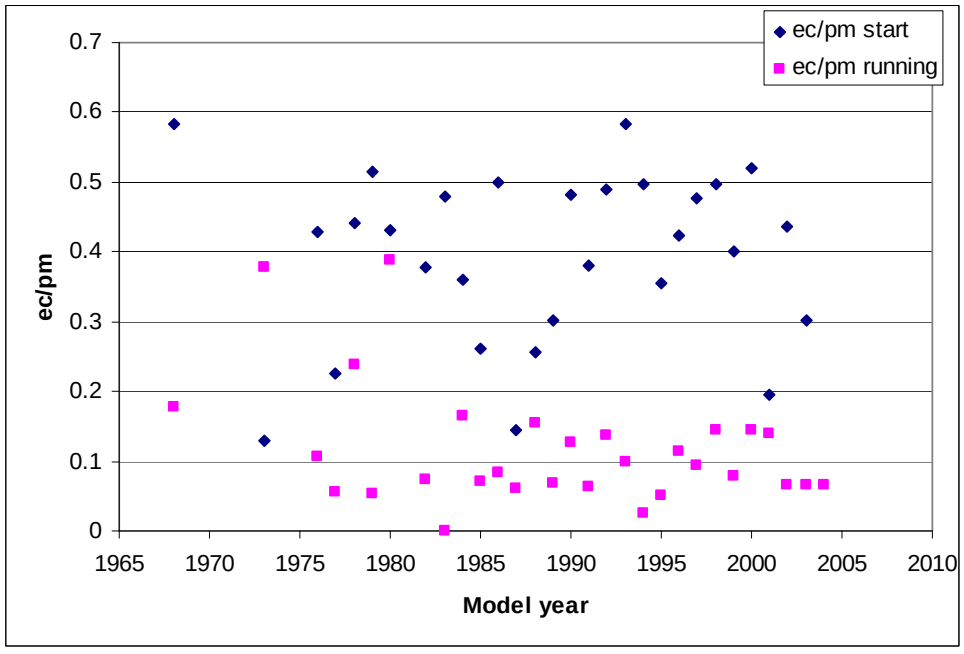


Figure 43. Elemental Carbon to Total PM ratio as a function of vehicle model year.

Figure 44 shows the EC/PM ratio as function of vehicle weight. This plot shows a clear trend of decreasing EC/PM ratio as weight increases. This could be a function of engine displacement (and peak power) as much as vehicle weight (the two tend to be correlated with each other). The trend may also be a function of the drive schedule since lighter (and possibly underpowered vehicles) may be more likely to go into enrichment than more powerful vehicles if driven on identical drive cycles. In subsequent modeling (in MOVES), cars and light trucks are modeled as separate vehicle types, which will capture some of this weight effect.

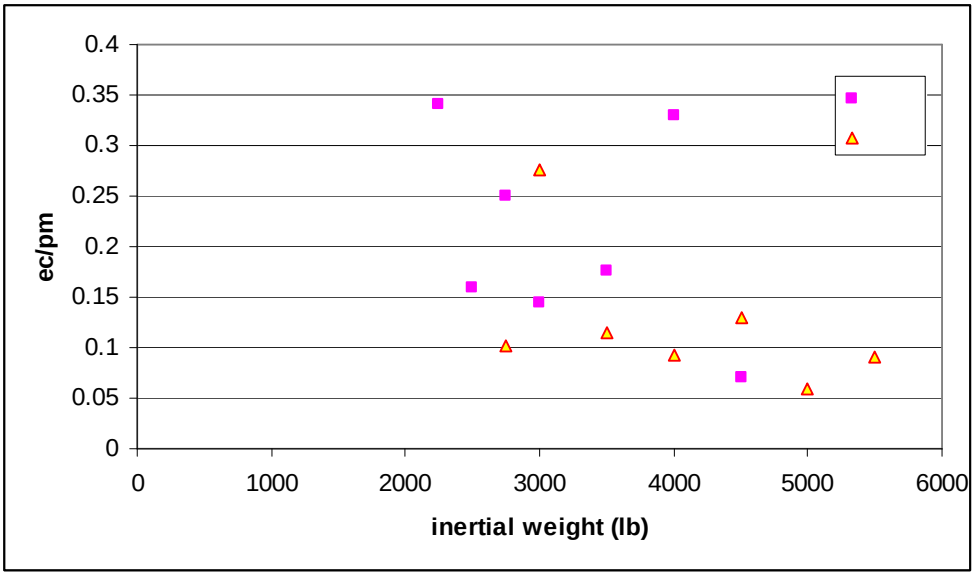


Figure 44. EC/PM ratio as a function of vehicle inertial weight.

An analysis shows the following statistics, with the breakdown of car vs truck in Table 14:

- avg Start EC/PM = 0.337
- avg Running EC/PM = 0.132
- Composite EC/PM ratio, it is 0.173
- The respective OC ratios can be calculated from the above by subtracting the fraction from 1.0.

The significantly higher levels of EC during starts is not surprising given the rich fuel conditions that exist during this mode of operation.

This is roughly consistent with past studies. Cadle et al., (1999) found OC/PM in Denver to range from 61-89%; in San Antonio to range from 53-93% and in South Coast, CA to range from 37-80%. For our analysis, we will use the values found in Kansas City, summarized in the table below. Non-carbon PM are wrapped in with OC and is assumed to be small.

Table 14. Elemental and Organic Carbon PM fractions in from vehicles in the KC study.

Process	EC/PM car	EC/PM Truck	OC/PM car	OC/PM truck
Start	0.345	0.325	0.655	0.675
Running	0.179	0.068	0.821	0.932

9 An Examination of the Representativeness of the Emissions Data

In section 7.1, we addressed the representativeness of the recruited fleet in Kansas City from a demographic point of view, and to the extent possible, assessed the effect of non-response on emissions results. However, these analyses do not definitively resolve the question of how representative the emissions are. Did the study recruit enough vehicles from the upper percentiles of the emissions distribution? This question is much more difficult to address, since there are no definitive data on the frequency of high PM emissions. However, there is a large body of data for the other criteria emissions. In this section, we compare the hydrocarbon results with a much larger measurement program. As noted previously conclusions drawn about PM based on HC data alone should be considered suggestive, though not necessarily conclusive.

The study report from EPA and ERG (2008) presents the results from the remote sensing (RSD) comparison to the dynamometer data. Overall, the results are inconclusive, though suggestive. The study seems to have not recruited enough NOx high emitters, but the CO emissions seems to match between the programs. The hydrocarbon comparisons are not shown. This is likely due to the limitations of remote sensing instrumentations in measuring HC; since the instrument is calibrated to single hydrocarbon specie (e.g. propane), a "correction" is required to estimate total HC (including aromatics), for which there is no true validation. Therefore these RSD data do not directly address the question of how the PM emissions in Kansas City compare to the national fleet. However, because some of the same processes that cause high CO emissions also cause high PM emissions (e.g. engines running rich), the comparison does suggest that because the CO emissions matched, KC may have recruited sufficient quantities of these types of high emitters.

We search for further evidence within the Inspection and Maintenance (I/M) database. A large set of HC measurements is available from evaluation samples collected in the Phoenix I/M program. For purposes of this analysis we used random evaluation tests performed during calendar year 2004, including 3,482 LDV and 3,036 LDT. We used the second and third of three replicate IM147s for each vehicle, for tests measured at ambient temperatures of 68-86 F, and appropriately weighted to reflect differential sampling frequencies for passing and failing tests in the stratified evaluation sample.

It is to be expected that the average emissions from Arizona (AZ) and Kansas City would differ for a variety of reasons. The following is a list of some of these differences:

- Test cycle, (IM147 for AZ and LA92 for KC)
- Vehicles in AZ are subject to I/M, which affects their HC emissions
- Temperature and climate differences (although we have attempted to neutralize these)
- Fuel
- Dynamometer and bench instruments
- Dynamometer loading

Most of these differences will affect the means (averages), therefore any comparison of these means will be qualitative until at least such time that the test cycle differences are taken into account (in a future publication). However, we do not expect that these differences will have a

large impact on the relative widths of the respective distributions, which reflects the length and weight of the tails of the skewed emissions distributions. The two largest potential exceptions to this could be temperature and cycle. To isolate the effect of temperature, we compare bag 2 of the KC LA92 (which is stabilized) to the I&M tests on the FTP temperature range.

Before comparing the two programs, it is first necessary to draw the analogy between HC and PM distributions. Figure 45 shows that the width (standard deviations) of natural-log transformed HC and PM distributions are quite similar (within 10%) for the model year 1996-99+ group.

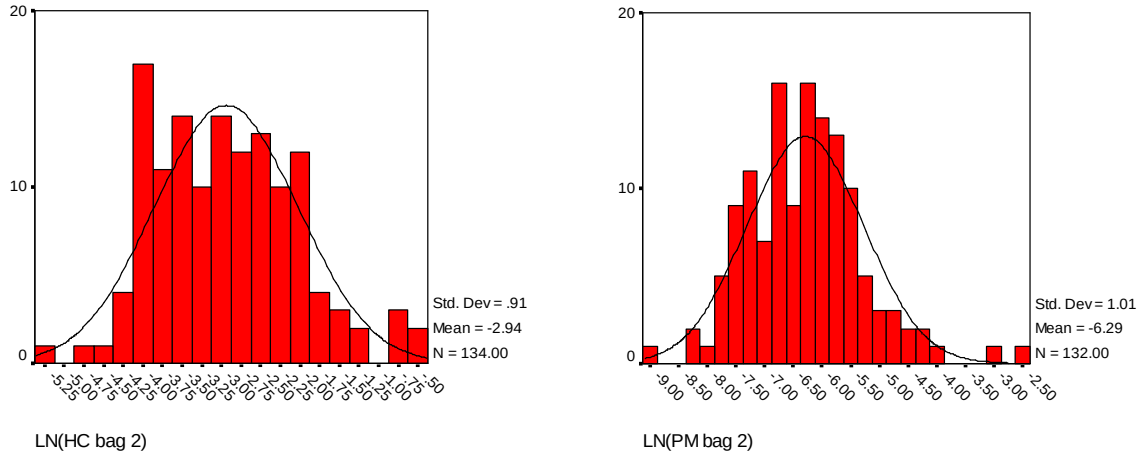


Figure 45. HC and PM distributions in Kansas City for model years 1996+ with normal curves overlaid.

Table 15 shows that all of the model year groups (with slight adjustment to the model year groups) are similar though there appear to be some random differences. These similarities seem to hold regardless of whether the PM results are temperature-adjusted or not. The results also seem to indicate that the widths of the distributions are consistent across the model year and car/truck divisions. Though not shown, if we were to plot the standard deviations of PM vs HC for the KC results, we would have a slope of 1.12 and an R^2 of 0.44 (corresponding to a correlation coefficient of 0.73).

Table 15. bag 2 HC and PM comparisons in Kansas City.

		ln (hc bag 2)			ln (pm (bag 2)			ln (pm bag 2 temp adjusted)		
car/truck	**MY group**	**mean**	**st dev**	**n**	**mean**	**st dev**	**n**	**mean**	**st dev**	**n**
car	**<84**	0.65	1.36	32	-3.71	1.31	33	-4.09	1.36	33
car	**85to89**	-0.51	1.14	54	-4.37	1.22	53	-4.55	1.20	53
car	**90to94**	-1.45	1.19	81	-5.16	1.33	80	-5.58	1.36	80
car	**95to99**	-2.52	0.88	108	-5.94	1.24	104	-6.33	1.18	104
car	**2000+**	-3.36	0.81	51	-6.44	0.86	52	-6.78	0.83	52
truck	**<84**	0.91	1.23	17	-3.48	1.42	16	-4.01	1.44	16
truck	**85to89**	-0.19	0.98	32	-4.11	1.28	32	-4.72	1.38	32
truck	**90to94**	-1.08	1.15	38	-4.95	1.23	37	-5.55	1.10	37
truck	**95to99**	-2.01	0.99	70	-5.4	1.1	68	-5.99	1.17	68
truck	**2000+**	-3.09	0.79	46	-5.78	0.94	47	-6.38	0.98	47

Having established that PM and HC distribution widths seems to be correlated, we now compare the HC from KC to the AZ I&M data. The comparison is shown on Table 16. Due to the exemption that most newer vehicles obtain from I&M tests, there is very little data from model years 2000 and later in the 2004 calendar year, so this bin is left out of the comparison. Despite the differences, it is interesting to compare the linear means from the two programs. The linear means differ from the logarithmic means by a factor of $\exp(\sigma^2/2)$, and this comparison is show on Figure 46. Surprisingly the two programs compare well with the notable exception of the two oldest model years, where KC is significantly higher. A detailed comparison of these means will be saved for a future publication.

We can now, however, compare the widths of these distributions. It is clear from the table, that the width of the HC distributions are getting smaller (narrower) for the later model years, whereas the opposite is occurring in Arizona. These are both statistically significant trends (in the opposite direction) to within a p-value of 0.05. This is a rather remarkable result, it implies that in Kansas City, there may have been an excessive number of high emitters recruited in the older model years, but an insufficient number in the later model years. Here "high-emitters" is a relative term, meaning high compared to the mean, so that a high emitter in model year 2000 may very well have lower emissions than a low emitter from model year 1980. The Arizona dataset is known to have a small number of incidence of data "plateaus" throughout the dataset (EPA, in publication). For this reason, we cannot draw any conclusions about whether Kansas City recruited excess high emitters, however we may be able to conclude that the KC study recruited a sufficient number of them in the older model years.

Table 16. Logarithmic means and standard deviations HC from bag 2 of the LA92 in KC vs the IM147 in Arizona. Due to the logarithmic calculation "means" are actually "medians".

		KC ln (HC bag 2 LA92)			AZ ln (HC IM147)		
car/truck	**MY group**	**mean**	**st dev**	**n**	**Mean**	**st dev**	**n**
car	**<84**	0.65	1.36	32	-0.35	1.17	202
car	**85to89**	-0.51	1.14	54	-0.95	1.17	763
car	**90to94**	-1.45	1.19	81	-1.59	1.35	1130
car	**95to99**	-2.52	0.88	108	-3.56	1.77	1379
car	**2000+**	-3.36	0.81	51			
truck	**<84**	0.91	1.23	17	0.46	0.89	233
truck	**85to89**	-0.19	0.98	32	0.005	0.97	709
truck	**90to94**	-1.08	1.15	38	-0.90	1.43	744
truck	**95to99**	-2.01	0.99	70	-2.78	1.69	1334
truck	**2000+**	-3.09	0.79	46			

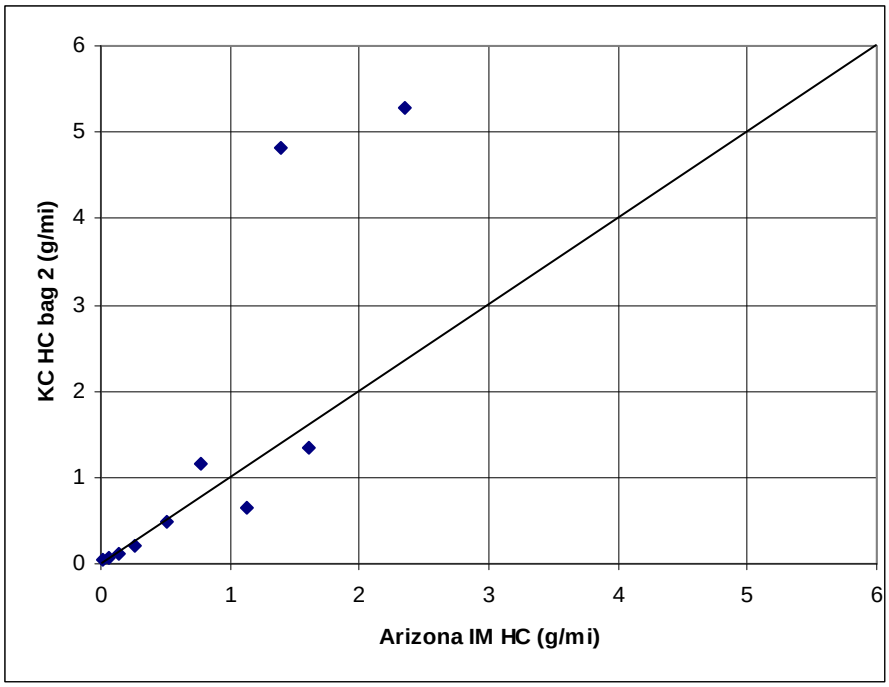


Figure 46. A comparison of the HC emissions (by model year group and vehicle type) between Kansas City and Arizona (with a 1:1 line included).

It is not unreasonable to assume that emissions distributions can broaden with newer model years. Tight fuel controls in combination with effective aftertreatment have reduced emissions by more than a factor of 100 compared to pre-control technologies (in the 60's and 70's). Because of this, if older vehicles emission control systems malfunction (and become high emitting), their emissions could increase by at most several factors of 10. However, newer malfunctioning vehicles can have emissions increases by factors of 100, or more. Thus it is plausible that emissions distributions broaden with newer model year vehicles.

However, it is possible that the instrumentation and testing is different enough to cause this increasing width trend. To explore this hypothesis, we present the results of the other pollutants from the two programs in Table 17. The comparison of the means are shown in Figure 47, where CO reflect the same conclusions as HC, but the NOx correlation is decent. Carbon monoxide (CO) trends often mirror that of HC due to the similar mechanisms of formation during combustion and exhaust aftertreatment. Comparing the widths, in KC the slope of the standard deviation from CO is not statistically discernable from zero, however, in Arizona, there is a significant increasing slope with model year (similar to HC). Hydrocarbons were measured with flame ionization detectors (FIDs) in both programs, whereas, the carbon monoxide was measured with non-dispersive infra-red detectors (NDIR). FIDs have significant issues with background corrections and instrument drift, which, could bias low level measurements however, the fact that both the CO and HC distributions show increasing widths in Arizona suggests that the background issue is likely not the root cause of the increasing variability. However, it is possible that both the HC and CO instruments are limited in their ability to measure low emissions. The measurement methods are very different between the two programs. In KC, HC was measured from a bag (cumulative emissions) and I&M programs measure modal (or second by second) emissions. The bag measurements can be extremely accurate, to within 1 ppm (CFR40, part 86).

The modal measurements have an analyzer accuracy of 12% at the most sensitive setting and for extremely low HC measurements, the uncertainty in each second of data is compounded for the entire test (CFR40, part 51m subpart S, Appendix D). At this time, it is impossible to prove whether the instrument limitations can cause this width widening at lower levels, or whether it is a real phenomenon, or both. However, we proceed with a brief discussion of the effect this width difference might have on the KC data if it does indeed reflect a lack of high emitter recruitment in the newer model year groups.

Table 17. Logarithmic comparison of the means and standard deviations of CO, NOx and CO2 from bag 2 of the LA92 in KC vs the IM147 in Arizona.

		KC						AZ					
		lnCO	lnCO SD	lnNOx2	lnNOx2 SD	lnCO2	lnCO2 SD	lnCO	lnCO SD	lnNOx2	lnNOx2 SD	lnCO2	lnCO2 SD
car	**<84**	3.47	1.23	0.58	0.75	6.06	0.24	2.50	1.25	0.12	0.89	6.00	0.22
car	**85to89**	2.46	0.96	0.36	0.67	5.95	0.21	1.68	1.55	0.04	0.83	5.86	0.23
car	**90to94**	1.58	1.16	0.02	0.76	5.96	0.18	1.23	1.38	-0.13	0.95	5.81	0.21
car	**95to99**	0.78	1.16	-0.71	0.74	5.9	0.16	-0.07	1.65	-1.20	1.11	5.78	0.16
car	**2000+**	-0.3	1.36	-2.22	0.9	5.93	0.16						
truck	**<84**	3.56	1.27	0.76	0.64	6.23	0.29	3.20	1.12	0.95	0.84	6.29	0.28
truck	**85to89**	2.18	1.47	0.85	0.43	6.06	0.17	2.64	1.30	0.78	0.88	6.28	0.33
truck	**90to94**	1.86	1.16	0.45	0.63	6.09	0.18	1.82	1.35	0.46	0.93	6.19	0.23
truck	**95to99**	0.9	1.05	-0.32	0.88	6.16	0.1	0.42	1.54	-0.54	1.03	6.16	0.22
truck	**2000+**	-0.18	1.17	-1.53	0.82	6.16	0.1						

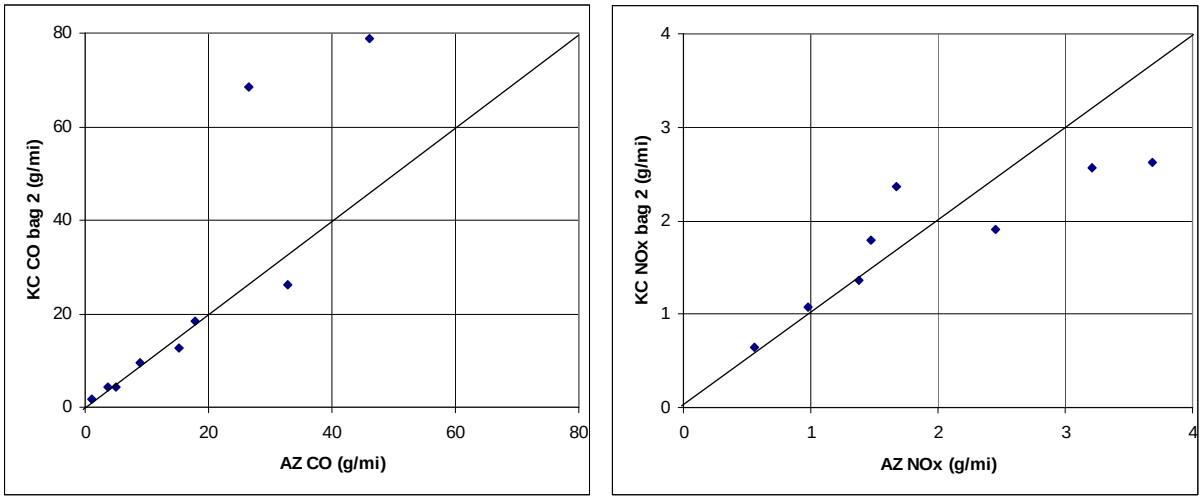


Figure 47. Comparisons of the CO and NOx emissions (by model year group and vehicle type) between Kansas City and Arizona (with 1:1 lines included).

The HC log-normal distributions for the 1995-1999 model years are compared in Figure 48, with the logarithmic means set equal. Of all of the model year groups, this group differs the most between the two programs. On a natural-log transformed scale, the distributions appear "normal," With the Arizona distribution approximately twice as wide as its counterpart from KC. Figure 49 shows the same distributions on a linear scale. The differences in the tails of this distributions show up much more clearly, where the mean of the AZ emissions is approximately

3 times larger than that for KC. At this time, we are not advocating this adjustment to the data since this is a modeling exercise reserved for the modeling paper, and it should have a more significant effect on future inventory projections rather than the present estimates shown later in this paper (Nam, 2008).

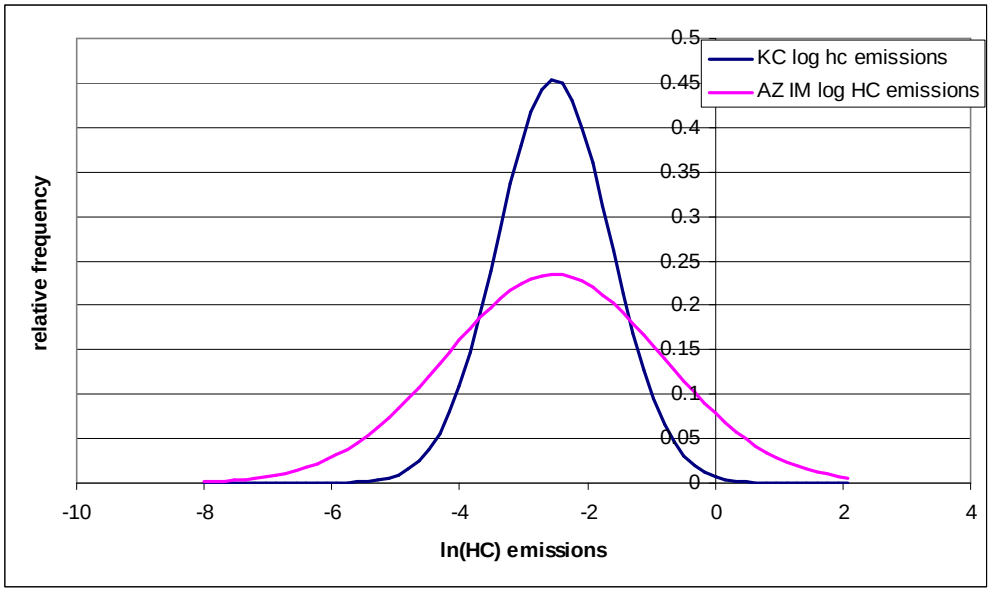


Figure 48. A comparison of the log-"normal" distributions of hot running HC in Kansas City vs Arizona.

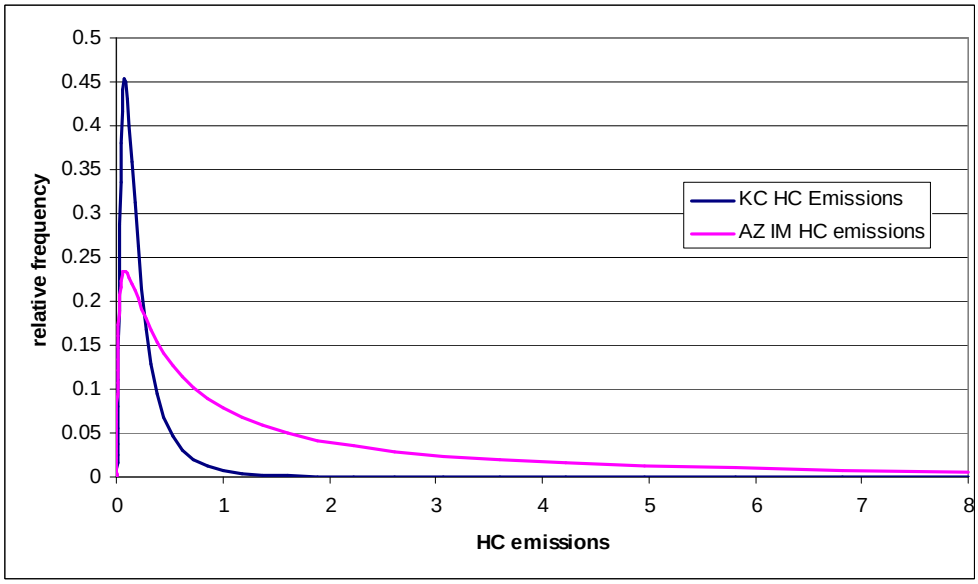


Figure 49. log-normal distributions of hot running HC from KC and AZ on a linear scale.

If the emissions differences between the two programs are real, then it is quite likely that the recruitment in Kansas City failed to capture of the upper reaches of the distribution (higher emitters) for the newer vehicles. Though it should be reiterated that the emissions in these "high-emitters" are still lower than the average emissions from the older vehicles, so to call them "high-emitters" is merely a relative term in comparison to their corresponding means. Another conclusion we may draw from this analysis is that it may be extremely difficult capturing these newer high-emitters by random sampling; from the fleet as a whole; the efficiency of sampling

may be substantially improved by use of a screening tool that provides a reasonably accurate index of emissions (e.g. I&M lanes, visible smoker, remote sending, etc). On the other hand, the random recruitment of older high-emitters seems to have been quite successful, and this issue should have little impact of present-day inventories.

10 Preliminary Inventory Results for Calendar Year 2002

Much of the emission rates from this report have been converted into emission rates for a recent calendar year for comparison with previous inventory models. An inventory result based on Kansas City data would only be valid for calendar years near 2004 or 2005 when the data were collected. Any other year would require estimations of deterioration rates and thus would be a modeling exercise described in another paper (Nam, 2008). The purpose of this analysis is to give a preliminary look at how the Kansas City results would translate to emission inventories. It should be noted that the rates that go into MOVES will have updated this work, including factors such as vehicle specific power-based rates and updated fuel effects; hence this work can give a sense for general trends but should not be considered the draft MOVES rates

For the purposes of this comparison, the MOVES model was recoded to accept aggregate emission rates in gram per mile units. The cold start and the hot running emission rates were separated in the model as they are in this report with the added detail that they are distinguished by the MOVES model year and age groups. The cold soak adjustments (for soak times less than 24 hours) were done by using the HC soak curves in MOVES (EPA, 2008 in publication). The temperature adjustments are conducted on each test to 75˚F in order to establish the baseline rates, which are in the appendix. Finally the EC, and OC rates were split from PM using the ratios presented earlier in this paper.

The remainder of this chapter compares MOVES results incorporating the new PM2.5 model based on the Kansas City data with NMIM runs based on MOBILE6.2.03. The MOVES runs were performed at the state level for all states. The NMIM runs were performed for the 2002 National Emissions Inventory (NEI) for all counties. NMIM (National Mobile Inventory Model) essentially runs MOBILE6.2 (EPA, 2005). The comparisons to NMIM, therefore, are comparisons to MOBILE6.2. Both models were executed for all months, 2002 temperatures, and P5VClasses LDGV, LDGT1, LDGT2. Fuel properties were the same for both models. MOVES runs were performed separately for each state, including the District of Columbia, and for each month, pre-aggregated to the state and day level. Other states used identical parameters, except for the state.

These MOVES runs are preliminary. Ongoing analysis is expected to result in changes to the emission factors. Furthermore, we have not assessed the effects of aggregating county temperatures to the state level or of aggregating 24 hourly temperatures to a single daily temperature. The MOVES RunSpec resulted in an Execution Database that contained a zonemonthhour table with a single temperature for each state-month. The aggregation method in MOVES weights by VMT (EPA, 2007).

The NMIM runs were executed for the 2002 NEI Version 3 (EPA, 2007). These runs were performed separately for each county and each month. The version of NMIM was NMIM20061128. The NMIM County Database (NCD) was NCD20061227pf02v3. This database is the same as NCD20061227, except for fuels files developed for the 2002 NEI (EPA, 2007). All of the RFS and 2002 NEI changes are incorporated into NCD20070912.

In the following comparisons, the emissions of LDGV, LDGT1, and LDGT2 were summed. To insure meaningful comparisons, MOVES inventories were produced by multiplying MOVES emission factors times NMIM VMT. MOVES emission factors were calculated by dividing MOVES emissions by MOVES VMT. (Although the two models agree with respect to VMT nationally, there are significant differences between them for some states, which we have not yet resolved.) The national annual ratio of MOVES to NMIM PM2.5 is 1.56. Ratios of MOVES to NMIM inventories on different spatial and temporal scales are shown in Table 18.

Table 18. Ratios of MOVES to NMIM inventories on different spatial and temporal scales.

	Minimum	**Maximum**
National Monthly	0.76 (July)	2.75 (December)
State Annual	0.76 (Florida)	3.38 (Alaska)
State Monthly	0.58 (Florida, July)	6.20 (Minnesota, January)

The groups of figures below show that MOVES emissions are usually higher than NMIM emissions for cold months and lower for warm months. This pattern results from the fact that MOVES PM2.5 is highly temperature sensitive and MOBILE6 PM2.5 is independent of temperature. Therefore, when plotting emission factors against months, MOVES presents a U-shaped pattern (cold at the beginning and end of the year and warm in the middle), and NMIM emission factors plot as a horizontal line. The slight deviations of NMIM from a straight line are the result of the emissions being a composite of LDGV, LDGT1, and LDGT2, which have different PM2.5 emission factors and slightly different usage patterns over the year.

Note that these results alone cannot be used to predict actual local impacts of new PM emission estimates using MOVES in a state implementation plan (SIP) or transportation conformity analysis. While they do incorporate some estimation of local inputs, they are not based on the actual local inputs that states will use in this analysis. As noted above, they are only valid for 2002, the year of analysis. The shape of the curves and degree of difference from MOBILE6.2 may be different in future years. Finally, MOVES will also incorporate other changes, such as new heavy duty diesel PM emission factors, which further complicate any attempt to predict the overall impact of the analysis described here on future SIP and conformity analyses.

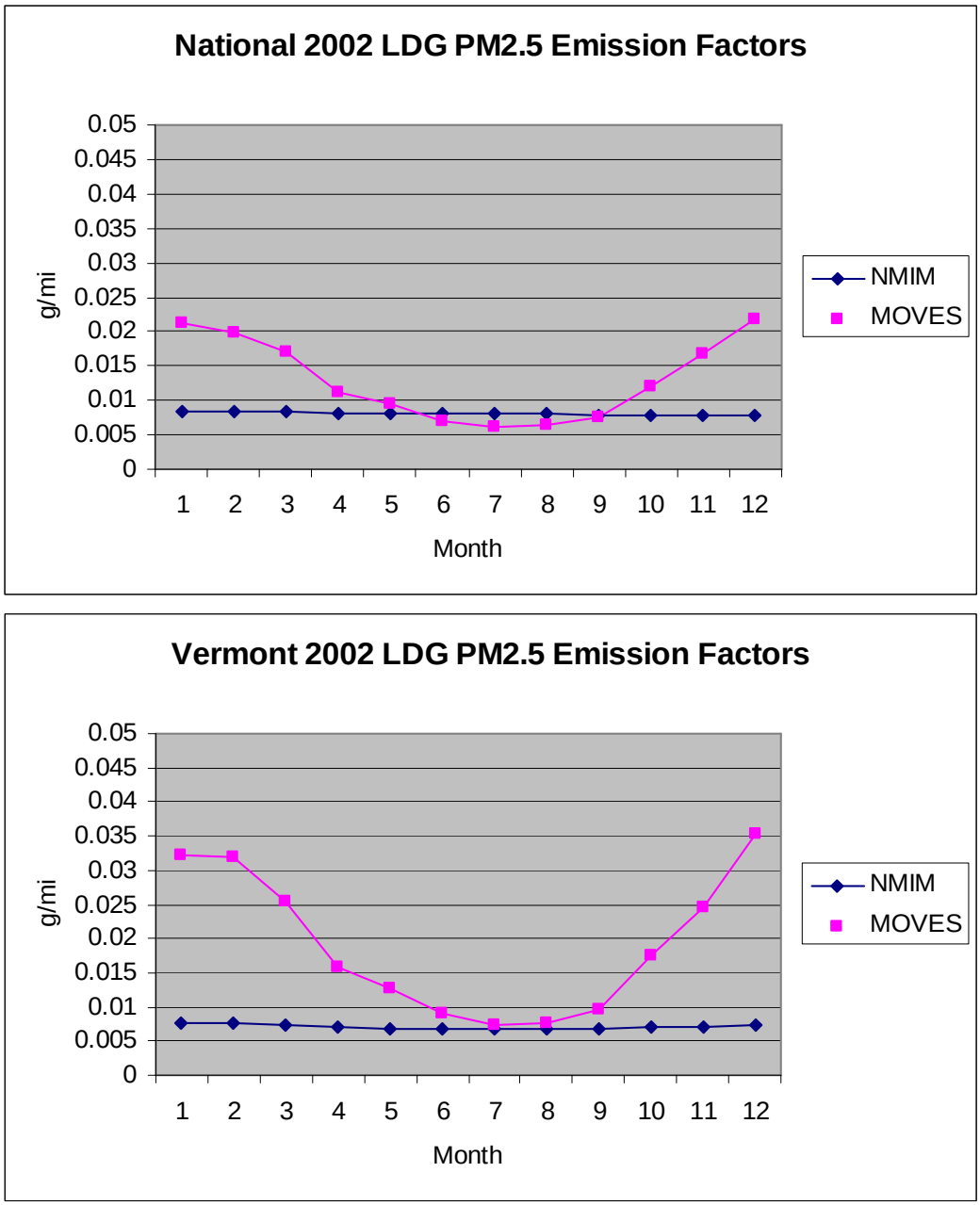
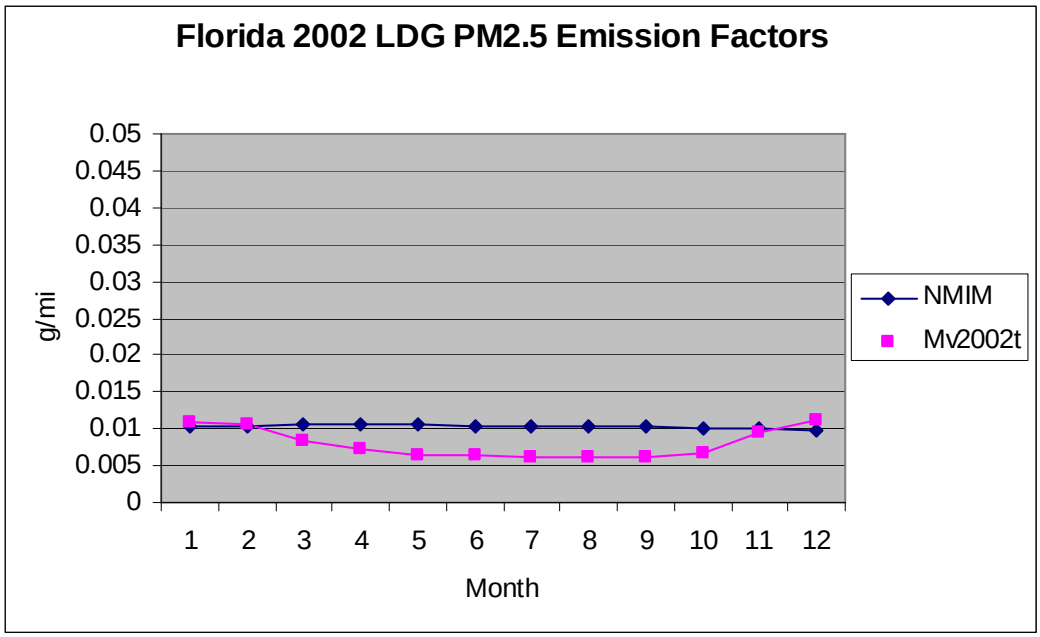


Figure 50. MOVES-NMIM Comparison: National Average and a Cold State.



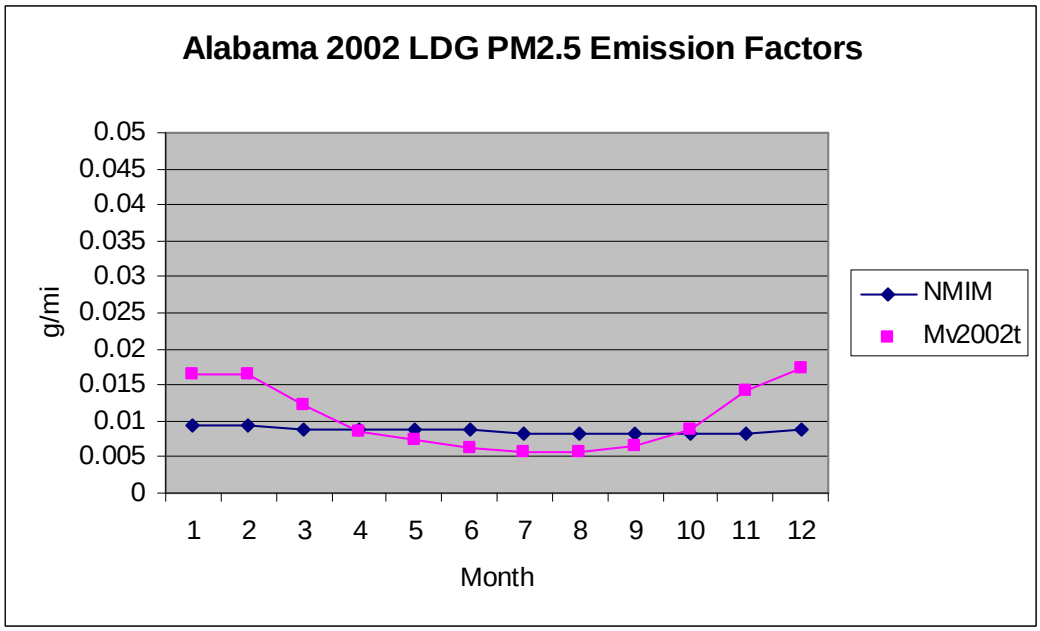


Figure 51. MOVES-NMIM Comparison: Two Southern States.

11 Conclusions

The results in this publication describe a small, but significant, portion of the KC dataset. It should be noted that particulate matter 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.

ajor efforts were taken to try to establish a random sampling methodology that would be representative of the KCMA and therefore allow EPA to conduct further investigation into its application for use in modeling emissions. The first goal was to recruit a random representative sample of vehicles from the fleet. The KC cohort used had a very good correlation with Census 2000 socioeconomic demographics. Our analysis between respondents and people who were converted from non-respondents failed to disprove a difference in PM emissions between the two groups. We can conclude that non-response was motivated largely by reasons other than perceived high emissions from their vehicle.

A comparison between the hydrocarbon emissions from Kansas City and Arizona (Inspection and Maintenance) data provided further evidence that sufficient "high emitters" were recruited from the older vehicles. However, there is some question as to whether this can be concluded about the newer (and cleaner) vehicles. If the spread in the Arizona emissions with later model years is not solely an instrument artifact, then it is likely that the study failed to capture the higher PM emitters from the newer vehicles. Further analysis will be conducted to explore this phenomena and its effect on the fleet representativeness and emission rates.

The emissions analysis in this study explored several dimensions of the dataset. These included: general result comparison with past studies, high emissions results, temperature trends, model year/age trends, PM correlation with other pollutants, cold start vs hot running trends, cold start and hot running trends, PM/HC as well as EC/PM (elemental carbon PM to total carbon) ratio trends. In general the vehicles in Kansas City were found to emit about the same level of emissions as equivalent model year vehicles in previous studies. This is rather surprising given the time difference between the studies.

One of the original goals of the study was to define and quantify high PM emitters. There are several potential ways to define a high emitter. The most common definition employed in the literature is based on the degree to which the vehicles are emitting visible smoke. The results from Kansas City indicate that 5% of the vehicles tested (in the summer phase) were found to be smokers. The average (population weighted) emissions of these vehicles is 43 mg/mi. Though smoking vehicles have higher emissions on average, it is found that not all of the highest emitters are smoking, and not all of the smokers were emitting very high emissions. Of the 14 highest emitters in the summer, only 2 of them are visibly smoking. However, it is important to note that vehicles that smoke at idle may or may not smoke when running and vice versa. This poses potential problems with using "smokers" as a classification scheme for "high emitters". This clearly requires further examination. The analysis for defining a break point for a high PM emitter will be reserved for a future study.

In this test program, it is found that PM emissions are highly sensitive to ambient temperatures. The emissions increase exponentially as temperature decreases so that for a 20ºF drop, PM doubles. This relationship is especially pronounced in cold start and not as severe during hot running operation. This behavior is consistent with what other test programs have shown. Since temperature is an uncontrolled variable in this dataset, the effects of this variable were isolated through a systematic analysis of the temperature effects on PM. A model is developed so that adjustments to any temperature are possible. This allows for subsequent analysis where temperature is effectively "controlled".

Further analysis indicates that the correlation of PM with hydrocarbons (HC) is stronger than for the other criteria pollutants. Despite this, measurements for individual vehicles are scattered very widely around the trend in averages. Thus, it does not appear that HC serves as direct surrogate for PM for individual vehicles, but it can be useful on a limited basis to understand trends if there is a lack of data on which to make further judgments. It is interesting to view emissions trends with respect to model year and age. For cold start PM, the decline in emissions up to model year 1990 appears more rapidly than for HC. In contrast, hot running PM emissions decline more slowly than HC. To explain the results, for cold start emissions, it is likely that improvements in emissions controls technologies have had a greater impact on HC during starts than on PM. For hot running emissions, this result is likely due to the fact that HC emissions have been significantly controlled (through regulations and technology changes), while PM emissions have not. This trend may also be indicative of a relatively constant level of oil consumption (and burning) in the fleet over the years. It should be noted though, that while PM is best correlated with HC compared to the other pollutants, there is significant scatter and any extrapolations of PM trends through the use of HC data should be done with care.

An analysis of the elemental carbon PM ratio to total carbon found that the average EC/PM ratio is about 20%. The ratio is higher during start emissions than hot running. It is also higher for cars compared to light trucks. Interestingly however, the ratio was not found to be highly sensitive to temperature or model year (technology).

A PM model combining the effects of temperature, model year and age is presented for calendar years 2002. Compared to MOBILE based (or NMIM) inventories, PM inventories are significantly higher in the colder weather months and regions of the countries for MOVES compared to MOBILE. On average for the nation (over the year), the inventories are estimated to be about 1.6 higher than MOBILE. It is important to keep in mind that PM from light-duty gasoline sources only form a fraction of the overall PM inventory, where stationary, non-road, diesel, road dust, wood-burning, and many other sources (natural and man-made) also play a significant role. However, even for light-duty gasoline PM, there is much work to be done before a final estimate of inventory impacts can be determined.

There is still much analysis that can be conducted with the data. EPA will be continuing to investigate factors that contribute to or reduce the formation of PM within its test facilities. EPA has documented the PM variability on other emission studies conducted between different laboratories and also exhibited with its reference vehicle when it was correlated between the two different testing facilities. EPA continues to explore these factors that can contribute to or

reduce PM formation especially testing methodologies, procedures and equipment site differences such as dilution rates, tunnel structure, sample probe location, resident times, dilution temperatures, etc) to achieve more consistent results. It is also important to resolve the differences between Kansas City and the more numerous (and presumably more representative) inspection and maintenance data. These factors may have a significant impact on both regional and national PM inventory estimates from gasoline mobile sources. In the future, it would also be important to examine trends in the speciated hydrocarbons and organic PM from the standpoint of toxic emission and also quantifying the PM emissions due to oil consumption. This is likely to expand the scientific understanding of PM formation and why some gasoline fueled vehicles emit more PM than others under certain conditions. For the modeling, it is important to understand the modal or load-based behavior of PM. An analysis of speed or modal effects on PM is not discussed in this paper but is critical for a full integration into a model like MOVES. Finally a full model including model year and age (deterioration) effects is required in order to generate inventories from the past and into the future.

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13 Appendix. PM2.5 Base Emission Rates for Passenger Cars and Trucks for calendar year 2005.

Table 19. The hot running PM2.5 rates for passenger cars & trucks circa calendar year 2004-2005. The rates beyond model year 2005 are identical to 2005. g/hour running rates can be converted to g/mi by multiplying by the average speed in LA92 (27.6mph) to obtain a constant g/mi rate across all speeds.

	g/hr	g/hr	g/start	g/start		g/mi	g/mi	g/mi	g/mi	g/mi	g/mi
Model Yr	car hot	truck hot	car start	truck start		car hot	truck hot	car start	truck star	car	truck
1980	1.1445	1.2245	0.1318	0.1816		0.0415	0.0444	0.1089	0.1500	0.0546	0.0625
1981	1.1305	1.2045	0.1308	0.1791		0.0410	0.0436	0.1081	0.1480	0.0540	0.0616
1982	1.1174	1.1860	0.1299	0.1768		0.0405	0.0430	0.1073	0.1461	0.0535	0.0607
1983	1.1053	1.1687	0.1290	0.1747		0.0400	0.0423	0.1066	0.1444	0.0529	0.0598
1984	1.0940	1.1527	0.1282	0.1727		0.0396	0.0418	0.1060	0.1427	0.0525	0.0590
1985	1.0836	1.1379	0.1275	0.1709		0.0393	0.0412	0.1054	0.1412	0.0520	0.0583
1986	0.5024	0.4432	0.0454	0.0391		0.0182	0.0161	0.0376	0.0323	0.0227	0.0200
1987	0.4934	0.4304	0.0448	0.0375		0.0179	0.0156	0.0370	0.0310	0.0224	0.0193
1988	0.4851	0.4185	0.0442	0.0360		0.0176	0.0152	0.0365	0.0298	0.0220	0.0188
1989	0.4773	0.4075	0.0437	0.0347		0.0173	0.0148	0.0361	0.0286	0.0217	0.0182
1990	0.4701	0.3973	0.0432	0.0334		0.0170	0.0144	0.0357	0.0276	0.0214	0.0177
1991	0.2312	0.2561	0.0155	0.0322		0.0084	0.0093	0.0128	0.0266	0.0099	0.0125
1992	0.2250	0.2472	0.0150	0.0311		0.0082	0.0090	0.0124	0.0257	0.0097	0.0121
1993	0.2193	0.2391	0.0146	0.0301		0.0079	0.0087	0.0121	0.0249	0.0094	0.0117
1994	0.2139	0.2315	0.0142	0.0292		0.0078	0.0084	0.0118	0.0241	0.0092	0.0113
1995	0.2090	0.2244	0.0139	0.0283		0.0076	0.0081	0.0115	0.0234	0.0090	0.0110
1996	0.1329	0.1080	0.0096	0.0165		0.0048	0.0039	0.0079	0.0136	0.0058	0.0056
1997	0.1286	0.1019	0.0093	0.0158		0.0047	0.0037	0.0077	0.0130	0.0056	0.0053
1998	0.0523	0.0963	0.0090	0.0151		0.0019	0.0035	0.0075	0.0125	0.0028	0.0050
1999	0.0486	0.0911	0.0088	0.0144		0.0018	0.0033	0.0072	0.0119	0.0026	0.0047
2000	0.0452	0.0862	0.0085	0.0138		0.0016	0.0031	0.0070	0.0114	0.0025	0.0045
2001	0.0420	0.0817	0.0083	0.0133		0.0015	0.0030	0.0069	0.0110	0.0024	0.0043
2002	0.0380	0.0541	0.0027	0.0067		0.0014	0.0020	0.0022	0.0055	0.0016	0.0026
2003	0.0353	0.0502	0.0025	0.0062		0.0013	0.0018	0.0020	0.0051	0.0015	0.0024
2004	0.0328	0.0466	0.0023	0.0057		0.0012	0.0017	0.0019	0.0047	0.0014	0.0023
2005	0.0304	0.0433	0.0021	0.0053		0.0011	0.0016	0.0018	0.0044	0.0013	0.0021