Final Report

CRC Project E-64

EVALUATION OF THE U.S. EPA MOBILE6 HIGHWAY VEHICLE EMISSION FACTOR MODEL

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ACRONYMS/ABBREVIATIONS

A/C	air conditioning
API	American Petroleum Institute
ATP	anti-tampering program
BER	basic emission rate
CA	California
CARB	California Air Resources Board
CBD	Central Business District
CE-CERT	College of Engineering – Center for Environmental Research and Technology
CIFER	Colorado Institute for Fuels and High Altitude Engine Research
CO	carbon monoxide
CO_2	carbon dioxide
CRC	Coordinating Research Council
CSHVR	City Suburban Heavy Vehicle Route
DI	direct injection
DRI	Desert Research Institute
EPA	Environmental Protection Agency
EROS	Earth Resources Observation Systems
FHWA	Federal Highway Administration
FID	flame ionization detector
FTP	Federal Test Procedure
GVW	gross vehicle weight
GVWR	gross vehicle weight rating
HC	hydrocarbon
HDDV	heavy-duty diesel vehicle
HDGV	heavy-duty gasoline vehicle
HDV	heavy-duty vehicle
HEI	Health Effects Institute
HHDDV	heavy heavy-duty diesel vehicle
HI	heat index
IDI	indirect injected diesel
IL	Illinois
I/M Program	Inspection and Maintenance Program
LADCO	Lake Michigan Air Directors Consortium
LDDT	light-duty diesel truck
LDDV	light-duty diesel vehicle
LDGT	light-duty gasoline truck
LDGV	light-duty gasoline vehicle
LDT	light-duty truck
LDV	light-duty vehicle
LHDDV	light heavy-duty diesel vehicle
LULC	Land Use/Land Cover
MC	motorcycles
MHDDV	medium heavy-duty diesel vehicle
MWCOG	Metropolitan Washington Council of Governments
MY	model year

NAMVECC	North American Motor Vehicle Emissions Control Conference
NCDC	National Climate Data Center
NDIR	nondispersive infrared
NEI	National Emissions Inventory
NIPER	National Institute for Petroleum and Energy Research
NLCD	National Land Cover Dataset
NLEV	National Low Emission Vehicle
NMHC	non-methane hydrocarbon
NOx	nitrogen oxides ($NO + NO_2$)
NOy	total oxidized nitrogen
NOz	the difference between NOy and NOx (NOy – NOx)
NREL	National Renewable Energy Laboratory
OTAQ	Office of Transportation and Air Quality
PAMS	Photochemical Assessment Monitoring Station
PM	particulate matter
PM10	particulate matter less than 10 microns in diameter
RFG	reformulated gasoline
RSD	remote sensing device
RVP	Reid vapor pressure
SAF	spatial allocation factor
SCAQMD	South Coast Air Quality Management District
SCAQS	South Coast Air Quality Study
SCC	source classification code
SEMCOG	Southeast Michigan Council of Governments
SFTP	Supplemental Federal Test Procedure
SIP	State Implementation Plan
SOS	Southern Oxidants Study
SwRI	Southwest Research Institute
THC	total hydrocarbons
TIGER	Topologically Integrated Geographic Encoding and Referencing system
TIUS	Truck Inventory and Use Survey
TNMOC	total non-methane organic compound
UCB	University of California, Berkeley
UDDS	Urban Dynamometer Driving Schedule
USGS	United States Geological Survey
VIN	vehicle identification number
VIUS	Vehicle Inventory and Use Survey
VMT	vehicle miles traveled
VOC	volatile organic compound
VSP	vehicle specific power
WVU	West Virginia University
ZML	zero-mile level

EXECUTIVE SUMMARY

The US Environmental Protection Agency (EPA) released the final version of its on-road mobile source emission factor model, MOBILE6, in January 2002. This version contains numerous updates of data as well as methodology from the prior model version (MOBILE5, originally released in 1993) for estimating emission factors for current and future year vehicles. Since all states are required to use MOBILE6 in their State Implementation Plan and conformity emissions inventory development (except California, which has its own model), it is important to understand the relationship of model predictions to real-world observations.

The overall purpose of this project was to conduct top-down assessments of MOBILE6 emission factors using "real-world" data, and to use available data on vehicle emissions collected in a controlled manner such that the vehicle sources are well-characterized and can be attributed to a test fleet that can be reasonably duplicated using MOBILE6. This report describes the results of five different types of MOBILE6 model evaluation studies. The methods and results of each are briefly described here.

TUNNEL STUDY COMPARISONS

Methods

MOBILE6 emission factor estimates were compared to data from tunnel studies in order to evaluate the model under a range of operating conditions. A number of tunnel studies were available for analysis, all of which were conducted during summer months. Three levels of evaluation were carried out: fleet average emission factors, light-duty vehicle emission factors, and heavy-duty vehicle emission factors. Table ES-1 shows the tunnel studies used, and the assessments performed for each tunnel study. Both emission factors and ratios of pollutants were evaluated for the tunnel studies in comparison to model predictions.

Table ES-1. Tunnel study data.

Tunnel	Year of Study	Fleet Average	Light-duty	Heavy-duty
Fort McHenry	1992	x	x	X
Tuscarora	1992, 1999	x	x	X
Callahan	1995	x		
Caldecott	1997			X

MOBILE6 modeling included the use of local data where available (e.g., speed, temperature, age distribution, and fleet mix). Although each specific experimental run was modeled as a separate scenario, vehicle class comparisons were ultimately made using weighted averages of the run-specific results. This was required because the 'observed' light- and heavy-duty emission factors were derived from fleet average data using regression analyses. The result was a single estimated emission factor for each tunnel study.

Results

The results indicate that the model's accuracy varies with pollutant. Note that accuracy here refers to a comparison of modeled results to measured results. There is no absolute standard by which either set of results can be judged. Even though measurements are assumed to better reflect actual conditions, known sources of uncertainty exist such as the assumptions made to facilitate derivation of vehicle class-specific emission factors. Factors that seem to exert strong influence on the ability of MOBILE6 to accurately predict emission factors are speed and age distribution. Road grade may also be an important factor for emission factors, but the effects of road grade are not modeled in MOBILE6 (modal emission models currently under development do incorporate the effects of road grade).

The major findings and conclusions from the tunnel study comparisons are as follows:

- Fleet average NOx predictions at Fort McHenry and Tuscarora generally agreed with observed data as well as MOBILE5 estimates. The models underpredict at bore 3 (which restricted traffic to light-duty vehicles) for runs with relatively high observed emission factors. Closer examination of these experimental runs shows lower total vehicle counts as well as high heavy-duty presence (on a percentage basis.) (Not all trucks complied with the restriction on bore 3). The presence of heavy-duty vehicles will inevitably increase the observed NOx emission rates, but because there were so few, their exact behavior and contribution cannot be modeled with high certainty.
- Fleet average NMHC estimates are slightly above observed values at Fort McHenry and Tuscarora. Once again, the model underpredicts for higher observed values.
- Fleet average CO emission factors are well overpredicted at all tunnels used for fleet level comparisons. The greatest deviation from previous model results is seen for CO, with MOBILE6 being considerably higher. This may be due to the revised effects of off-cycle operation, sulfur, and facility-specific speed correction factors.
- The fleet average predictions at Callahan are all overestimated. Factors that differentiate this tunnel from the other two are older fleet, lower speed, and larger speed variation among the experimental runs. Speed corrections seem to be responsible for the MOBILE6/MOBILE5 comparison results but do not explain the large differences between modeled and observed. The older fleet distribution, if responsible, would imply that deterioration of older vehicles is overestimated in the models.
- MOBILE6 overpredicts the light-duty emission factors at both Fort McHenry and Tuscarora. Except for CO, the new model shows more accurate predictions than MOBILE5 at both tunnels.
- For heavy-duty vehicles, the modeled NMHC and CO emissions are higher than those observed, especially for CO. In this case, MOBILE5 has the better agreement with the observed data. The situation for NOx was special in that two additional studies were available for analysis. 1999 Tuscarora data showed NOx to be overpredicted while the emission factor derived at Caldecott is significantly higher than the model prediction. The relationship between observed and modeled estimates at Tuscarora may be due to the excess NOx corrections within the model, which affect model years 1988-2000. At Caldecott, one reason for the high observed NOx is that the tunnel is constant uphill unlike Fort McHenry (both up and downhill) and Tuscarora (flat). The underprediction is further

compounded by corrections made to model outputs to lower emissions based upon certification standards.

Overall, MOBILE6 updates generally resulted in overpredictions of fleet average emission factors, most noticeably for CO. This is despite the lack of explicit accounting for the effects of road grade.

RECONCILIATION OF HC/NOx AND CO/NOx RATIOS IN MOBILE6 BASED EMISSION INVENTORIES WITH AMBIENT DATA

Methods

Ratios of species in emission inventories prepared using MOBILE6 were compared with corresponding ratios in ambient monitoring data during morning commute hours at urban locations with significant mobile source impacts. While this "ambient-inventory reconciliation" approach does not allow the evaluation of accuracies of the absolute magnitudes of emissions estimates, it does allow evaluation of the degree to which MOBILE6 based emission inventories reproduce the observed pollutant mixture.

Ambient-inventory reconciliation analyses were performed for five locations with Photochemical Assessment Monitoring Stations (PAMS) in the mid-western and eastern U.S.: two sites in Chicago (Jardine and Northbrook) and one each in Detroit MI, Washington DC, and Lynn MA. Mean ambient HC/NOx and CO/NOx ratios at these monitoring sites were compared with corresponding ratios in local-scale emission inventories specifically compiled for the study. Extensive inventory development efforts were undertaken for each location to obtain inventories suitable for comparison with ambient data. Inventory data were processed for an 80 x 80 km region centered on each ambient monitoring site at 4 km grid resolution. County-level point, area and off-road emissions were obtained from the 1999 National Emissions Inventory (NEI, Version 2). Suitable gridding surrogates and temporal profiles and associated source category cross-reference files were assembled from the latest available data sources for spatial and temporal allocation of area source emissions. Spatially and temporally disaggregated on-road mobile source emissions were estimated by combining county level vehicle miles traveled (VMT) data with daily emission factors computed using county-specific MOBILE6.2 inputs. Spatial allocation of mobile source emissions was based on geographic distributions of road length by roadway (facility) type within each county. Temporal allocation of mobile source emissions was designed to account for different patterns of light and heavy-duty vehicles by time of day and day of week in the region around each ambient monitoring site: hourly VMT distributions by day of week were used to apportion the total daily running non-start emissions to individual hours. Diurnal profiles of the daily start emissions were constructed that account for the pattern of all starts and the pattern of cold vs. hot starts.

VOC emissions were adjusted to reflect the overall fraction of reported VOC accounted for by the 56 target species quantified in the PAMS ambient data used in this study. The PAMS fraction of the reported VOC was determined from the results of previous ambient-inventory reconciliation analyses in which a full VOC speciation was performed on the inventory. Effects of uncertainties in the assumed PAMS fraction on the results were quantified.

Hourly CO data were not available at the monitoring sites used in this study. These data were, therefore, obtained from the CO monitoring site closest to each study site. CO sites were found within 5 km of the Chicago-Jardine and Washington DC sites; CO monitors nearest to the other sites were located at distances ranging from 9 to 20 km and, while useful, are less likely to be representative of CO levels at those sites.

Only NO_2 data were available during the time period (summer of 2001) for which HC data were available in Detroit – the NOx measurements were not saved for some reason. However, NOx and NO_2 data were available from this site during 2000. We, therefore, made a very rough estimate of NOx for summer of 2001 based on a regression of NOx against NO_2 using the 2000 data. The resulting estimated NOx values for Detroit are subject to significant uncertainties and are likely biased but no other suitable estimates were available for this study.

Results

Results of the ambient-inventory reconciliation are summarized in terms of the weekday morning ratios of ambient HC/NOx and CO/NOx to inventory HC/NOx and CO/NOx, respectively, in Table ES-2.

Table ES-2. Ratio of average ambient ratios to inventory ratios (ratio of ratios) for weekday mornings, all wind direction quadrants and subgrid lengths combined.

	HC/NOx*	CO/NOx+
Detroit	1.26	0.33
Chicago-Jardine	1.16	3.70
Lynn	2.37	4.65
Washington DC (McMillan)	1.15	1.58
Chicago-Northbrook	1.06	2.22

* HC/NOx ratio calculated as ratio of sum of PAMS target hydrocarbons to NOx.

\+ CO data are from nearby monitors not co-located with NOx monitors and may, therefore, not be representative of CO levels at the NOx and HC monitoring sites (see text). Care should be taken in interpreting these results.

Ambient and inventory HC/NOx ratios agree reasonably well except at Detroit and Lynn. The discrepancy at Detroit may be at least partially due to underestimation of ambient NOx in the regression model used at this site as described above. We found HC/NOx emission ratios at Lynn to be lower than at the other sites, whereas the ambient ratios are roughly the same as at the other sites. Since mobile source HC/NOx emission ratios at Lynn are on par with those at the other sites, this result suggests that the area and/or point source HC/NOx emissions ratio at Lynn is too low. This is consistent with either an underestimation of VOC emissions or an overestimation of NOx emissions from these sources (or both).

CO/NOx ambient ratios exceed corresponding emissions ratios by a wide margin at all sites except Detroit. This difference may be at least partially due to the fact that CO and NOx data were obtained from different locations in each city as discussed above, especially at Detroit, Northbrook, and Lynn where CO and NOx monitors were relatively far apart. Taken together, one cannot conclude from these results that there is necessarily a problem with

CO/NOx ratios in the inventory in general or mobile sources in particular. There is no solid evidence in these results of an overestimate of CO relative to NOx by MOBILE6 as suggested by the tunnel study comparisons and the remote sensing data comparisons discussed elsewhere in this report, but these results by themselves cannot be used to rule out this possibility.

Comparisons of ambient to inventory ratios on weekends reveal that ambient HC/NOx ratios on weekends exceed the inventory ratio to a greater extent than on weekdays because the weekend increase in ambient ratios is only partially matched by the weekend increase in the inventory ratios (see Table ES-3). The weekend morning increase in ambient HC/NOx is due to a decrease in NOx, consistent with results from other studies. Thus, adjustments to the emissions inventory on weekends either decrease VOCs too much or do not decrease NOx enough. Significant reductions in on-road mobile source NOx emissions on weekend mornings associated with decreased heavy-duty vehicle activity were included in the inventory estimates as described above. In contrast, examination of the point and area source NOx emissions shows weekend morning levels are estimated to be almost equal to those on weekday mornings. Further analysis of weekend vs. weekday activity levels for all source categories will be needed to better estimate weekend emissions.

Table ES-3. Ratio of ambient HC/NOx to inventory HC/NOx (ratio of ratios) for weekday and weekend mornings, all quadrants and subgrid lengths.*

Site	Weekday	Weekend
Detroit	1.26	1.39
Jardine	1.16	1.46
Lynn	2.37	3.64
McMillan	1.15	1.42
Northbrook	1.06	1.19

* HC/NOx ratio calculated as ratio of sum of PAMS target hydrocarbons to NOx.

Results from the above analyses are subject to numerous sources of uncertainty. We estimate that uncertainty in the assumed fraction of reported VOC emissions accounted for by the 56 PAMS target species results in a potential error in the HC/NOx ratio of ratios of at most ± 25%. This is not a particularly large difference given the other uncertainties involved in making these sorts of ambient/inventory comparisons. The potential influence of background sources (i.e., those not included in the region around each monitor covered by the emission inventories) must also be considered, particularly for CO. For example, at Northbrook, correcting for an assumed background CO level of 500 ppb reduces the mean CO/NOx ratio by about 33%. However, even with this adjustment, the ambient CO/NOx ratio still exceeds the emissions ratio by a factor of nearly 1.5. At McMillan, where ambient CO levels are lower than at Northbrook, adjusting for an assumed background CO of 200 ppb reduces the mean weekday morning ambient CO/NOx ratio to 10.6 which is very close to the emissions ratio for all sources (10.4). Since the CO and NOx monitors are located much closer to each other at McMillan than at Northbrook, there is somewhat less uncertainty about the results at McMillan.

Other sources of uncertainties in ambient to inventory ratio comparisons are discussed in Section 3.

Results of the ambient-inventory reconciliation analyses presented above cannot be used to directly infer the accuracy of MOBILE6 emission estimates since the mobile source contributions to ambient concentrations cannot be separated from those of other source categories. However, the generally good agreement in weekday HC/NOx ambient and inventory ratios is consistent with the conclusion that HC/NOx ratios predicted by MOBILE6 are reasonably accurate. Although we found that ambient CO/NOx ratios generally exceed inventory CO/NOx ratios, it is not possible to conclude from this that MOBILE6 underpredicts CO (or overpredicts NOx), given the potential influence on the comparisons of background CO and the differences in locations between the CO and NOx monitors in each city. It is interesting to note, however, that there is no indication in these results of any tendency for MOBILE6 to over predict CO relative to NOx as has been suggested by recent tunnel studies and remote sensing data analyses. This issue will require further investigation.

COMPARISON OF EMISSION RATIOS FROM REMOTE SENSING MEASUREMENTS IN CHICAGO AND DENVER WITH MOBILE6 PREDICTIONS

Methods

A vast amount of roadside remote sensing measurement of in-use vehicle tailpipe emissions has been collected in recent years. Under CRC Project E-23, remote sensing device (RSD) measurements of vehicle exhaust plumes have been collected over a period of years in Denver (1999 – 2001) and Chicago (1997 – 2000) as well as other cities. Each year's measurements were made over a period of a few days with the location and time of year held constant from one year to the next. Analyses of these multi-year data sets generally suggest that they provide an accurate and consistent portrayal of light-duty vehicle exhaust emissions for the fleet and driving conditions observed at each monitoring site. The Denver and Chicago RSD data were compared with corresponding vehicle exhaust emission factors predicted by MOBILE6.

There are fundamental differences between emission factors derived from RSD data and factors predicted by MOBILE6. A RSD measures volumetric ratios of CO, NO, and HC to CO_2 in the tailpipe effluent of a moving vehicle during a brief (approximately half second) interval. Using a few reasonable assumptions about the combustion process, the RSD data can be converted to fuel specific emission factors (e.g., grams CO per kilogram of fuel consumed). The relative frequency with which different types of vehicles are observed is influenced by the location chosen for data collection and other factors. The data collection location also heavily influences the operating mode (acceleration, deceleration, cruise) of the vehicles as they are being measured. MOBILE6, on the other hand, predicts average tailpipe emission factors for each of several vehicle classes and roadway (facility) types in units of g/mile.

The comparisons of MOBILE6 predictions with RSD data were designed to account as best as possible for the inherent differences between the RSD measurements and MOBILE6 predictions described above. Factors which were taken into consideration included:

- Rather than making absolute comparisons of RSD data with MOBILE6 emission factors (which would have introduced large uncertainties associated with converting between g/mile and g/kg of fuel), comparisons were limited to CO/NO and HC/NO emission

ratios and relative changes in CO, NO, and HC mass emission factors with vehicle age and other factors.

- A rough vehicle classification was performed for the RSD observations based on license plate derived vehicle registration data and information derived from a vehicle identification number (VIN) lookup performed for a single year of RSD data in each city. These results were used to compute a weighted average of MOBILE6 predictions over vehicle classes for comparison with the RSD measurements. Local fuel composition, I/M program parameters, and meteorological conditions measured in conjunction with the RSD data collection were accounted for in the MOBILE6 runs.

- Comparisons were made with and without correction for differences in the distribution of vehicle specific power (VSP) between the RSD data and the freeway on and off ramp driving cycle used as the basis for emission calculations in MOBILE6. Emissions are known to vary as a function of VSP which can be reasonably approximated from road grade, vehicle speed and acceleration. Near instantaneous speed and acceleration were determined contemporaneously with the RSD measurements; these data were used to compute the VSP frequency distribution associated with emission factors computed from the RSD data.

- RSD data contain measurements of %NO while MOBILE6 reports emission factors for NOx. Since nearly all NOx in the exhaust of light-duty vehicles is released as NO, we assumed the MOBILE NOx mass emission factors were equivalent to NO mass emission factors.

- RSD HC measurements are based on a nondispersive infrared (NDIR) measurement that has been shown to produce a response equal to one-half the equivalent flame ionization detector (FID) measurement (Singer et al., 1998). Thus, the RSD %HC values recorded in the E-23 data were doubled and the MOBILE6 runs specified that HC be output as THC (which is representative of the FID response).

Results

Results of the comparison of MOBILE6 predictions with RSD measurements revealed some areas of reasonably good agreement and some areas of significant disagreement. Major findings are as follows:

- In comparison to RSD CO/NO ratios, MOBILE6 overestimates CO relative to NO for newer vehicles by up to a factor of three. This appears to be a result of the fact that MOBILE predicts a much greater increase in CO with vehicle age than is evident in the RSD data; there appears to be much better agreement between MOBILE6 and the RSD data regarding the dependence of NO emissions on vehicle age.

- MOBILE6 HC/NO ratios for broad vehicle classes (defined using registration records) that were found from the limited VIN lookup results to be composed mostly of light-duty gasoline vehicles (LDGV's) are in much better agreement with the RSD data than

is the case for CO/NO ratios. For vehicle registration classes more heavily weighted towards light-duty trucks (LDT's), the MOBILE6 HC/NO ratios consistently exceed the RSD ratios (by up to a factor of four). For both types of vehicle classes, however, the dependence of HC/NO ratios (and of HC emission factors) on vehicle age predicted by MOBILE6 tracks reasonably well with the RSD data, although there is less of a relative difference in HC emission factors between 1-5 year old vehicles and 6-10 year old vehicles in the MOBILE6 predictions than is found in the RSD data.

- In Chicago, where temperature and fuel RVP changed more significantly over the course of the four year measurement program than was the case over the three years of measurements in Denver, MOBILE6 predicted significantly lower CO and HC emissions in 2000 as compared to 1997. For a fixed model year group (i.e., 1986 – 1991 model years which represent vehicles that were 6 – 10 years old in 1997 and 9 – 13 years old in 2000), the RSD data showed essentially no change in CO emissions between 1997 and 2000, whereas MOBILE6 predicted emissions in 2000 that were less than half of the 1997 prediction. Since MOBILE6 was run with temperature, humidity, and fuel parameters representative of actual conditions during each measurement year, this suggests that the MOBILE6 temperature/RVP correction factors may not be appropriate for the vehicles and driving conditions captured in the RSD data.[1] Similar discrepancies were found for HC emissions and, to a lesser extent, for NO.

As noted above, the distribution of VSP values associated with the RSD data differs from the VSP distribution associated with the ramp driving cycle used as the basis for the MOBILE6 emission factor calculations. It would, therefore, be expected that emission factors from the RSD data will differ to a certain extent from the MOBILE6 predictions. We analyzed this VSP effect by binning the RSD data by VSP, computing mean emission factors in each bin, and computing VSP adjusted averages of the RSD emissions using the MOBILE6 ramp cycle VSP bin frequencies. Results of this analysis produced several key findings:

- The ramp driving cycle includes significantly higher frequencies of negative VSP modes than was observed in the RSD data. This is not unexpected as the ramp cycle is intended to represent driving behavior over the entire length of a ramp, whereas the RSD data collection sites were specifically chosen to capture vehicles during acceleration events. On the other hand, the RSD data included a small fraction of events (less than 0.5% in Denver) with VSP's above 28 kW/tonne, whereas the ramp cycle does not include any VSP's above this level.

- Adjusting the RSD data according to the MOBILE6 ramp cycle VSP distribution produces a 61% increase in the overall mean CO/NO ratio in Denver (31% in Chicago) and a 93% increase in the HC/NO ratio in Denver (70% in Chicago). Applying these adjustments decreases the degree to which MOBILE6 overpredicts the CO/NO ratios relative to RSD values for 1 – 5 year old vehicles. For example, in Denver for vehicles in the registration category dominated by LDGVs, the overprediction is reduced from a factor of three to a factor of two. However, the ratios for the oldest vehicles are underpredicted when the adjustment is applied.

[1] Inspection of the MOBILE6 results and output of sensitivity runs suggested that the humidity differences did not play a major role.

- Differences in VSP distributions between vehicle age bins were found to be minor and making the VSP adjustment on a vehicle age bin basis had little effect on the dependence of CO/NO ratio on vehicle age seen in the RSD data.

- Increasing the RSD HC/NO ratio to account for the VSP adjustment results in RSD HC/NO ratios for the LDGV dominated registration category in Denver that are much larger than the corresponding MOBILE6 predictions.

COMPARISON OF HEAVY-DUTY DIESEL CHASSIS EMISSIONS DATA WITH MOBILE6

Methods

For the development of MOBILE6 emission factors for light-duty vehicles, test data from whole vehicle testing of emissions using chassis dynamometers were used. Chassis dynamometers are equipment that allows the entire vehicle to be driven on rollers that can provide the resistance through the wheels that a vehicle experiences when driven on the road including rolling resistance, wind resistance, grade, and inertia. For MOBILE6 heavy-duty vehicle emission factors, however, whole vehicle testing of emissions using chassis dynamometers were not used. Instead, emission factor estimates for heavy-duty vehicles in MOBILE6 rely on engine emission testing as a function of work, where work is defined as the mechanical energy developed at the flywheel of the engine. (In operation with a whole vehicle, the engine work would be converted through the transmission to the wheels to propel the vehicle along the road.) An energy conversion factor is used to translate engine work (in g/hp-hr) to vehicle activity in terms of miles traveled using survey information on vehicle and engine efficiency.

There is a growing database of emissions data taken by running in-use heavy-duty vehicles on chassis dynamometers. Such test data allow for a direct and independent verification of the MOBILE6 estimates. A database of individual vehicle results was compiled from test results from all groups in North America known to have tested whole heavy-duty vehicles on chassis dynamometers. The data included a variety of sources and grouped according to like vehicle types and emission standards and compared with the MOBILE6 estimates. It is particularly important to validate heavy-duty NOx emissions because heavy-duty vehicles are more significant in MOBILE6 than in MOBILE5, now representing up to half of total NOx emissions for an urban area.

Results

In general, the results indicate that while the MOBILE6 emissions estimates for heavy-duty diesel HC and CO are similar and usually within the data uncertainty for most model years, the NOx emissions could be overpredicted by 50 percent to 100 percent for earlier model years (1978 and earlier), and underpredicted by up to 50% for late model vehicles (1994 and later) compared with the available data. These conclusions were reached using the emission trends by model year and weight of evidence based on the data available for similar average

speed test cycles; however, when the data were disaggregated by individual test cycle, vehicle weight class, and model year, the conclusions were less clear. The low number of tests for any one type of truck or bus and test cycle would be insufficient to clearly conclude whether MOBILE6 emission factors were accurate or to develop alternative emission factors. The high NOx emissions for late model vehicles highlight a need for further investigation because these vehicles will be used for many years to come. Individual high THC and CO emitters (up to 20 times the average emission levels of normal emitters) were identified and implied that high PM emitters also exist. An emissions effect was found to be statistically significant for cold starts with even older diesel vehicles and indicated that a start methodology for diesel vehicles should be investigated for inclusion in emissions modeling if activity information indicates a sufficient number of starts occur with these vehicle types to affect overall emission estimates.

COMPARISON OF MOBILE6 DIESEL FUEL CONSUMPTION ESTIMATES WITH FUEL SALES

Methods

MOBILE6 uses fuel consumption rates to derive heavy-duty diesel emission factors. EPA uses fuel consumption rates to estimate the work required per mile of vehicle travel. MOBILE6 highway-diesel fuel consumption estimates were compared with fuel sales information. In order to estimate the fuel consumption rates predicted by MOBILE6, ENVIRON combined the national and state vehicle miles traveled (VMT) estimates with MOBILE6 fuel consumption rates. These calculated national and state fuel consumption rates were compared with fuel sales information available from the Department of Energy's Energy Information Administration for the calendar year 1999. This comparison provides an analysis of the accuracy of the national heavy-duty diesel vehicle activity estimates, and whether individual state estimates can be considered accurate for state or regional inventories.

Results

Diesel fuel consumption estimates using the MOBILE6 fuel consumption rates were slightly lower ($<10\%$) than the fuel sales estimates on a national level, but were significantly higher or lower for many individual states. Because the national estimates of fuel consumption using MOBILE6 and VMT were comparable with fuel sales estimates, one can be reasonably confident that MOBILE6 is accurately predicting fuel consumption and CO_2 emission rates. Because MOBILE6 uses fuel consumption estimates to convert engine specific emissions rates to per mile vehicle emission rates, this work provides confidence that EPA has used accurate figures reflecting fuel consumption rates in its calculation of emission factors. Diesel vehicle activity within a given state could not be predicted because the fuel consumption calculated using MOBILE6 combined with state VMT was not comparable with fuel sales for most states, often differing by up to factor of three. More research is needed to determine whether MOBILE6+VMT or fuel sales estimates are more accurate reflections of an individual state's heavy-duty vehicle activity.

OVERALL SUMMARY

The methods employed in this work to validate the MOBILE6 model vary widely in their nature, scope, and focus. Some are based on the operation of vehicles at a specific location and under narrow operational parameters, while others capture an entire geographic area with its particular mix of operational parameters such as speed, acceleration, load, etc. Some methods focus on the emission rate of particular vehicles, each with its own maintenance status, while others are limited to ascertaining emission from large groups of vehicles. Furthermore, each methodology has different sources of uncertainty. Given these differences, it is not unreasonable to expect that the various MOBILE6 validation efforts would produce seemingly contradictory results. Nevertheless, it is interesting to compare the results of each method to attempt to identify strong coherent themes.

Light-duty vehicles dominate hydrocarbon emissions. For hydrocarbons, most of the tunnel study comparisons and the remote sensing comparisons indicate a tendency of MOBILE6 to slightly overpredict emission factors. The analyses of ambient HC/NOx ratios, on the other hand, indicate that the model underpredicts on-road HC inventories. However, the ambient ratios analyses include evaporative emissions from on-road vehicles along with exhaust emissions and emissions from point, area, nonroad, and biogenic sources, and the inaccuracies in these other emissions sources play a major part in the accuracy of the validation process. Also, the ambient ratio analyses encompass a range of vehicle operations, whereas both the tunnel studies and the remote sensing data techniques cover a relatively narrow segment of vehicle operational conditions. The model may overpredict for some operating conditions and underpredict for others.

Light-duty vehicles also dominate carbon monoxide emissions. Both the tunnel study comparisons and the RSD comparisons imply that MOBILE6 significantly overpredicts light-duty vehicle emissions in recent years; this result cannot be extrapolated to future years. Results from these two comparisons likely corroborate because of the narrow range of operating conditions each captures. Results from the ambient ratios analyses for CO are suspect because hourly CO data were not available at the monitoring sites used in the analyses, and instead were taken from the closest monitor and are, therefore, not necessarily representative of CO levels at the sites evaluated.

Both light-duty and heavy-duty vehicles have significant NOx emissions, though the proportions change over time as NOx emissions controls occur earlier in the light-duty fleet. For NOx, the significant factor affecting the comparisons of “real-world” data to the MOBILE6 model appears to be the age composition of the heavy-duty fleet. Comparisons of heavy-duty chassis dynamometer test data revealed MOBILE6 overprediction for older model years (pre-1979) and underprediction for 1994 and newer model year emission rates. NOx comparison results for the tunnel studies results are mixed, suggesting that heavy-duty NOx is overpredicted except when significant load (i.e., grade) is present.

1. INTRODUCTION

The MOBILE model, developed by EPA's Office of Transportation and Air Quality (OTAQ), is EPA's regulatory model for estimating on-road mobile source emissions. For many years, MOBILE5, released in 1993, was the regulatory model. Over a period of several years, EPA developed significant changes to the model and publicly released MOBILE6 in January 2002.

Validation of the MOBILE model is a major topic with significant implications for air quality management. This was recognized and emphasized in the National Research Council review of EPA's mobile source modeling program (NRC, 2000). The NRC report found that

> "Model evaluation and validation have not been addressed adequately by EPA during MOBILE development. MOBILE's *predictions* of the benefits of air quality programs (e.g., vehicle emissions inspection and maintenance, oxygenated fuels, and reformulated gasoline) are often taken as *measurements* of the benefits of these programs. Confidence in the model has been undermined when large discrepancies have been observed between the model's predictions and field measurements. Proper testing and evaluation would improve the accuracy of mobile source emissions modeling in estimating emissions, estimating the effects of emissions on human health and the environment, and estimating the effectiveness of control strategies."

The NRC report recommended that "*Enhanced model evaluation studies should begin immediately and continue throughout the long-term evolution and development of mobile source emissions models*."

In response to the NRC report and recommendations from the user community, the Coordinating Research Council and the EPA OTAQ jointly funded this MOBILE6 evaluation study. CRC/EPA solicited proposals from contractors, without specifying the exact nature of the evaluation studies to be done. Rather, CRC/EPA solicited ideas for such evaluations from interested contractors. ENVIRON proposed several different types of model evaluation approaches, and the results of these different efforts are presented in this report.

Below we discuss the significant changes in MOBILE6 from the previous version of the model, and provide some model comparisons. We then describe the different types of evaluation tasks that were performed.

COMPARISON OF MOBILE5 AND MOBILE6

MOBILE5 and MOBILE6 both estimate emission factors for on-road vehicles for NO_x, VOC, and CO. During the course of this project, EPA released two updated versions of MOBILE6 that also include emission factors for particulate matter (based on the older PART5 model), and mobile source toxics (based on the older MOBTOX model). Exhaust, evaporative, and refueling emission factors are estimated in units of grams per mile. The MOBILE emission factors are then multiplied by an estimate of vehicle miles traveled (VMT) to estimate total on-road emissions. The public release version of MOBILE6, along with detailed technical

documentation on the MOBILE6 updates, may be found on the EPA MOBILE6 web site at http://www.epa.gov/OMSWWW/m6.htm.

MOBILE5 estimated emission factors for the following eight vehicle classes:

- Light-duty gas vehicles (LDGV – passenger cars), up to 6000 lb gross vehicle weight (GVW)
- Light-duty gas trucks (pick-ups, minivans, passenger vans, and sport-utility vehicles), up to 6000 lb GVW (LDGT1)
- Light-duty gas trucks of 6001-8500 lb GVW (heavier versions of LDGT1s; the categories are modeled separately because numerically different emission standards are established under the Clean Air Act for LDGT1s and LDGT2s)
- Light-duty diesel vehicles (LDDV – passenger cars), up to 6000 lb GVW
- Light-duty diesel trucks (LDDT), up to 8500 lb GVW (unlike gasoline powered LDTs, the same emission standards are applicable to all diesel LDTs up to 8500 lb GVW)
- Heavy-duty gas vehicles, 8501 lb and higher GVW, that are equipped with heavy-duty gas engines (HDGV)
- Heavy-duty diesel vehicles (HDDV), vehicles of 8501 lb and higher GVW equipped with heavy-duty diesel engines
- Motorcycles (MC, all of which are gasoline powered; highway-certified motorcycles only are included in the model, off-road motorcycles such as "dirt bikes" are modeled as a nonroad mobile source)

Significant updates in MOBILE6 from the MOBILE5 model include:

- Emission factor estimates for 28 instead of 8 vehicle classes (primarily for disaggregating HDDV emissions).
- Update of base emission rate equations to account for new data and analytical methods that better characterize in-use deterioration rates for light-duty vehicles (LDV).
- Incorporation of gasoline sulfur impacts and revised gasoline oxygenate impacts.
- Revisions to speed correction factors to better reflect off-cycle operation for LDVs.
- Revisions to the air conditioning algorithm to reflect data and analytical methodologies developed in the last several years for LDVs.
- Revisions to vehicle activity estimates (e.g., annual mileage accrual rates, VMT distributions by vehicle class, etc.) to better reflect more recent data.
- Addition of off-cycle NO_x impacts for HDDVs as a result of fueling strategies that optimize fuel economy (i.e., the “defeat device” issue).
- Low NOx rebuilds for HDDVs to correct the “defeat device” issue.
- Wholesale revision of evaporative emission factor estimates, using real-time evaporative emissions testing data from several major test programs conducted by both EPA and industry.
- Updated I/M algorithms, including emissions impacts of the second-generation on-board diagnostics (OBD II) regulations.
- Incorporation of new engine, vehicle, and fuel standards:
  - National Low Emission Vehicle (NLEV) standards for LDVs, beginning with model year 2001;

- Tier 2 emission standards for passenger cars and light-duty trucks (LDT), beginning with model year 2005, with low sulfur gasoline beginning in the summer of 2004;
- Supplemental Federal Test Procedure (SFTP) requirements (i.e., control of "off-cycle" and air conditioning impacts);
- Heavy-duty vehicle (HDV) emission standards beginning with model year 2004; and
- HDV emission standards beginning with model year 2007, with low sulfur diesel beginning in the summer of 2006.

With all of these changes, MOBILE6 emission factors are significantly different from MOBILE5 emission factors. Figures 1-1 through 1-3 show EPA's comparison of MOBILE5 and MOBILE6 emission factors for a national fleet for NO_X, VOC, and CO. In general, MOBILE6 emission factors are higher than MOBILE5 in past years, and lower than MOBILE6 in future years, because revised emission factor estimates increase overall but more stringent emissions standards are now incorporated into MOBILE6.

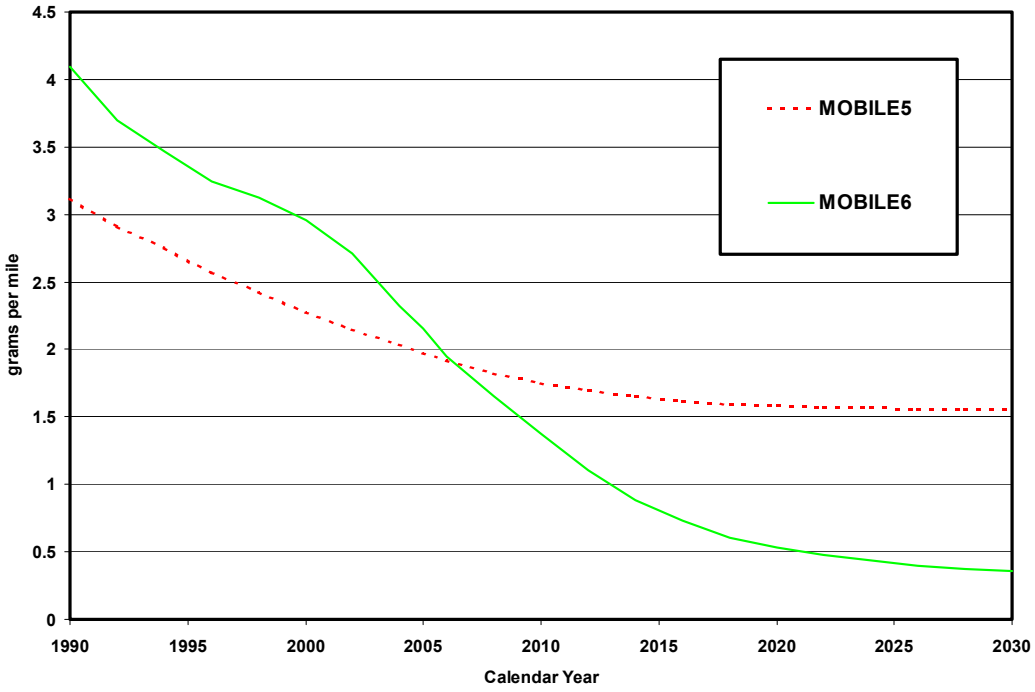


Figure 1-1. Comparison of MOBILE5 and MOBILE6 NO_X emission factors. Source: Beardsley, 2001.

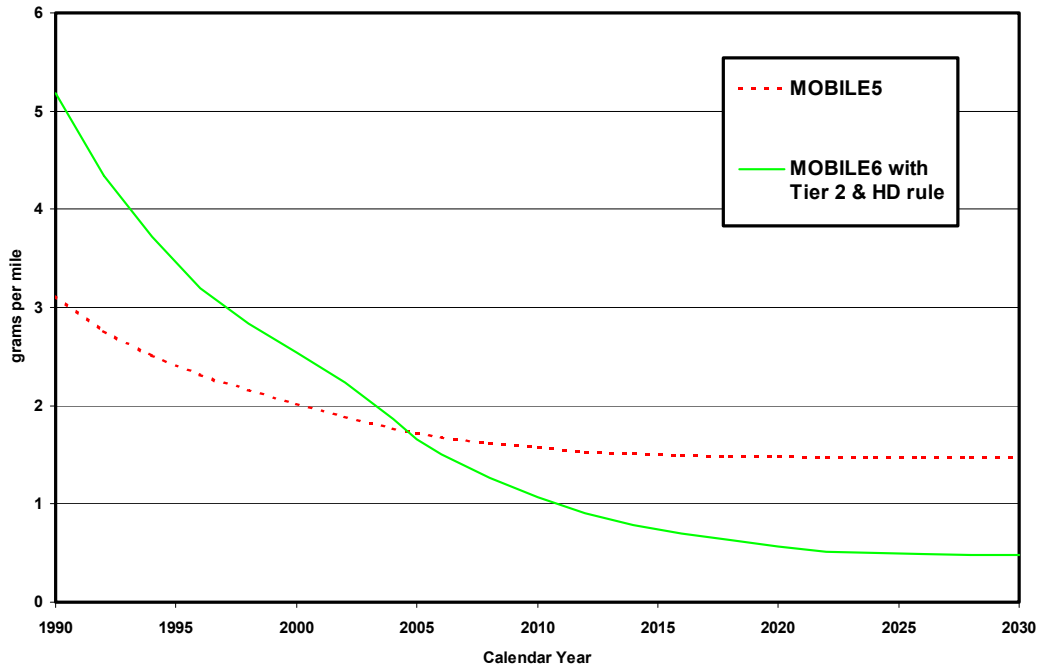


Figure 1-2. Comparison of MOBILE5 and MOBILE6 VOC (exhaust + evaporative) emission factors. Source: Beardsley, 2001.

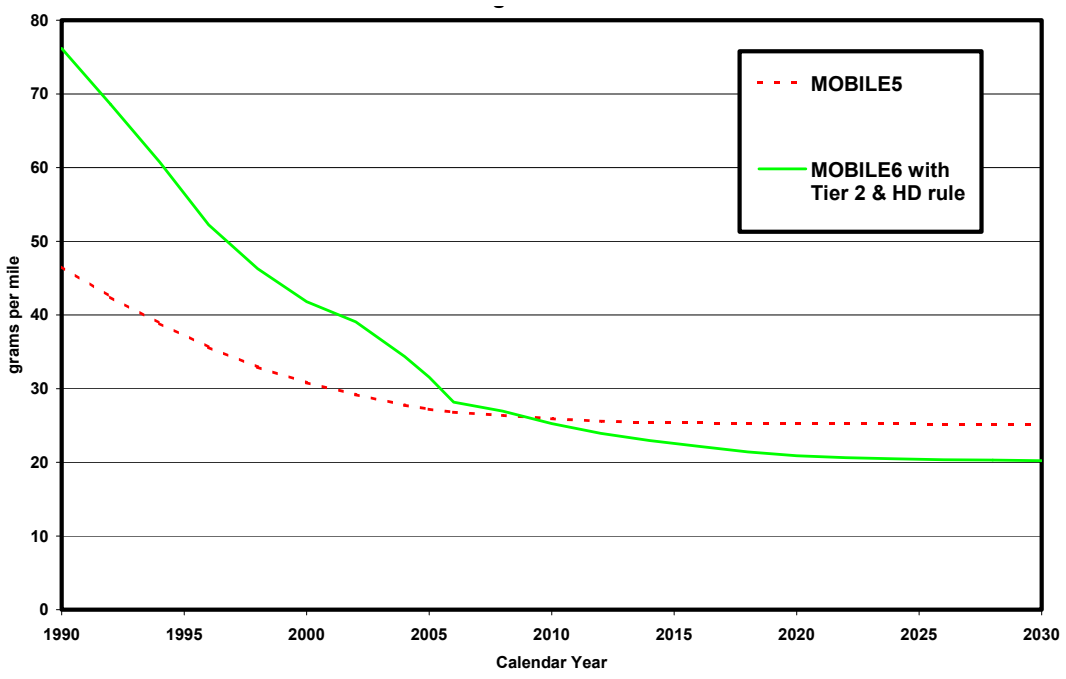


Figure 1-3. Comparison of MOBILE5 and MOBILE6 CO emission factors. Source: Beardsley, 2001.

The distributions of NOx and VOC emissions by vehicle type for MOBILE5 and MOBILE6 for years 1990, 2000, and 2010 are shown in Figures 1-4 and 1-5, respectively; these results use model inputs for the Houston area, and the default VMT mix. Figure 1-4 shows that HDDVs account for a large proportion of NOx emissions, especially in year 2000, in part because of the NOx defeat device adjustment applied to the relevant model years. Figure 1-5 shows that LDTs account for a large proportion of hydrocarbon emissions relative to passenger cars in MOBILE6 than in MOBILE5.

MOBILE6 EVALUATIONS PERFORMED

- Tunnel study comparisons: Emission factors derived from tunnel studies were compared to MOBILE6 emission factors to evaluate MOBILE6 model performance under a range of operating conditions. LDV, HDV, and fleet average emission factors were compared.

- Ambient Ratio Analyses: Ratios of species in emission inventories prepared using MOBILE6 were compared with corresponding ratios in ambient monitoring data in locations with significant mobile source impacts. This ambient-inventory reconciliation analysis was performed for five locations in the mid-western and eastern U.S.: two sites in Chicago and one each in Detroit MI, Washington DC, and Lynn MA. Mean ambient HC/NOx and CO/NOx ratios at these monitoring sites were compared with corresponding ratios in local-scale emission inventories.

- Comparison with HDDV chassis dynamometer data: MOBILE6 HDDV emission factors are based on engine certification test data converted to g/mile emission rate. Comparisons of MOBILE6 HDDV emission factors with chassis dynamometer data were made for THC, CO, NOx, and PM emissions; by model year (emission standard), vehicle type (weight class), and average speed.
- Comparison with remote sensing measurements: Denver and Chicago remote sensing data from CRC Project E-23 were compared with corresponding vehicle exhaust emission factors predicted by MOBILE6. Remote sensing data were converted to g/gal based on fuel properties, and comparisons were made by vehicle class. Comparisons were made for CO/NO and HC/NO ratios, and for relative changes in HC, CO, and NO with vehicle age.

- Comparison of MOBILE6 highway diesel fuel consumption estimates with fuel sales information: MOBILE6 uses fuel consumption rates to estimate the work required per mile of vehicle travel as part of the derivation of HDDV emission factors. National and state fuel consumption rates calculated from MOBILE6 were compared with fuel sales information available from the Department of Energy's Energy Information Administration for the calendar year 1999.

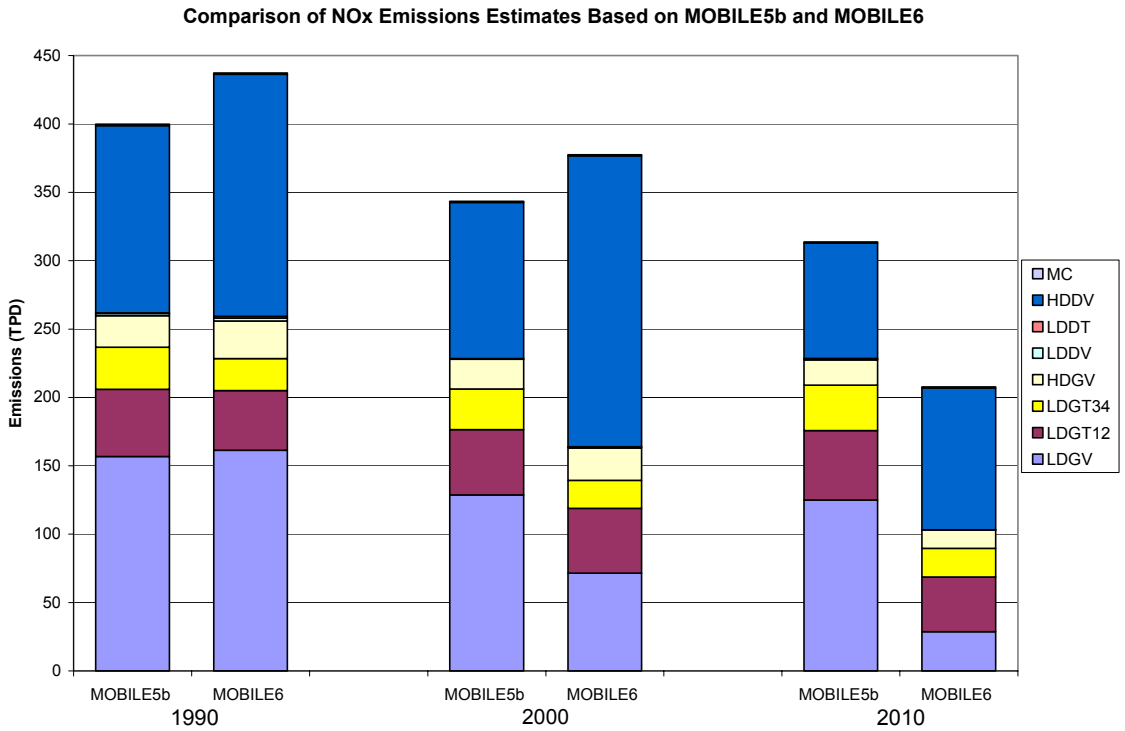
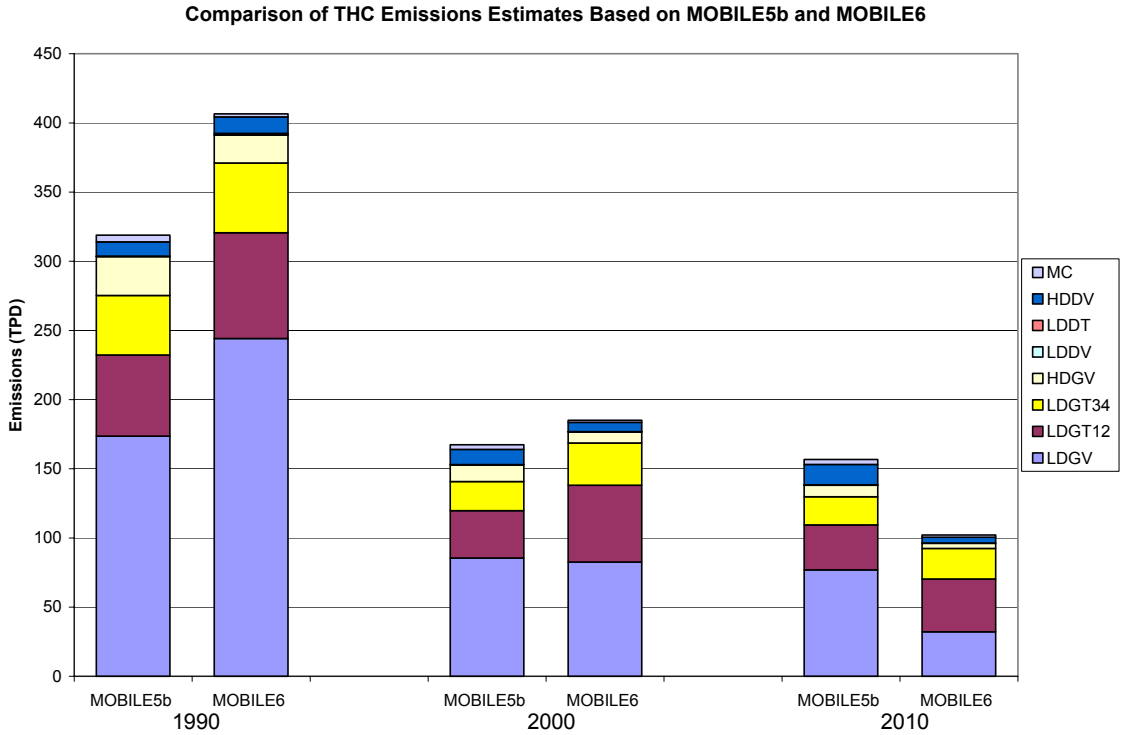


Figure 1-4. Distribution of Houston area MOBILE5b and MOBILE6 NOx emissions by



vehicle type for years 1990, 2000, and 2010.

Figure 1-5. Distribution of Houston area MOBILE5b and MOBILE6 THC emissions by vehicle type for years 1990, 2000, and 2010.

2. COMPARISON OF ON-ROAD TUNNEL STUDY EMISSION FACTORS WITH MOBILE6

INTRODUCTION

Tunnel studies have historically served as a major means of validating emission factor models. In the late 1980s and throughout the 1990s, tunnel studies were used to validate both California's and US EPA's emission factor models. The 1987 study performed at the Van Nuys Tunnel in Southern California as part of the Southern California Air Quality Study (SCAQS) was the first study to show the discrepancy between model predictions and observed data. In general, nitrogen oxides (NOx) predictions agreed well with tunnel data, but carbon monoxide (CO) and hydrocarbon (HC) emission rates were typically underpredicted by the models. At the time of the Van Nuys study, the existing version of EMFAC underestimated CO and HC emission factors by at least half (Ingalls, 1989). Later assessments of MOBILE (Robinson et al., 1996) showed that versions 4.1 and 5 underpredict under complex traffic conditions and overpredict when vehicles are operating under steady speeds.

Tunnel studies are typically conducted by taking pollutant concentration measurements during several discrete runs throughout a day. The runs are principally designed to capture varying fleet mix, and oftentimes they capture fluctuating temperature, humidity and speed as well. Emission rates are back-calculated from concentrations, air flow rates, vehicle counts, and other physical parameters. From these emission factors, ratios of pollutants can be directly computed.

Emission rates and ratios of pollutants obtained from tunnel measurements contain a combination of in-use effects. In some instances, it is possible to gauge the influence of individual factors. An illustration is the Caldecott Tunnel studies, which were performed before and after the implementation of California Phase II RFG. This allowed the effects of this fuel to be studied without interfering factors other than fleet turnover. As discussed below, through regression or apportionment analysis, tunnel data also provide a means to validate light-duty (LD) and heavy-duty (HD) emission factors separately. Finally, speciation of hydrocarbon measurements yields estimates of exhaust and evaporative fractions. The latter portion is generally relatively small in tunnel experiments.

In most tunnels, the LD and HDVs are not routed through separate bores. Thus, the emission rates derived from raw data are representative of the overall fleet. Vehicle class-specific emission factors can be obtained by regressions performed on the fleet emission rate as a function of LD and HD fractions. The regressions are then extrapolated back to zero to determine the complementary emission factor. For example, a regression of fleet average emission factors against LD fraction, when extrapolated to zero LD fraction, yields the HD emission factor and vice versa. This method was first employed by Pierson et al. (Pierson et al., 1996). Some tunnels exclude HD traffic in designated bores so that LD emission factors can be estimated directly and compared with model predictions. HD emission factors at such tunnels are obtained by 'subtracting' the LD portion from the total observed.

Despite their usefulness, tunnel study data present inherent problems for model validation. First, the data often include the effects of road grade and vehicle loads which are both very

difficult to accurately quantify. The MOBILE6 model does account for off-cycle and air conditioning effects, but these may not specifically reflect the tunnel conditions. In addition, since tunnels involve smaller samples of the overall fleet, the effect of high emitters may not only be more pronounced but is also more uncertain. Another difficulty is encountered when attempting to quantify the penetration of in-use controls such as inspection/maintenance (I/M) programs or low Reid Vapor Pressure (RVP) fuels. This is significant when vehicles passing through a tunnel come from areas with different fuels and control programs. Finally, it may be difficult to assess the combination of modes (cold start, hot start, hot stabilized) under which the vehicles are operating.

AVAILABLE TUNNEL STUDIES

Table 2-1 lists the tunnel study data available for use in validating the MOBILE6 emission factor model predictions. Some tunnels were studied in one year only, and two were repeatedly studied in several years. In each tunnel study (by which we mean one tunnel in one year), there are always multiple "runs" at different times of day and different days. These studies have all been performed by either Desert Research Institute (DRI) or UC Berkeley. Brief descriptions of each tunnel are provided in Appendices A (DRI tunnel studies) and B (UC Berkeley Caldecott tunnel studies). Where appropriate, grades and other special tunnel conditions that affect operation within the measurement zone are noted in these appendices to serve as caveats qualifying the resulting comparisons. Finally, with the exception of a January Deck Park study, all these data were collected during summer months.

Table 2-1. Summary of available tunnel studies.

Tunnel	Location	Length (m)	Fleet	Year(s)
Non-California tunnels				
Fort McHenry Tunnel	Baltimore, Maryland	2174	Highway	1992, 1993, 1995
Tuscarora Mountain Tunnel	Pennsylvania Turnpike, Pennsylvania	1623	Highway	1992, 1999
Cassiar Connector	Vancouver, British Columbia	730	Urban	1993
Callahan Connector	Boston, Massachusetts	1545	Urban	1995
Deck Park Tunnel	Phoenix, Arizona	804	Urban	1995
Lincoln Tunnel	New York/New Jersey	2440	Urban	1995
California tunnels				
Caldecott Tunnel	San Francisco Bay Area, California	965	Urban	1994-1997, 1999, 2001
Sepulveda Tunnel	Los Angeles, California	582	Urban	1995, 1996
Van Nuys Tunnel	Los Angeles, California	222	Urban	1995

There are four tunnel studies listed in Table 2-1 that were excluded from consideration for MOBILE6 comparisons:

- The 1995 Fort McHenry study focused on measuring dioxin and furan emissions from the in-use fleet. Emissions of CO, HC, and NO_x were not measured.

- The 1993 Fort McHenry study quantified only PM_{10} emissions. MOBILE6.1 will include PM_{10} emission factors, but was not available in time for use in this project.

- The Cassiar connector is a Canadian tunnel. It is not being considered for comparison to MOBILE6 because of the large number of changes that would be required to MOBILE6 to reflect Canadian fleet and fuel differences.

In addition, only the Caldecott study performed in 1997 includes HD emission factors; all other Caldecott tunnel studies measured LD emissions only.

Tunnel Studies Selected for Comparison with MOBILE6

The tunnel studies that were used to compare fleet average (i.e., cars and trucks combined) emission factors to model predictions are:

- Fort McHenry, 1992;
- Tuscarora, 1992 and 1999;
- Callahan, 1995.

These tunnels and years were chosen to ensure that a relatively wide range of operating parameters is included in this study. These include effects of newer technologies, grades, speeds, fleet mix, and ambient conditions. They were also used because the measured emissions were readily available and reliable. Finally, for the Fort McHenry and Tuscarora (1992) tunnels, MOBILE5 results were also available.

For the purpose of validating LDV emission factor predictions the same tunnel studies identified above can be used. However, regression analysis is required for tunnels that do not separate LD and HD. Run-specific results are lost and one cannot develop a single modeling scenario whose results are directly comparable (because the regressions implicitly include the effects of changing temperature, speed, humidity, and other factors). Tunnels where the vehicle classes are separated are thus most useful because vehicle class-specific emission rates can be derived with less uncertainty. The 1992 Fort McHenry tunnel data were preferred for this analysis since LD vehicles were essentially the only occupants in one of the bores measured (bore 3). The Tuscarora (1992) and 1992 Fort McHenry (bore 4) data were also added because of the wide-ranging fleet mix among the runs which enhances the regression technique. This is discussed further below.

To assess the accuracy of MOBILE6 estimates of HD emission factors California tunnels were used as well as non-California tunnels with appropriate adjustments to by-model-year emission factors for differences in certification standards. We relied on the 1997 Caldecott and the 1992 Fort McHenry tunnel studies, as these both have bores in which HD vehicles are restricted. (Note that for HD vehicles, only NOx and $PM_{2.5}$ data were derived from the 1997 Caldecott Tunnel measurements.)

In addition, we utilized the regression approach described above to obtain HD emission factors from the Tuscarora, Callahan, Lincoln, and Deck Park data. (The Lincoln and Deck Park results were ultimately excluded. See discussion below.) The HD truck emission factors estimated from the regression analyses were compared to MOBILE6 HD-specific emission factors, with the comparisons being made to the weighted average.

Table 2-2 summarizes major characteristics of the tunnels and fleets used in this assessment. The information shown is of particular importance in subsequent discussions of the comparison results. Some noteworthy observations are (1) the Callahan tunnel has the largest speed variation; (2) measurements at the Fort McHenry and both Tuscarora studies captured a wide range of LD/HD fractions; and (3) Fort McHenry and Callahan results include the effects of an uphill and downhill operation while only one direction (uphill) is captured at Caldecott. The observations for multiple slope tunnels presented in subsequent sections are averages except where noted.

Table 2-2. Selected characteristics of tunnels chosen for comparison with MOBILE6.

Tunnel	Grade	Speeds (mph)	LD Fraction
Fort McHenry	-3.76%/+3.76%	38 to 53	0.28 to 0.99 (bore 4)
Tuscarora 1992	Flat (<0.3%)	55 to 60	0.20 to 0.94
Tuscarora 1999	Flat (<0.3%)	54 to 62	0.14 to 0.88
Callahan	-3.8%/+3.25%	14 to 35	0.94 to 0.98
Caldecott	+4.0%	41 to 56	0.95 to 0.97

MOBILE6 MODELING

General Approach

MOBILE6 requires a number of input parameters to specify a run scenario. At a minimum, these include

- fleet composition data – model year registration, vehicle class distribution;
- operating conditions – speed, operating mode (controlled via the SOAK DISTRIBUTION and STARTS PER DAY commands);
- ambient conditions – temperature, humidity;
- fuel parameters – RVP, sulfur content, RFG status; and
- control program status – I/M and anti-tampering program (ATP).

We used inputs derived from local data where available. When local data are not available from the existing tunnel studies, we attempted to obtain the most representative data available from local agencies and other publicly available sources of historical data. Fleet composition, ambient conditions, and operating conditions were available for most of the tunnel studies. Fuel parameters and I/M controls were obtained from local regulatory or SIP documentation. Operating modes fractions and facility class selection must be developed based upon engineering judgment; DRI and UCB were consulted to define these parameters.

MOBILE6 Meteorological Input Parameters

In the MOBILE6 modeling of the tunnel study runs, historical meteorological data were used whenever possible for the input parameters. We chose to use MOBILE6 defaults for two meteorological parameters, (cloud cover and peak sun), because no reliable historical data could be found. Default cloud cover is assumed to be a 100% clear day. To gain an understanding of the effect of using the default value, we note that above a heat index (HI) of about 100, there is no difference in air conditioning (A/C) demand between 0 and 100% cloud cover. Below that HI, this difference varies as a function of the HI, with the maximum being about 20% demand. (EPA, 2001) Thus, on average, we may expect less than a difference of 10% in demand between using the default and the actual cloud cover. The default peak sun period is indicative of early summertime so its use is appropriate for the modeling scenarios in this work. The following sections discuss the sources of data for meteorological parameters that were modified for each run.

Temperatures

Temperatures for the Tuscarora and Callahan Tunnel study runs were obtained from DRI's data. In general, a specific temperature was reported for the hour of each run. The Caldecott Tunnel 1997 temperatures were obtained from historical Oakland airport readings. In all cases, MOBILE6 was run at constant temperatures throughout the day, but only the hour corresponding to the experimental run was used.

Sunrise/Sunset

Historical sunrise and sunset times for each test run were obtained from the US Naval Observatory web site, found at http://aa.usno.navy.mil/data/docs/RS_OneDay.html. Data for the nearest available city for each tunnel were used. In accordance with MOBILE6 sunrise/sunset input structure, the times were all rounded to the nearest hour.

Absolute Humidity

MOBILE6 accepts a daily average absolute humidity value that is calculated from barometric pressure and relative humidity readings. For all runs, except for the 1999 Tuscarora Tunnel and the 1997 Caldecott Tunnel studies, pressure and relative humidity values were obtained from the National Climate Data Center (NCDC) web site, at http://www4.ncdc.noaa.gov/, using the nearest available weather station. Values for each specific test day were read from historical monthly data graphs. 1999 data were not available at the NCDC web site, so an alternate data source was found for the 1999 Tuscarora runs. Daily pressure, average temperature and dewpoint temperature values for 1999 were found at http://www.wunderground.com/cgi-bin/findweather/getForecast?query=huntingdon%2C+PA. The average and dewpoint temperatures were used to calculate relative humidity using the calculator found at http://www.weatherlord.com/weather/calculator/humidity/. The Caldecott Tunnel 1997 pressures and relative humidities were obtained from historical Oakland airport readings.

Using the MOBILE6 methodology detailed at http://www.epa.gov/otaq/m6.htm, the pressure and relative humidity data were combined with ambient temperature for each run to calculate absolute humidity.

MOBILE6 Time And Geographical Input Parameters

Month of Year

MOBILE6 has the capability of modeling either a January 1 or July 1 run for any given year. The tunnel studies used in this work had performance periods that ranged from May to September. Thus, the July 1, or summer, setting was used in all cases.

Weekday/Weekend

MOBILE6 was set to use either weekday or weekend vehicle activity rates, depending on the historical day of week of each experimental run.

Altitude

MOBILE6 has a low and a high altitude setting. The low altitude setting translates to approximately 500 feet above mean sea level while the high altitude setting represents areas of about 5,500 feet above mean sea level. In all cases in this work, the elevation of the areas around the tunnels was much closer to 500 than 5,500 feet above mean sea level. Thus, the low altitude setting was always used.

Facility Type

The MOBILE6 facility type was designated as “Freeway” for all tunnels except Callahan. For that particular tunnel, the speed range is relatively wide from 14 to 35 mph with a corresponding low average. For these reasons, we believed that neither the Freeway nor Arterial cycle correctly represents the tunnel conditions. We chose to model the tunnel as “Arterial”. (To check the effects of this assumption, we modeled the tunnel under both designations.) For the conditions at this tunnel, the maximum differences (within this speed range) between freeway and arterial fleet average results are about 5% for CO, 3% for NMHC, and 8% for NOx).

Fuel and I/M Program Inputs

Fuel Inputs

All fuel inputs were obtained from National Institute for Petroleum and Energy Research (NIPER) data. We had access to NIPER data for the summer of 1993 and the summer of 1995. The nearest available year was chosen for each test run. NIPER provided RVP, sulfur,

and oxygenate data. Oxygenate data of about 2% volume MTBE or less was determined to be insignificant and was not used as MOBILE6 input.

Additionally, each tunnel area was checked for federal RFG status at the website http://www.eia.doe.gov/emeu/steo/pub/special/rfg2.html. Only two tunnel and year combinations fell within federal RFG areas, Lincoln and Callahan Tunnels for 1995. For these two cases, the RFG flag was set in MOBILE6, which automatically defines RVP and oxygenate content.

I/M and ATP Inputs

I/M and ATP program status for each tunnel area was determined based on data available at http://www.epa.gov/oms/epg/state.htm which summarizes the latest programs (i.e., enhanced I/M). According to the programs and start years specified at this web site, only one state was affected by enhanced I/M: California (CA). However, the San Francisco Bay Area is exempt from CA enhanced I/M program. Thus MOBILE6 I/M and ATP inputs for Caldecott were set according to the latter information. (Note that this does not influence the results since only HD emissions are included from this tunnel.) In addition, the Maryland, Pennsylvania, and Massachusetts state I/M offices were polled regarding historic programs for their respective states. Since the Tuscarora Mountain Tunnel was distant from any Pennsylvania I/M areas, no I/M was modeled. Maryland and Massachusetts operated basic I/M programs starting in 1984 and 1983, respectively. These were modeled as two-speed idle programs.

MOBILE6 Output Processing

MOBILE6 database output was used to obtain emission factors for each specific test hour. Only running exhaust and evaporative emissions were used, as all other start and evaporative emissions were assumed to be insignificant under the conditions of each study. The output was delineated by both vehicle class and model year. Fleet-average values were calculated from these model year and vehicle class-specific emission factors, using observed and MOBILE6 default (for vehicle classes with no observed data) age distribution and fleet mix data to appropriately weight the emissions.

Fleet-average emission factors obtained in the manner described above were compared directly to values observed in the tunnels. The corresponding pollutant ratios were also evaluated. Additional calculations were necessary before the vehicle class-specific comparisons can be made. The MOBILE6 LD and HD factors for each experimental run were computed by combining the appropriate vehicle classes' emission factors (i.e., vehicle classes 1 through 5, 14, and 15 were combined into LD and classes 6-13, 16-23, and 25-27 were combined into HD). Then, the emission factors for the individual runs at a single tunnel/bore were combined into a weighted average using the number of vehicles in each run as the weights.

For HD diesel vehicles, an additional issue must be resolved before comparisons can be made. This is the issue of NOx defeat devices. These devices purportedly increase NOx emissions from HD diesel trucks under steady-state operating conditions. MOBILE6 assumes that certain model years' experience increased NOx emissions due to the presence of such

mechanisms. Thus, it is important to determine whether the traffic conditions within a tunnel are conducive to these devices being in operation. If not, the MOBILE6 emission factors associated with the tunnel/run must be adjusted to reduce NOx emissions. We attempted to determine the exact operational criteria under which increased NOx would result (so that these can be compared to the tunnel conditions) but were unsuccessful. EPA documentation of this feature did not clearly specify the precise parameters. It stated that the on/off status of these devices for particular fleets and facility class/operational scenarios was determined using "proprietary and confidential data submitted by the engine manufacturers, limited testing of affected engines, and engineering judgment by experts in engine control and emission control software." (EPA, 2002a) Thus, no adjustment for excess NOx was made to the default model outputs.

Caldecott HD diesel NOx results were obtained in a different manner than the federal tunnels used in this study. Using a carbon balance and the observed concentration, the emission factor was originally calculated on a fuel-specific (g/kg fuel) basis. To convert to a g/mile basis, the fuel density (0.77331 g/ml) and the fuel economy (4.8 mile/gal) were required. Both of these values were taken from (Pierson et al., 1996), with the fuel economy representing HD vehicles moving uphill at the Fort McHenry Tunnel.

MOBILE6 results for the Caldecott Tunnel were also adjusted to reflect differences in CA and Federal HD NOx emission standards. In particular, a ratio-of-standards approach was used to correct model years 1987 to 1989. The CA and Federal standards for these model years were 6 and 10.7 g/bhp-hr, respectively. No adjustments were made to other model years because the standards were equivalent.

RESULTS AND DISCUSSION

Graphical results are presented below. A discussion follows at the end of the presentation of results.

Fleet-Average Emission Factors (Federal Area Tunnels)

Model-predicted emission factors as well as pollutant ratios are compared to observed data. In addition to the direct comparison between MOBILE6 and tunnel study data, analogous MOBILE4.1 and MOBILE5 predictions are also shown to assess model changes. Figures 2-1 through 2-14 show the predicted run-specific fleet average emission factors plotted against the corresponding observed value for Fort McHenry, both years of Tuscarora, and Callahan (1999 Tuscarora NMHC data are faulty and thus are omitted). Table 2-3 and Figures 2-15 and 2-16 present the pollutant ratios, where available.

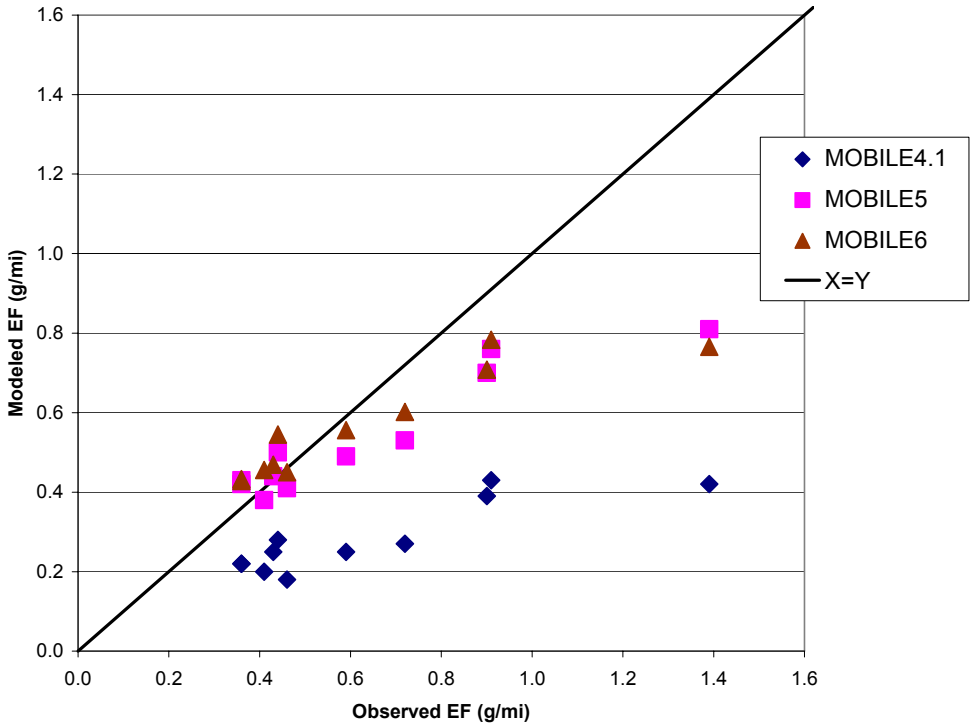


Figure 2-1. Comparison of observed to modeled fleet average NMHC emission factors at Fort McHenry (1992), Bore 3.

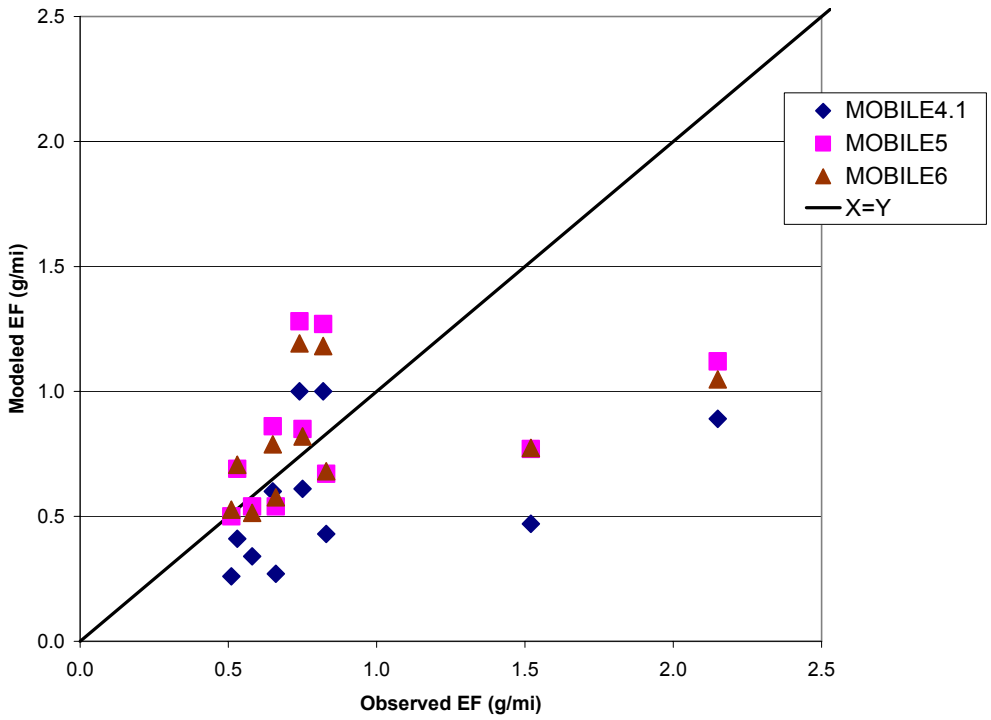


Figure 2-2. Comparison of observed to modeled fleet average NMHC emission factors at Fort McHenry (1992), Bore 4.

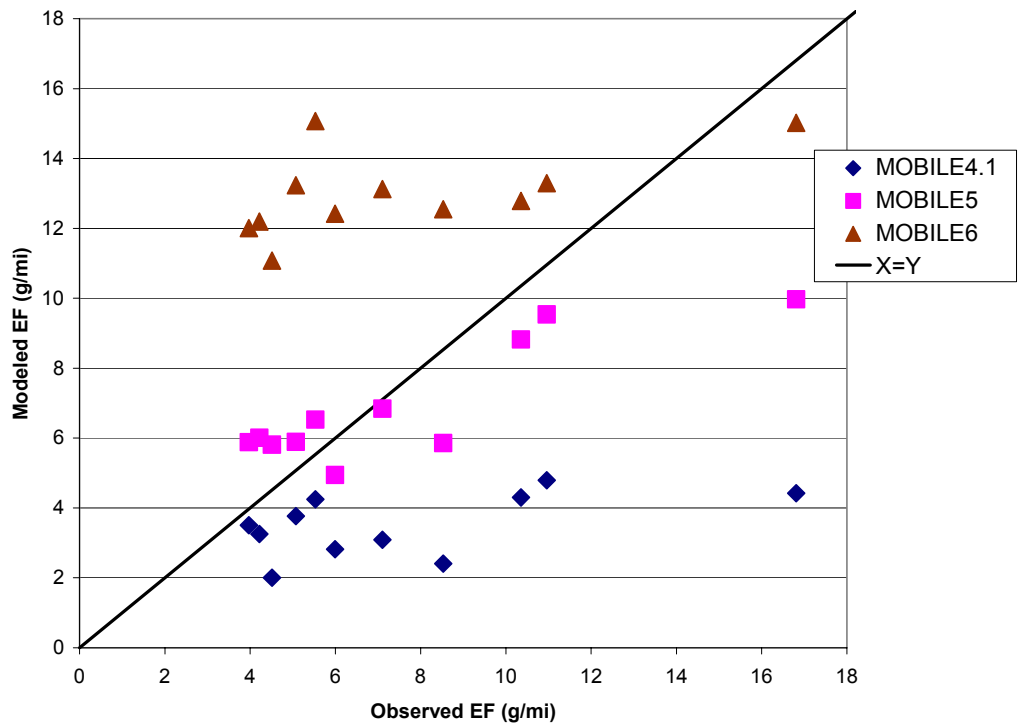


Figure 2-3. Comparison of observed to modeled fleet average CO emission factors at Fort McHenry (1992), Bore 3.

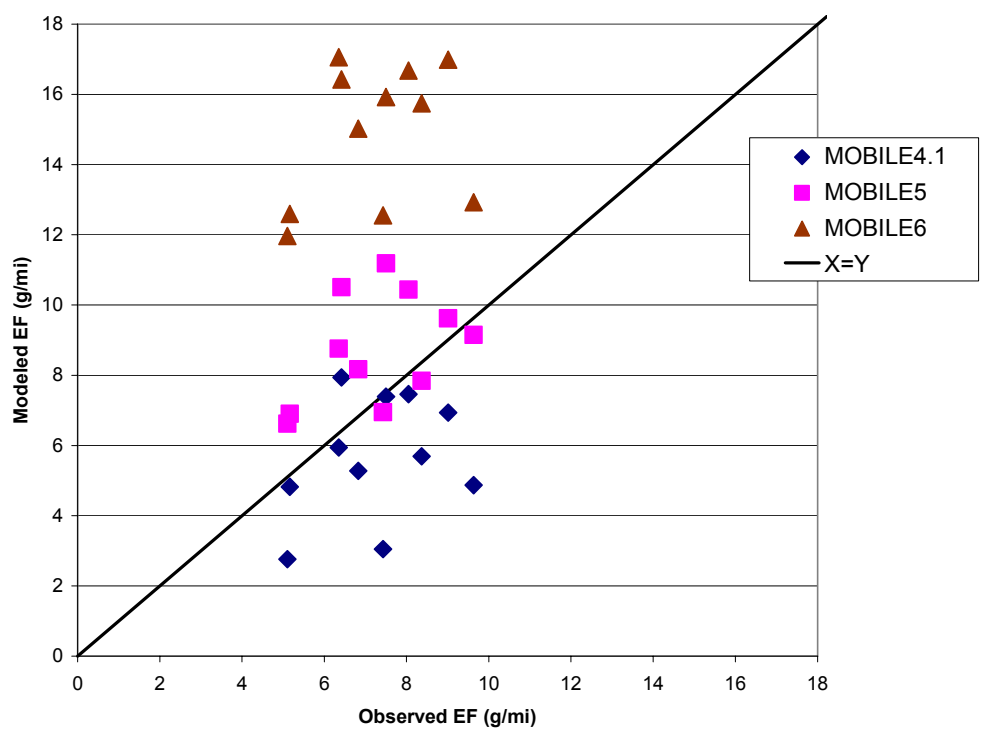


Figure 2-4. Comparison of observed to modeled fleet average CO emission factors at Fort McHenry (1992), Bore 4.

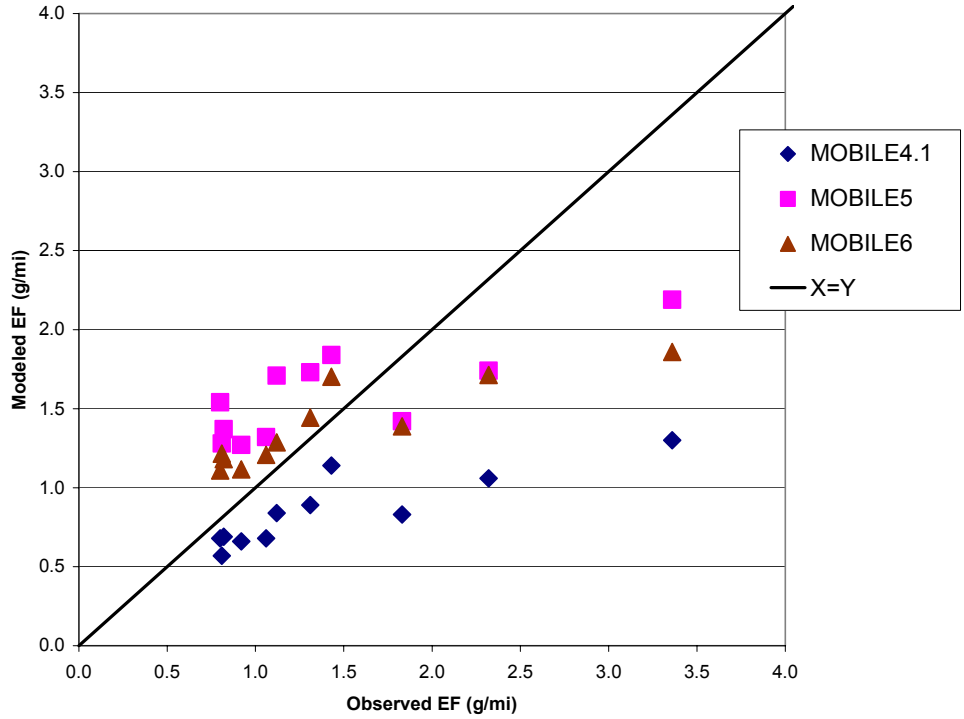


Figure 2-5. Comparison of observed to modeled fleet average NOx emission factors at Fort McHenry (1992), Bore 3.

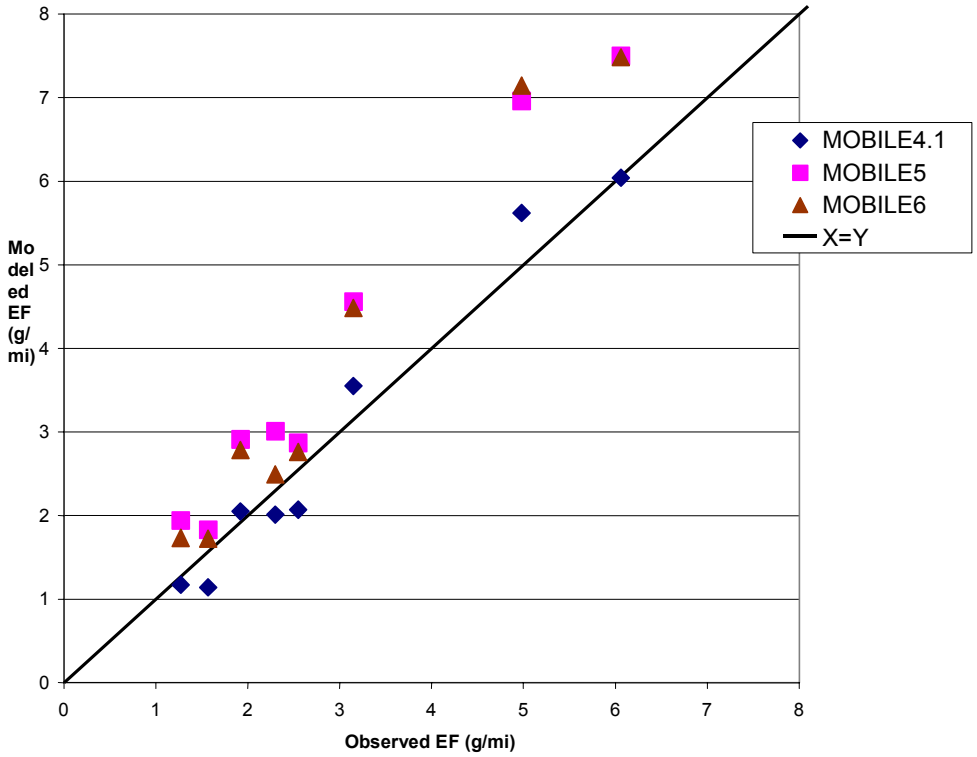


Figure 2-6. Comparison of observed to modeled fleet average NOx emission factors at Fort McHenry (1992), Bore 4.

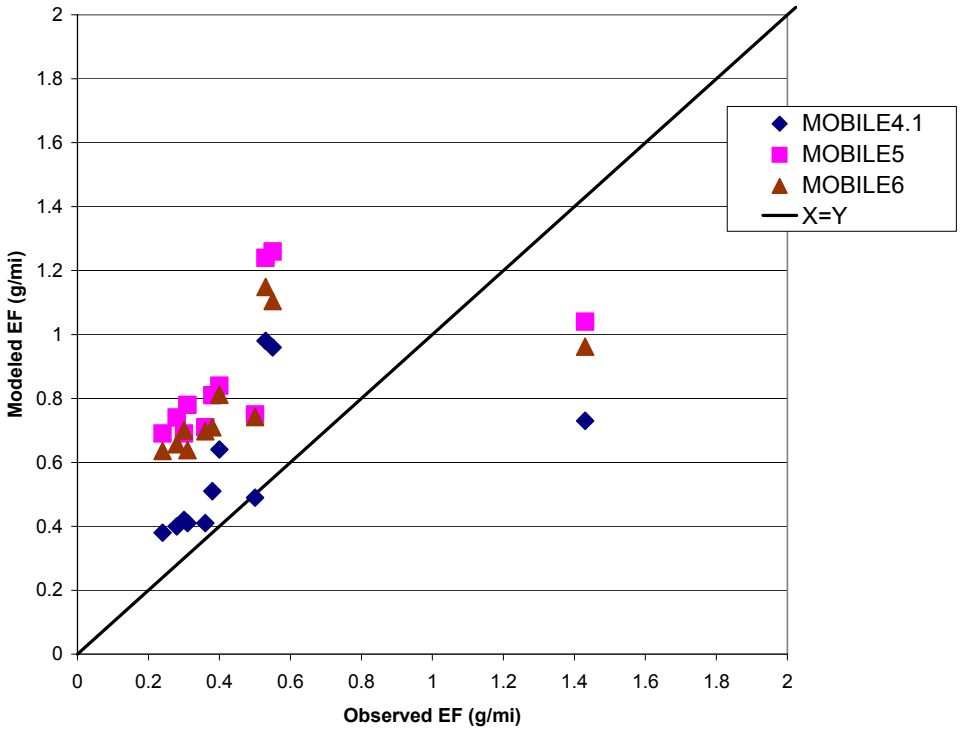


Figure 2-7. Comparison of observed and modeled fleet average NMHC emission factors at Tuscarora Mountain (1992).

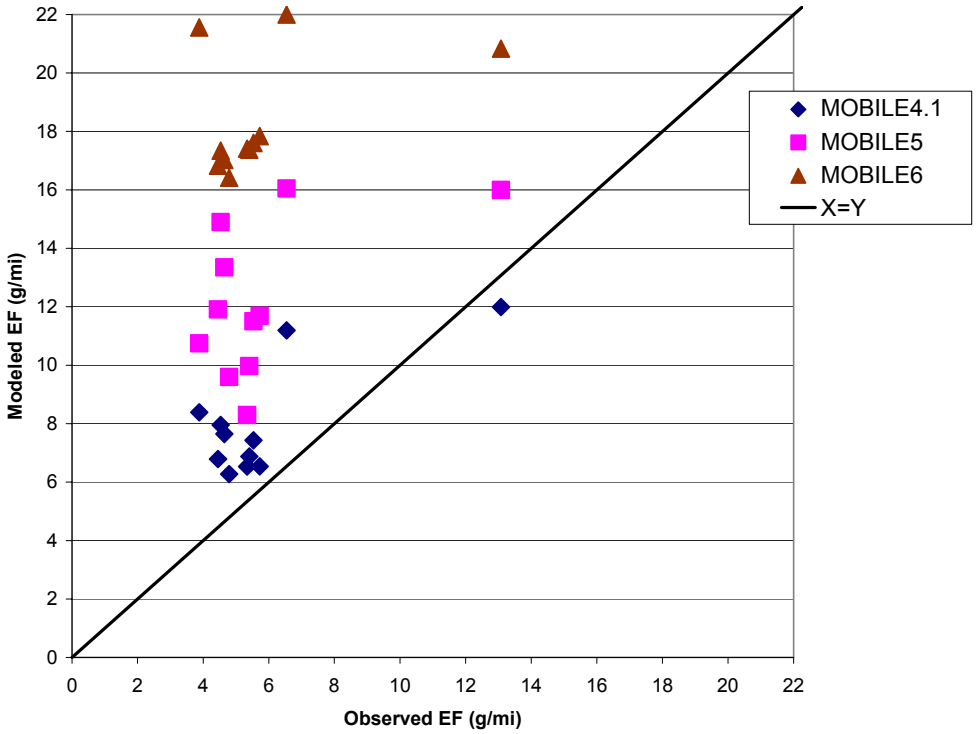


Figure 2-8. Comparison of observed and modeled fleet average CO emission factors at Tuscarora Mountain (1992).

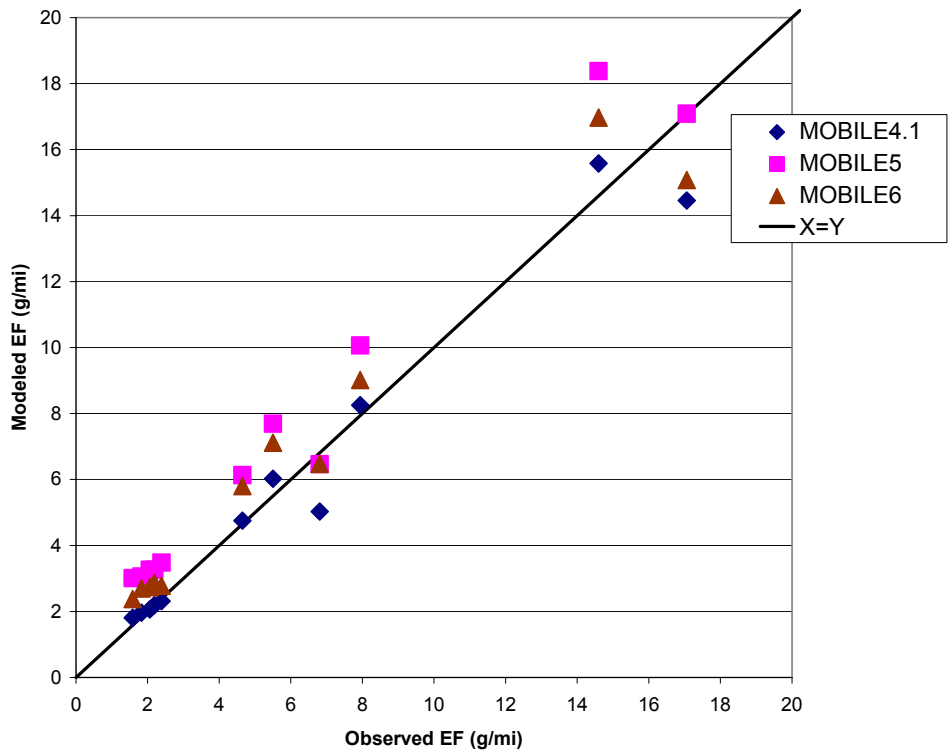


Figure 2-9. Comparison of observed and modeled fleet average NOx emission factors at Tuscarora Mountain (1992).

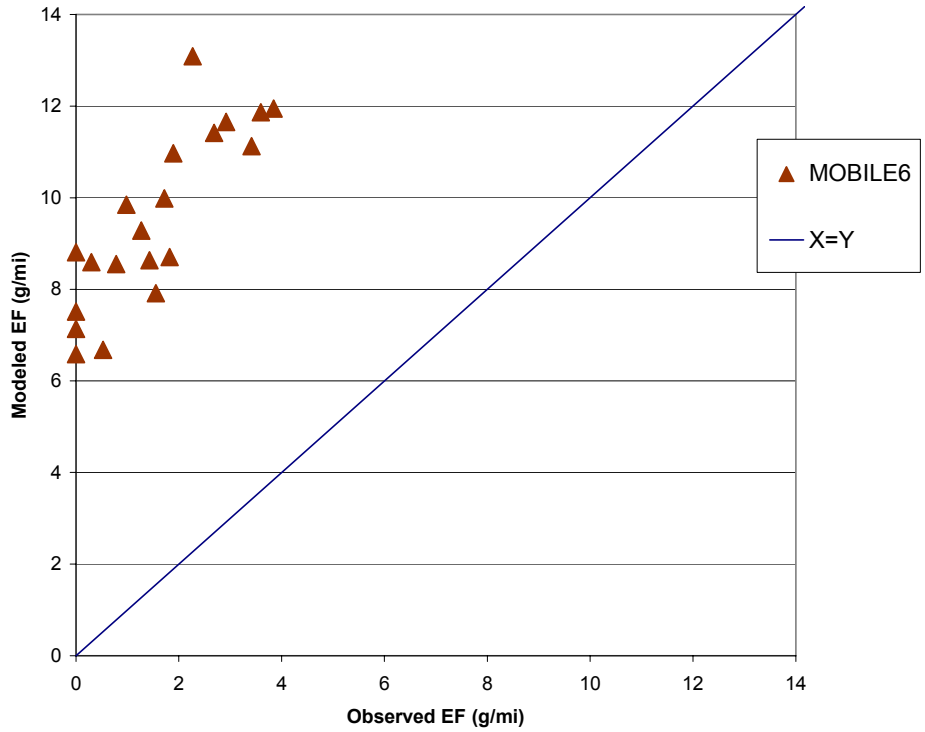


Figure 2-10. Comparison of observed and modeled fleet average CO emission factors at Tuscarora Mountain (1999).

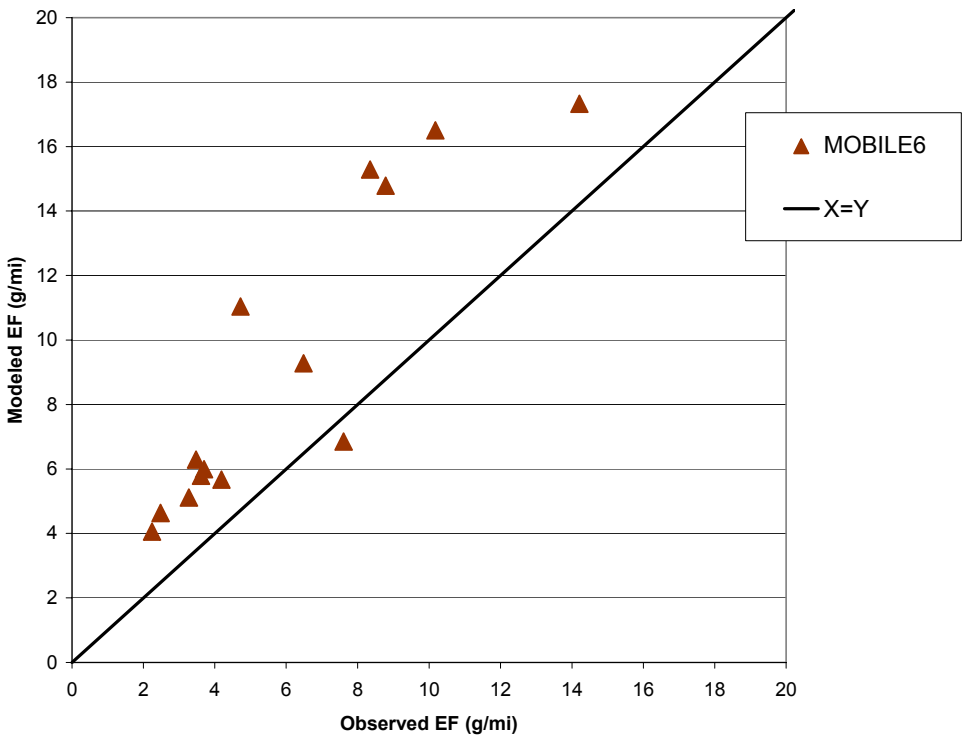


Figure 2-11. Comparison of observed and modeled fleet average NOx emission factors at Tuscarora Mountain (1999).

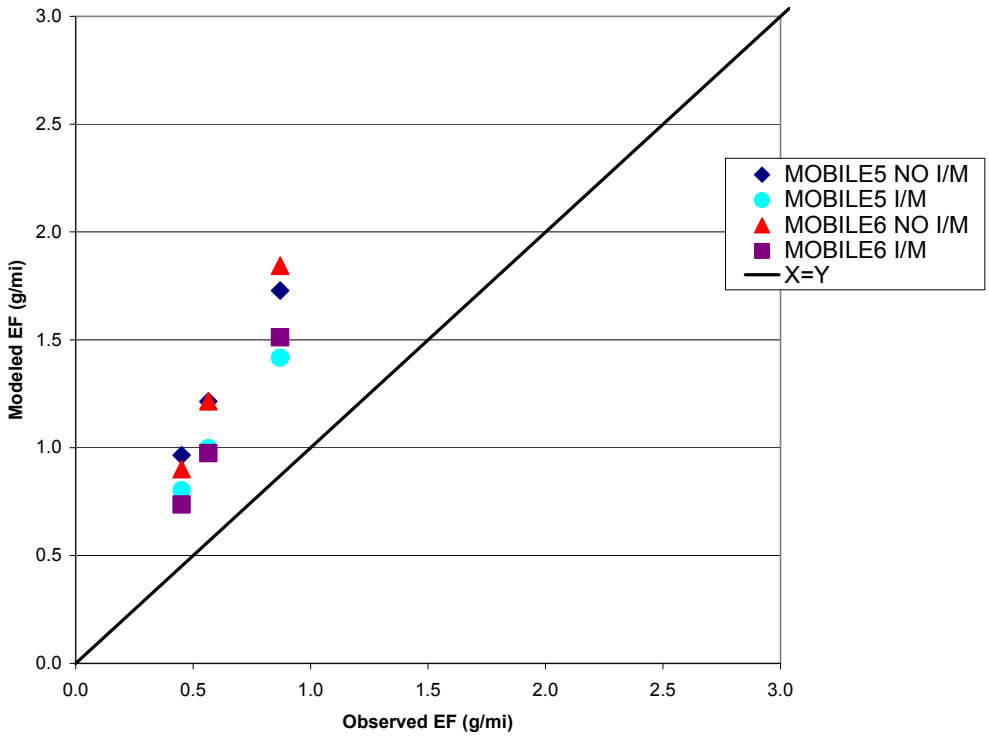


Figure 2-12. Comparison of observed and modeled fleet average NMHC emission factors at Callahan Tunnel (1995).

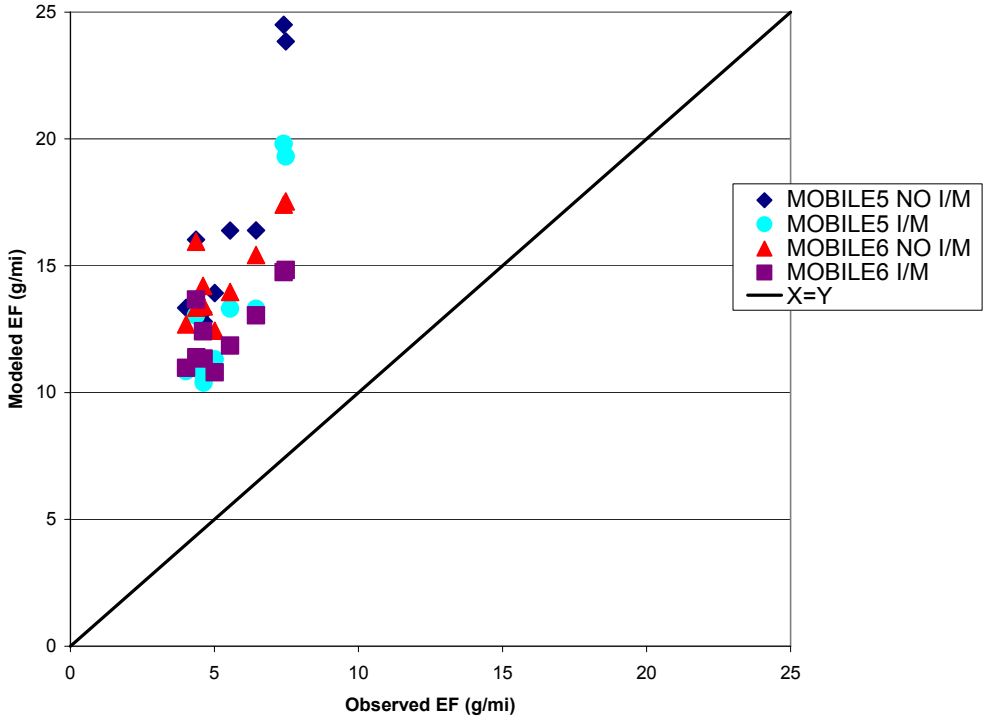


Figure 2-13. Comparison of observed and modeled fleet average CO emission factors at Callahan Tunnel (1995).

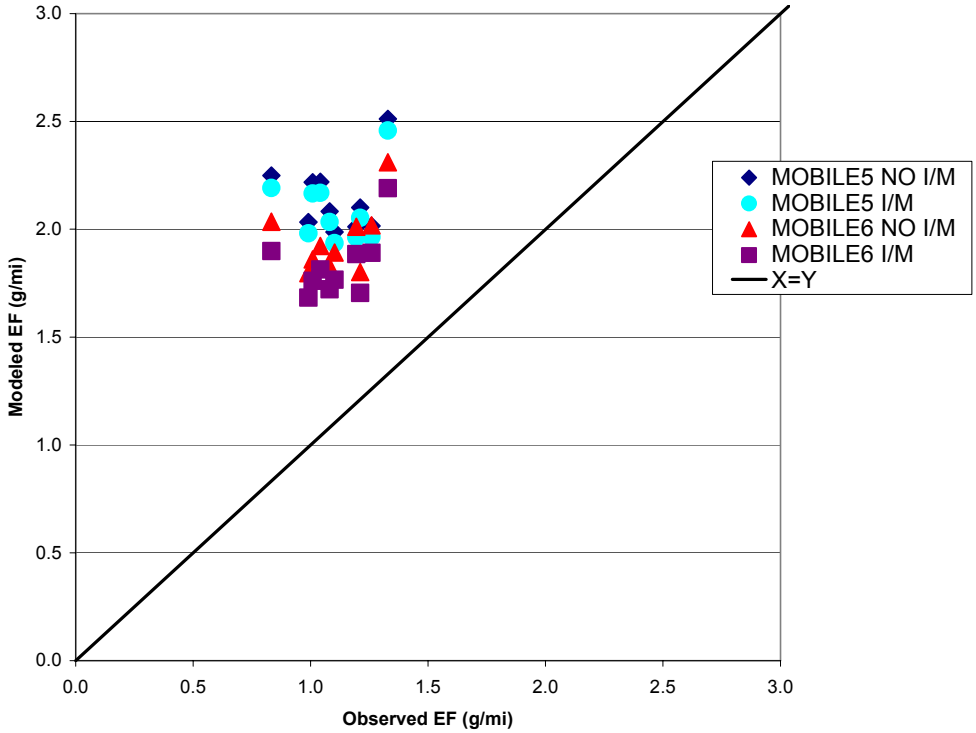


Figure 2-14. Comparison of observed and modeled fleet average NOx emission factors at Callahan Tunnel (1995).

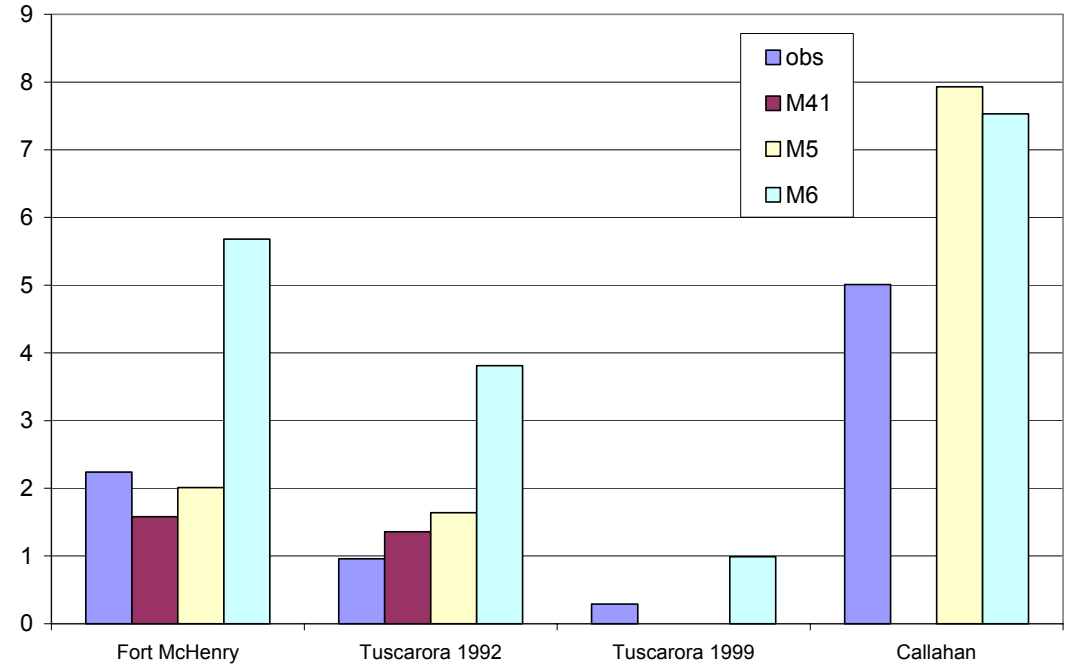


Figure 2-15. Observed and predicted CO/NOx ratios.

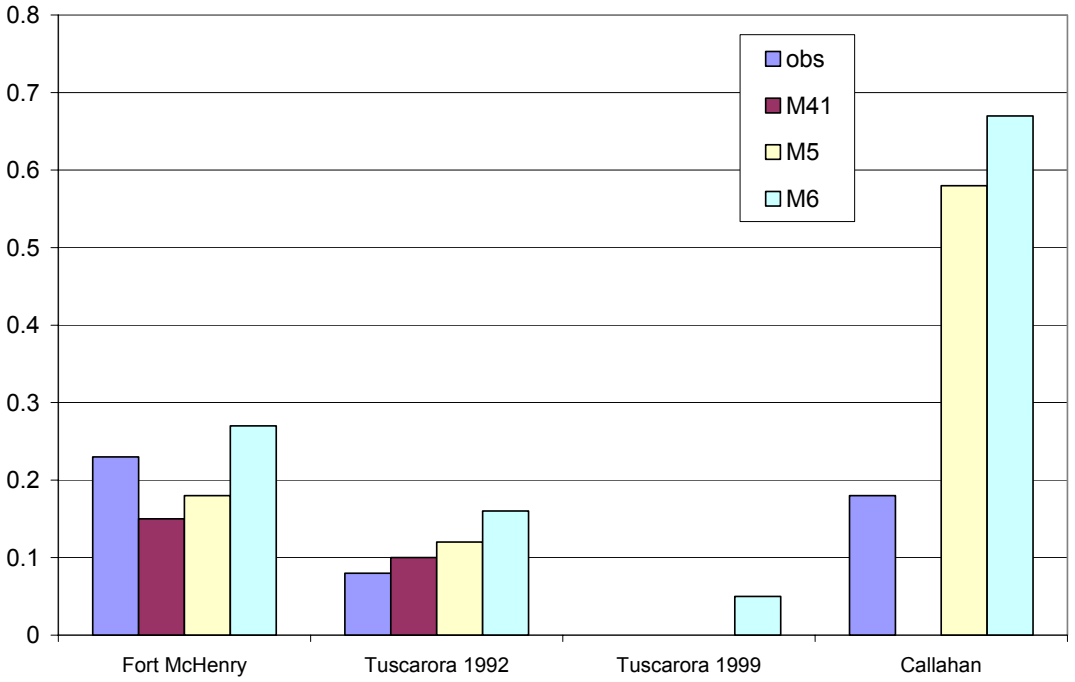


Figure 2-16. Observed and predicted NMHC/NOx ratios.

Table 2-3. Ratio of pollutants for the overall fleet.

CO/NOx	Fort McHenry	Observed	2.24
		MOBILE4.1	1.58
		MOBILE5	2.01
		MOBILE6	5.19
	Tuscarora 1992	Observed	0.96
		MOBILE4.1	1.36
		MOBILE5	1.64
		MOBILE6	3.89
	Callahan	Observed	5.01
		MOBILE4.1	na
		MOBILE5	7.93
		MOBILE6	6.93
	Tuscarora 1999	Observed	0.29
		MOBILE4.1	na
		MOBILE5	na
		MOBILE6	0.99
NMHC/NOx	Fort McHenry	Observed	0.23
		MOBILE4.1	0.15
		MOBILE5	0.18
		MOBILE6	0.23
	Tuscarora 1992	Observed	0.08
		MOBILE4.1	0.10
		MOBILE5	0.12
		MOBILE6	0.16
	Callahan	Observed	0.18
		MOBILE4.1	na
		MOBILE5	0.58
		MOBILE6	0.57
	Tuscarora 1999	Observed	na
		MOBILE4.1	na
		MOBILE5	na
		MOBILE6	0.05

Light-Duty Vehicle Emission Factors (Federal Area Tunnels)

The observed MOBILE4.1, and MOBILE5 LD emission factors were derived from fleet average values via regression analysis. Pierson et al. derived the MOBILE4.1 and MOBILE5 LD emission factors from fleet average values using weighted regressions in order to attenuate the influence of high emitters (Pierson et al., 1996). The standard errors associated with the regressions are shown below as error bars. (Note that MOBILE6 factors used in this work were not derived but rather came directly from the model. We felt that using the direct vehicle class specific model results would give a clearer assessment of the model's estimates. As such, these emission factors do not have predicted errors since these errors would be associated solely with the error in the model, the determination of which is beyond the scope of this work.) Figures 2-17 through 2-19 depict comparisons of observed and modeled emission factors for Fort McHenry and 1992 Tuscarora data. Table 2-4 and Figures 2-20 and 2-21 summarize the corresponding NMHC/NOx and CO/NOx ratios. Fort McHenry LD data, as shown, were combined for both bores. As discussed in the following section, the MOBILE6 values are weighted averages, with the total number of vehicles in each run as the weighting factors.

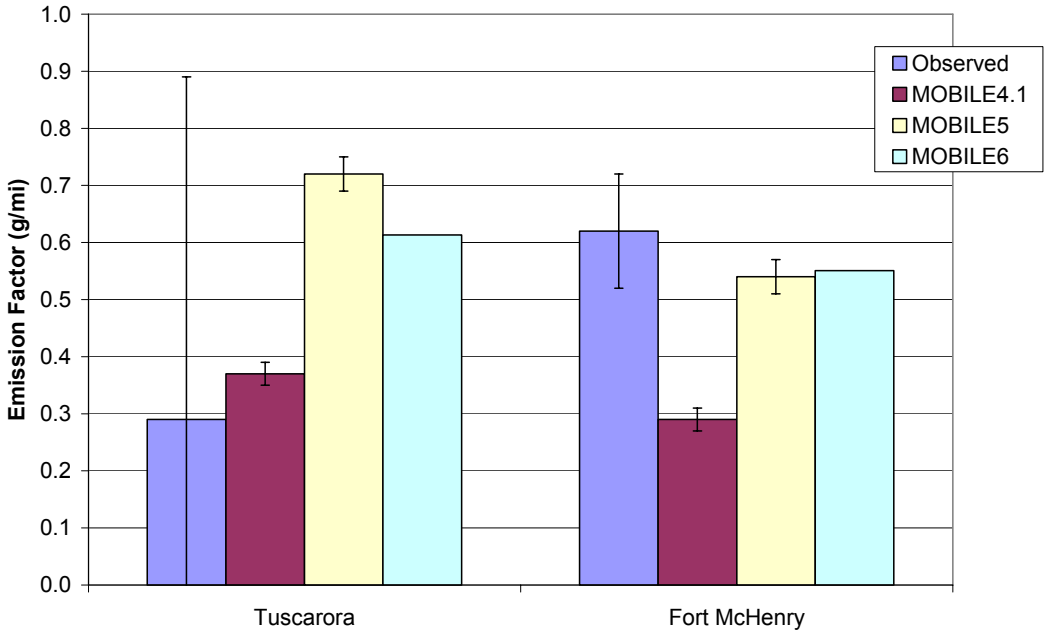


Figure 2-17. Comparison of observed and modeled light-duty NMHC emission factors at Fort McHenry and Tuscarora Mountain (both 1992).

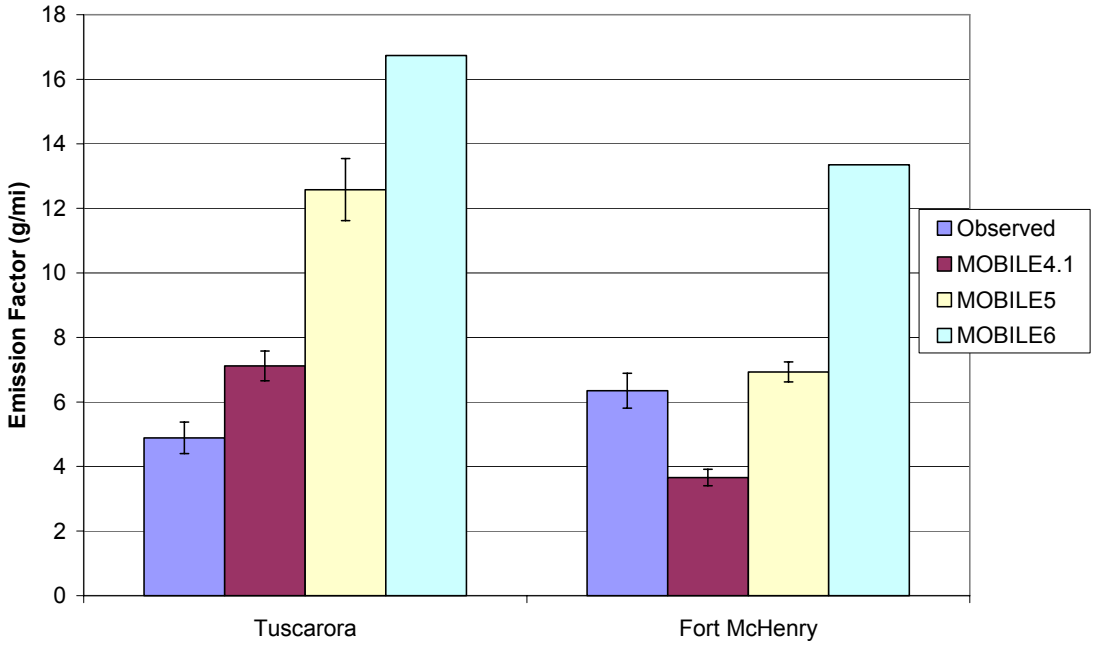


Figure 2-18. Comparison of observed and modeled light-duty CO emission factors at Fort McHenry and Tuscarora Mountain (both 1992).

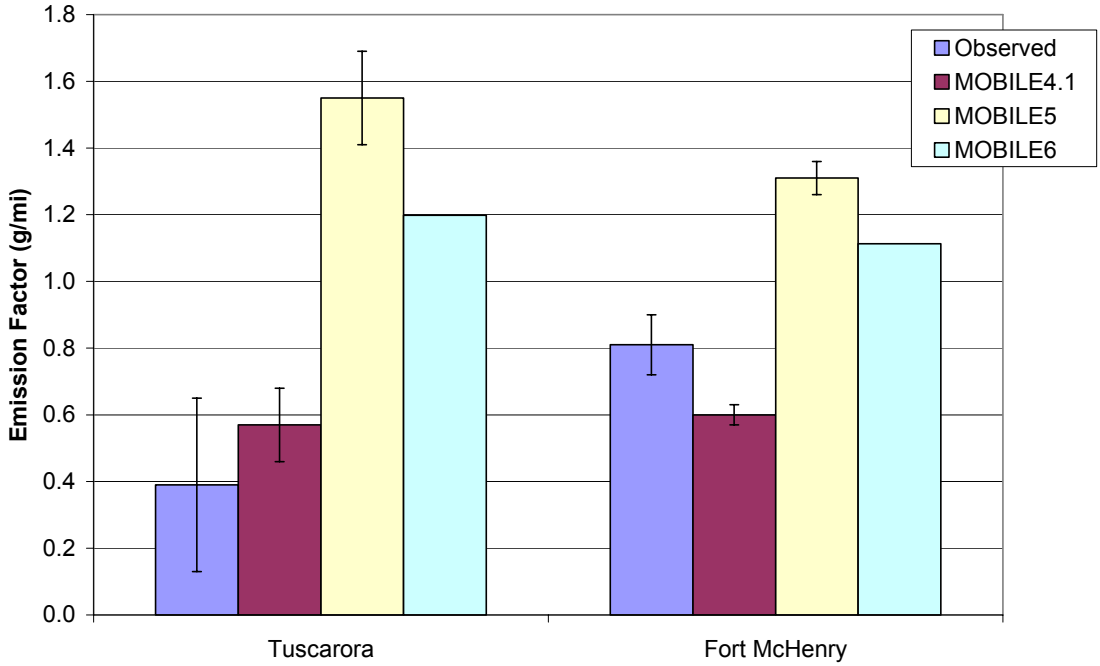


Figure 2-19. Comparison of observed and modeled light-duty NOx emission factors at Fort McHenry and Tuscarora Mountain (both 1992).

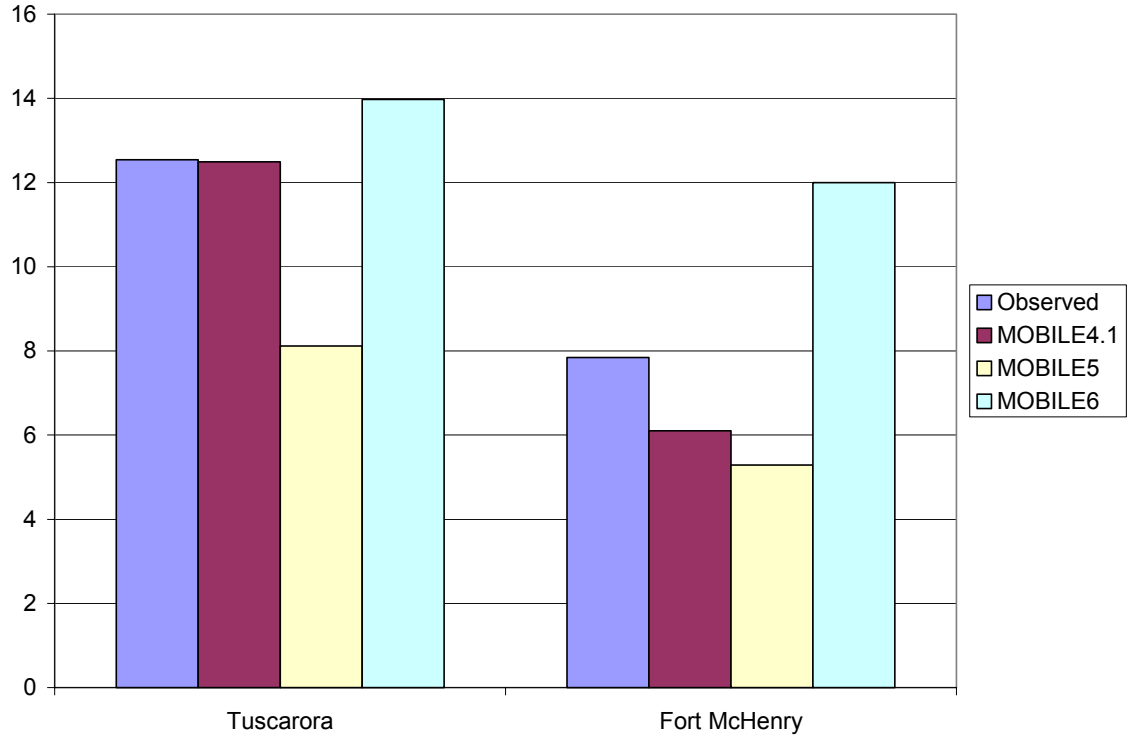


Figure 2-20. Observed and predicted light-duty CO/NOx ratios.

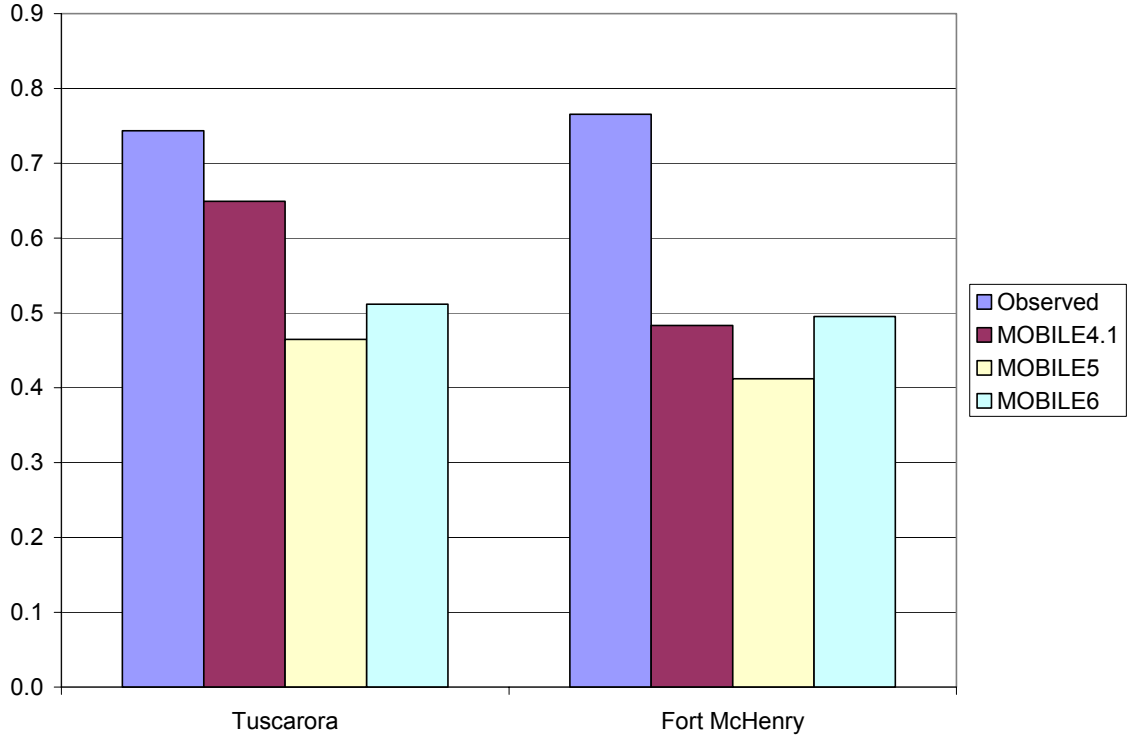


Figure 2-21. Observed and predicted light-duty NMHC/NOx ratios.

Table 2-4. Ratio of pollutants for LDVs.

CO/NOx	Fort McHenry	Observed	7.8 ± 1.15
		MOBILE4.1	6.1 ± 0.5
		MOBILE5	5.3 ± 0.3
		MOBILE6	12.0
	Tuscarora 1992	Observed	12.7 ± 8.5
		MOBILE4.1	12.5 ± 2.5
		MOBILE5	8.1 ± 1.0
		MOBILE6	14.0
NMHC/NOx	Fort McHenry	Observed	0.76 ± 0.14
		MOBILE4.1	0.48 ± 0.04
		MOBILE5	0.41 ± 0.03
		MOBILE6	0.51
	Tuscarora 1992	Observed	0.76 ± 0.53
		MOBILE4.1	0.65 ± 0.13
		MOBILE5	0.46 ± 0.05
		MOBILE6	0.50

Heavy-Duty Vehicle Emission Factors (Federal Area and CA Tunnels)

The components of this analysis are similar to those for LD vehicles described above. However, both Federal and CA tunnel data were used. The emission factor results and ratio of pollutants are shown in Figures 2-23 through 2-25 and Table 2-5, respectively. Uncertainties in the emission factors were estimated similarly to the LD case discussed above. Figures 2-26 and 2-27 present the ratios graphically. Because data collected at the Lincoln and Deck Park Tunnels reflect a very narrow range of fleet mixes, the regression method cannot be reliably applied to derive HD emission rates. Note that although the fleet mix at the Caldecott Tunnel shows a similar narrow variation, the HD emission factor was derived using a carbon mass balance approach. Thus the result was not nullified by limitations of a regression approach. Figure 2-22, which shows NMHC results for Deck Park, is an illustration of this unreliability; all of the LD fractions are between 0.9 and 1.0 and extrapolating back to zero LD fraction to estimate the HD emission factor would be highly uncertain (in fact in this case, it is negative). 1999 Tuscarora CO readings were very low and therefore also adversely affected our ability to resolve LD/HD contributions. Thus, although some of these results are available, they are not used in the assessments of model performance with regard to HD vehicles.

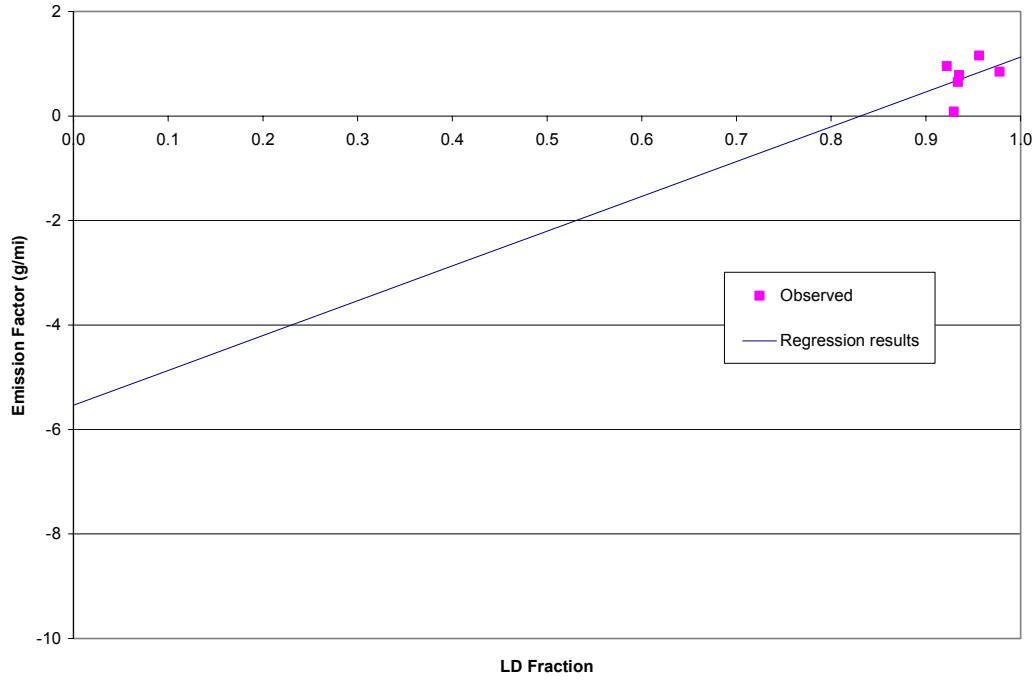


Figure 2-22. Illustration of inappropriate results obtained via regression.

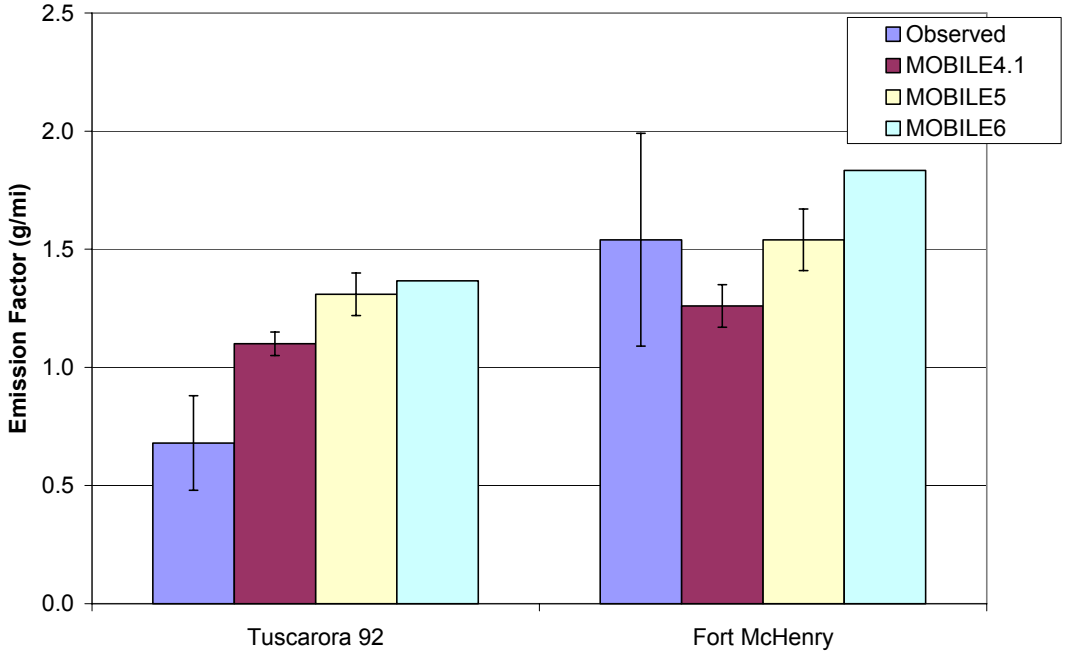


Figure 2-23. Comparison of observed and modeled heavy-duty NMHC emission factors at Fort McHenry and Tuscarora Mountain (both 1992).

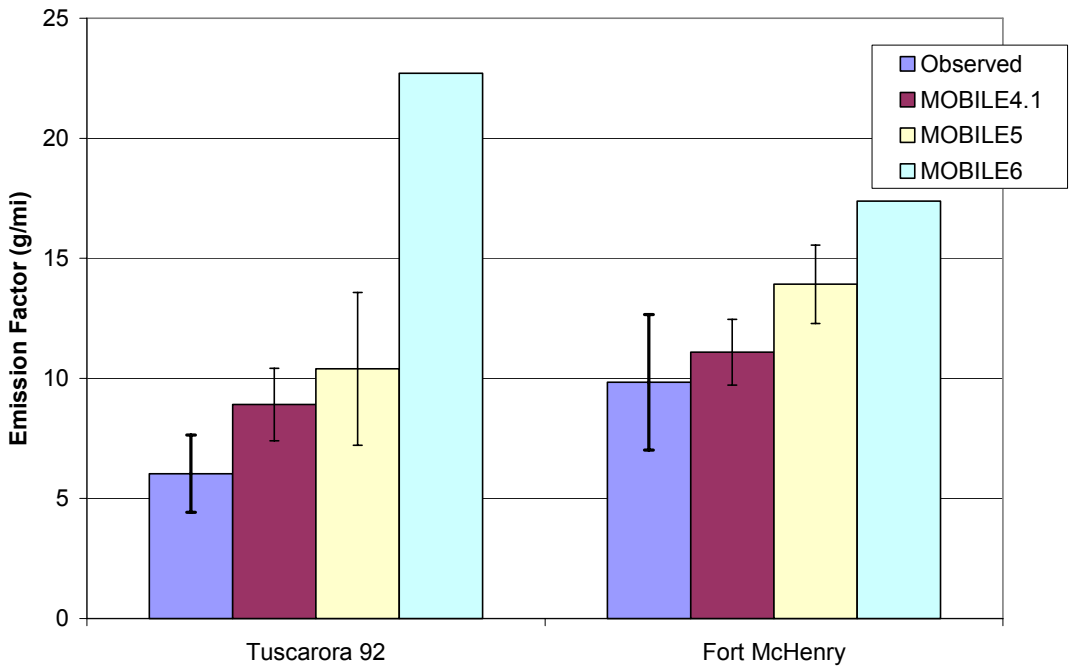


Figure 2-24. Comparison of observed and modeled heavy-duty CO emission factors at Fort McHenry and Tuscarora Mountain (both 1992).

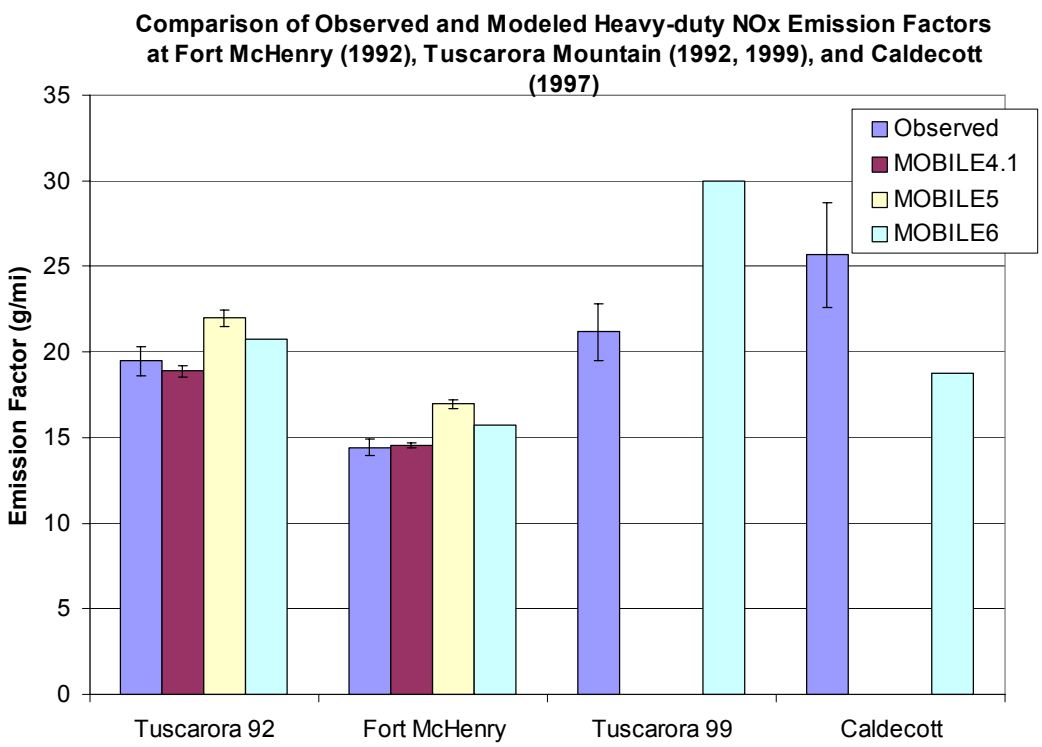


Figure 2-25. Comparison of observed and modeled heavy-duty NOx emission factors at Fort McHenry (1992), Tuscarora Mountain (1992, 1999), Lincoln and Deck Park (both 1995), and Caldecott (1997).

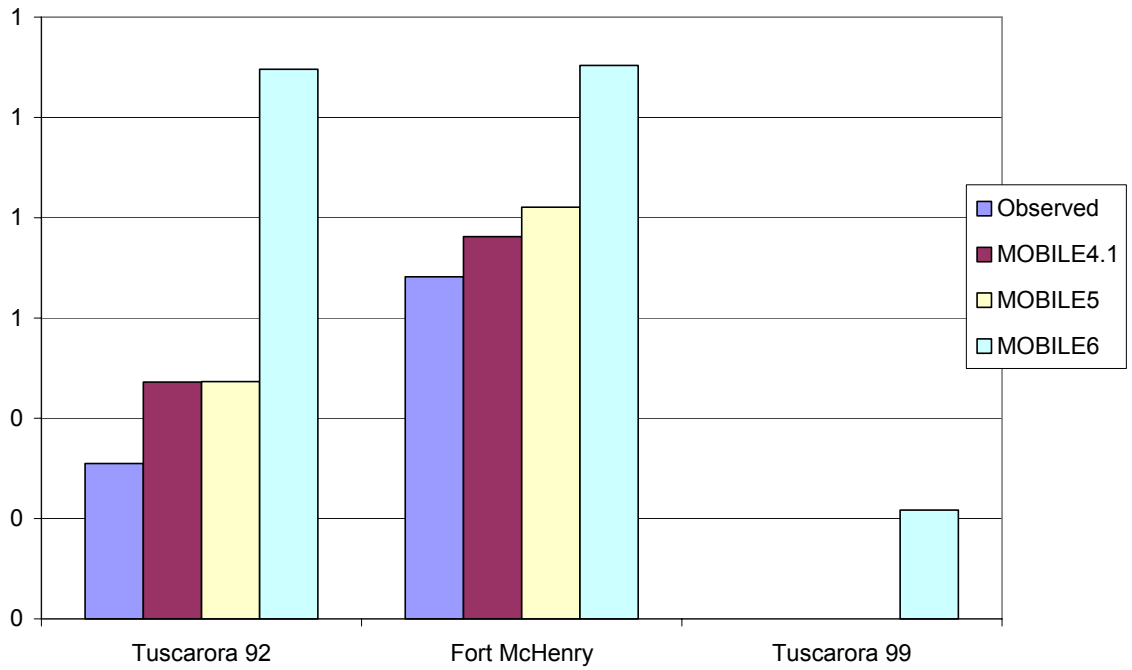


Figure 2-26. Observed and predicted heavy-duty CO/NOx ratios.

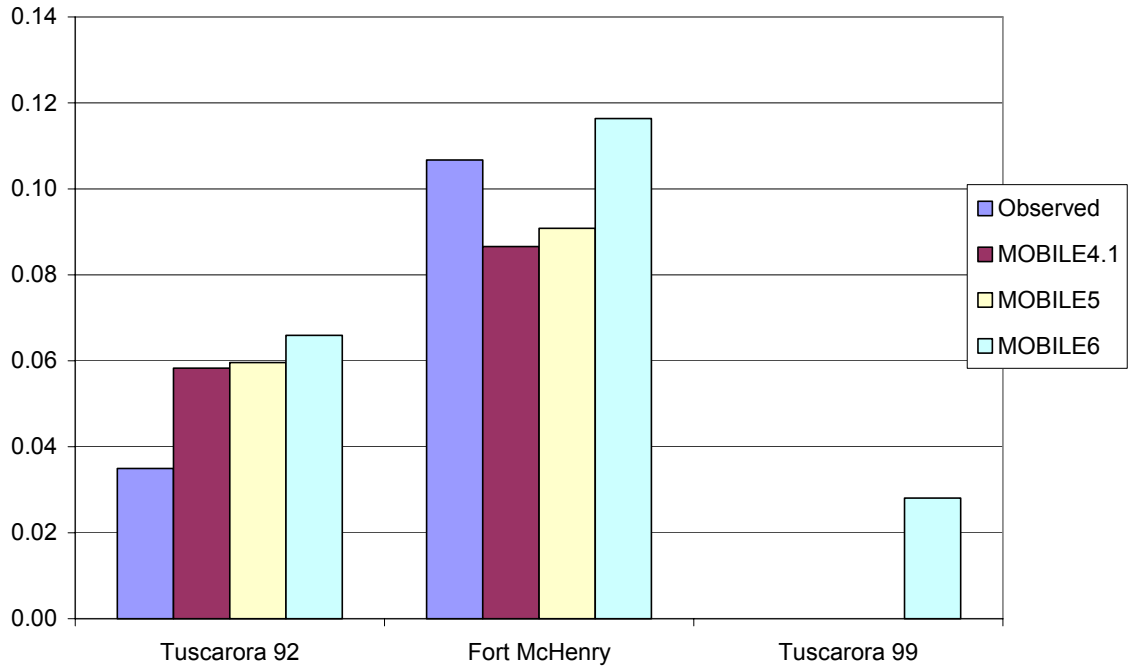


Figure 2-27. Observed and predicted heavy-duty NMHC/NOx ratios

Table 2-5. Ratio of pollutants for HDVs.

CO/NOx	Fort McHenry	Observed	0.68 ± 0.20
		MOBILE4.1	0.76 ± 0.09
		MOBILE5	0.82 ± 0.10
		MOBILE6	1.10
	Tuscarora 1992	Observed	0.31 ± 0.08
		MOBILE4.1	0.47 ± 0.08
		MOBILE5	0.47 ± 0.14
		MOBILE6	1.10
	Tuscarora 1999	Observed	na
		MOBILE4.1	na
		MOBILE5	na
		MOBILE6	0.22
	Caldecott	Observed	na
		MOBILE4.1	na
		MOBILE5	na
		MOBILE6	na
NMHC/NOx	Fort McHenry	Observed	0.107 ± 0.032
		MOBILE4.1	0.086 ± 0.006
		MOBILE5	0.091 ± 0.008
		MOBILE6	0.12
	Tuscarora 1992	Observed	0.035 ± 0.010
		MOBILE4.1	0.058 ± 0.003
		MOBILE5	0.059 ± 0.004
		MOBILE6	0.07
	Tuscarora 1999	Observed	na
		MOBILE4.1	na
		MOBILE5	na
		MOBILE6	0.03
	Caldecott	Observed	na
		MOBILE4.1	na
		MOBILE5	na
		MOBILE6	na

Discussion

Due to competing factors, it is difficult to predict MOBILE6 results relative to previous versions for any *particular* set of conditions. We approach this analysis by first identifying the general trends due to changes between versions and then seek probable explanations for deviations from these trends.

Major factors updated in MOBILE6 that affect exhaust emissions include:

- Off-cycle driving and air conditioning
- Sulfur on catalysts
- HD excess NOx (only on MY 1988-2000)
- Newer technologies' deterioration

For reference, Table 2-6 shows national fleet-average increases (relative to MOBILE5), incorporating all changes in MOBILE6.

Table 2-6. National fleet level increases in emission factors from MOBILE5 to MOBILE6.

Year	CO	NOx	VOC
1992	60%	25%	50%
1995	50%	25%	45%

Source: EPA presentation on MOBILE5/MOBILE6 (EPA, 2001b).

Updated speed corrections also have significant impacts and the directional effects depend upon the speed and pollutant. For the speeds involved in the tunnels above, the following approximate effects (relative to MOBILE5) are noted for LD vehicles:

Table 2-7. Selected speed effects changes from MOBILE5 to MOBILE6.

Tunnel	Average Speed (mph)	CO	NOx	VOC
Fort McHenry	48	+100%	-25%	+40%
Tuscarora 1992	58	+100%	-40%	+15%
Callahan	26	+20%	-15%	+15%

Source: EPA MOBILE6 documentation of speed corrections, Figures 6a-c.

According to EPA's recent analysis of MOBILE6 model sensitivity (available at http://www.epa.gov/ttn/chief/conference/ei11/mobile/giannelli.pdf), age distribution, temperature, and speed are the three most influential factors. As an illustration of the effects of age distribution, according to the above reference, a 20 percent shift to older vehicles results in approximately 50%, 50%, and 40% increases in HC, CO, and NOx, respectively.

Fleet-average Results

Fleet-average MOBILE6 NOx predictions are generally lower than MOBILE5 results but not by much, and with the exception of the Callahan Tunnel, they still remain within the vicinity of the observed data. This continues the historic trend (observed by Gertler et al. 1997, 1997b) that NOx is generally the pollutant most accurately predicted by these models.

Comparisons of NMHC results indicate small differences between MOBILE6 and MOBILE5. In some instances, these differences lead to slightly better agreement with observed data and in others, they do not. From the tables above, MOBILE6 LD results are expected to be higher; however, the presence of a sizeable HD fleet acts to reduce the increases predicted in Tables 2-6 and 2-7. In all these cases, MOBILE5/6 still tends to overpredict when the observed emission factors are small and underpredict when these are large. Upon examining the experimental data corresponding to the high observed emissions, we note that three out of the four runs have much lower total vehicle counts than the other experimental runs in the same tunnel. Noteworthy is that neither extreme speed nor temperature was present in these three runs. (Even if there were, these effects, along with fleet mix, should have been accounted for in the model.) A plausible explanation is that high emitters might have been present and strongly affected the observed emission factors, and in fact according to DRI, three high

emitters were observed during Run 8 at Tuscarora Tunnel through use of remote sensing. Table 2-8 summarizes the experimental runs with high emission factors. (Run 11 in Bore 4 at Fort McHenry seems to have experienced congestion).

Table 2-8. NHMC results and other information related to experimental runs with high emission factors.

Run Description	Number of Vehicles*	Avg. Speed* (mph)	Temperature* (F)	EF* (g/mi)	MOBIL5.0/ MOBILE6 (g/mi)
Ft. McHenry Bore3, Run8	102 (1133)	45 (48)	64 (70)	1.39 (0.63)	0.81/1.00
Ft. McHenry Bore4, Run2	279 (1291)	46 (48)	70 (70)	2.15 (0.89)	1.12/1.10
Ft. McHenry Bore4, Run 11	1836 (1291)	38 (48)	70 (70)	1.52 (0.89)	0.77/0.90
Tuscarora 1992,Run8	79 (539)	58 (58)	65 (67)	1.4 (0.48)	1.0/0.96

* Average values across all runs for the particular tunnel study are shown in parentheses.

MOBILE6 CO results are much higher than MOBILE5 values (and hence observed values) for Ft. McHenry (both bores) and Tuscarora Tunnel (1992). From the speed effects noted above in Table 2-7, this is not surprising. However, they are slightly lower for the Callahan Tunnel. The lower humidity (61 grains/lb air vs. 79 and 91 for Ft. McHenry and Tuscarora) which decreases A/C usage contributes to this observation in a minor way. More importantly, the speed assumed in Table 2-7 is an average. Speeds at the Callahan Connector show the largest variation (see Table 2-2) and Figure 2-28 shows that MOBILE5 has larger speed correction factors at the lower speeds. These facts corroborate to yield the lower MOBILE6 predictions.

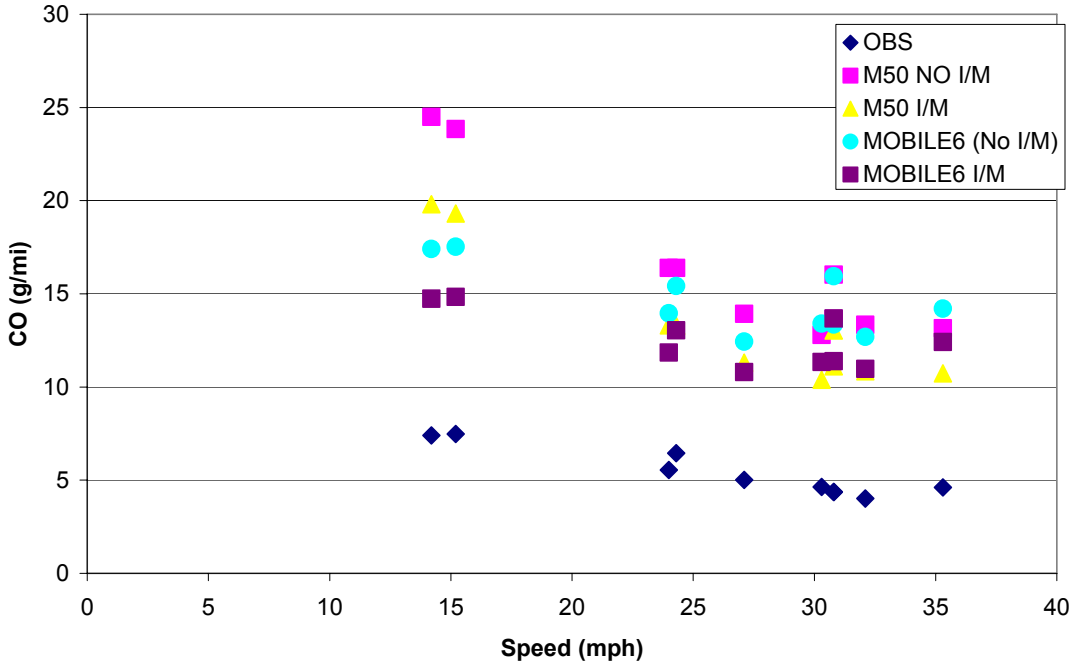


Figure 2-28. Speed effects on CO emission factors at Callahan (1995).

MOBILE also overpredicts NOx and NMHC at Callahan. The fleet at this tunnel is the oldest of the three, with 27.2 percent being older than ten years while the next oldest fleet (Tuscarora 1992) has only 17.8 percent older than ten years. (Model year distributions were all obtained by matching video license plate data.) This seems to suggest that the emission factors from the older model years are overestimated. Another factor is that no toll plaza exists so that traffic flow is smooth, albeit slow (i.e., very little acceleration inside the tunnel).

In the foregoing discussion, all observed data presented were a combination of uphill and downhill measurements (except Tuscarora, which is flat). Thus, the effects of grades were implicit. Robinson et al., (1996) explicitly reported the effects of grades at the Fort McHenry Tunnel. (A sampler was placed at a mid-tunnel point in order to separate the uphill and downhill portions.) The average results are presented in Table 2-9. (MOBILE6 results are from this study.) Note that the differences between uphill and downhill are more pronounced in Bore 4, which has a considerable number of HD trucks. In other words, grades have a larger impact on the HD vehicles. Also important is the fact that MOBILE6 predictions can be greater than the ascending value despite the fact that the model does not account for the effects of grades.

Table 2-9. Effects of grades at the Fort McHenry Tunnel. Based on Tables 5 and 6 of (Robinson et al, 1996).

		Bore 3	Bore 4
CO (g/mi)	DESCEND	5.06	5.11
	ASCEND	9.28	9.90
	M41	3.67	5.01
	M50	6.80	8.32
	M60	14.79	16.39
NMHC (g/mi)	DESCEND	0.54	0.55
	ASCEND	0.64	1.17
	M41	0.28	0.43
	M50	0.52	0.69
	M60	0.69	0.89
NOx (g/mi)	DESCEND	0.81	2.04
	ASCEND	1.70	4.79
	M41	0.82	2.97
	M50	1.57	3.94
	M60	1.44	5.98

The two studies performed at the Tuscarora Mountain Tunnel provide some insight into the trends in fleet-average emissions as well as the MOBILE6's ability to predict those trends. Table 2-10 summarizes the observed and modeled CO and NOx emission factors. The most striking change is the decrease in average CO emissions which is by a factor of about three. In fact, the raw data show several runs where the derived CO emission factor is below the detection limit. Not surprisingly, the observed NOx increased between 1992 and 1999. This is expected due to the purported heavy-duty off-cycle NOx. Overall, modeled emission factors seem to match the observed values more closely in 1992 than 1999 for both pollutants, with CO being the weaker match.

Table 2-10. Changes in fleet-average observed and modeled emission factors between 1992 and 1999 at Tuscarora Mountain Tunnel.

	CO				NOx			
	1992		1999		1992		1999	
Description	OBS	M6	OBS	M6	OBS	M6	OBS	M6
Minimum	3.88	16.42	0.00	6.58	1.5	2.38	2.25	4.06
Maximum	13.08	22.00	3.84	13.09	17.06	16.97	20.23	26.04
Average	5.81	18.39	1.55	9.52	6.06	6.72	9.14	13.39

Light-duty Results

MOBILE6 results for NOx are lower than for MOBILE5, probably due to the speed effects noted above in Table 2-7. Note, though, that the fleet-average increases shown in Table 2-6 are strongly affected by HD vehicles so they are not as directly applicable here.

NMHC emission factors seem to agree well with the observed data (if the large standard error is taken into account at Tuscarora).

CO emission factors are consistently higher in MOBILE6 than MOBILE5. However, there is little difference between the two tunnels for MOBILE6 while MOBILE5 results show a large difference. This is consistent with a large upturn in the MOBILE5 LD CO speed correction curve for 1981-1992 model years which only affects the Tuscarora speed.

Heavy-duty Results

MOBILE6 seems to agree well with NOx observations at Fort McHenry, Tuscarora (1992), and Caldecott. (Recall that Lincoln and Deck Park results are not suitable for inclusion in this discussion due to reasons given above.) The observed NOx at Tuscarora (1999) is considerably lower. Examination of the by-model-year outputs indicates that the assumptions regarding excess NOx were implemented from model year 1988 onward. This is the major driving force behind the 1999 Tuscarora NOx prediction. Note also that MOBILE6 predicts higher NOx at Tuscarora and Fort McHenry but underpredicts at Caldecott. This is because travel at the latter is one-way uphill while the other tunnels have averaged results or no significant grade. As mentioned above, the effects of grades on the HD vehicles are more pronounced, and in this situation, the inability of the model to account for slopes is clearly shown.

With respect to NOx, there are small differences between the two latest versions of the model. MOBILE6 yields slightly lower estimates for the tunnels for which MOBILE5 predictions are available. However, this may simply be due to the different manners in which these values were derived. MOBILE5 HD emission factors used herein were obtained through regression analysis of experimental run-specific fleet average predictions while the MOBILE6 values are weighted averages of run-specific vehicle class-specific values. (Using vehicle class-specific factors gives a more direct assessment of the model's accuracy. A weighted average was used to combine all runs, with the vehicle count in each run as the weights.)

For CO and NMHC, MOBILE6 predicts the highest emission factors, with NMHC still tracking the observed values better than CO. Since there are no speed effect changes in MOBILE6 for HD, these increases are due to basic emission rate changes (including deterioration).

Summary and Conclusion

The use of tunnel data to assess MOBILE6 model performance has some limitations that must be accounted for before drawing conclusions from result comparisons. Notwithstanding these difficulties, tunnel data used as described above produce good insights into the accuracy of model predictions as well as factors that drive these results. In particular, the fleet average comparisons showed that NOx continues to be reasonably well predicted under most circumstances. However, the age distribution assumed for these calendar years play major roles in determining whether the model will overpredict. Light-duty emission rates are also being overpredicted, with speed being a major factor. Heavy-duty NOx is influenced by assumptions on defeat device operation, which is most clearly seen in the 1999 Tuscarora results. The effects of grades are not observed except perhaps in the Caldecott data. Taken together, the CO and NMHC results for all vehicle classes suggest that MOBILE6 tends to

overpredict even more than MOBILE5 for these calendar years and tunnels. For NOx, the predictions for these precise operating conditions have decreased and more closely approximate the observed values.

In addition, the US EPA has released a draft version of MOBILE6.1, which estimates emission factors for on-road PM. This version is currently available for review at the Office of Transportation and Air Quality (OTAQ) web site at http://www.epa.gov/otaq/m6.htm#extens. Since a few of the tunnel studies discussed in this report also examined particulate matter emissions (e.g., Caldecott and Tuscarora 1999), it is possible to use them to validate the MOBILE6.1 emission factors as well.

3. COMPARISON OF HC/NOx AND CO/NOx RATIOS IN MOBILE6-BASED EMISSION INVENTORIES WITH AMBIENT DATA

INTRODUCTION

One way of evaluating MOBILE6 is to compare ratios of species in emission inventories prepared using MOBILE6 with corresponding ratios in ambient monitoring data. While this "ambient-inventory reconciliation" approach does not allow one to evaluate the accuracy of the estimated absolute magnitudes of emissions, it does allow one to evaluate the ability of MOBILE6 to reproduce the observed composition of the mobile source pollutant mixture. Obtaining accurate estimates of the relative composition of species in emissions is critical for effective air quality management since predictions of ozone and secondary PM formation are sensitive to species ratios in the inventory. The primary advantage of this method of validating MOBILE6 is that it provides a direct comparison of the inventory estimates for an area around a given ambient monitoring site with data from that site. Unlike tunnel studies that are limited to a few specific facilities, each with their own unique fleet and operating mode characteristics, comparisons with ambient data can be conducted at a wide variety of locations where suitable ambient data are available and thus provide a broader perspective on the overall accuracy of the inventory estimates.

There are, of course, a number of limitations inherent in ambient-inventory reconciliation analyses in addition to the fact that it is limited to an evaluation of species ratios rather than the absolute magnitude of emissions. In the context of the current study, it must be recognized that species ratios in the monitoring data represent a mixture of source categories and are not limited to just on-road mobile sources. Thus, discrepancies between ambient and inventory ratios cannot be definitively tied to inaccuracies in the MOBILE6 portion of the inventory. This ambiguity can be minimized by selecting monitoring sites in locations that are dominated by on-road mobile sources.

In general, one must recognize that there are several reasons why ratios of, for example, NMHC/NOx in the inventory may differ from ratios in the ambient data:

- NMHC emissions may be over (under) estimated in the inventory.
- NOx emissions may be under (over) estimated in the inventory.
- Emissions of NMHC or NOx may not be properly spatially allocated across different sources or properly temporally allocated to different times of the day or may not represent actual emissions during the ambient monitoring period (day specific effects). This may be particularly important in the case of diesel truck activity which can strongly influence NOx levels and exhibits distinctive diurnal and day-of-week variations. Irregular activity levels at industrial point sources can also have large impacts on NOx.
- The definition of what range of HCs are included in the inventory definition of "NMHC" may not correspond well with the range of HCs captured by the ambient NMHC measurement. Previous studies have used mobile source speciation profiles to break down the VOC calculated by MOBILE into its individual component species and then include in the comparison only the range of species represented in the ambient data (typically the sum of the 56 PAMS target species or the TNMHC reported from the GC/FID or GC/MS

analysis). Since the objective of the proposed study is to validate the MOBILE6 model and not necessarily the speciation profiles used to perform the validation, care must be taken to use the most appropriate speciation profiles and to estimate the potential uncertainties introduced into the analysis as a result of this extra step.

- Ambient NMHC or NOx concentrations may be lower than they would otherwise be due to chemical reactions between time of emissions and when material is observed at the monitoring site. The rate of reaction will in general be different for different chemical species, different pollutant mixtures, and different meteorological conditions.
- Air parcels sampled at the monitoring site may represent a different source mixture than is contained in the area-wide average emission inventory. This is particularly important for NOx emissions from elevated sources such as power plant smoke stacks since the extent to which smoke stack plumes mix to the ground at the monitoring site is highly variable.
- Errors may occur in the ambient measurements due to concentrations below instrumentation detection limits, NMHC species misidentification, calibration errors, etc.

Previous ambient-inventory reconciliation studies have employed a number of techniques to reduce the influence of some of these confounding factors, including the potential confounding influence of nonroad, area, and point sources, reactions of emitted species prior to observation at the monitoring site, diurnal variations in emission patterns and meteorological conditions, etc. Similar techniques were used in the present study. Primarily, comparisons were limited to the weekday morning commute period when atmospheric reactivity is low, mobile source emissions are high and mixing limited. Observations were screened to exclude periods of very low VOC and/or NOx concentrations. These restrictions were intended to insure that observations included in the comparison are strongly if not overwhelmingly influenced by fresh on-road mobile emissions. Wind direction observations were used to allow average ambient ratios to be computed by wind direction sector for comparison with emissions ratios for emission grid cells falling within the sector and comparisons were made using grids covering several different size areas around the monitoring site to determine sensitivity of the comparisons to spatial inhomogeneities in the inventory. Comparisons were also done with and without the inclusion of elevated point sources in the inventory. Absent dispersion modeling or tracer studies, it is impossible to determine if elevated point sources were significantly impacting the ambient data used for comparisons. The potential influence of these sources were further reduced by restricting attention to the early morning period when vertical mixing is limited.

Previous Ambient/Inventory Reconciliation Studies

A capsule summary of previous ambient/inventory reconciliation studies conducted in the United States over approximately the past decade is provided in Table 3-1. With the exception of the LADCO analysis, these studies (which primarily focused on the HC/NOx ratios) concluded that the inventory HC/NOx ratio was lower than the ambient ratio by factors ranging from 1.2 to 6. Studies which also examined CO/NOx ratios concluded that the ambient CO/NOx ratio generally exceeded the inventory CO/NOx ratio although CO/NOx ratios were found to be in better agreement in the latest Houston study (Stoeckenius et al., 2002). Although all of these studies concluded that the discrepancies in HC/NOx ratios are due either to underestimation of HC emissions and/or over estimation of NOx emissions in the inventory, researchers were generally more suspicious of the accuracy of the HC inventory in general and of the on-road mobile source HC inventory in particular.

Table 3-1. Summary of selected inventory reconciliation studies.

Study	Reference	Location	Date	Ambient Data	Inventory Data	Ratio of Ratios: (Ambient HC/NOx) / (Inventory HC/NOx)	Additional Results
TXAQS 2000	Stoeckenius et al. (2002)	Houston, TX	2000	PAMS, TXAQS (locations and wind directions with max on-road mobile source impacts)	Gridded, fully speciated, temporally allocated version of Houston SIP modeling inventory (MOBILE6)	HC/NOx: 2.4 to 3.7 CO/NOx: 0.7 – 1.7	Both HC/NOx and CO/NOx ratio of ratios higher at other (less mobile source dominated) locations and wind directions
MARAMA	Stoeckenius and Jimenez (2000)	Mid-Atlantic	1997	PAMS: McMillan Res., Washington DC; Essex, Baltimore MD	New gridded, speciated, MOBILE 5b based inventory (no excess NOx adjustment)	Washington: 1.2 to 1.6 Baltimore: 1.5 to 3.7	Relative abundance of aromatics slightly higher in inventory relative to ambient
CA-PAMS	Haste-Funk and Chinkin (1999)	Central and Southern California	1996	PAMS: Fresno, Sacramento	ARB county-level inventory for 1996	HC/NOx: 1.5 to 4.0 CO/NOx: 1.5 to 2	Ambient paraffins slightly higher than in inventory; olefins and aromatics lower
NARSTO-NE	Haste et al. (1998)	Northeastern U.S.	1995	PAMS and NARSTO-NE: Bronx, NY (New York City), Lake Clifton, MD (Baltimore, MD), Lynn, MA (near Boston).	OTAG 1990 grown to 1995 using 1995 OTAG modeling inventory (MOBILE 5)	1.5 to 3.5	Ambient NMHC composition similar to composite of mobile and area inventory composition
LADCO	LADCO, 1998	Lake Michigan, New York City, Washington, DC	1995	PAMS (Jardine-Chicago; IITRI-Gary, IN; UWM-North-Milwaukee; Northbrook, IL)	OTAG 1995 modeling inventory with local adjustments (MOBILE 5)	1 ± .25	
COAST	Korc et al. (1995)	Southeast TX	1993	PAMS-Houston (Clinton Dr., Galleria)	COAST inventory from TNRCC (MOBILE4.1)	2 to 6	
SCAQS	Fujita et al. (1992)	South Coast (Los Angeles)	1987	SCAQS	SCAQS (EMFAC7E)	CO/NOx: 1.1 to 2.7 NMOG/NOx: 1.8 to 3.2	

Selection of the Study Sites

Hourly ambient hydrocarbon, NOx, CO and wind direction data are needed for the reconciliation analysis. Hourly speciated hydrocarbon and NOx data are available at selected PAMS monitoring sites; hourly CO data are available at some PAMS sites or from nearby sites. For this study, we are interested in data from sites where on-road mobile sources dominate the anthropogenic emissions budget. Since we are using emissions estimates for 1999, it is desirable to use ambient data from the same time period to avoid introducing biases into the comparison resulting from year-to-year changes in the inventory. Using just data from 1999 would result in fairly small sample sizes when looking at data for individual times of the day under specific wind direction sectors, but extending the data window to before 1998 or after 2000 would introduce potential uncertainties arising from differences in emission ratios for years other than 1999. We, therefore, focused on data from the period 1998 – 2000. Based on the availability of hourly PAMS data, information from previous ambient/inventory reconciliation analyses, and input from the project sponsors, we selected the following PAMS monitoring sites for use in this analysis:

Jardine Water Filtration Plant: located on Navy Pier just east of the Loop in downtown Chicago, this Type II PAMS site is representative of core urban area emissions impacts and was included in a previous ambient/inventory reconciliation analysis (LADCO, 1998).

Northbrook Water Plant: located in a northern suburb of Chicago, this Type III PAMS site is representative of city suburb impacts with less elevated point and industrial impacts than at Jardine and was included in the LADCO (1998) study.

Detroit, MI (East Seven Mile Road): this site was established as an air toxics monitoring site representative of urban core emissions; its use in this study was recommended by LADCO.

McMillan Reservoir in Washington DC: this site was included in two previous ambient/inventory reconciliation analyses (Stoeckenius and Jimenez, 2000; Haste et al., 1998) and is known to be strongly influenced by mobile sources with relatively little industrial source impact.

Lynn: is a PAMS Type II site located in a suburb north of Boston, MA and represents a mix of suburban and industrial sources. This site was included in a previous ambient/inventory reconciliation analysis (Haste et al., 1998).

DEVELOPMENT AND PREPARATION OF EMISSION INVENTORY AND AMBIENT DATA

Extensive emission inventory development efforts were undertaken for each of the five urban regions included in this study to obtain an inventory suitable for comparison with ambient data. These activities, along with steps involved in gathering and preparation of the ambient data are described in this section.

Emission Inventory Development

County-level point, area and off-road emissions were obtained from the 1999 National Emissions Inventory (NEI), Version 2).[1] County-level on-road mobile source emissions were developed by applying the MOBILE6 emission factor model to the NEI99 VMT data as described under "On-Road Mobile Sources" below. Spatial and temporal allocation factors for other source categories were applied to the county level inventory as described in the following subsections. All emissions were processed using the EPS2 system (EPA, 1992). The resulting spatially and temporally disaggregated inventory was developed for 80 x 80 km regions centered on the five ambient monitoring sites selected for this study. These five sites are located in Chicago (Jardine and Northbrook) IL, Detroit MI, Washington DC, and Lynn MA. For consistency, gridding was performed on the National Unified LCP 4 x 4 km grid. Figure 3-1 depicts the locations of the five emission grids used in this study. Note that a portion of the Detroit grid lies over Canada. Since the NEI does not include Canadian emissions data, ambient/inventory reconciliation for the Detroit monitor was limited to time periods during which monitored winds indicate a low likelihood of significant impacts from Canada.

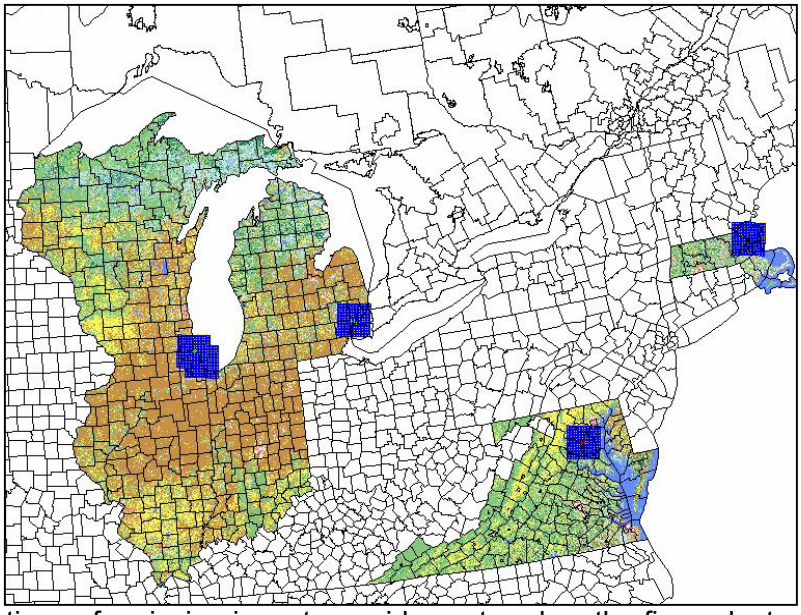


Figure 3-1. Locations of emission inventory grids centered on the five selected ambient monitoring sites included in the study. Each grid consists of 20 cells in the N-S and E-W directions; each cell is 4 km on a side. Colors indicate different land use/land cover types used for inventory development.

Emission Gridding Surrogate Development

Spatial allocation of regional or county-level emission estimates was accomplished through the use of gridding surrogates or spatial allocation factors (SAFs) for each emission source category or group of source categories. Spatial surrogates are typically based on the proportion of a known region-wide characteristic variable that exists within the modeling domain grid cells. Traditionally the development of spatial gridding surrogates has been

[1] http://www.epa.gov/ttn/chief/net/index.html#1999.

performed by a variety of methods depending on the emission source category being considered, the required spatial resolution, the geographic extent of the domain, and the particular characteristics of the geospatial data available. Spatial surrogates must define the percentage of regional or county level emissions from a particular source category that is to be allocated to some spatial region, typically a modeling grid cell. For most area and off-road sources, these percentages are based on areas of a particular land use/land cover type while for on-road mobile source categories, the percentages are usually based on total length of a certain road type or a transportation network. Often human population is also used as a spatial surrogate for certain emission source categories.

Gridding surrogates were developed from several sources of spatial data describing the Land Use/Land Cover (LULC), transportation networks and population characteristics. Land use data were obtained from the United States Geological Survey (USGS) Earth Resources Observation Systems (EROS) Data Center web site and are a subset of the National Land Cover Dataset (NLCD).[2] This dataset provides dominant land use data for each state at a spatial resolution of 30 meters. The 21 LULC categories and codes utilized in the NLCD are presented in Table 3-2. More detailed descriptions of the NLCD land use types are available from the USGS web site.[3]

Transportation networks, including inland waterways, were obtained from the US Census Bureau TIGER/Line data files.[4] Population data were obtained in the form of a global 1-km gridded GIS dataset.[5] Additional spatial surrogate information, specifically information on airport and shipping port locations were obtained from spatial surrogate data developed by the EPA.[6] Processing and development of gridding surrogates were performed using the Arc/INFO Geographic Information System.

Spatial Surrogate Assignments

To apply the EPS2 emissions processing system using the spatial gridding surrogates developed as described above, the LULC codes listed in Table 3-2 needed to be aggregated and re-mapped to the surrogate codes recognized by EPS2. Table 3-3 displays the mapping of NLCD codes to EPS gridding surrogate codes.

The US EPA's source classification code (SCC)-spatial surrogate cross-reference files were evaluated for use in our analysis. In most cases, the EPA's surrogate assignments are based on fairly broad surrogate categories (i.e., population, rural land, agricultural land, etc.). As EPS2 allows surrogates to be user-defined using more detailed categorization of LULC classifications for specific application, the EPA-defined surrogate assignments were compared with those typically used by ENVIRON when developing modeling inventories using EPS2. It was determined that the EPA's surrogate assignments were considerably less detailed than the

[2] http://edcftp.cr.usgs.gov/pub/data/landcover/states
[3] landcover.usgs.gov/nationallandcover.html
[4] www.census.gov/geo/www/tiger/tigerua/
[5] Center for International Earth Science Information Network (CIESIN), Columbia University; International Food Policy Research Institute (IFPRI); and World Resources Institute (WRI). 2000. Gridded Population of the World (GPW), Version 2. Palisades, NY: CIESIN, Columbia University (http://sedac.ciesin.columbi.edu/plue/gpw).
[6] ftp://ftp.epa.gov/EmisInventory/emiss_shp/

most recent allocation assignments typically used by ENVIRON. Therefore, the more refined SCC-surrogate assignments developed by ENVIRON were used. The use of these assignments result in improved spatial allocation of various emission source, particularly off-road sources, which EPA's assignment allocates mostly to population, rather than specific land use types for which the activity data associated with these sources are more appropriate.

Table 3-2. Land use categories and codes utilized in the National Land Cover Dataset (NLCD).

NLCD Category Code	NLCD Category Description
11	Open Water
12	Perennial Ice/Snow
21	Low Intensity Residential
22	High Intensity Residential
23	Commercial/Industrial/Transportation
31	Bare Rock/Sand/Clay
32	Quarries/Strip Mines/Gravel Pits
33	Transitional
41	Deciduous Forest
42	Evergreen Forest
43	Mixed Forest
51	Shrubland
61	Orchards/Vineyards/Other
71	Grasslands/Herbaceous
81	Pasture/Hay
82	Row Crops
83	Small Grains
84	Fallow
85	Urban/Recreational Grasses
91	Woody Wetlands
92	Emergent Herbaceous Wetlands

Table 3-3. Mapping of EPS2 surrogate codes to NLCD LULC codes.

Surrogate Name	EPS2 Surrogate Code	NLCD LULC Codes
County area	1	Sum all NLCD codes
Population	2	N/A
Households	3	Sum NLCD codes 21 and 22
Urban	4	Sum NLCD codes 21-23 and 85
Agriculture	5	Sum NLCD codes 61 and 81-84
Range	6	Sum NLCD codes 51 and 71
Railways	7	Sum TIGER road types B11-B52
Waterways	8	Sum TIGER road types H11-H22
Forest	9	Sum NLCD code 41-43
Bodies of Water	10	Sum NLCD codes 11 and 12
Barren	11	Sum NLCD codes 31-33
Ports	12	Ports from EPA's surrogate database
Commercial/Industrial	13	NLCD code 23
Ports	14	Ports from EPA's surrogate database
Rural	15	Sum NLCD codes 31-33, 41-43, 51, 61, 71, 81-84 and 91-92

Temporal Allocation

In addition to spatial allocation of county-level emission estimates, the inventory was temporally allocated by hour of day, day of week and month of year. Temporal allocation of point and area sources is discussed here; temporal allocation of mobile source emissions is discussed in the next sub-section.

EPS2 makes use of temporal profiles for area sources identified by unique codes cross-referenced to emission source categories by SCC. The EPS2 modeling systems allows for the use of user-defined temporal profiles and cross-references. For the present project, the EPS2 default temporal profiles, a set of custom profiles recently used for work for the State of Texas, and EPA's temporal profiles[7] and SCC cross-reference assignments were all reviewed. Our review focused on the specific emission source categories present in the NEI database and the geography and types of activity within the emission inventory domains. Based on this review, it was determined that the EPA's temporal profiles and SCC cross-reference assignments were most appropriate for the current application.

Temporal allocation of point sources was based on information contained in the NEI database. These data specify the number of hours of operation per day, a start hour for operation, and the number of days per week and weeks per year of operation of each individual point source. Based on this information, specific temporal profiles were developed for each source.

[7] http://www.epa.gov/ttn/chief/emch/temporal/index.html

On-Road Mobile Sources

Spatially and temporally disaggregated on-road mobile source emissions were estimated by combining county level VMT data and MOBILE6 input files (registration distributions, speed distributions, I/M program details and other inputs needed to generate county-level emissions) from the NEI99 together with county-specific daily emission factors computed using MOBILE6.2, land use data, and temporal activity profiles from various sources as described below.

Mobile Source Spatial Allocation

Spatial allocation of mobile source emissions was based on geographic distribution of road length by roadway (facility) type within each county as determined from the U.S. Census Bureau TIGER files. Activity on each facility type was assumed to be uniform over the specific type within the county. This procedure provided the best possible spatial allocation of on-road emissions short of using link level output from a travel demand model.

Facility type classifications were based on general groupings of facility classes found in the emission inventory database and the available roadway types contained in the TIGER Line data files. Table 3-4 provides the definition of the surrogate codes used for spatial allocation of on-road mobile sources. Table 3-5 shows the correspondence between the EPS surrogate codes and the associated MOBILE model facility types. TIGER codes shown in these tables are defined in the TIGER/Line technical documentation (Census Bureau, 2002). Note that the surrogate codes were used just for spatial allocation; MOBILE6 emission factors were calculated for each of the facility classes listed in Table 3-5 and multiplied by the county-wide VMT for the corresponding facility class. In particular, the TIGER coding system did not allow us to distinguish between the Local and Collector facility types. Although emissions were computed separately for Locals and Collectors, the spatial allocation of the Collectors emissions was based on the spatial distribution of "Locals/Collectors" (EPS2 Surrogate Code 3) because this code represents a better spatial distribution of activity on Collectors than, say, Code 2 (Arterials).

Table 3-4. On-road mobile source spatial surrogate definitions.

Surrogate Name	EPS2 Surrogate Code	TIGER Line Codes
Interstate/Freeway	1	Sum TIGER codes A11-18 and A63
Arterials	2	Sum TIGER codes A21-A38 and A64
Locals	3	Sum TIGER codes A41-A48 and A60-A62

Table 3-5. On-road mobile source facility classes and surrogate assignments.

Facility Class	EPS2 Surrogate Code
Collector Urban	3
Interstate Urban	1
Interstate Rural	1
Local Urban	3
Local Rural	3
Major Collector Rural	3
Minor Collector Rural	3

Facility Class	EPS2 Surrogate Code
Minor Arterial Urban	2
Minor Arterial Rural	2
Other Freeways and Expressways	1
Other Principal Arterial Rural	2
Principal Arterial Urban	2

Mobile Source Temporal Allocations

Hourly VMT distributions, developed separately for each region and for light- and HDVs, were used in the MOBILE6.2 emission factor modeling. Weekday and weekend hourly activity (VMT) distributions for LD- and HDVs, respectively, were developed and assigned to each of five geographic areas. These were used to apportion the total daily running emissions to individual hours. Three unique sets of distributions were created corresponding to data from the South Coast (Los Angeles) area, the Southeast Michigan (Detroit) area, and the Washington D.C. area. Profiles from Los Angeles were used to represent the diurnal pattern of emissions in the two Chicago emissions grids since data from Chicago were not readily available and the Los Angeles data were the best representations of congested urban area activity profiles available to this study. Profiles for the Detroit area were also assumed to be representative of the Lynn MA emissions grid as they are the best available representation of diurnal patterns in somewhat smaller and less congested urban areas. Data sources and analysis methods for each geographic region are briefly described in the following paragraphs.

Southeast Michigan (SEMCOG) - Profiles were developed using hourly VMT mix data provided by SEMCOG in combination with hourly total (across all vehicle classes) traffic counts from stations which represent urban interstates, rural interstates, urban arterials, and rural arterials. Profiles are weekday only and were used for both the Detroit and Lynn emissions grids as noted above.

Metropolitan Washington COG (MWCOG) - Hourly activity distributions for freeways, arterials, and collectors were obtained from MWCOG. These were combined with the hourly VMT mixes from SEMCOG (above) to arrive at separate activity profiles for LD and HD vehicles. The arterial and collector profiles are very similar and were, therefore, combined using VMT as weights. Again, these represent weekday only and were used for the Washington DC (McMillan) emissions grid.

Los Angeles - Activity profiles were obtained from data assembled by the California Air Resources Board in support of their weekday/weekend air quality research program. The LD profiles are for arterials and collectors as well as freeways, the latter representing four different areas in the basin. The arterial and collector profiles are nearly identical so just one profile can be used to represent both facility types. The HD profiles are for "surface" streets and freeways, the latter from the same four areas of the basin as for the LD. The "Interior Basin" freeway profiles were used for both LD and HD because they are most likely to be generally representative of profiles in congested urban areas. The other sites (Long Beach, Inyo, and Castaic) are likely to be more specific to these localized areas and were, therefore, not used. As noted

above, profiles from Los Angeles were used to represent the diurnal pattern of emissions in the two Chicago emissions grids.

The only reliable weekend profiles that contrast LD activity with HD activity that were available for use in this study are from the Los Angeles data described above. Separate profiles are available for each day of the week. However, to keep the size of the ambient samples used for comparison with the inventory reasonably large, a combined profile for both weekend days was used rather than keeping Saturday and Sunday profiles separate. Factors derived from the Los Angeles data that were used to adjust the ozone season day average emissions to weekday and weekend emissions are listed in the Table 3-6. The weekday and weekend diurnal profiles described above are shown in Figures 3-2a - 3-2c.

Table 3-6. Factors used to adjust the ozone season day average emissions to weekday and weekend emissions.

Vehicle Class	Emissions Process	Weekday	Friday	Saturday	Sunday	Avg (Sat, Sun)
LD	Exhaust, hot soak, running loss	1.023	1.13	0.95	0.83	0.890
	Diurnal and resting loss	0.899	0.87	1.11	1.43	1.267
HD	Exhaust, hot soak, running loss	1.213	1.21	0.52	0.41	0.467
	Diurnal and resting loss	0.622	0.62	1.47	2.42	1.944

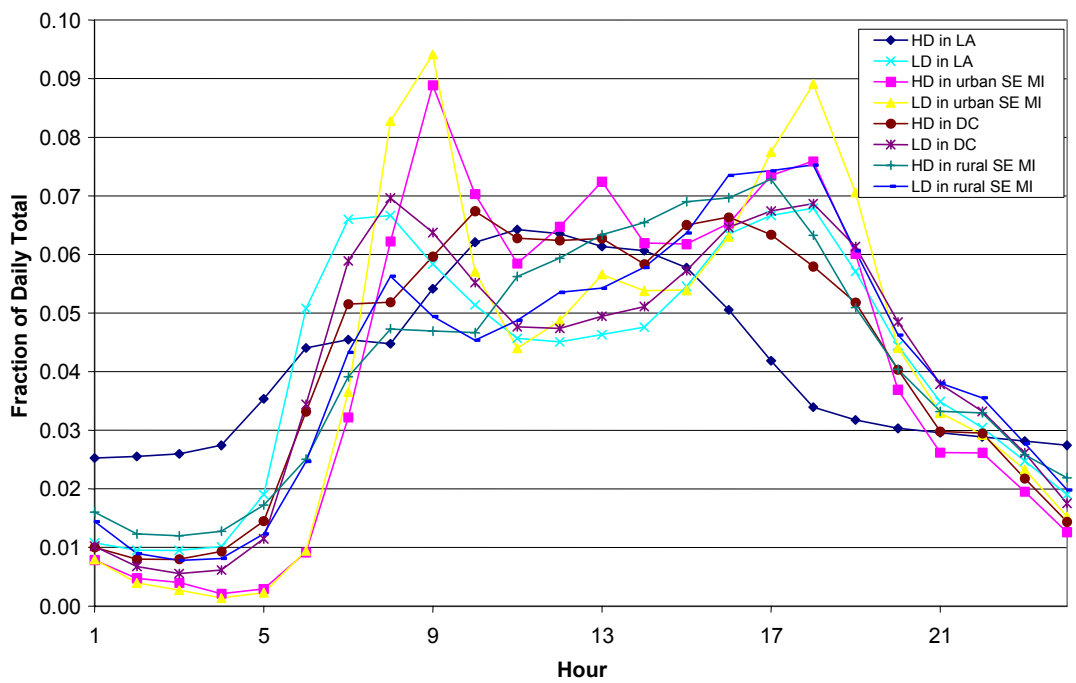


Figure 3-2a. Weekday hourly VMT distributions for freeways and interstates.

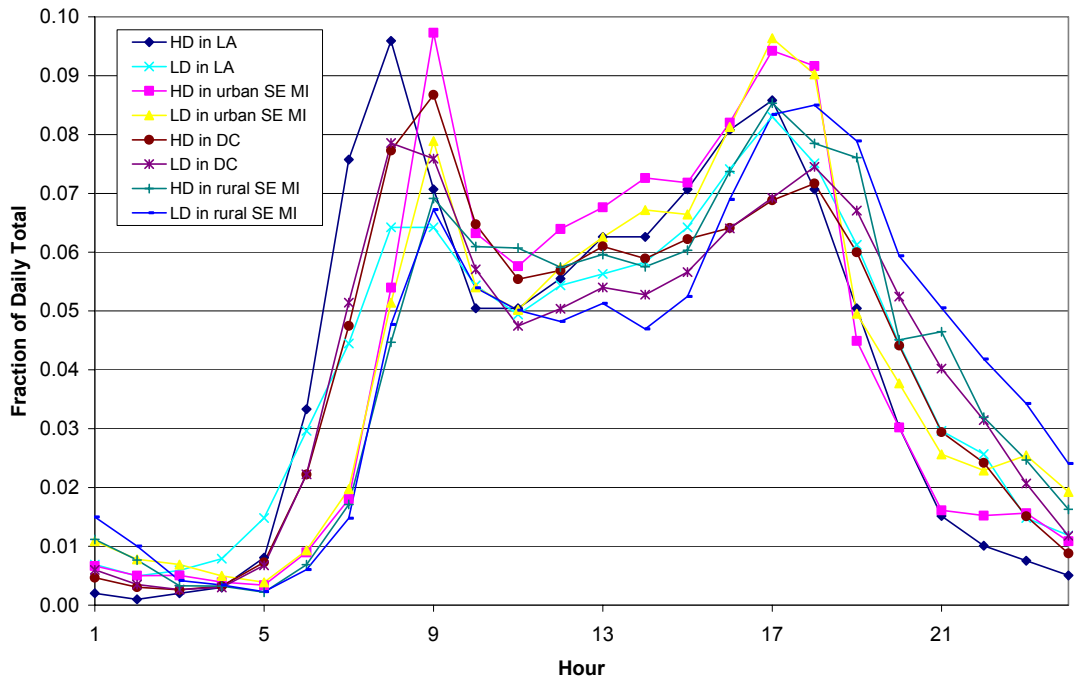


Figure 3-2b. Weekday hourly VMT distributions for arterials and collectors.

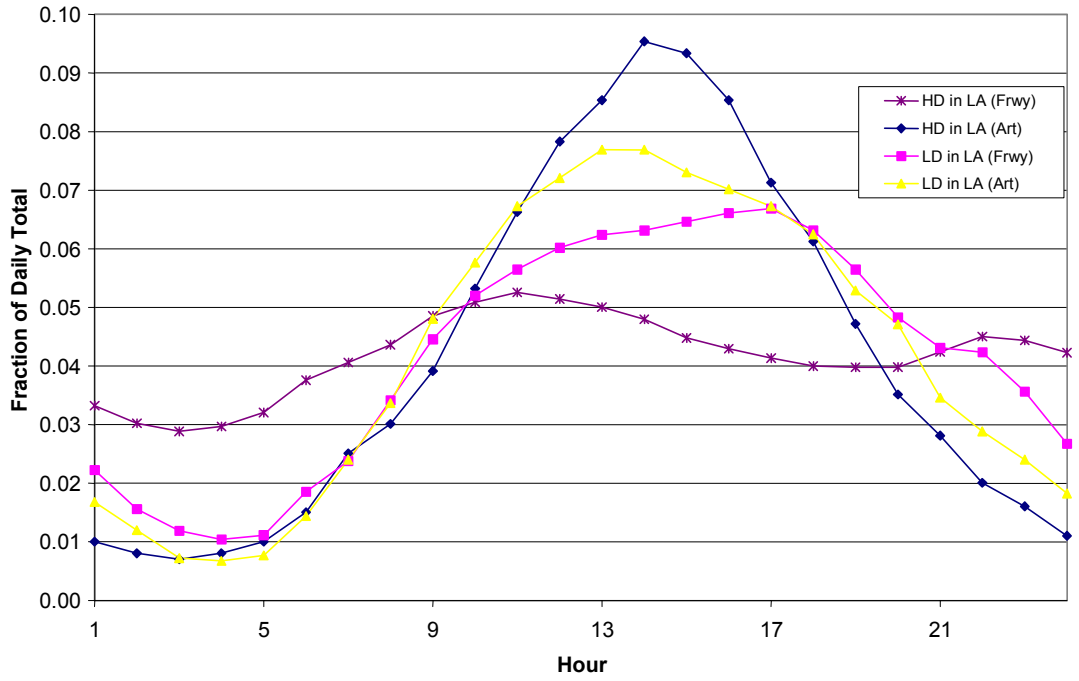


Figure 3-2c. Weekend hourly VMT distributions.

Morning cold starts are likely to have an influence in differentiating morning HC/NOx and CO/NOx emission ratios from daily average ratios. Unfortunately, direct hourly estimates of hot and cold start emissions (in g/hour) are not readily available from MOBILE6. Therefore, a suitable diurnal profile of the daily start emissions that accounts for the pattern of all starts and the pattern of cold vs. hot starts was applied. Consistent with the methodology previously outlined, this profile was constructed by combining the diurnal profile of hourly start emission factors (which takes into account the pattern in the cold start fraction) with the diurnal profile of the number of starts per hour. The combined profile was then applied to temporally allocate the daily start emissions estimated by MOBILE6.

Emissions Processing

Temporally and spatially allocated emissions estimates developed via the methods described in the previous section were aggregated into four major source categories:

- Elevated point sources
- Low-level point sources
- Area sources (including off-road mobile sources)
- On-road mobile sources

Elevated point sources were defined as sources with estimated plume heights greater than 25 m. The distinction between low-level and elevated point sources is useful because the degree to which elevated point source emission plumes impact a nearby surface monitoring site can be quite variable, and these plumes typically have very low VOC/NOx ratios compared to other source categories.

Since the auto-GC instruments used at the ambient monitoring sites are not designed to detect all of the organic compounds included in the emission inventories, emissions of species not included in the PAMS target list described above were excluded from the comparison. Resource constraints for this project precluded us from performing a full speciation of the VOC emissions. However, good estimates of the fraction of VOC emissions within a typical urban area that are accounted for by the PAMS species are available from a previous study (Stoeckenius et al., 2002). For the inventory as a whole, the sum-of-PAMS species is estimated to account for 67 percent of the total VOC emissions (on a mass basis). This result is consistent with similar analyses conducted for other cities (LADCO, 1998; Haste et al., 1998; Korc et al., 1995). The potential impact on ambient/inventory comparisons of uncertainties in the fraction of VOC emissions accounted for by PAMS species is discussed below under Summary and Conclusions.

All emissions were converted from g/hour to moles/hour for comparison with ambient measurements expressed as mixing ratios. NOx emissions were converted to moles NO_2. VOC g/hour emissions adjusted to account for the PAMS species fraction as discussed above were assumed to have a composite molecular weight of 13.9 g/moleC. This value is based on analysis of a fully speciated summer weekday 2000 inventory developed for Houston (Stoeckenius et al., 2002). For the Houston inventory, the composite molecular weight was

found to have a standard error associated with spatial and temporal variations in the source mix of just ±1%.

For any given time period, meteorological conditions will determine the size and shape of the emissions source region impacting a monitoring site. In addition, chemical transformation and deposition of pollutants between the time they are emitted and subsequently detected at the monitoring site depends on meteorology as well. Application of a photochemical air quality model to estimate the impact of a group of sources on a monitoring site over a specific time interval (accounting for chemical transformations along the way) was beyond the scope of this study. Instead, we simply defined a square region approximately centered on each monitoring site consisting of twenty of the 4 km emission grid cells on a side and divided this region into four wind direction quadrants centered on NE, SE, SW, and NW compass points. Three different size squares ("subgrids"), with side lengths of 8, 16, and 20 km measured from the center were used as shown in Figure 3-3. Emissions were subtotaled for each quadrant for each of the different subgrids and then integrated over all quadrants for each subgrid. The subgrid lengths (8, 16, 20 km) correspond roughly to one-hour transport times to the monitoring site under straight-line winds at speeds of 2, 4, and 6 m/s, respectively. Such light winds are typical of early summer mornings with high ozone precursor concentrations.

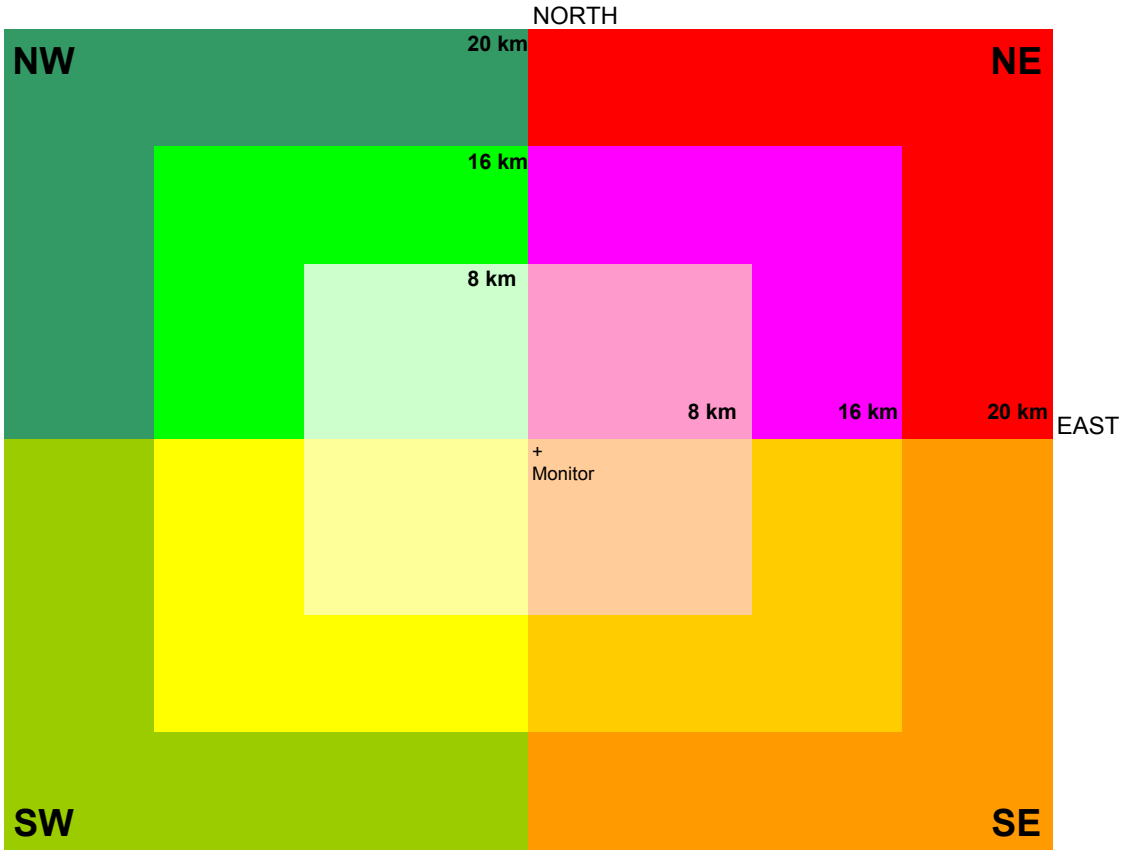


Figure 3-3. Orientation of emission quadrants with respect to monitoring site.

Table 3-7 lists total mobile, area, and point source emissions for hours 6 – 8 within the 20 km box centered on each monitoring site. Note that point source emissions are not a large fraction of the inventory total for any of the three species around the McMillan or Northbrook sites, but significant point source NOx exists around Jardine and Lynn and significant point source emissions of all three species exist around the Detroit monitoring site. Point sources typically have emission characteristics significantly different from area and mobile sources. Furthermore, the impacts of emissions from elevated point sources on a ground-level monitoring site are highly variable and difficult to quantify. These factors make it difficult to use ambient/inventory comparisons to evaluate mobile and area source inventories at locations with significant point source activity, i.e.; at the Lynn and Detroit monitoring sites. Point source NOx is also significant around the Jardine site, and a large percentage of this NOx is from elevated point sources.

Table 3-7. Sum of pollutant mass emissions (kg/hr) for hours 6, 7, and 8 am at distance of 20 km in all quadrants.

Monitor Site	VOC			NOx			CO		
	Mobile	Area	Point	Mobile	Area	Point	Mobile	Area	Point
Detroit	18,770	17,999	10,937	26,157	1,810	31,192	194,988	1,250	28,673
Jardine	14,989	27,697	6,945	21,073	1,014	19,066	141,530	901	6,383
Lynn	9,951	11,354	2,863	16,582	1,781	15,936	104,841	3,289	2,467
McMillan	19,195	19,589	744	29,934	3,120	6,739	200,671	5,253	771
Northbrook	16,520	20,483	3,466	23,358	822	1,439	156,969	1,217	582

Ambient Air Quality Data

Hourly ambient hydrocarbon, NOx, CO and wind direction data were obtained for the five monitoring sites selected for inclusion in this analysis. Since we are using emissions estimates for 1999, it was desirable to use ambient data from the same time period to avoid introducing biases into the comparison resulting from year-to-year changes in the inventory. Using just data from 1999 would result in fairly small sample sizes when looking at data for individual times of the day under specific wind direction sectors but extending the data window to before 1998 or after 2000 would introduce potential uncertainties arising from differences in emission ratios for years other than 1999. We, therefore, focused on data from the period 1998 – 2000. Validated data from the Detroit monitor were only available for 2001 in time for use in our study. All data were downloaded from EPA's AQS database; validated hydrocarbon data for Detroit were provided by the Michigan Department of Environmental Quality.

Detroit NOx Data

In Detroit, only NO_2 data were available for 2001. However, both NO_2 and NOx data were available for 2002. A linear regression model was, therefore, developed using the 2002 data to allow us to roughly estimate NOx from NO_2 in 2001; the model fit is shown in Figure 3-4. This fit is based on all observations between 6 and 8 am with NOx $>$ 10 ppb so as to match the conditions under which the ambient/inventory comparisons described in the next section were performed. The model fit is reasonable overall although there is a notable tendency towards underprediction above 40 ppb NOx, as one would expect. The possibility of using

more complex regression models to explain this behavior was discarded since there is no clear indication that the overall uncertainty would be reduced by a significant amount: there is simply no good way to predict occurrences of high NO fractions given just NO_2 measurements.

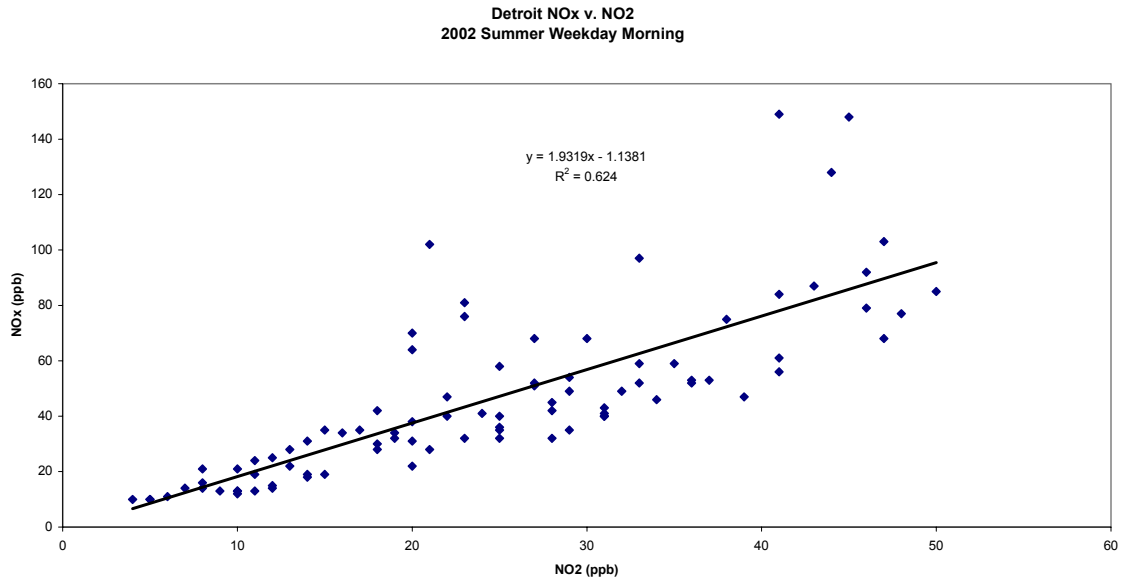


Figure 3-4. Regression of ambient hourly summer (June – August), weekday morning (6 – 8 am LST) NOx to NO_2 for Detroit, 2002.

CO Data

CO data were not available at any of the PAMS monitoring locations included in our analysis. A search of AQS was undertaken to identify the CO monitoring sites closest to each PAMS site. Monitors identified by this search are shown in Table 3-8.

Table 3-8. Nearest CO monitoring site associated with each PAMS site.

	PAMS Site	CO Site	
City	Location	AIRS ID	Distance from PAMS Site (km)
Detroit, MI	E. 7-Mile	26-099-1003	9.1
Chicago, IL	Jardine	17-031-0063	3.0
Lynn, MA	Lynn	25-025-0021	11.6
Washington, DC	McMillan Res	11-007-0023	3.4
Chicago, IL	Northbrook	17-031-3103	20.3

Figure 3-5 shows the locations of the CO monitors relative to the PAMS sites. CO monitors were located fairly close to the Jardine and McMillan PAMS monitors but the closest CO monitors to the other PAMS sites are further away and, therefore, may not be representative of conditions at the PAMS site. This may be especially true for Northbrook where the nearest CO monitor is over 20 km from the PAMS monitor.

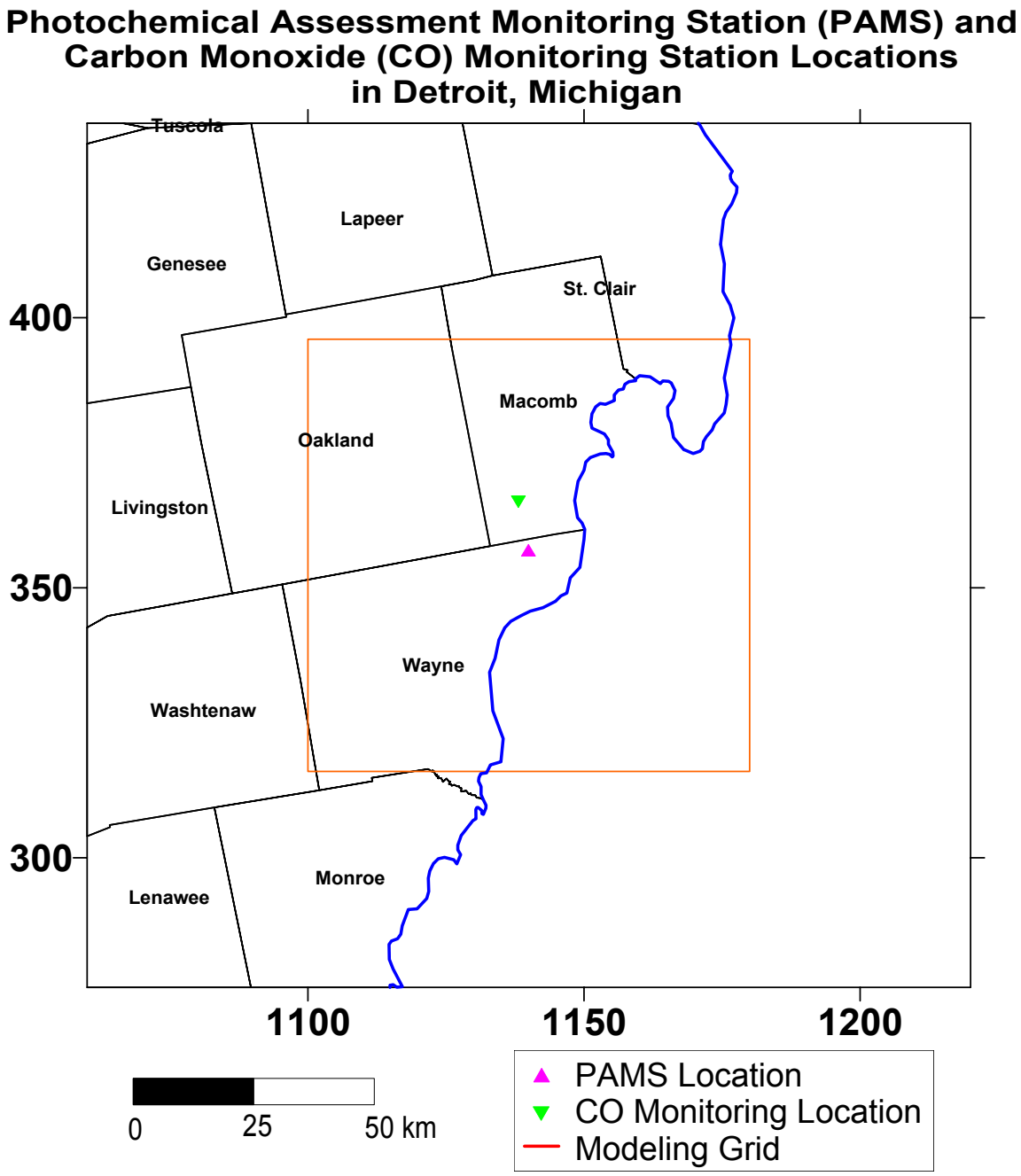


Figure 3-5a. PAMS and CO monitoring station locations (yellow square indicates 40 x 40 km emissions grid area): Detroit.

Photochemical Assessment Monitoring Station Carbon Monoxide (CO) Monitoring Station Locations in Jardine, Illinois

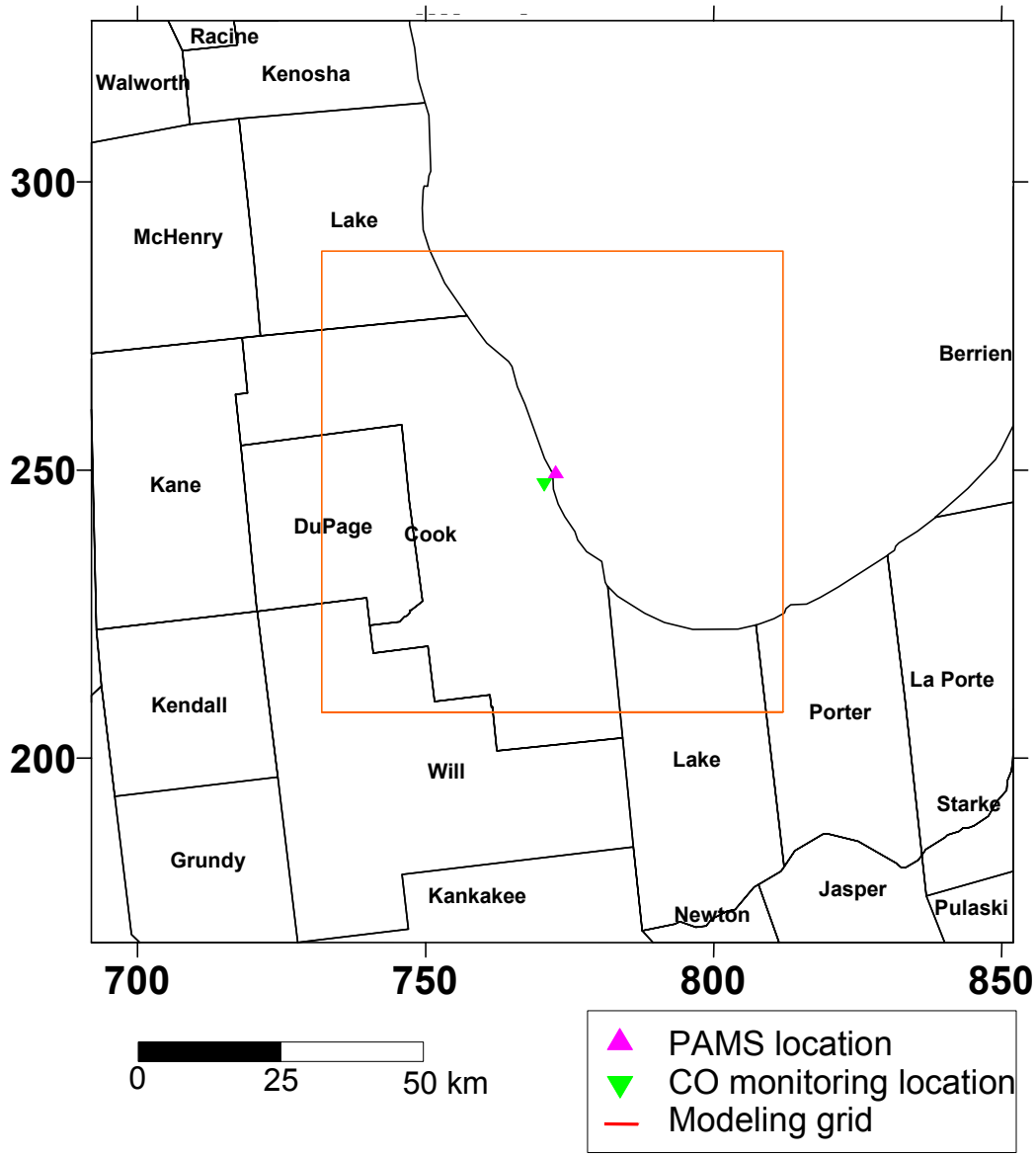


Figure 3-5b. PAMS and CO monitoring station locations (yellow square indicates 40 x 40 km emissions grid area): Jardine.

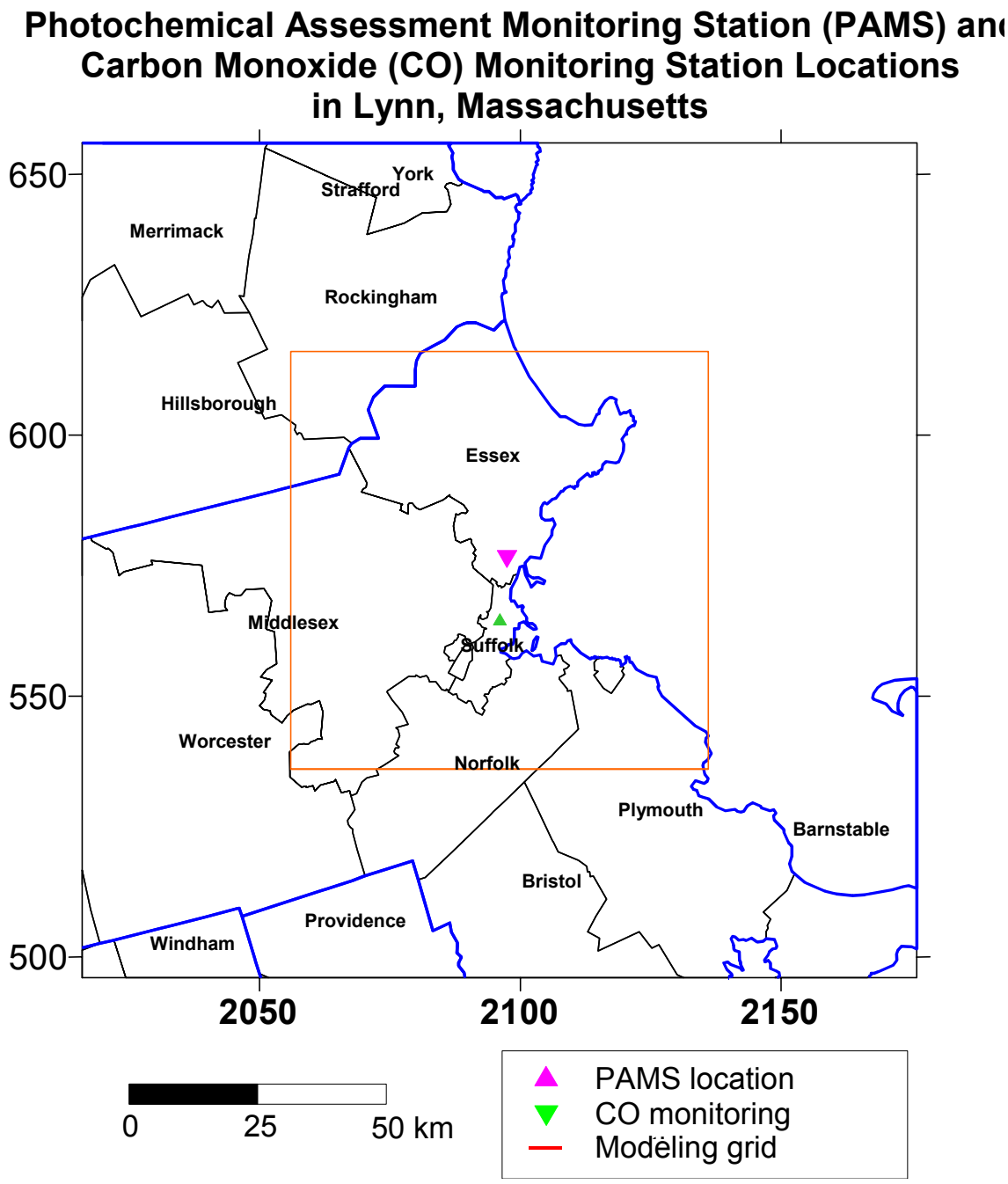


Figure 3-5c. PAMS and CO monitoring station locations (yellow square indicates 40 x 40 km emissions grid area): Lynn.

Photochemical Assessment Monitoring Station (PAMS) and Carbon Monoxide (CO) Monitoring Station Locations in McMillan, D.C.

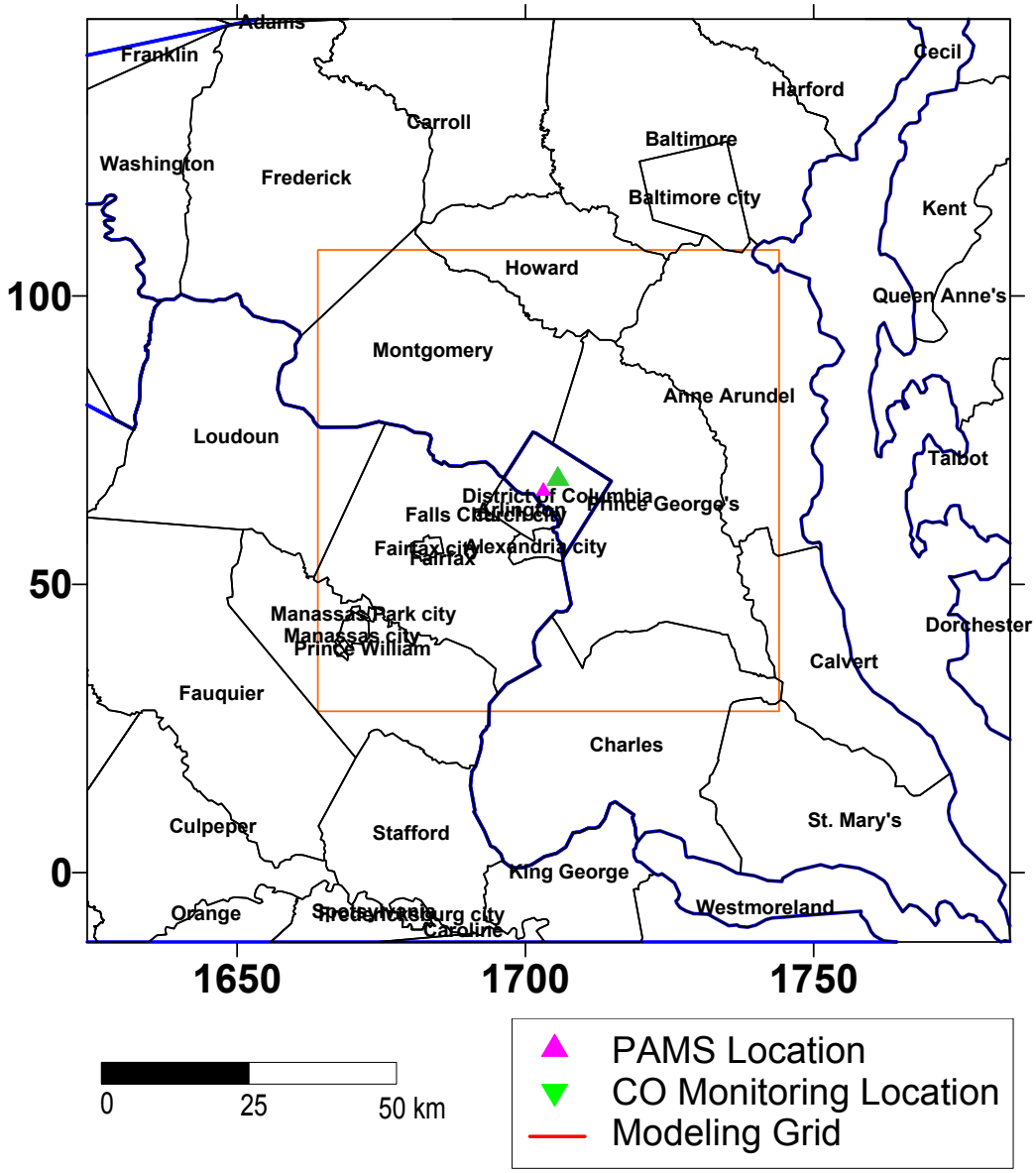


Figure 3-5d. PAMS and CO monitoring station locations (yellow square indicates 40 x 40 km emissions grid area): McMillan.

Photochemical Assessment Monitoring Station (PAMS) and Carbon Monoxide (CO) Monitoring Station Locations in Northbrook, Illinois

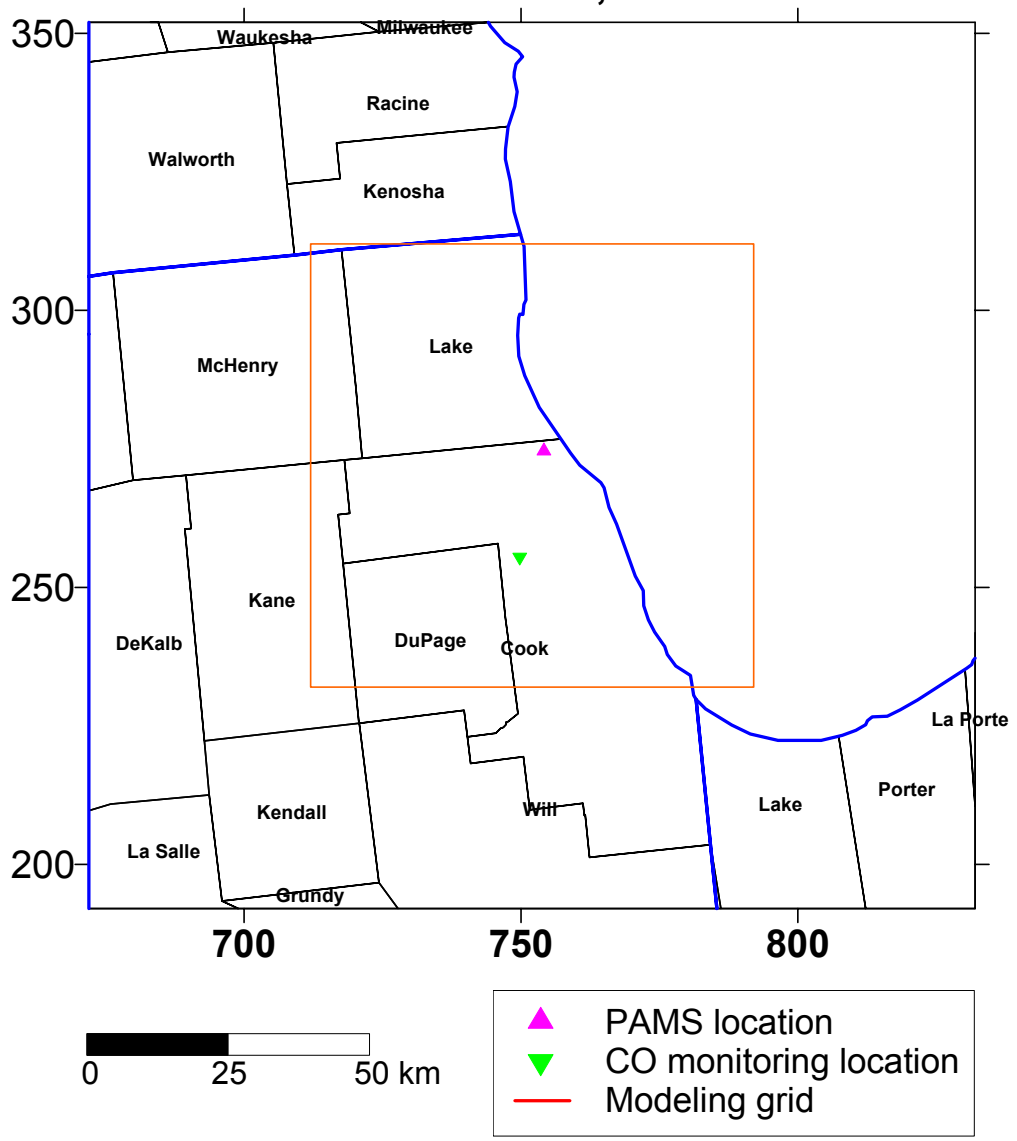


Figure 3-5e. PAMS and CO monitoring station locations (yellow square indicates 40 x 40 km emissions grid area): Northbrook.

Ambient Data Processing

Hourly PAMS hydrocarbon data are collected by auto-GC equipment that is capable of identifying a range of individual C2 – C10 hydrocarbons (alkanes, alkenes, alkynes, aromatics) and undecane, including 56 species identified as PAMS target species list (see Table 3-9). Most PAMS sites report results for between 70 – 100 individual species but the list of quantified species varies from site to site. For the sake of consistency, we therefore used the sum of the PAMS target species reported by each PAMS monitoring site for making comparisons with the inventory estimates.

Table 3-9. PAMS target species.

1,2,3-Trimethylbenzene	Isopentane
1,2,4-Trimethylbenzene	Isoprene
1,3,5-Trimethylbenzene	Methylcyclopentane
1-Butene	Methylcyclohexane
1-Pentene	m-Diethylbenzene
2,2,4-Trimethylpentane	m-Ethyltoluene
2,2-Dimethylbutane	m,p-Xylene
2,3,4-Trimethylpentane	n-Butane
2,3-Dimethylbutane	n-Decane
2,3-Dimethylpentane	n-Heptane
2,4-Dimethylpentane	n-Hexane
2-Methylheptane	n-Nonane
2-Methylhexane	n-Octane
2-Methylpentane	n-Propylbenzene
3-Methylheptane	n-Pentane
3-Methylhexane	n-Undecane
3-Methylpentane	o-Ethyltoluene
Acetylene	o-Xylene
Benzene	p-Diethylbenzene
c-2-Butene	p-Ethyltoluene
c-2-Pentene	Propane
Cyclohexane	Propylene
Cyclopentane	Styrene
Ethylbenzene	t-2-Butene
Ethane	t-2-Pentene
Ethylene	Toluene
Isobutane	1-Hexene
Isopropylbenzene	N-Dodecane

Samples with low sum-of-PAMS (less than 50 ppbC) or low NOx (less than 10 ppb) concentrations were eliminated from the analysis since such samples are not likely to have PAMS/NOx ratios representative of significant fresh mobile source emissions. Excluding the low NOx samples also avoided issues related to measurement uncertainties at low NOx

concentrations. Table 3-10 lists the fraction of morning hours with valid PAMS and NOx (or valid CO and NOx) measurements that were retained after applying the PAMS and NOx cutoffs. Retention of data was somewhat more limited at Northbrook (for PAMS/NOx) and at Lynn (for PAMS/NOx and CO/NOx) than at the other sites due to low PAMS concentrations and missing data.

Table 3-10. Fraction of morning hours (6, 7, 8 am) with valid PAMS and NOx (or CO and NOx) measurements retained after applying minimum concentration cutoffs (50 ppbc for PAMS, 10 ppb for NOx).

	% Samples Retained	
Site	**PAMS/NOx Pairs**	**CO/NOx Pairs**
Detroit	82	80
Jardine	80	71
Lynn	55	35
McMillan	71	56
Northbrook	62	100

AMBIENT/INVENTORY COMPARISONS

Ratios of PAMS/NOx and CO/NOx computed from the hourly ambient monitoring data collected at each location as described in the previous section were compared with corresponding ratios in the temporally and spatially allocated emissions inventories. Diurnal patterns in the ambient and emissions data were examined to ensure that the ambient observations and inventory data were temporally aligned and consideration was given to the impact of biogenic emissions on the comparisons as discussed in the following two subsections. This in turn is followed by subsections describing results of the ratio comparisons for weekdays and weekends.

Diurnal Patterns

Average weekday ambient total non-methane organic compound (TNMOC) and NOx concentrations by hour of day are shown for each monitoring site in Figure 3-6. A distinct peak in NOx and TNMOC concentrations between the hours of 6 – 8 am is evident at each site, indicating peak commute activity emissions coinciding with limited dispersion. The TNMOC peak is slightly earlier at Northbrook (5 am) but the NOx peak is at 6 am. Examination of weekday diurnal emission patterns for mobile sources and for all sources combined (Figures 3-7 – 3-11) indicated a peak in mobile source emissions during the 6 – 8 am period coinciding with the ambient peaks. Therefore, to capture the morning peak in mobile source emissions and to keep the time interval consistent from site to site, we chose to compare ambient and inventory PAMS/NOx and CO/NOx ratios using the ambient and emissions data with time stamps of 6, 7, and 8 am.

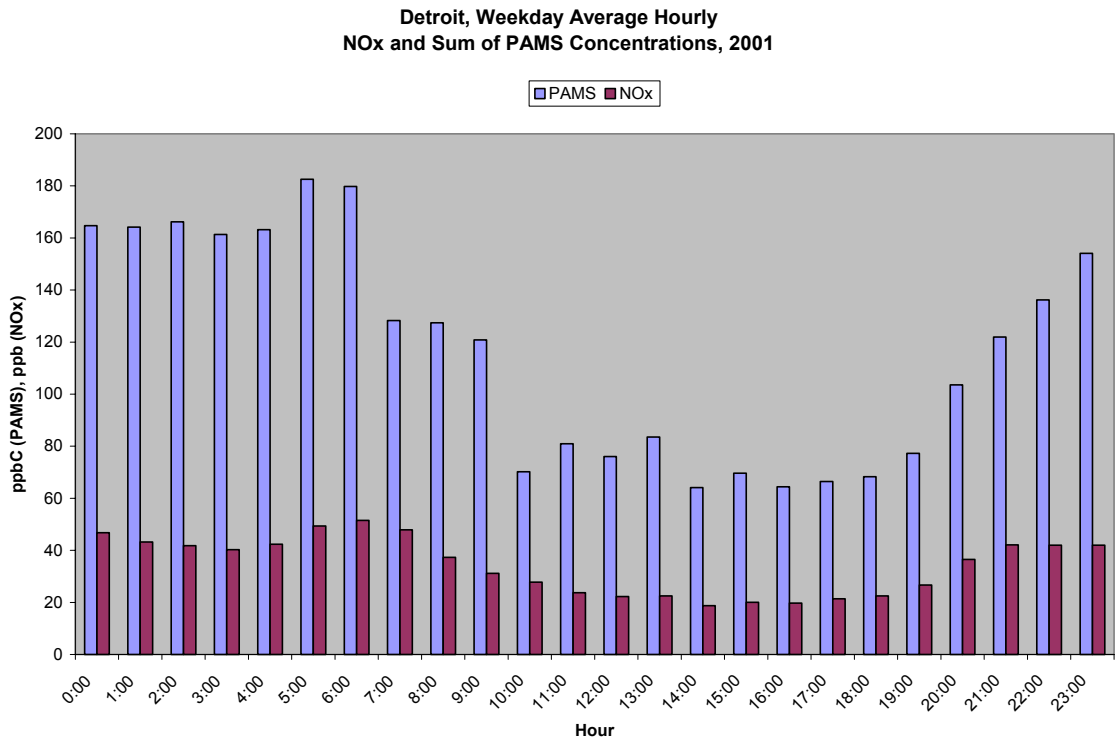


Figure 3-6a. Summer weekday average hourly TNMOC and NOx concentrations: Detroit.

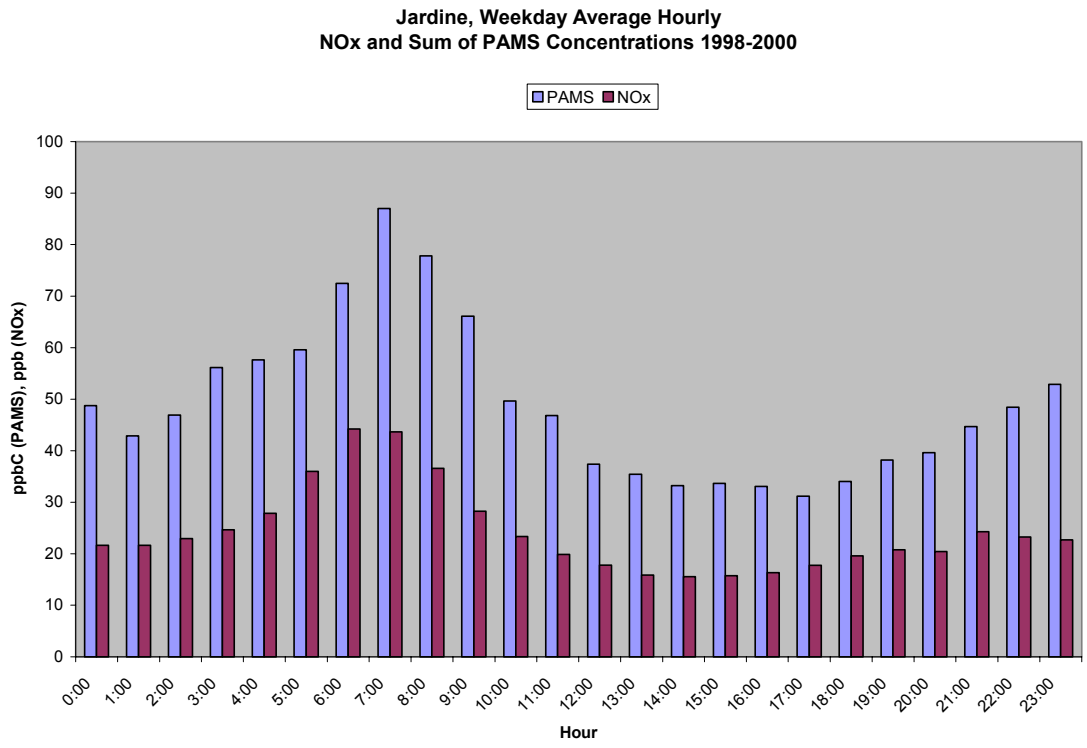


Figure 3-6b. Summer weekday average hourly TNMOC and NOx concentrations: Jardine.

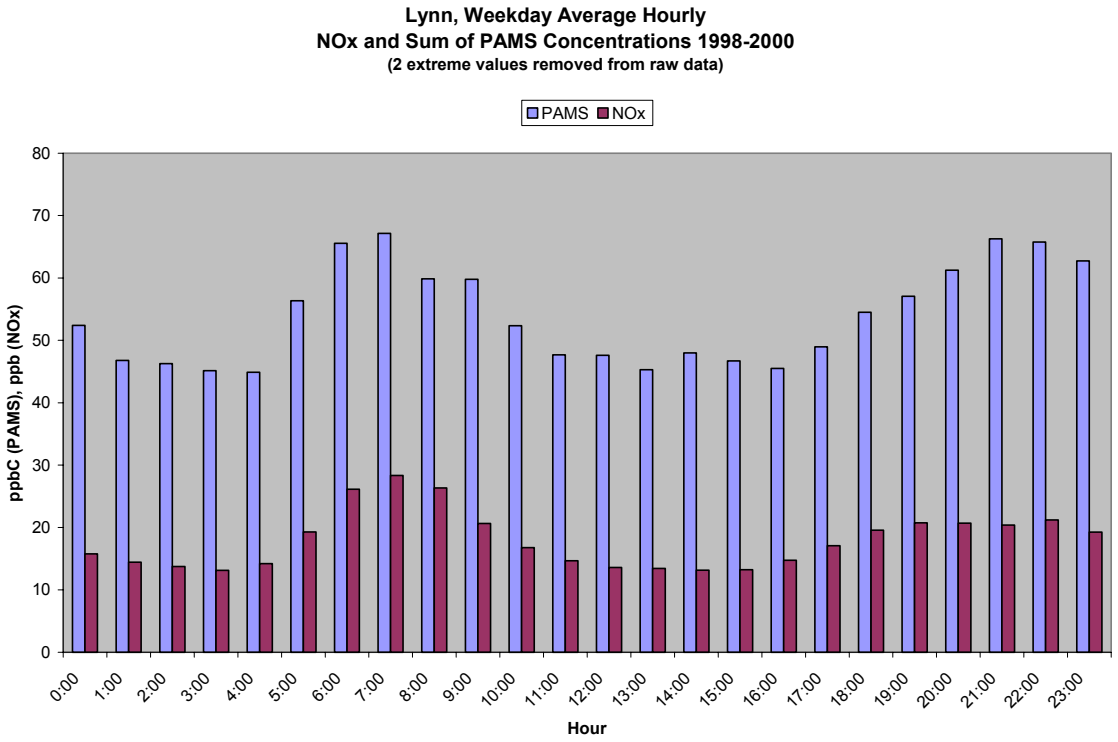


Figure 3-6c. Summer weekday average hourly TNMOC and NOx concentrations: Lynn.

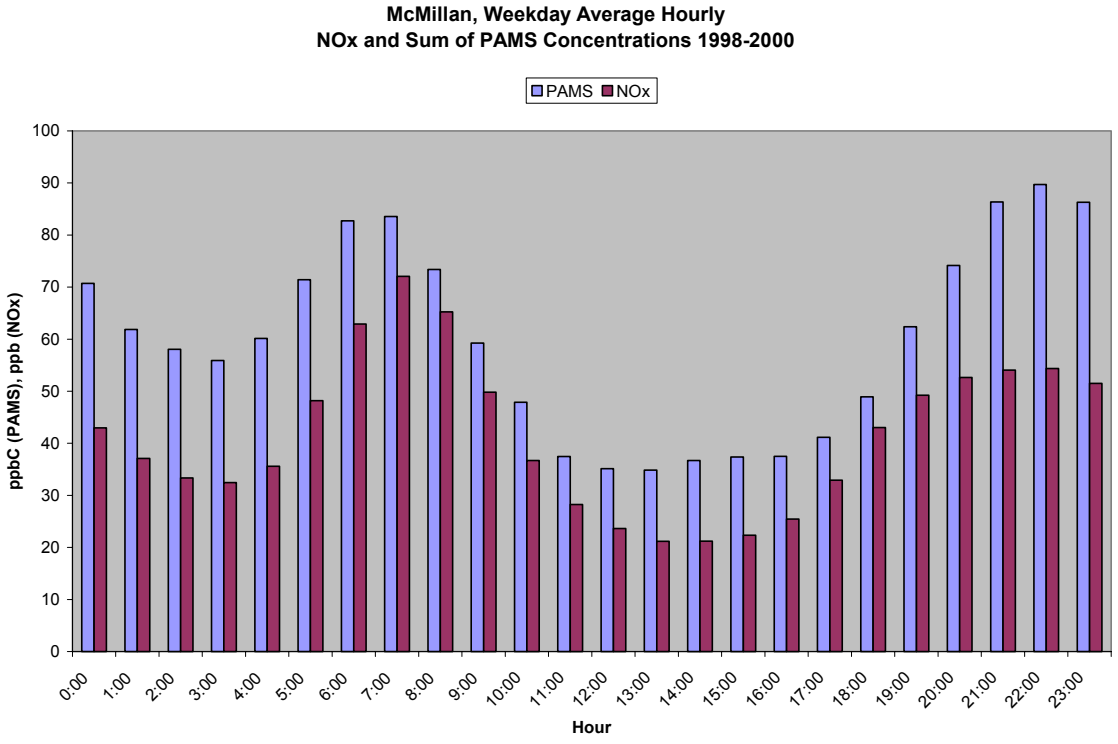


Figure 3-6d. Summer weekday average hourly TNMOC and NOx concentrations: McMillan.

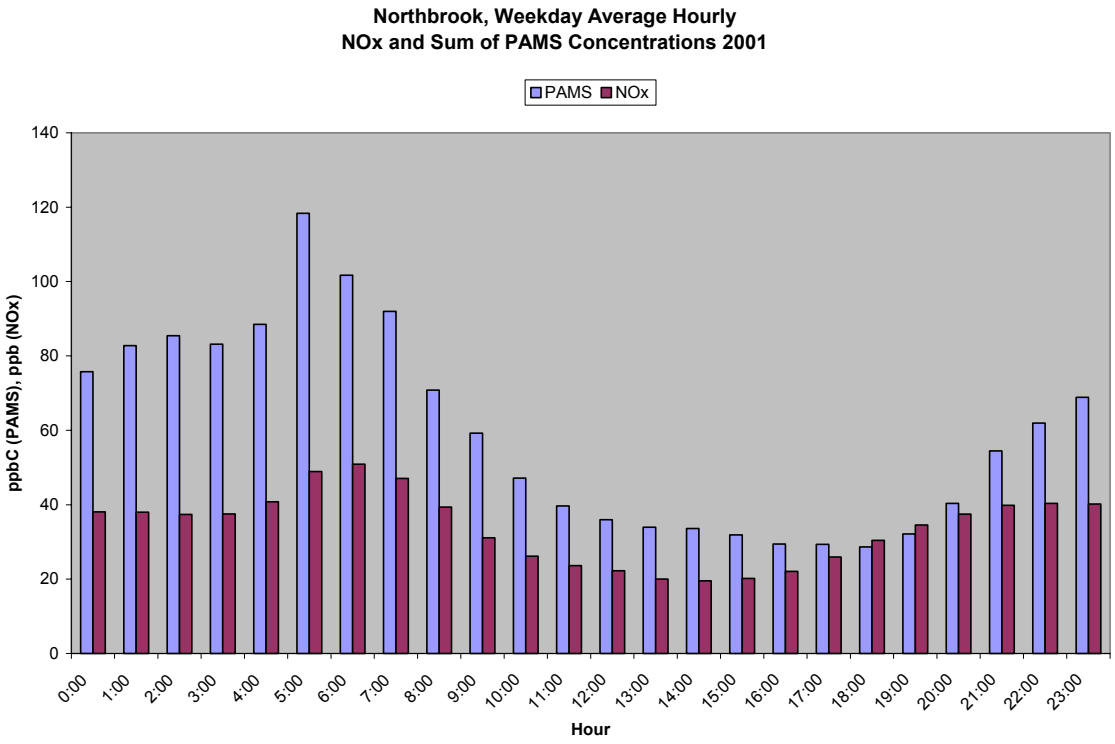
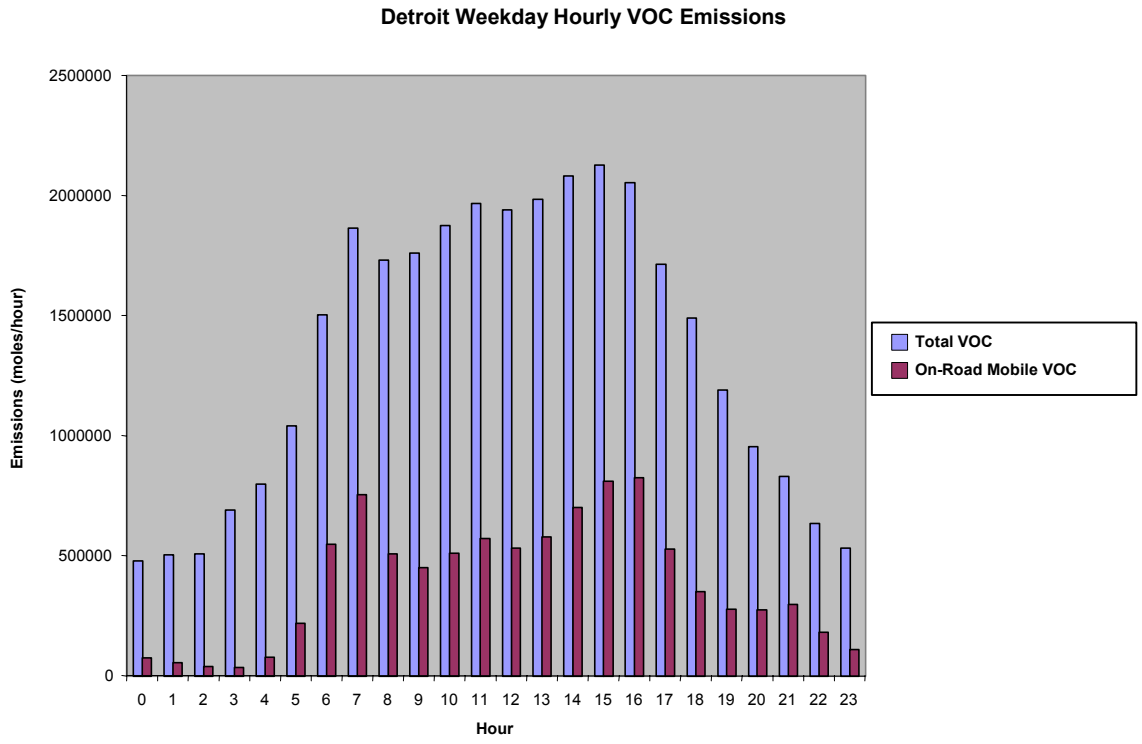


Figure 3-6e. Summer weekday average hourly TNMOC and NOx concentrations: Northbrook.



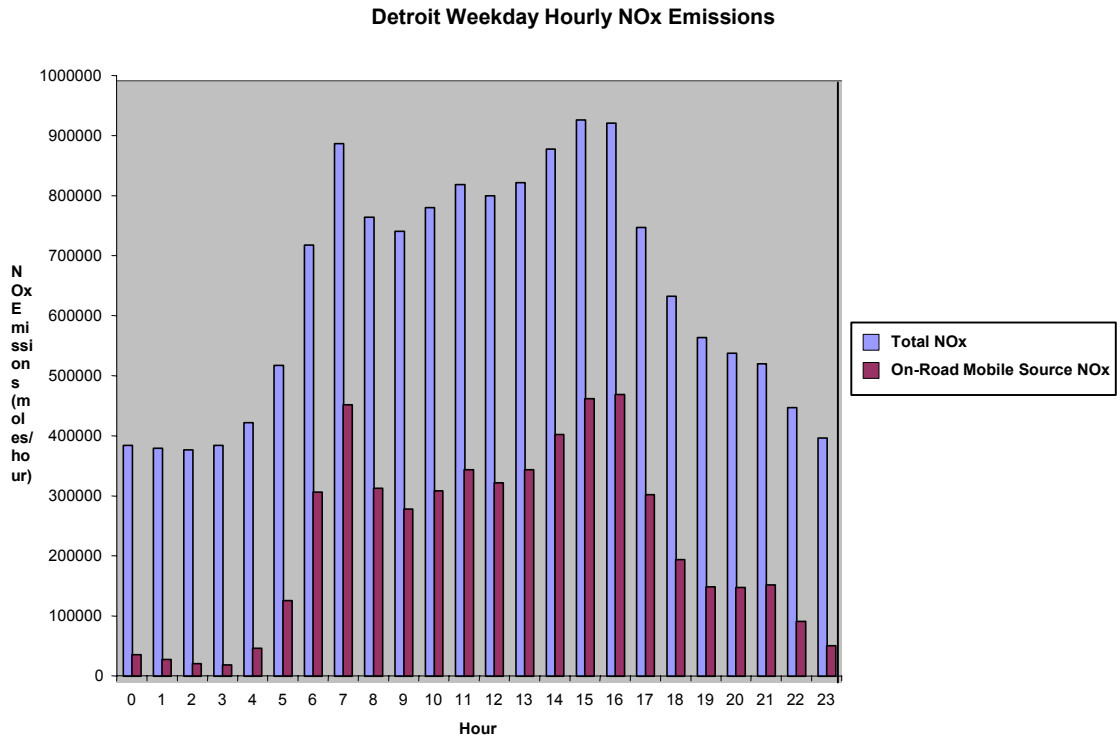
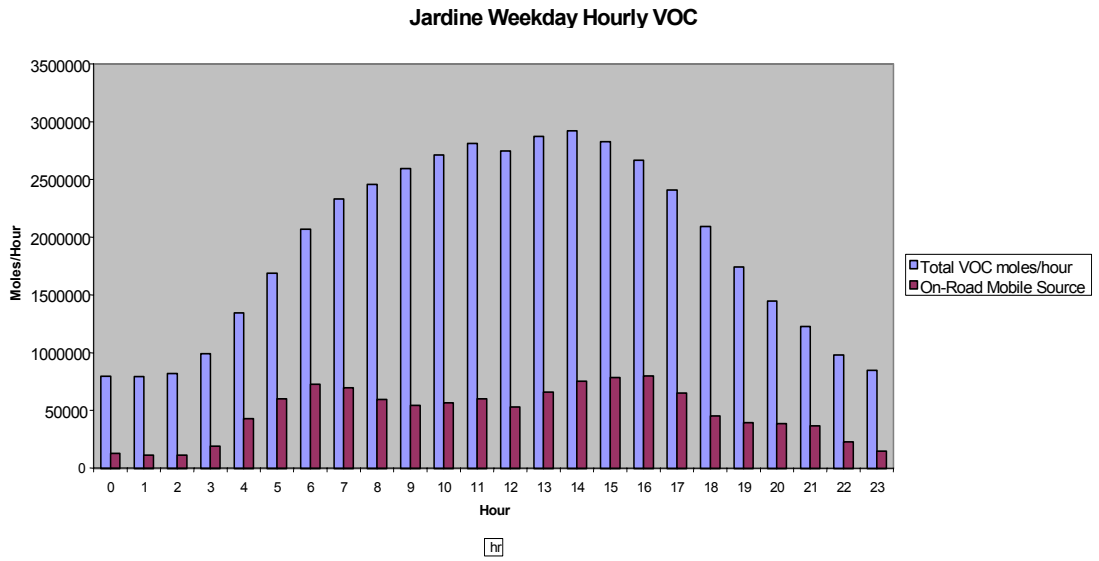


Figure 3-7. Weekday hourly emissions for Detroit: VOC (above), NOx (below).



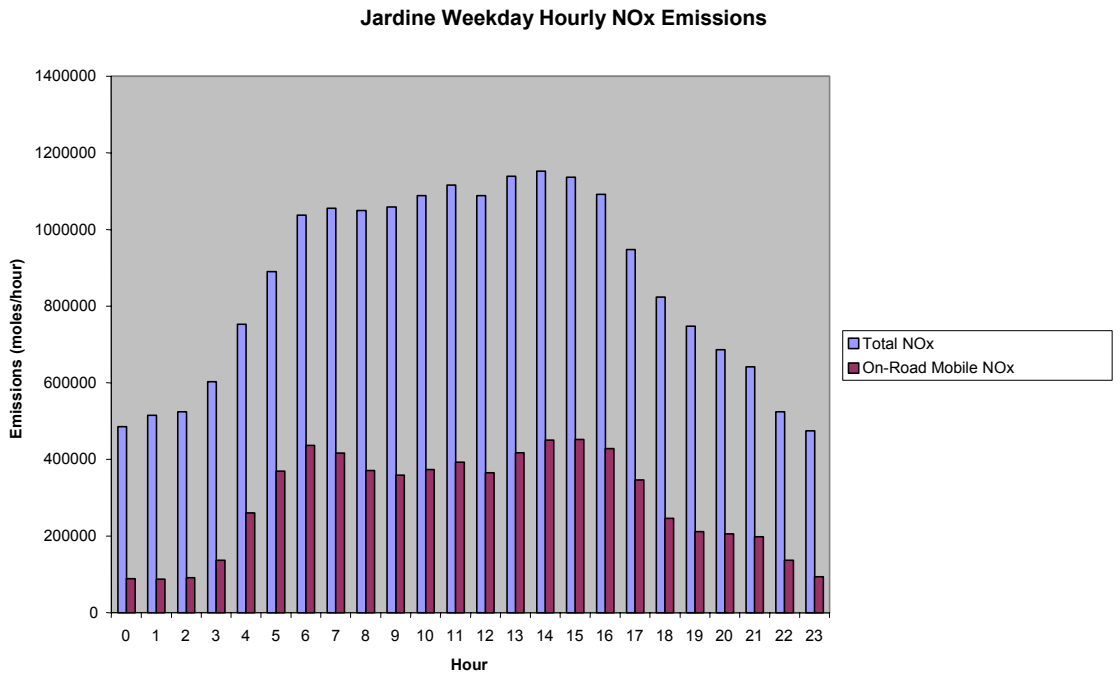
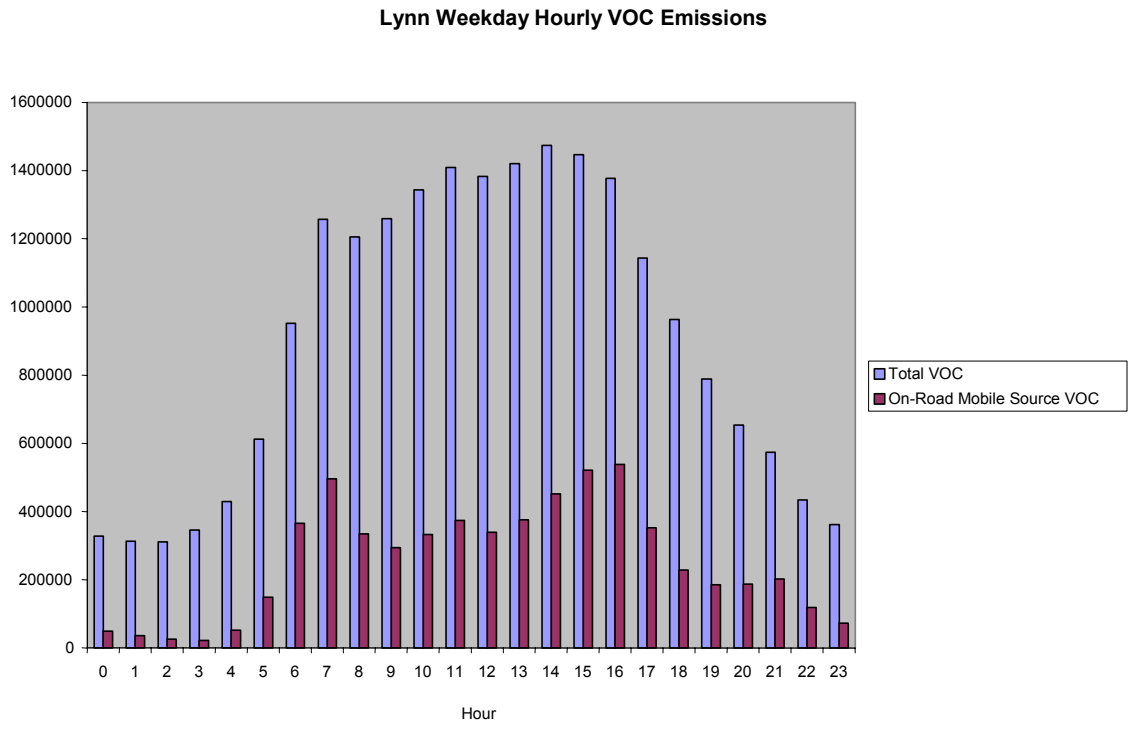


Figure 3-8. Weekday hourly emissions for Jardine: VOC (above), NOx (below).



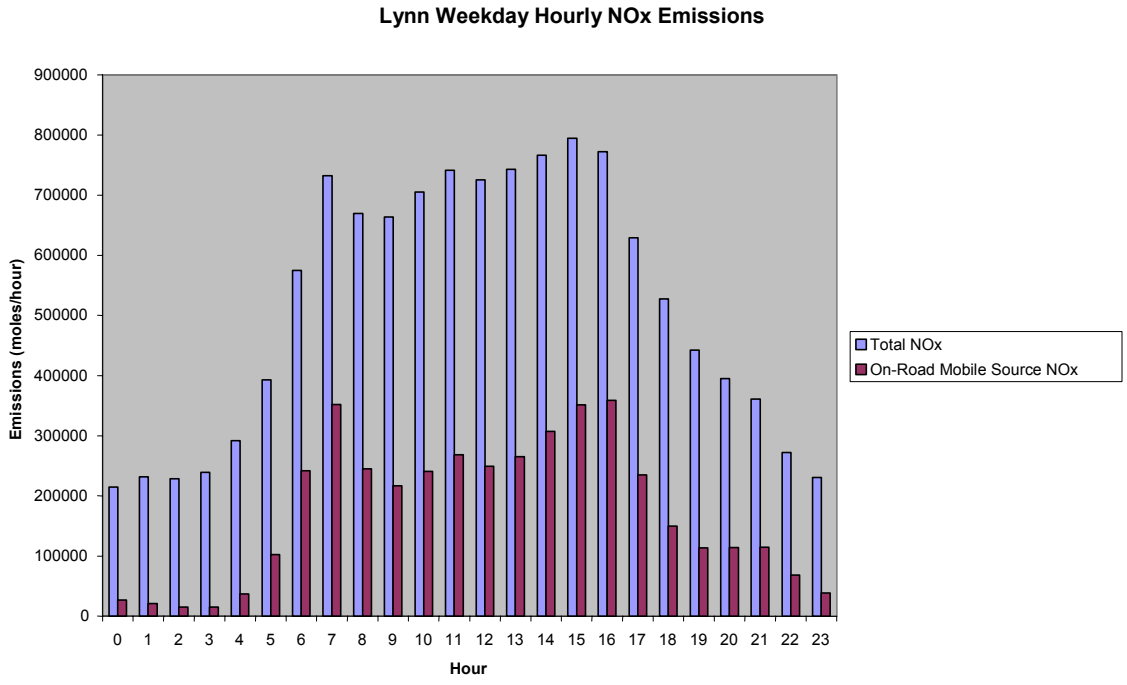
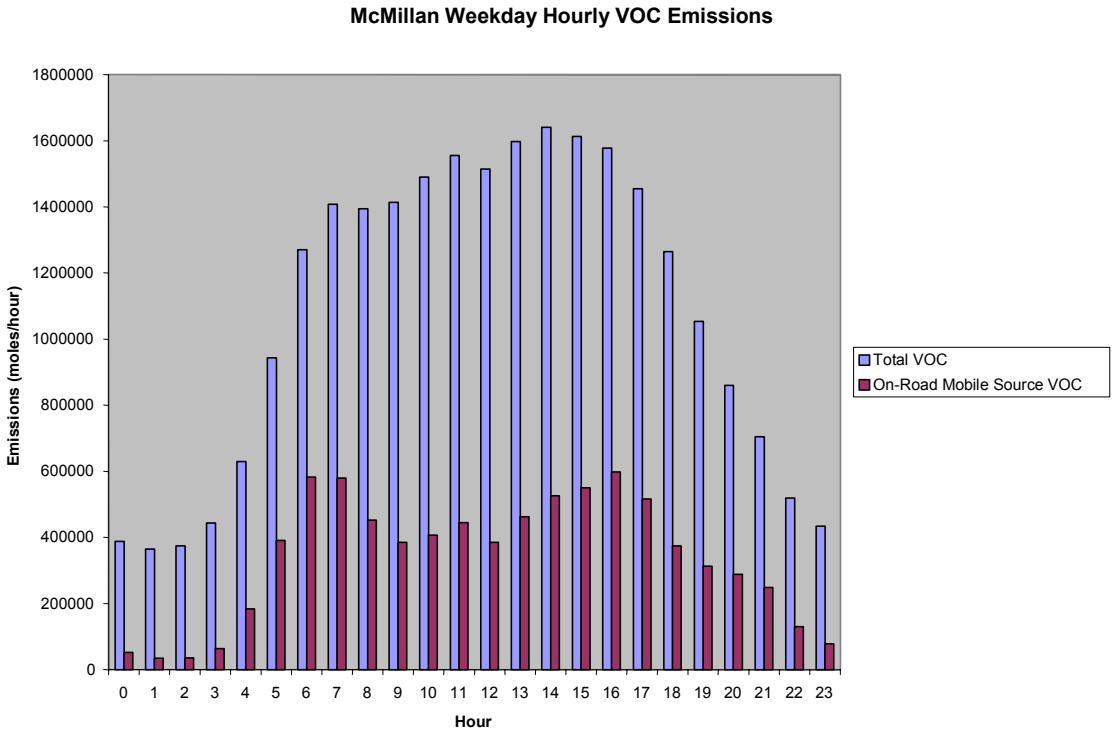


Figure 3-9. Weekday hourly emissions for Lynn: VOC (above), NOx (below).



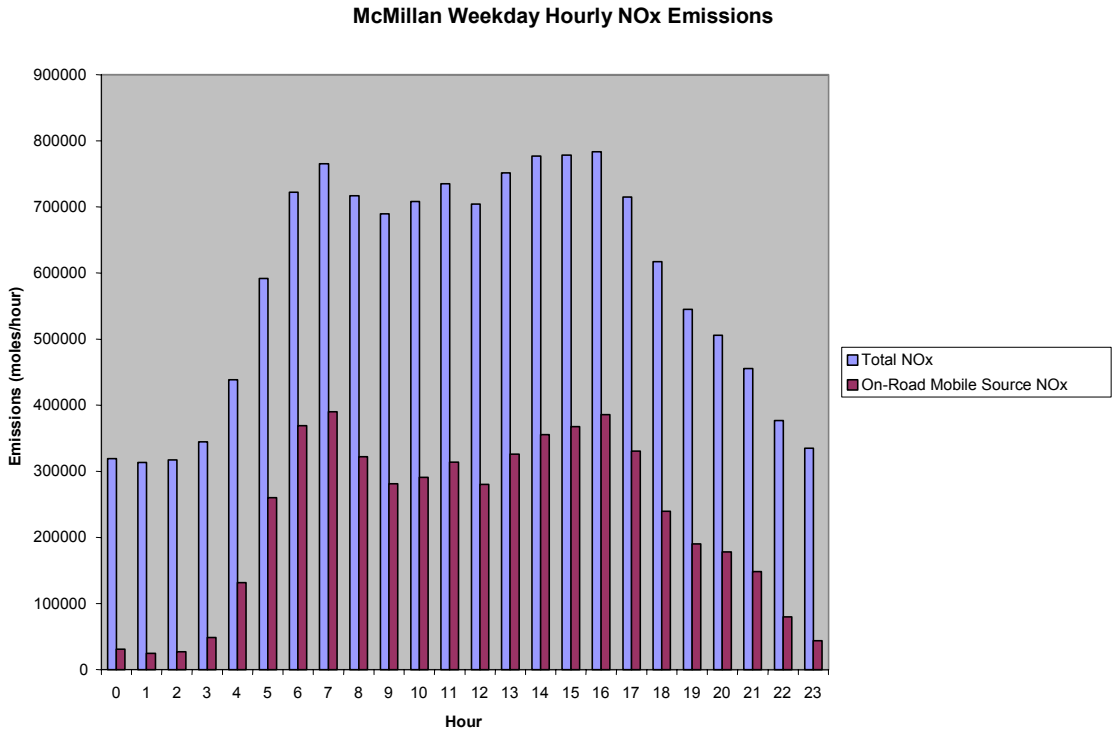
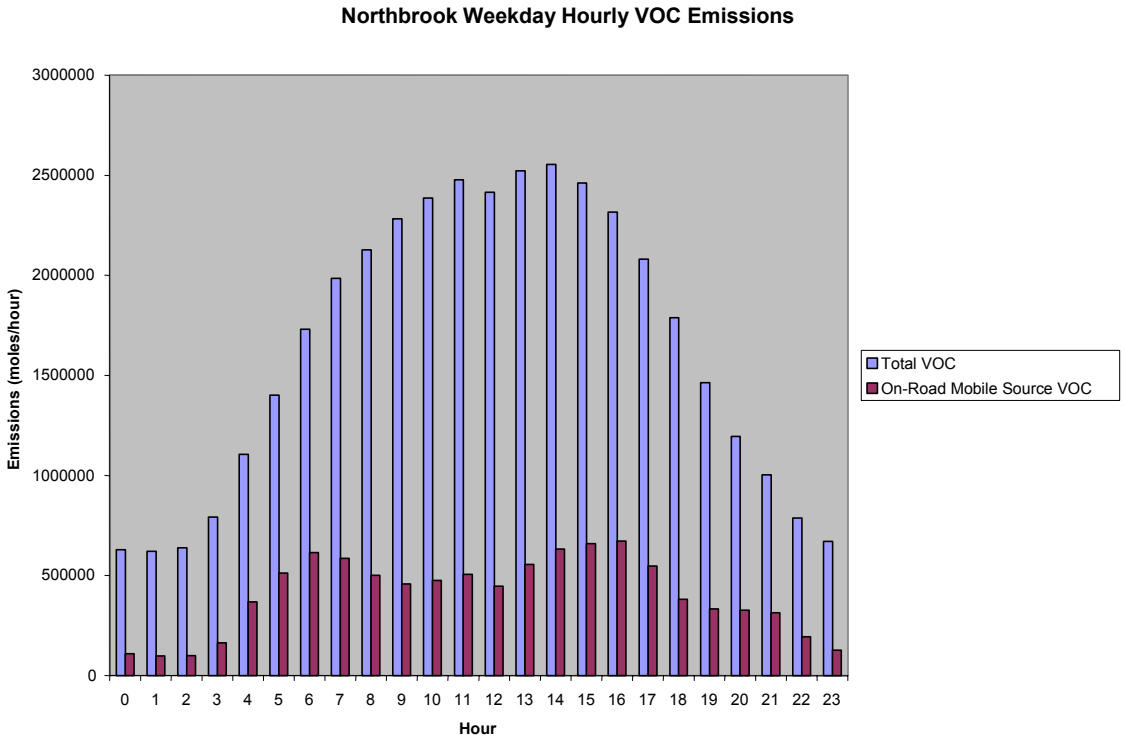


Figure 3-10. Weekday hourly emissions for McMillan: VOC (above), NOx (below).



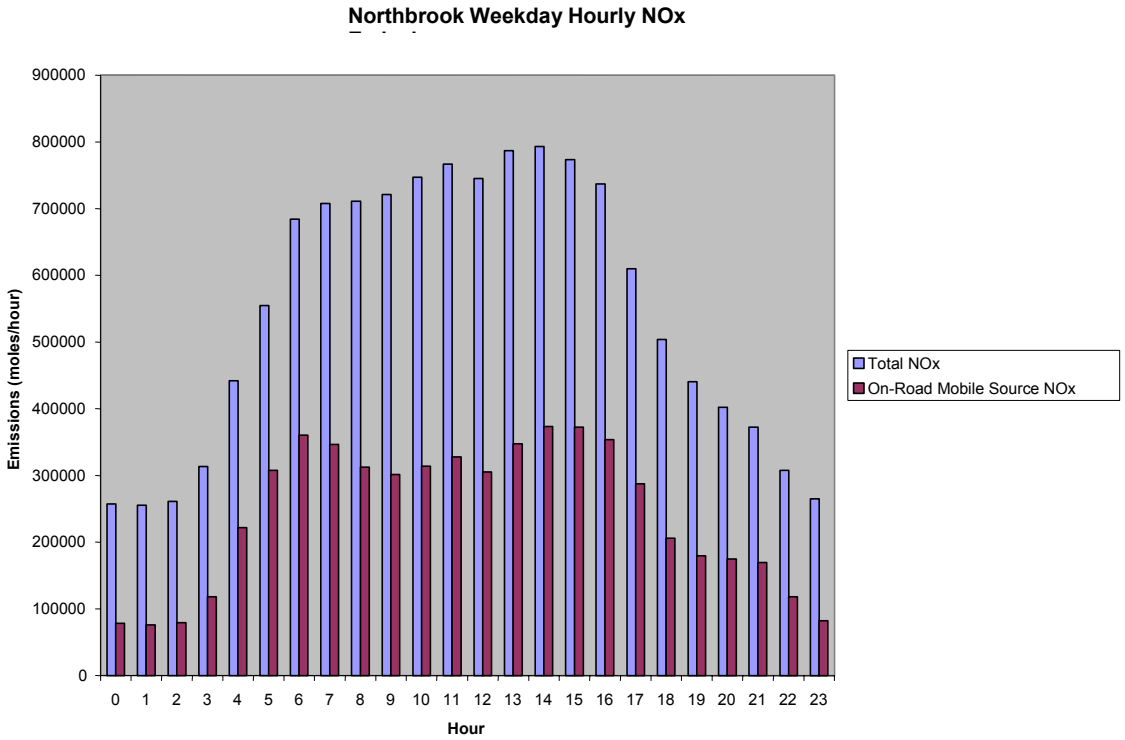


Figure 3-11. Weekday hourly emissions Northbrook: VOC (above), NOx (below).

Biogenics

Of the PAMS target species (see Table 3-9), only isoprene is of biogenic origin. Since the emission inventory developed for this study does not include biogenics, the biogenic fraction of ambient isoprene should be excluded from the ambient/inventory comparisons. Summer season daily isoprene emissions are typically dominated by biogenics. Aside from certain specific industrial processes, most anthropogenic isoprene is in small amounts from on- and off-road mobile sources. Roadway tunnel measurements indicate vehicle exhaust VOC is less than 0.5% isoprene on a mass basis (Yarwood, 2002). Our examination of the 6 – 8 am PAMS data shows that average isoprene fractions of the sum of PAMS species are 0.2, 10.7, 9.2, and 6.3 % at Jardine, Lynn, McMillan, and Northbrook, respectively. Given these relatively small fractions, we chose not to adjust the ambient sum of PAMS mixing ratios to account for biogenic isoprene in this analysis; a less than 10% (10.7% at Lynn) adjustment to ambient PAMS/NOx ratios is small compared to the influence of other uncertainties involved in comparing ambient ratios with inventory ratios.

Background Concentrations and Carryover of Emissions

In deriving species ratios for the ambient/inventory comparisons discussed below, no adjustments were made for either background concentrations or the potential impact of overnight carryover of emissions. Our comparisons were restricted to morning hours with relatively high concentrations most likely to be representative of fresh emissions so as to minimize the influence of background and carryover. This is certainly true for NOx, which is highly reactive and has low overnight emissions (except for some isolated large elevated point sources). The influence of background hydrocarbons is also likely to be minor given our restriction to morning hours with PAMS > 50ppbC, but the contribution of overnight carryover of hydrocarbons from local sources, especially of some of the less reactive PAMS species, is difficult to quantify. In the case of CO, a review of ambient concentrations during the hours selected for analysis indicated that background CO does have potential to produce a noticeable impact on our comparisons of CO/NOx ratios. The potential impact of background CO is discussed in more detail below under Summary and Conclusions.

Summary of Weekday Ratios

Inventory PAMS/NOx and CO/NOx molar ratios were computed for each quadrant around each ambient monitoring site as described in the section on ambient data preparation above and compared with corresponding ratios in the ambient data.

Weekday PAMS/NOx Ratios

Emissions PAMS/NOx ratios for the total inventory and with point sources excluded are compared with the ambient ratios for each monitoring site in Figure 3-12. Table 3-11 summarizes the numerical results in terms of the ratio of ambient PAMS/NOx to total inventory PAMS/NOx ("ratio of ratios") by hour and wind direction quadrant. With the exception of a few isolated cases, the ratio of ratios varies little from hour to hour. Overall,

emission and ambient ratios are in good agreement at McMillan and Northbrook and reasonably good agreement at Detroit. At Lynn, ambient PAMS/NOx ratios exceed the inventory ratios by roughly a factor of 2. At Jardine, the ambient ratio is approximately the same as the inventory ratio computed for all sources combined but the mobile plus area inventory PAMS/NOx ratio exceeds the ambient ratio by approximately 50%. The strong point source influence on PAMS/NOx ratios at Jardine complicates interpretation of results at this site.

Table 3-11. Ratios of weekday average ambient PAMS/NOx ratio to emissions inventory PAMS/NOx ratio (ratio of ratios) by site and upwind quadrant.

(Ambient PAMS/NOx)/(Inventory PAMS/NOx)					
		Hour			
Site	**Quadrant**	**6**	**7**	**8**	Average
Detroit	All	1.38	1.14	1.27	1.26
Detroit	NE	1.50	1.65	1.01	1.37
Detroit	NW	0.86	0.73	1.85	1.17
Detroit	SE	0.85	0.94	1.06	0.95
Detroit	SW	1.48	1.09	0.92	1.16
Jardine	All	1.04	1.14	1.28	1.16
Jardine	NE	21.09	17.52	24.27	20.95
Jardine	NW	0.71	0.86	0.89	0.83
Jardine	SE	0.82	1.02	1.06	0.97
Jardine	SW	0.96	1.07	1.26	1.10
Lynn	All	2.46	2.19	2.46	2.37
Lynn	NE	4.08	2.56	2.66	3.00
Lynn	NW	1.00	1.12	1.36	1.17
Lynn	SE	2.61	3.19	2.81	2.88
Lynn	SW	1.97	1.91	1.92	1.94
McMillan	All	1.16	1.11	1.17	1.15
McMillan	NE	0.69	0.80	0.79	0.76
McMillan	NW	0.98	0.90	0.91	0.93
McMillan	SE	1.10	1.17	1.48	1.25
McMillan	SW	1.10	0.95	1.04	1.03
Northbrook	All	1.05	1.01	1.11	1.06
Northbrook	NE	1.93	1.82	1.06	1.57
Northbrook	NW	0.82	0.73	0.92	0.83
Northbrook	SE	0.86	0.76	0.82	0.81
Northbrook	SW	0.90	1.04	1.18	1.05

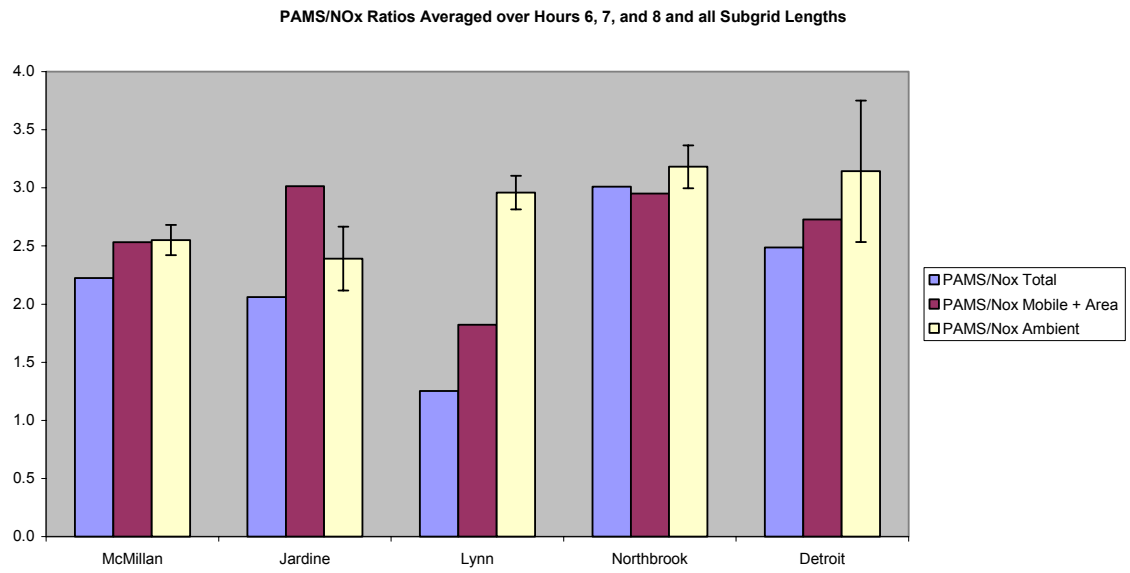


Figure 3-12. Emissions and ambient average summer weekday morning PAMS/NOx ratios.

Ambient/inventory comparisons were also made in which inventory ratios in each upwind quadrant were compared with ambient ratios averaged over hours in which the resultant wind direction fell within that quadrant (Figures 3-13a – e). Sample sizes are small in some cases leading to large uncertainties in the mean ambient ratios; 95% confidence intervals for the mean based on normal statistics are indicated by the error bars in each Figure 3-13.

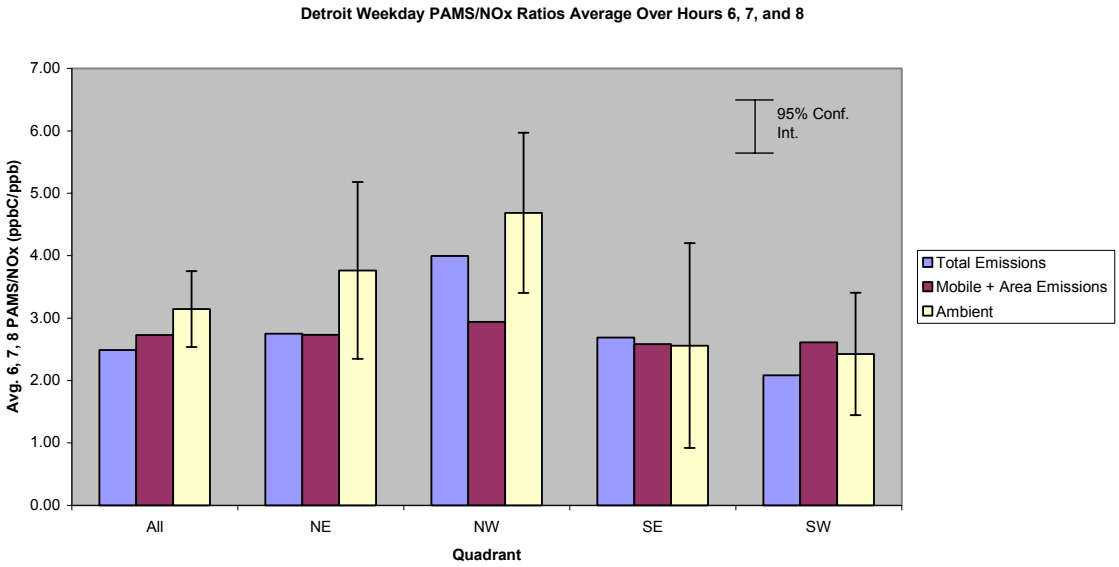


Figure 3-13a. Emissions and ambient average summer weekday morning PAMS/NOx ratios by wind direction quadrant: Detroit.

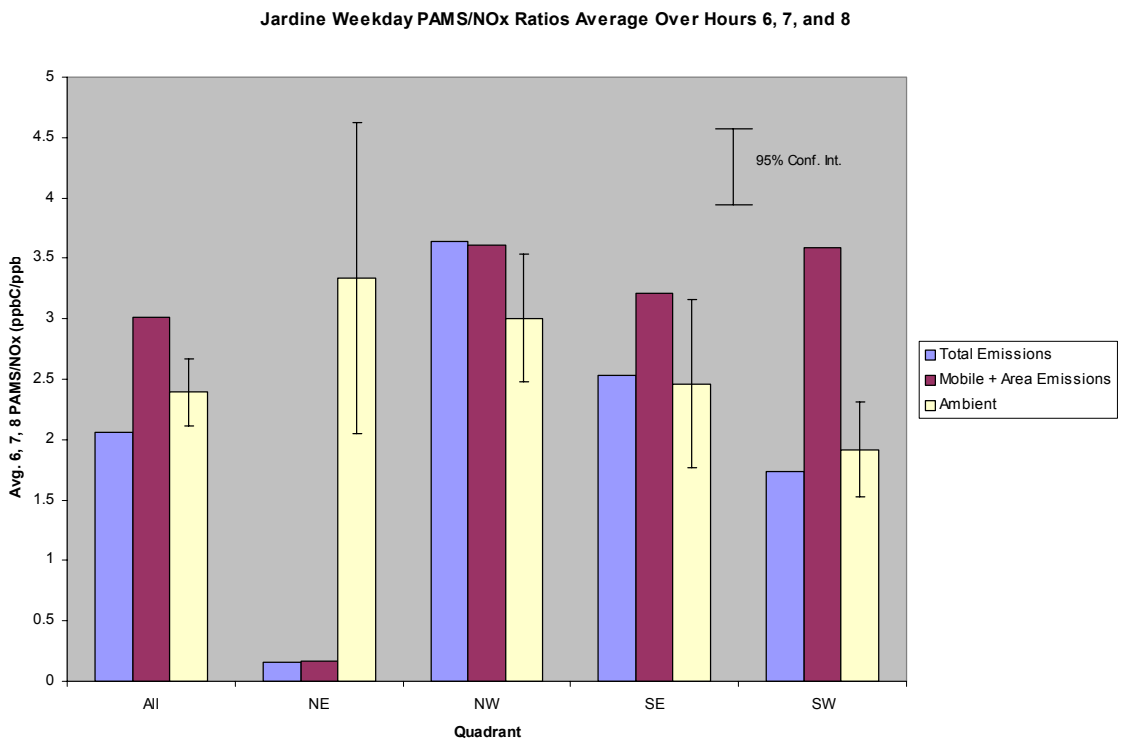


Figure 3-13b. Emissions and ambient average summer weekday morning PAMS/NOx ratios by wind direction quadrant: Jardine.

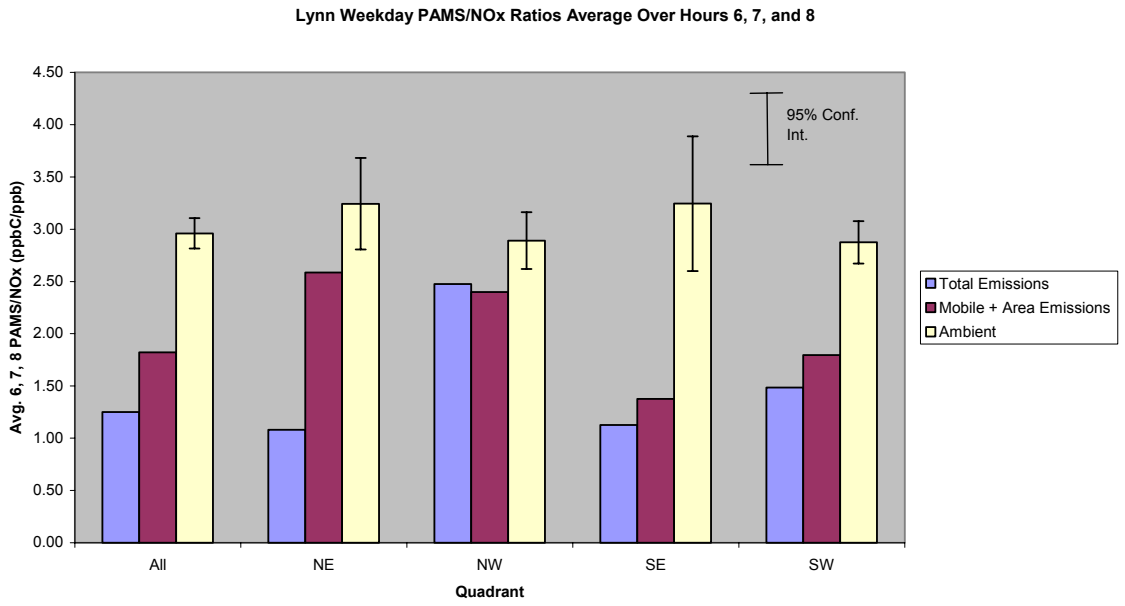


Figure 3-13c. Emissions and ambient average summer weekday morning PAMS/NOx ratios by wind direction quadrant: Lynn.

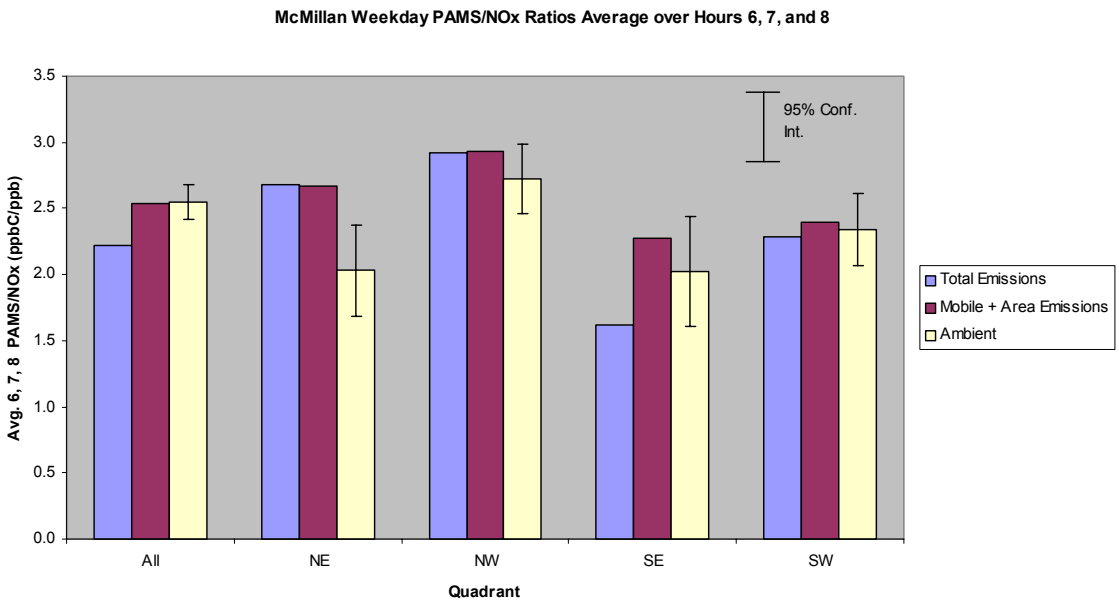


Figure 3-13d. Emissions and ambient average summer weekday morning PAMS/NOx ratios by wind direction quadrant: McMillan.

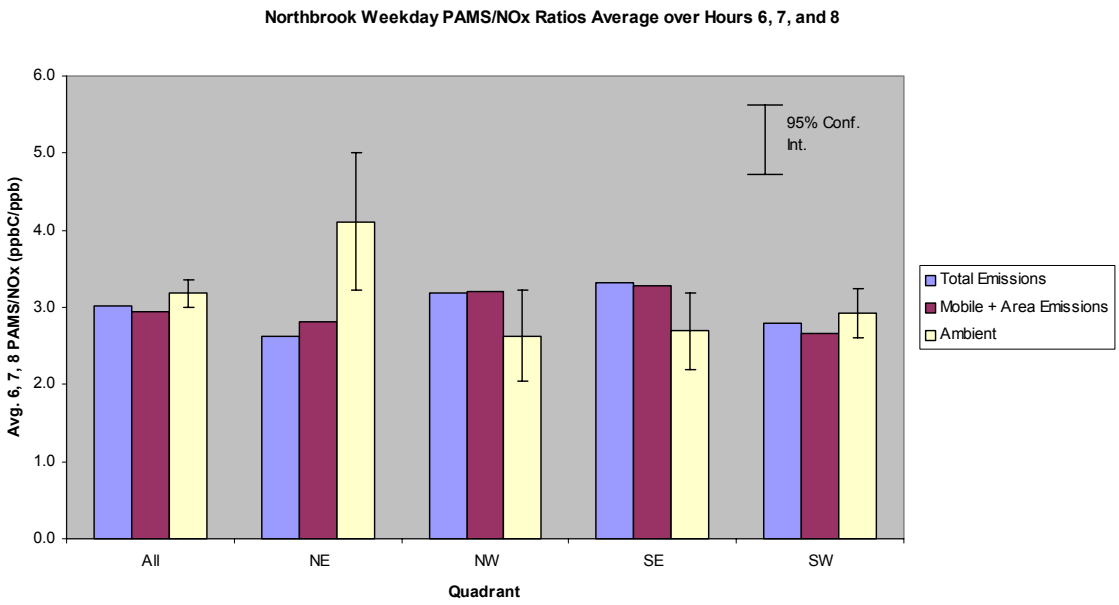


Figure 3-13e. Emissions and ambient average summer weekday morning PAMS/NOx ratios by wind direction quadrant: Northbrook.

For Detroit (Figure 3-13a), ambient and inventory ratios are in fairly good agreement for the SW and SE quadrants but agreement is not as good for the other quadrants. These variable results can be better understood if we examine the spatial distribution of emissions within the 20 km grid centered on the monitoring site (see Figure 3-14). Emissions in the SE and NE quadrants are very low due to the location of the PAMS monitor near the western shore of

Lake St. Clair and the fact that emissions for the Canadian portion of the grid (to the east and southeast of the monitor) were unavailable, making ambient/inventory comparisons for these quadrants not very meaningful. The biggest concentrations of emissions are to the southwest of the monitor, and the inventory ratio in this quadrant agrees well with the ambient ratio. NOx emissions in the NW quadrant are not very large but there are quite a few VOCs in this quadrant, resulting in a high PAMS/NOx emissions ratio. Judging from the fact that the emissions PAMS/NOx ratio with all sources combined is higher than the ratio for just the mobile plus area sources, it appears that a good proportion of the VOC emissions in the NW quadrant are from point sources. Ambient PAMS/NOx ratios under northwest winds are also higher than in other quadrants and are in reasonable agreement with the total inventory ratio.

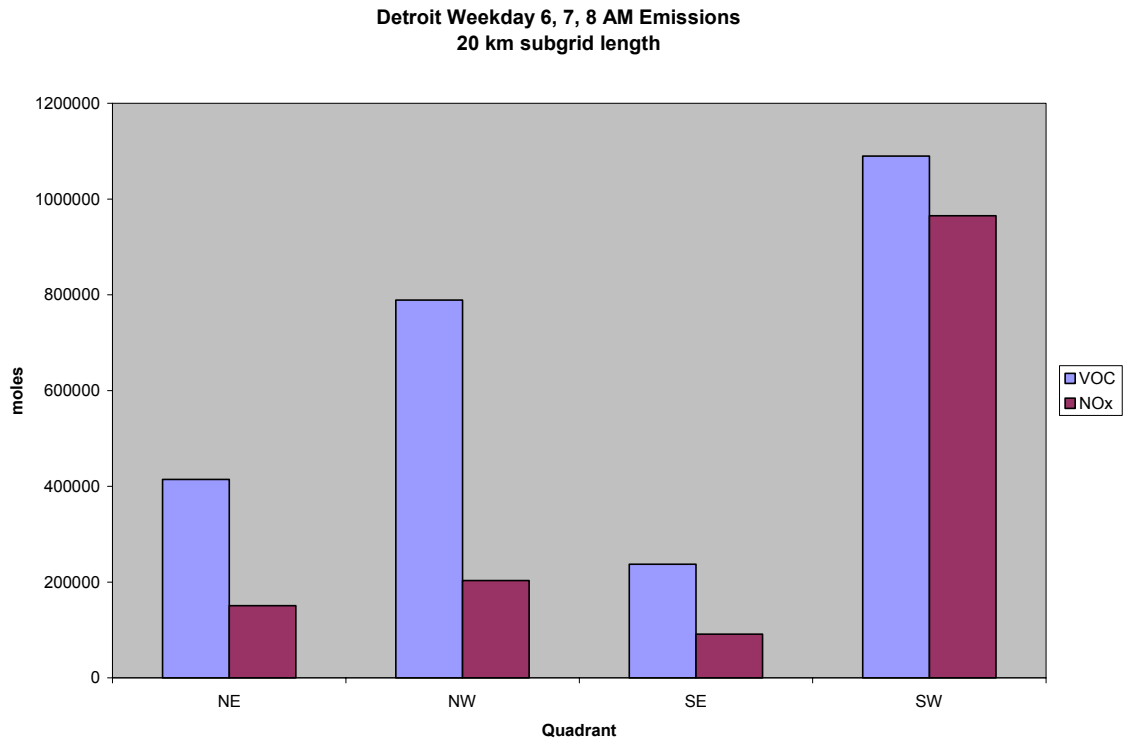


Figure 3-14. Summer weekday morning VOC and NOx emissions: Detroit.

For Jardine (Figure 3-13b), ambient PAMS/NOx ratios are less than the inventory ratios in the SE, SW, and NW quadrants, although the ambient ratios match the inventory ratio for all sources combined very well for the SW quadrant. Comparisons for the NE quadrant are not meaningful since this sector is out over Lake Michigan (the SE quadrant also includes a large over water segment – see Figure 3-5). The good agreement with the total inventory ratio in the SW quadrant must be viewed with caution because 91% of the point source NOx in this quadrant is from elevated point sources, and it is not clear to what extent these emissions actually impact the monitoring site. The PAMS/NOx emissions ratio for all sources combined except elevated point sources is 3.4 (averaged over all quadrants), just slightly higher than the mobile+area source emissions ratio. Point sources have little effect on the inventory PAMS/NOx ratio in the NW quadrant. The fact that inventory ratios for all quadrants combined are similar to ratios in the SW quadrant indicates that the highest concentration of

emissions are to the southwest of the monitoring site; this is confirmed by a summary of the inventory totals by quadrant (not shown). Inventory PAMS/NOx ratios for mobile plus area sources are higher at Jardine in the SE, SW, and NW quadrants than around any of the other sites. In contrast, ambient PAMS/NOx ratios at Jardine are lower than around any of the other sites.

For Lynn (Figure 3-13c), ambient ratios exceed the inventory ratios in all quadrants. Ambient ratios at this site are of roughly the same magnitude (roughly between 2.5:1 and 3:1) as at Detroit, McMillan, and Northbrook but inventory ratios are quite a bit lower (except in the NW quadrant where the ambient and inventory ratios are nearly in agreement). The mean ambient ratio in the NE quadrant is similar to the emissions ratio for mobile+area sources but point sources in this quadrant have a strong effect on the emissions ratio, making it difficult to interpret the ambient/inventory comparison. Overall, there is significantly greater heterogeneity in the mobile+area PAMS/NOx emissions ratio at Lynn than at the other sites and this heterogeneity is not evident in the ambient data.

For McMillan (Figure 3-13d), agreement between ambient and inventory PAMS/NOx ratios is very good overall. Emission ratios differ somewhat between the NE-NW and SE-SW quadrants. The SE quadrant is the only one in which point sources have a significant influence on the ratio. Ambient ratios are about 25% lower in the NE and SE quadrants, matching similarly lower inventory ratios in the SE but not in the NE quadrant. This difference may simply be an effect of using such a low resolution (90 deg.) for the direction-specific comparisons.

For Northbrook (Figure 3-13e), the comparison of ambient and inventory PAMS/NOx ratios differs between quadrants. Point sources do not have a significant impact on the PAMS/NOx ratio at this site. However, a review of the emissions data for this site shows that emissions in the NE and NW quadrants are very low due to the presence of Lake Michigan and lower source densities to the northwest. This makes comparisons with ambient ratios for the NE and NW quadrants of less importance. The somewhat higher emission PAMS/NOx ratios in the SE quadrant as compared to the SW quadrant are not seen in the ambient data, but this may simply be an effect of using such a low resolution (90 deg.) for the direction-specific comparisons. For data from all quadrants combined, the ambient and inventory ratios are in nearly perfect agreement.

Weekday CO/NOx Ratios

Table 3-12 summarizes results for CO/NOx in terms of the ratio of ambient CO/NOx to total inventory CO/NOx ("ratio of ratios") by hour and wind direction quadrant. The CO/NOx ratio of ratios varies somewhat more from hour to hour than the PAMS/NOx ratio of ratios discussed above. Outside of Detroit, all but two of the individual ratio of ratios are greater than 1:1, with many values greater (in some cases much greater) than 2:1.

Table 3-12. Ratios of weekday average ambient CO/NOx ratio to emissions inventory CO/NOx ratio (ratio of ratios) by site and upwind quadrant.

[Ambient CO/NOx]/[Inventory CO/NOx]					
		Hour			
Site	**Quad**	**6**	**7**	**8**	Average
Detroit	All	0.51	0.30	0.19	0.33
Detroit	NE	0.62	0.57	0.28	0.49
Detroit	NW	0.30	0.13	0.00	0.14
Detroit	SE	0.58	0.45	0.44	0.49
Detroit	SW	0.35	0.16	0.18	0.23
Jardine	All	3.50	3.22	4.36	3.70
Jardine	NE	59.84	31.65	66.69	52.68
Jardine	NW	2.29	2.16	3.08	2.52
Jardine	SE	2.46	3.05	4.18	3.24
Jardine	SW	3.47	3.20	3.40	3.36
Lynn	All	5.13	4.07	4.81	4.65
Lynn	NE	6.35	5.21	6.14	5.87
Lynn	NW	2.25	2.32	2.62	2.40
Lynn	SE	2.14	2.80	3.04	2.67
Lynn	SW	4.41	3.57	3.92	3.97
McMillan	All	1.56	1.68	1.51	1.58
McMillan	NE	1.06	1.42	0.96	1.14
McMillan	NW	1.36	1.37	1.17	1.30
McMillan	SE	1.26	1.03	1.77	1.35
McMillan	SW	1.44	1.50	1.61	1.51
Northbrook	All	2.14	2.23	2.30	2.22
Northbrook	NE	3.65	3.51	2.51	3.21
Northbrook	NW	2.02	2.26	2.39	2.23
Northbrook	SE	2.04	1.97	2.07	2.03
Northbrook	SW	1.95	2.17	2.26	2.13

Emissions CO/NOx ratios for the total inventory and with point sources excluded are compared with the ambient ratios for each monitoring site in Figure 3-15 (results by wind quadrant with 95% confidence intervals for the mean ambient ratios are shown in Figures 3-16a – e). Ambient ratios exceed the inventory ratios at all sites except Detroit. As previously noted, in Detroit, Lynn and Northbrook the closest CO monitors are located much further away from the PAMS monitor than in the other locations. It is possible that this resulted in higher ambient CO/NOx ratios than would have been observed at the PAMS monitors. In the case of Jardine, the CO monitor is located on-shore in downtown Chicago while the PAMS site is located on a pier extending out over Lake Michigan. As a result, it is possible that CO concentrations are lower at the PAMS site than at the CO monitor. At McMillan, the PAMS and CO sites are located relatively close to one another. At Lynn, the mobile+area CO/NOx emissions ratio is very low compared to the other sites and, surprisingly, the ratio for mobile+area sources is less than half the ratio for all sources combined. For mobile sources only, the CO/NOx ratio at Lynn averages 11.4 which is similar to that found around the other sites. In contrast, ambient ratios at Lynn are quite high. At McMillan, the ambient ratio exceeds the inventory ratio by 50%, a difference which does not necessarily indicate a problem with the inventory ratio given that the CO and NOx is being measured at different

monitors and that this comparison does not include any adjustment for background CO (see the Summary and Conclusions section below for a discussion of background CO adjustments).

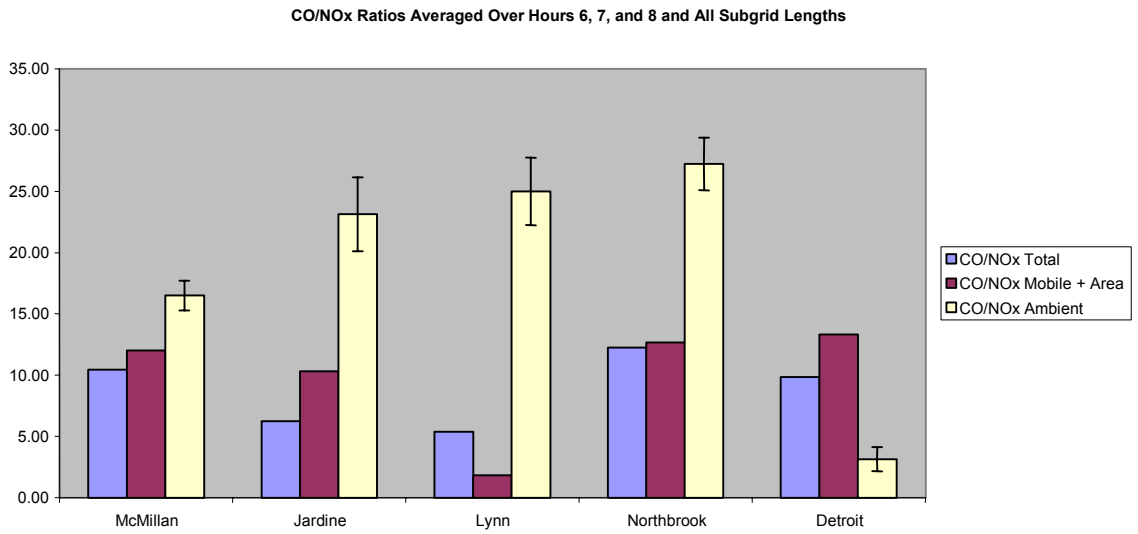


Figure 3-15. Emissions and ambient average summer weekday morning CO/NOx ratios.

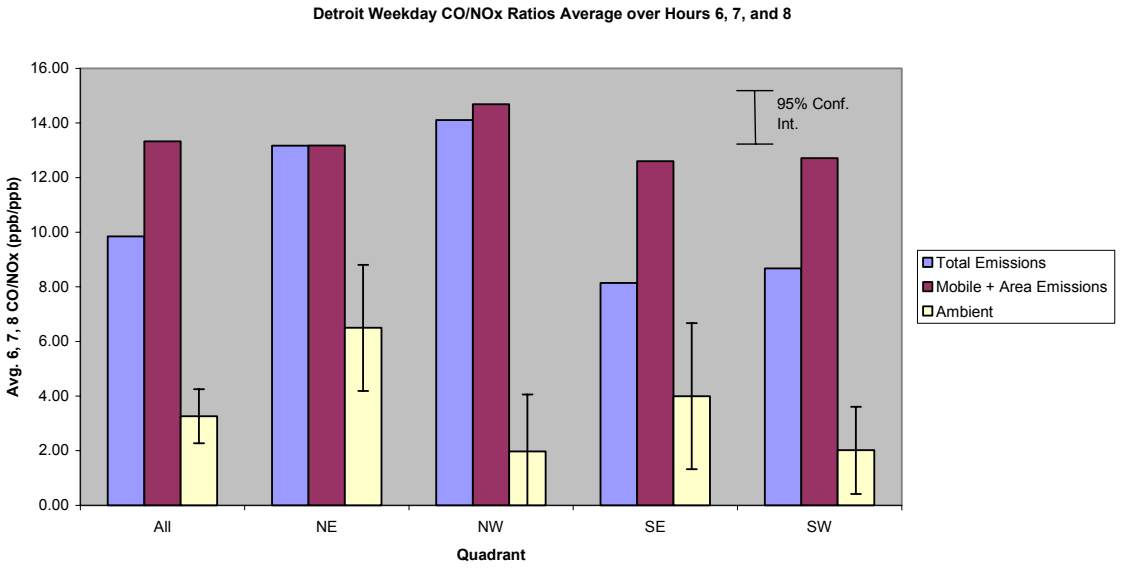


Figure 3-16a. Emissions and ambient average summer weekday morning CO/NOx ratios by wind direction quadrant: Detroit.

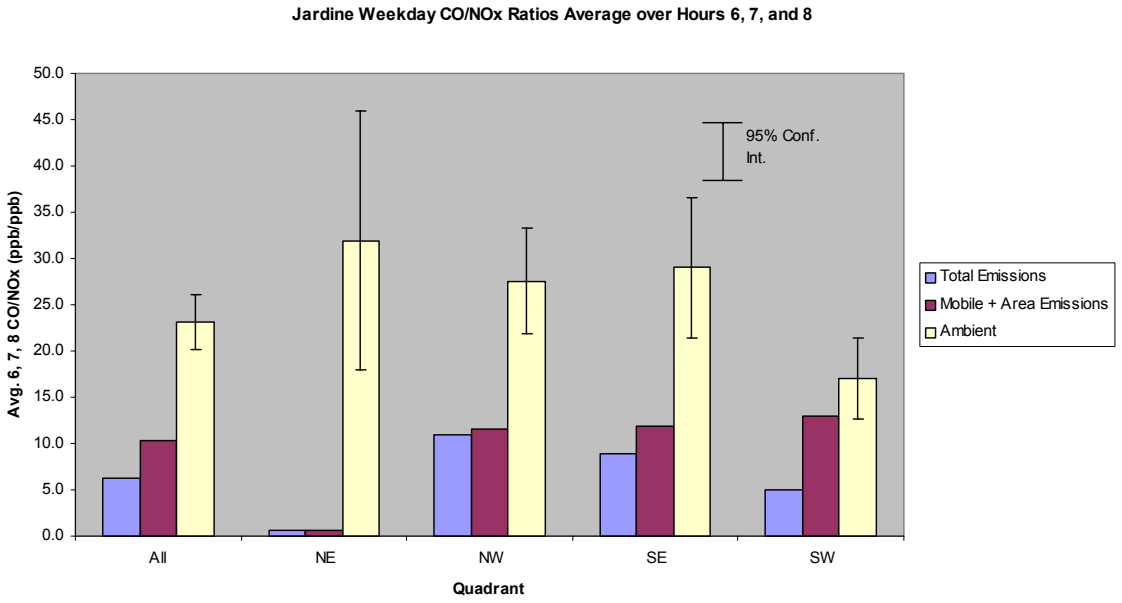


Figure 3-16b. Emissions and ambient average summer weekday morning CO/NOx ratios by wind direction quadrant: Jardine.

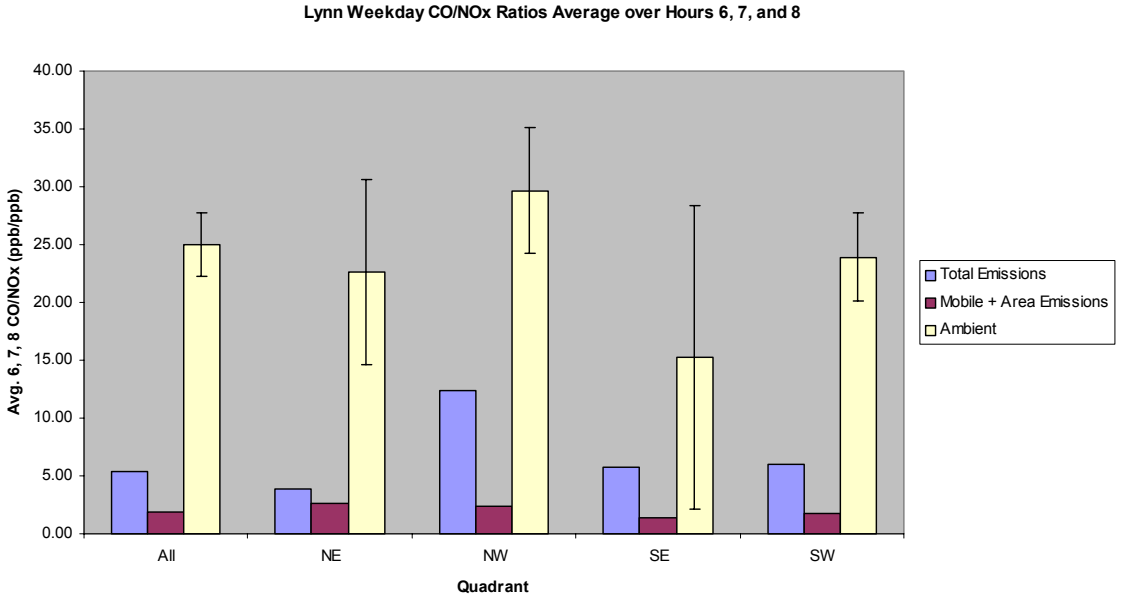


Figure 3-16c. Emissions and ambient average summer weekday morning CO/NOx ratios by wind direction quadrant: Lynn.

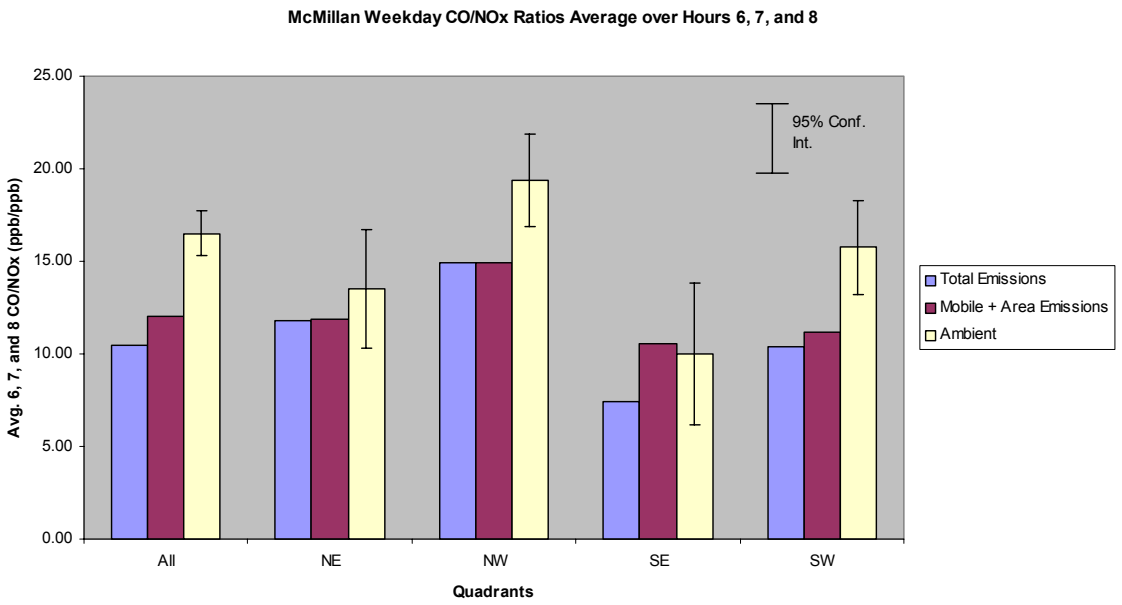


Figure 3-16d. Emissions and ambient average summer weekday morning CO/NOx ratios by wind direction quadrant: McMillan.

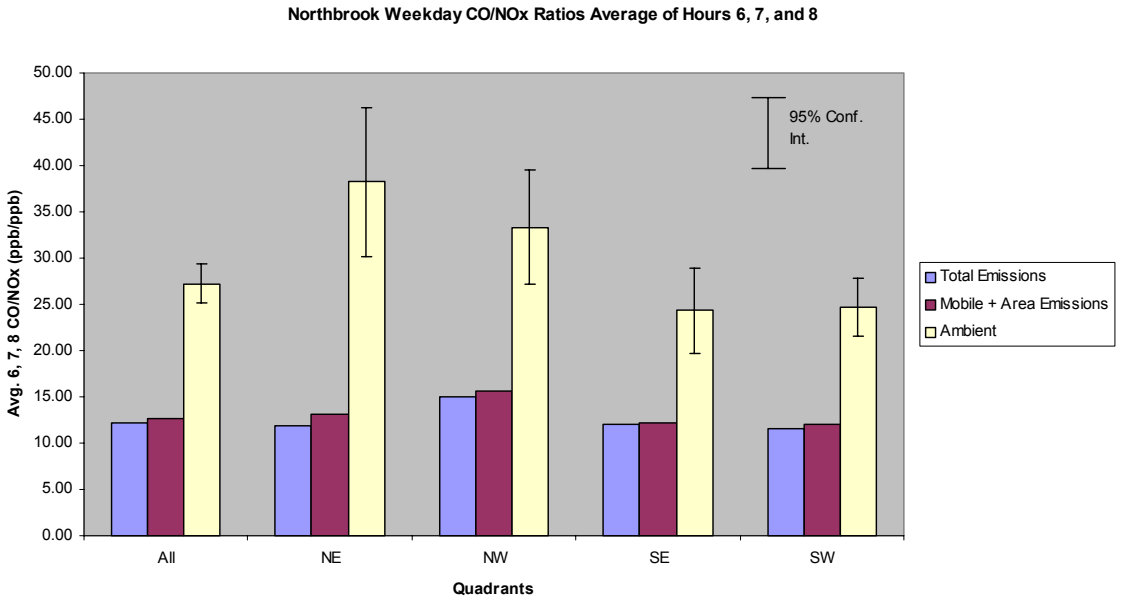


Figure 3-16e. Emissions and ambient average summer weekday morning CO/NOx ratios by wind direction quadrant: Northbrook.

Weekday/Weekend Comparisons

As noted above, development of the emission inventories used in this analysis took into account differences between activity levels on weekday mornings vs. weekend mornings, to the extent that such data were available. This allowed us to compare morning ambient and inventory PAMS/NOx and CO/NOx ratios on weekends with ratios on weekdays. Weekday/weekend comparisons for each location are illustrated in Figures 3-17a – e (for PAMS/NOx ratios) and Figures 3-18a – e (for CO/NOx ratios).

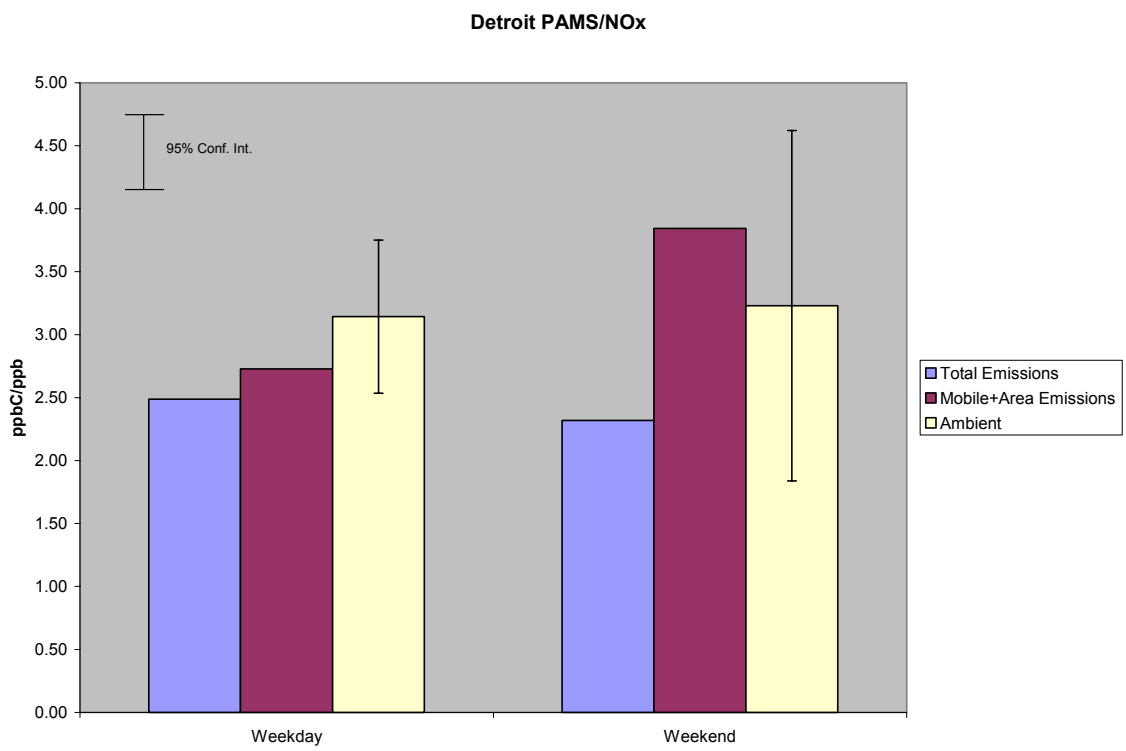


Figure 3-17a. Weekday and weekend morning PAMS/NOx ratios: Detroit.

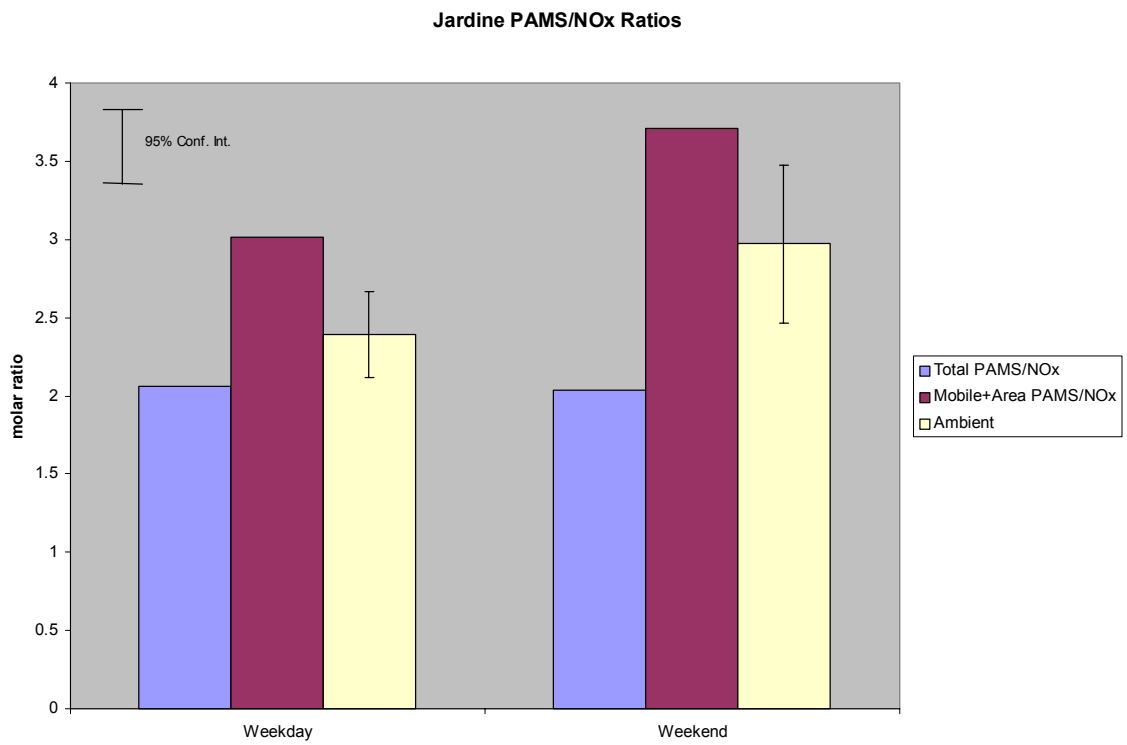


Figure 3-17b. Weekday and weekend morning PAMS/NOx ratios: Jardine.

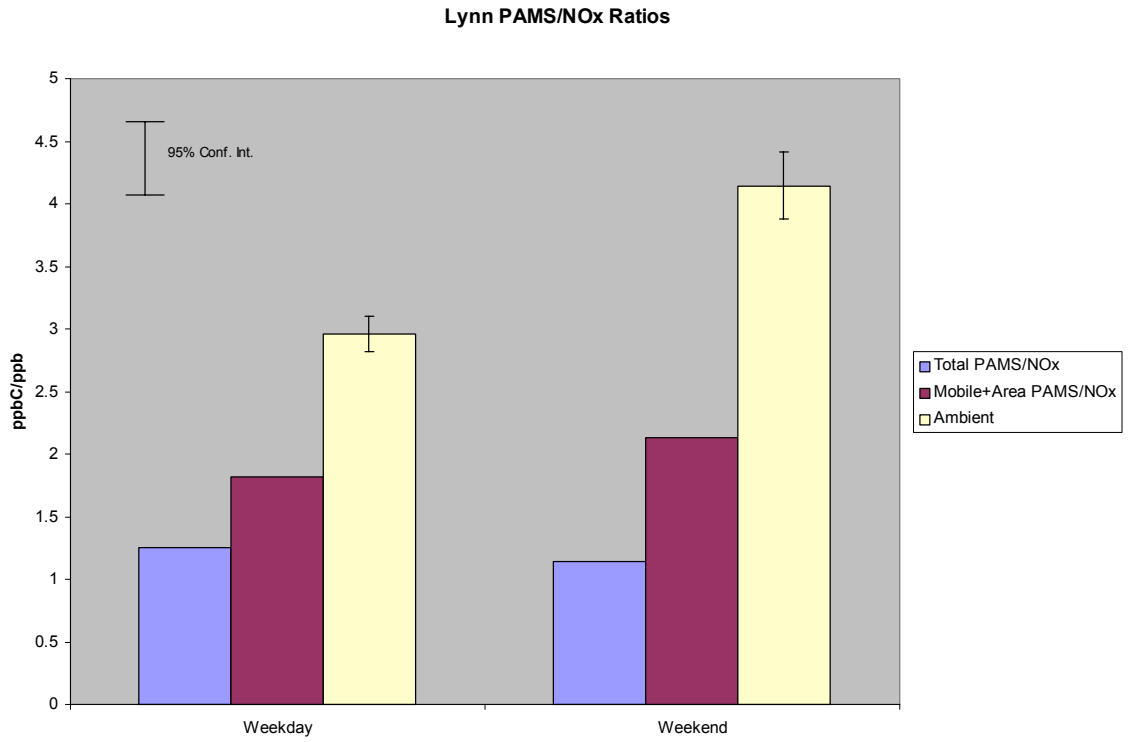


Figure 3-17c. Weekday and weekend morning PAMS/NOx ratios: Lynn.

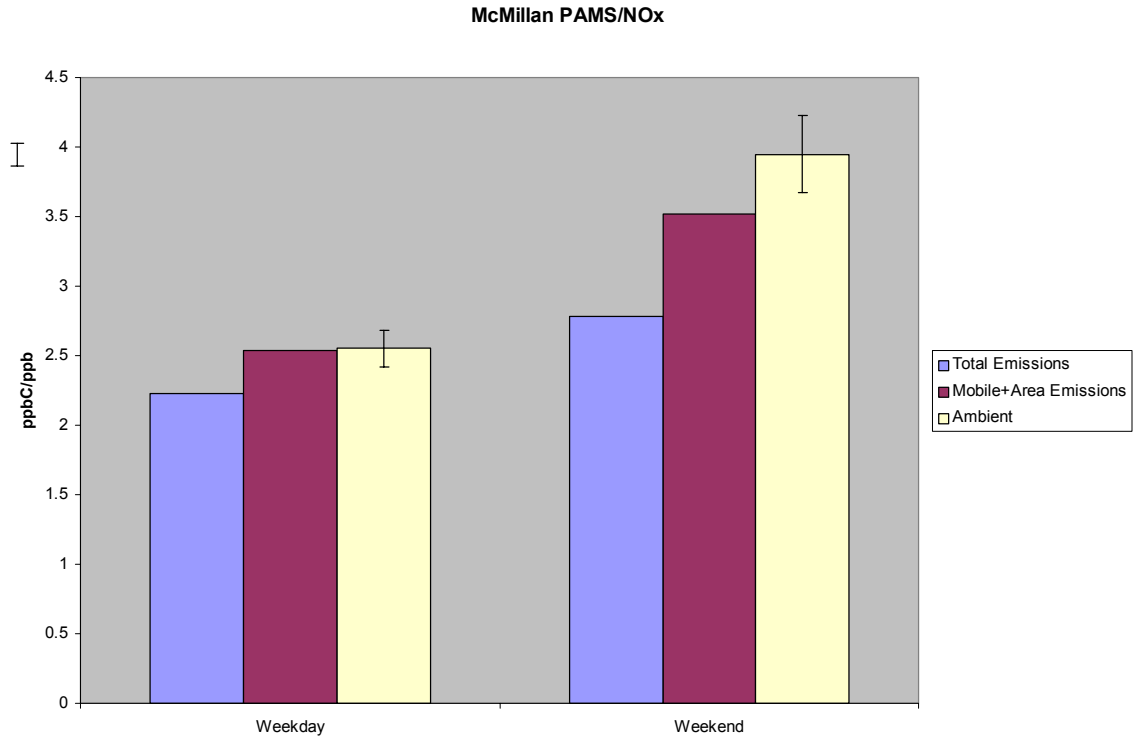


Figure 3-17d. Weekday and weekend morning PAMS/NOx ratios: McMillan.

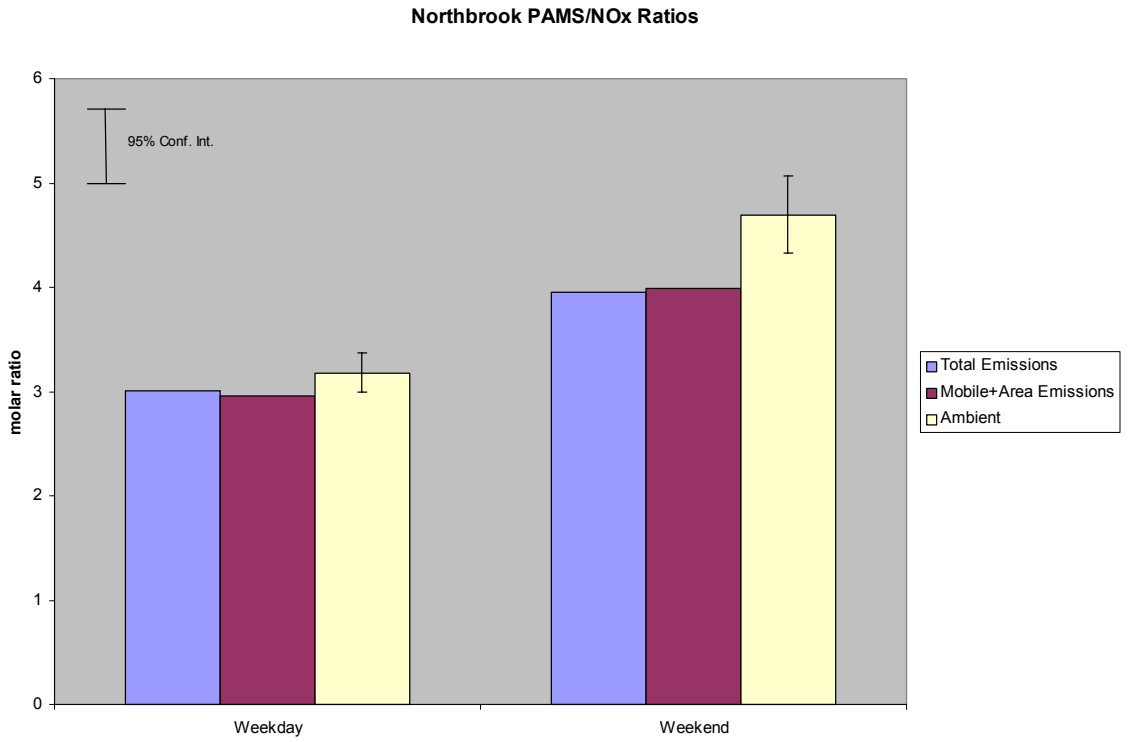


Figure 3-17e. Weekday and weekend morning PAMS/NOx ratios: Northbrook.

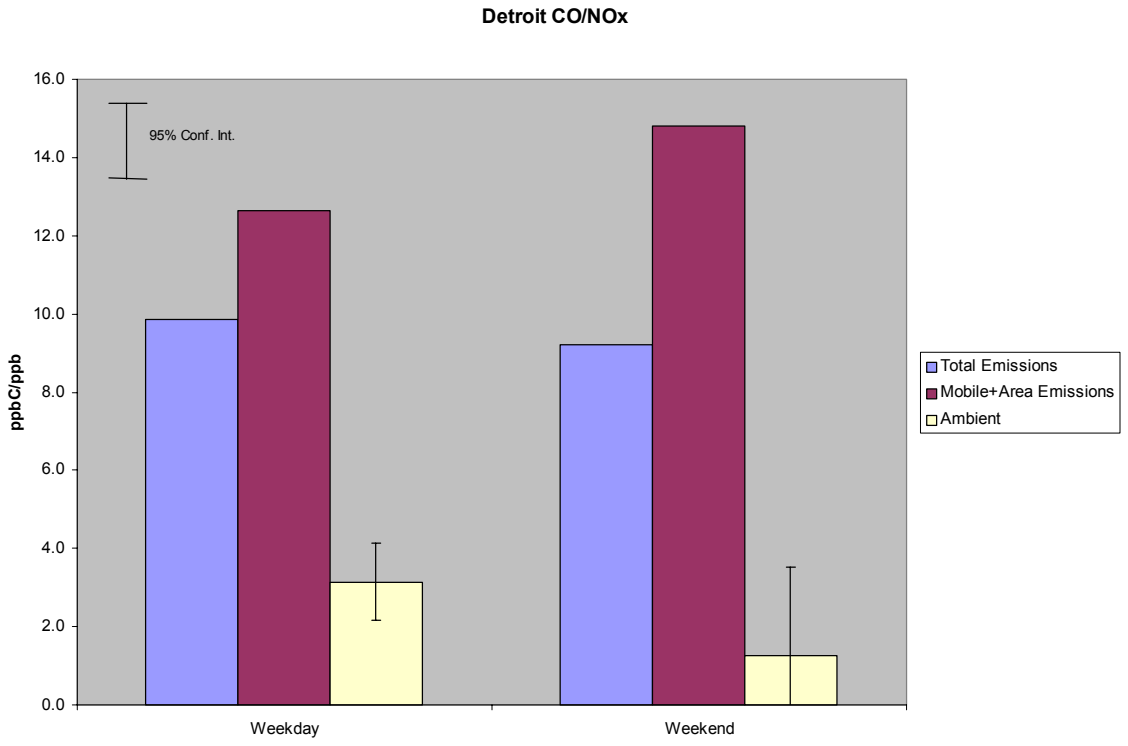


Figure 3-18a. Weekday and weekend morning CO/NOx ratios: Detroit.

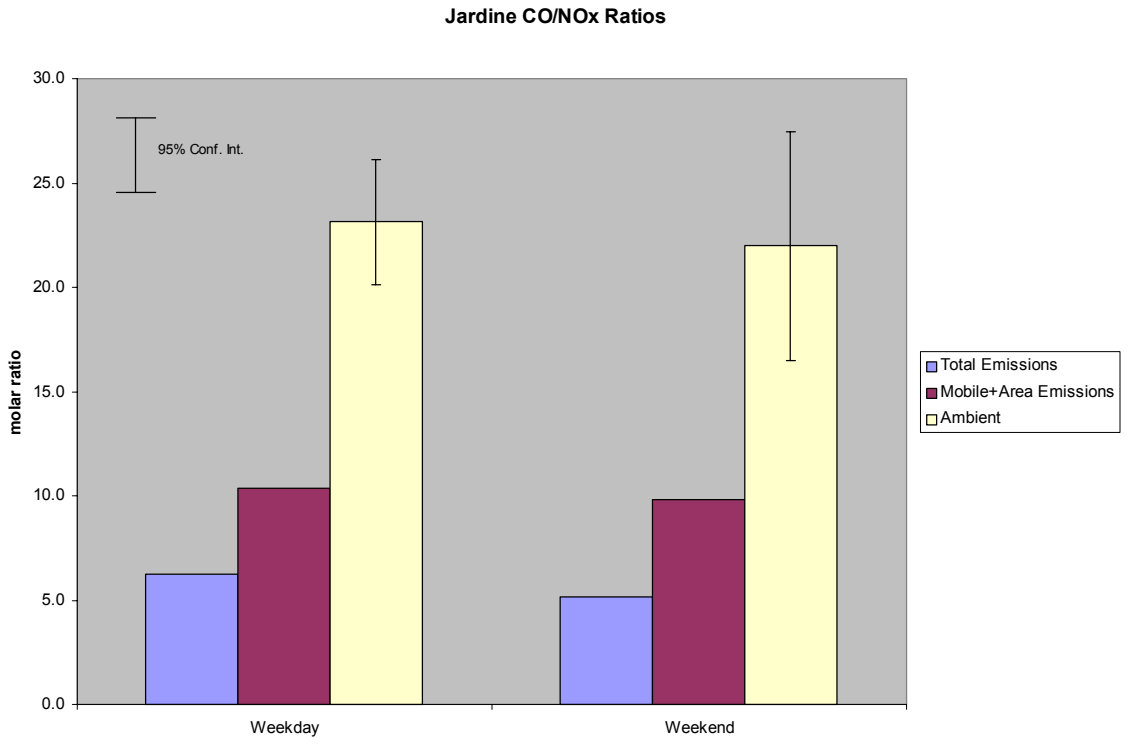


Figure 3-18b. Weekday and weekend morning CO/NOx ratios: Jardine.

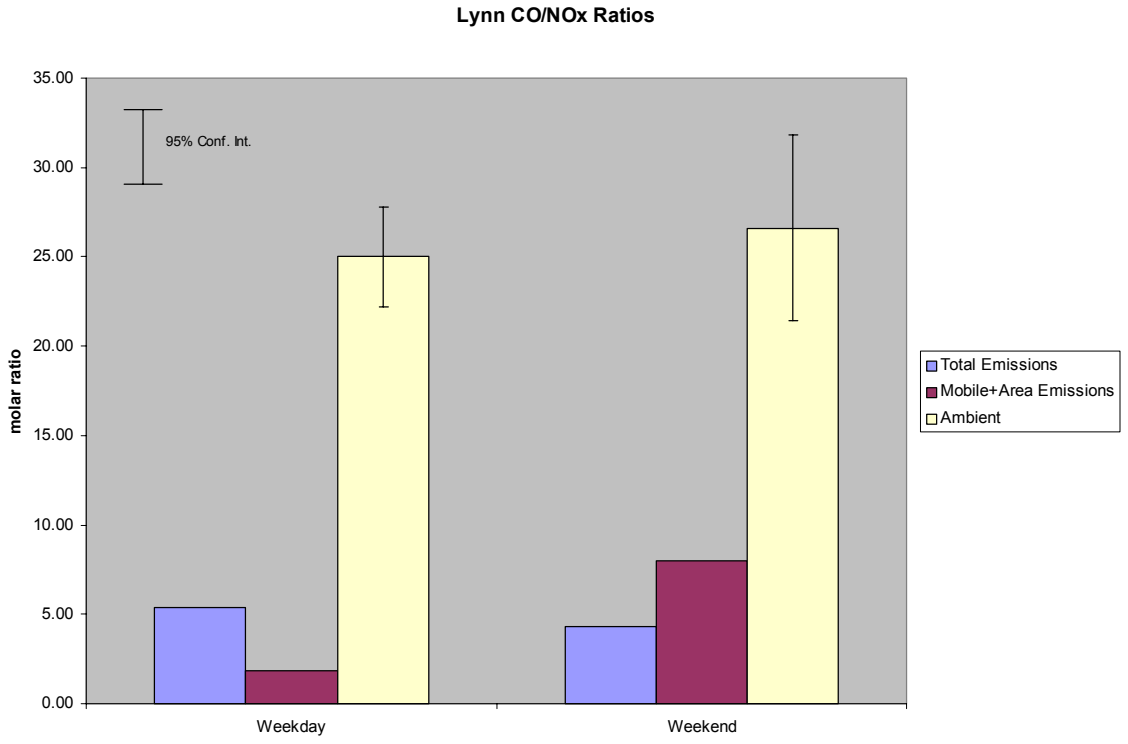


Figure 3-18c. Weekday and weekend morning CO/NOx ratios: Lynn.

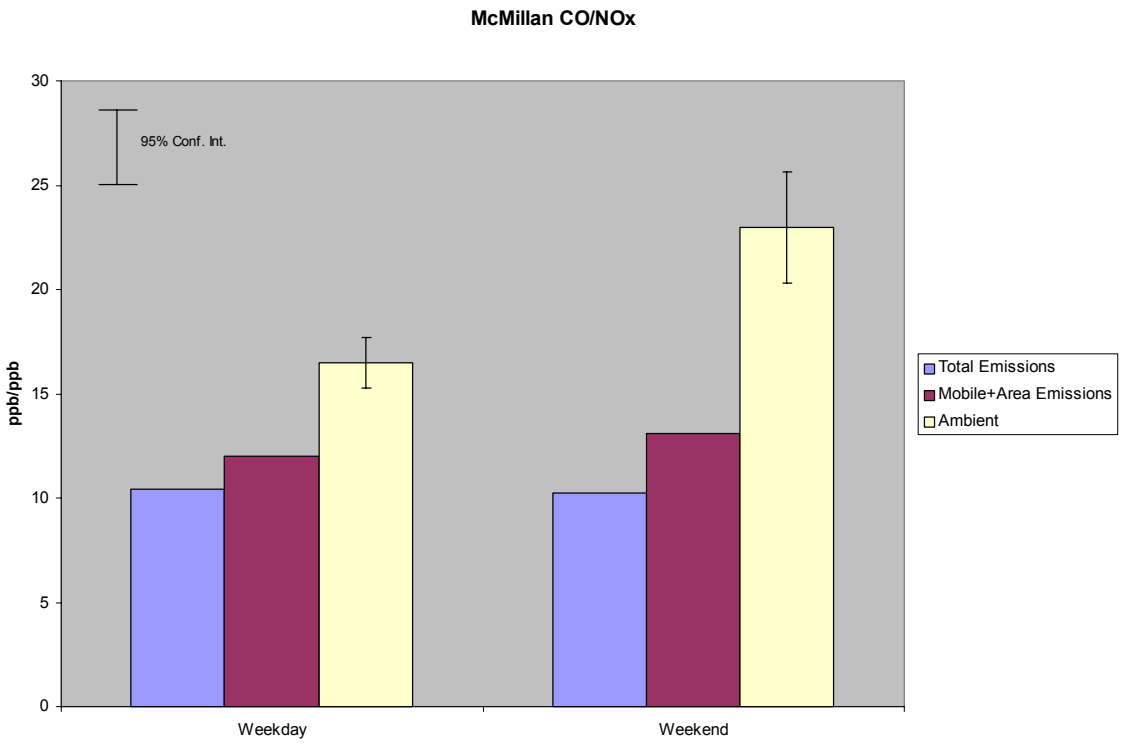


Figure 3-18d. Weekday and weekend morning CO/NOx ratios: McMillan.

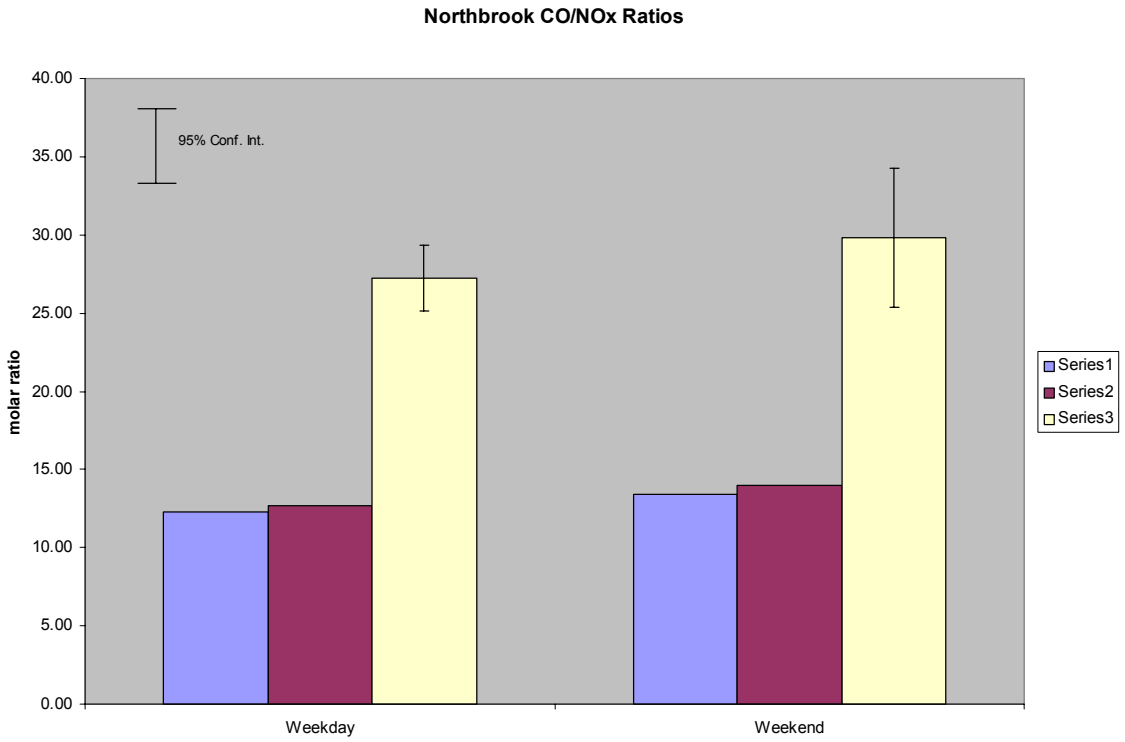


Figure 3-18e. Weekday and weekend morning CO/NOx ratios: Northbrook.

Ambient PAMS/NOx ratios are significantly higher on weekends than on weekdays at all sites except Detroit*. Inventory ratios for mobile plus area sources are also higher on weekends, reflecting the decrease in HDV activity on weekend mornings. At McMillan and Northbrook, the increase in emission ratios on the weekend is not as large as the increase in the ambient ratio, resulting in ratio of ratios greater than 1:1 on the weekends. In Detroit, ambient PAMS/NOx ratios are the same on weekends as on weekdays and the total inventory ratio is also largely unchanged despite the increase in the ratio for mobile+area sources. Examination of the inventory indicated that this is due to a large reduction in point source VOC emissions on weekends coupled with almost no change in point source NOx, thus resulting in a much smaller VOC/NOx ratio for point sources on weekends.

Ambient CO/NOx ratios are similar on weekends and weekdays at all sites except McMillan where weekend ratios are higher. There is also little change in emission ratios between weekdays and weekends at all sites. One possible explanation for the weekend effect in the ambient ratios at McMillan could be a change in local traffic patterns near the CO monitor. Examination of the inventories revealed that the expected weekend increase in mobile source CO/NOx ratios (due to reduced HDV activity) is offset by relatively large decreases in area source CO/NOx ratios. This is somewhat surprising and suggests that day-of-week adjustments to area source categories require careful scrutiny.

SUMMARY AND CONCLUSIONS

Summary Of Results

Weekday morning ratios of ambient PAMS/NOx and CO/NOx to inventory PAMS/NOx and CO/NOx, respectively ("ratio of ratios") are summarized in Table 3-13.

Table 3-13. Ratio of average ambient ratios to inventory ratios (ratio of ratios) for weekday mornings, all quadrants and subgrid lengths.

	PAMS/NOx	CO/NOx
Detroit	1.26	0.33
Jardine	1.16	3.70
Lynn	2.37	4.65
McMillan	1.15	1.58
Northbrook	1.06	2.22

Ambient and inventory PAMS/NOx ratios agree reasonably well at Jardine, McMillan, and Northbrook. Ambient PAMS/NOx ratios exceed inventory ratios somewhat at Detroit and much more so at Lynn. In no case is the ratio of ratios less than 1:1. The discrepancy at Detroit may be at least partially due to underestimation of ambient NOx in the regression

*Comparisons of the weekday and weekend 95% confidence intervals shown by the error bars in Figure 3-15 indicate that the "weekend effect" for PAMS/NOx ratios is statistically significant at Lynn, McMillan, and Northbrook. The weekend effect for CO/NOx ratios is statistically significant only at McMillan.

model used at this site as previously described. As noted in Figure 3-12, PAMS/NOx emission ratios at Lynn are lower than at the other sites, whereas the ambient ratios are roughly the same as at the other sites. Since mobile source PAMS/NOx emission ratios at Lynn are on par with those at the other sites, this result suggests that the area and/or point source PAMS/NOx emissions ratio at Lynn is too low. This is consistent with either an underestimation of VOC emissions or an overestimation of NOx emissions from these sources (or both).

CO/NOx ambient ratios exceed corresponding emissions ratios by a wide margin at Jardine, Lynn, and Northbrook, but much less so at McMillan. For Detroit, the relationship is reversed: the ambient CO/NOx ratio is less than one-third of the inventory ratio. Since CO was not measured at any of the PAMS monitoring sites, these CO/NOx results are based on CO measurements from the closest available monitoring locations. As previously discussed, the different monitoring locations may be partially or wholly responsible for the differences between ambient and inventory CO/NOx ratios, especially at Detroit, Northbrook, and Lynn. At Jardine and McMillan, the PAMS and CO monitors are relatively close together although the exposure to CO sources may still be different. Point and/or area source emission ratios at Lynn appear to be in error as noted in the discussion of PAMS/NOx ratios above. Taken together, one cannot conclude from these results that there is necessarily a problem with CO/NOx ratios in the inventory in general or mobile sources in particular. There is no evidence in these results of an overestimate of CO relative to NOx by MOBILE6 as has been suggested by a recent analysis of tunnel studies (Tran et al., 2002) and remote sensing data (Stoeckenius and Tran, 2003), but these results by themselves cannot be used to rule out this possibility.

Comparisons of ambient to inventory ratios on weekends reveal that ambient PAMS/NOx ratios on weekends exceed the inventory ratio to a greater extent than on weekdays because the weekend increase in ambient ratios is only partially matched by the weekend increase in the inventory ratios (see Table 3-14). The weekend morning increase in ambient PAMS/NOx is due to a decrease in NOx, consistent with results from other studies (Pun et al., 2001). Thus, adjustments to the emissions inventory on weekends either decrease VOCs too much or do not decrease NOx enough. Significant reductions in on-road mobile source NOx emissions on weekend mornings associated with decreased HD, and to a lesser extent LD, vehicle activity were included in the inventory estimates as described above. In contrast, examination of the point and area source NOx emissions shows weekend morning levels are estimated to be almost equal to those on weekday mornings. Further analysis of weekend vs. weekday activity levels for all source categories will be needed to better estimate weekend emissions.

Table 3-14. Ratio of ambient PAMS/NOx to inventory PAMS/NOx (ratio of ratios) for weekday and weekend mornings, all quadrants and subgrid lengths.

Site	Weekday	Weekend
Detroit	1.26	1.39
Jardine	1.16	1.46
Lynn	2.37	3.64
McMillan	1.15	1.42
Northbrook	1.06	1.19

Uncertainties

Results presented above are subject to numerous sources of uncertainty. Of particular note is the assumption that PAMS species account for 67% of the reported VOC emissions. If we assume an uncertainty range for this value of 50 – 85%, then the PAMS/NOx ratio of ratios reported above would change by at most ± 25% which is not a large difference given the other uncertainties involved in making these sorts of ambient/inventory comparisons.

Another source of uncertainty is the contribution of background sources to the ambient measurements, particularly for CO. For example, at Northbrook, where the average CO concentration for hours meeting the screening criteria used in this study is around 1 ppm, correcting for background CO reduces the mean CO/NOx ratio by about 33%. With this across-the-board adjustment, the average weekday ambient ratio is reduced to 18.2 but this still exceeds the emissions ratio by a factor of nearly 1.5. At McMillan, minimum CO levels are lower, possibly indicating a lower background level. Assuming a background of 200 ppb CO for this site, the mean weekday morning ambient CO/NOx ratio drops from 16.5 to 10.6 which is very close to the emissions ratio for all sources (10.4). Recall that the McMillan PAMS and CO monitors are located much closer together (3 km) than at Northbrook (20 km), suggesting that the McMillan CO/NOx ratio comparison is subject to less uncertainty. Another source of uncertainty to note in the CO/NOx analysis is that ambient CO measurements are only reported to the nearest 100 ppb, but the impact (if any) of this lack of precision on the ratio comparisons is not known.

Other sources of uncertainties in ambient to inventory ratio comparisons include:

- Comparisons of ambient to inventory ratios can also be affected by differences in the relative reactivity of NOx, CO and different VOC species. Reaction rates of these species are sensitive to ambient ozone and hydroxyl radical mixing ratios and temperature. However, VOC reactivity and loss of NOx to NOz should be minimal during the early morning, high emission periods focused on in this analysis and the ambient NOx measurements are typically biased high because some of the NOz is included in the reported NOx. Reaction rates for CO are much slower.

- Air parcels sampled at the monitoring site may represent a different source mixture than is contained in the area-wide average emission inventory. This is particularly important for NOx emissions from elevated sources such as industrial and utility boilers since the extent to which smoke stack plumes mix to the ground at the monitoring site is highly variable. This is why our ambient/inventory comparisons were made both with and without point sources included.

- Errors may occur in the ambient measurements due to concentrations below instrumentation detection limits, calibration errors, etc. We sought to minimize problems with detection limits by restricting the ambient samples analyzed to those with PAMS mixing ratios greater than 50 ppbC and NOx greater than 10 ppb.

Recommendations

Results from this study demonstrate the value of ambient/inventory reconciliation analyses and suggest a number of directions for further research, including:

- Prepare comparisons of ambient and inventory HC composition using a fully speciated version of the inventory and the speciated PAMS ambient data, including an analysis of receptor model source contribution estimates. Such comparisons will provide additional insight into the extent to which mobile sources impact each monitoring site, the accuracy of the inventory speciation, and (via a reactivity analysis) the potential implications of any speciation discrepancies for photochemical modeling.

- Perform ambient/inventory reconciliation at additional monitoring sites, especially sites where high resolution CO data are available.

- Perform inventory reconciliation analyses in conjunction with photochemical modeling to better estimate the relative impacts of different sources at the monitoring sites, taking into account transport, dispersion and chemical transformations. Use of a regional-scale model with nested grids would also reduce uncertainties due to background concentrations.

- Examine the sensitivity of the ambient/inventory comparisons to the choice of low concentration thresholds used to filter the ambient data.

- Examine the sensitivity of the ambient/inventory comparisons to the selection of the plume height cutpoint used to define elevated point sources.

4. COMPARISON OF EMISSION RATIOS FROM REMOTE SENSING MEASUREMENTS WITH MOBILE6

INTRODUCTION

Under the CRC E-23 project, remote sensing device (RSD) measurements of vehicle exhaust plumes have been collected over a period of years in Denver (1999 – 2001) and Chicago (1997 – 2000).[1] Each year's measurements were made over a period of a few days with the location and time of year held constant from one year to the next (Pokharel et al., 2001, 2002). These data have been analyzed by Pokharel et al. (*ibid.*) and by Slott (2002). Results of those analyses generally suggest that these multi-year RSD data provide an accurate and consistent portrayal of LDV exhaust emissions for the fleet and driving conditions observed at the monitoring sites.

As part of a series of analyses designed to evaluate EPA's MOBILE6 emission factor model under CRC project E-64, ENVIRON undertook a comparison of the Denver and Chicago RSD data with corresponding vehicle exhaust emission factors predicted by MOBILE6.

RSD measurements represent the ratios of hydrocarbons, carbon monoxide, and nitrogen oxide to carbon dioxide in the vehicle exhaust plume over approximately a one-half second time interval. These measurements, together with a few reasonable assumptions about the combustion process, can be used to determine the percent HC, CO, and NO in the exhaust plume and, from the carbon balance, mass emissions in grams per kg of fuel. MOBILE6, on the other hand, predicts tailpipe emission factors in units of grams/mile. MOBILE6 tailpipe emission factors are calculated by applying various adjustment factors to the basic exhaust emission rates (BER's). BER's contained in the model are derived from direct measurements of HC, CO, and NOx mixing ratios in the tailpipe exhaust of test vehicles. A number of factors must, therefore, be considered when making comparisons between RSD data and MOBILE6:

- Expressing MOBILE6 g/mile factors in g/kg of fuel (or converting RSD emission factors from g/kg to g/mile) requires an estimate of the instantaneous fuel economy which is not available from the RSD data used in this study. Since MOBILE6 only provides fleet average fuel economy figures and the instantaneous vehicle specific fuel economy may vary significantly from the average, it is only possible to compare ratios of mass emission factors rather than the individual factors.

- RSD HC measurements are based on a nondispersive infrared (NDIR) measurement that has been shown to produce a response equal to one-half the equivalent flame ionization detector measurement (Singer et al., 1998). Thus, the RSD %HC values must be doubled and the MOBILE6 runs must specify HC to be output as THC (which is representative of the FID response).

[1] Data collection in previous years at these sites was not performed under the umbrella of Project E-23.

- Emissions are known to vary as a function of vehicle specific power (VSP) which can be reasonably approximated from road grade and vehicle speed and acceleration (Jimenez et al., 1999). Near instantaneous speed and acceleration were determined contemporaneously with the RSD measurements. RSD %NO (and to a lesser extent %HC) are sensitive to VSP; %CO is relatively insensitive to VSP (Slott, 2002). MOBILE6 emission factors are based on facility-specific driving cycles with speed correction factors applied for some but not all facility types. RSD measurements in Denver and Chicago were made on freeway ramps so comparisons here are based on MOBILE6 output for ramps (MOBILE6 does not apply a speed correction factor for ramps). Comparisons of straight RSD data means (i.e., without re-weighting to account for differences in VSP distributions) with MOBILE6 results assume that the VSP effect has been properly accounted for by using output for the appropriate facility type in MOBILE6. It must be recognized, however, that the distribution of VSP observed in the RSD data (in which observations are each limited to a half second of driving at a single location along a single ramp in each city) may be significantly different from the distribution of VSP found in the MOBILE6 ramp driving cycle or the true population distribution of all travel on ramps (which the MOBILE6 ramp cycle attempts to represent). This issue is further considered below.

- RSD data contain measurements of %NO while MOBILE6 reports emission factors for NOx. Since nearly all NOx in the exhaust of LDVs is released as NO, we assumed the MOBILE NOx mass emission factors were equivalent to NO mass emission factors.

DATA

RSD Data

RSD data collected under CRC Project E-23 were downloaded from www.feat.biochem.du.edu/light_duty_vehicles.html for 1999-2001 for the Denver 6th Ave./I-25 site and for 1997-2000 for the Chicago Arlington Heights/Algonquin Rd.&I-290 site. Only the E-23 data from these two sites were used in the present analysis to avoid potential complications arising from use of different instrumentation and analysis protocols prior to the start of the E-23 program. These data sets include registration data from plate matching including vehicle model year and some limited information on vehicle type. Approximately 20,000 – 22,000 successful plate-matched observations were made during each year. These measurements included approximately 15,000 – 18,000 unique vehicles per year with the balance of the observations accounted for by multiple measurements of the same vehicle (Pokharel et al., 2001, 2002). All duplicate observations were removed prior to processing to avoid unequal weighting of vehicles when computing averages. Records with invalid %HC, %CO or %NO were also removed.

RSD %HC values suffer from a bias the origins of which are not fully understood (*ibid.*). The magnitude of this bias is significant relative to the overall mean HC so the HC offsets recommended by Pokharel and co-workers were subtracted from the reported values (see Table 4-1).

Table 4-1. HC offset values (ppm) recommended by Pokharel et al. (2001, 2002).

	1997	1998	1999	2000	2001
Denver			5	60	-50
Chicago	80	120	70	60	

MOBILE6 Modeling Inputs

MOBILE6.2 was run with inputs appropriate to each city and year. The basis for comparison with the RSD data, and thus the modeling results required, is by calendar year, vehicle class, and model year. Emission factors were estimated hourly using hourly temperatures averaged across all RSD sampling days. The daily average absolute humidity was acquired by averaging across all hours of RSD data; day-to-day variations in average humidity were ignored. Temperature and humidity data corresponding to observation times in each city were provided by Bishop (2003). Upon examination of the results, we determined that there was relatively little difference in the hot running exhaust emissions from hour to hour so results were simply averaged over all hours for comparison with the RSD data.

The discussion below is divided into separate sections for Chicago and Denver modeling since the procedures shared little other than the general approach outlined above.

Chicago

RSD data for Chicago encompass the years 1997 through 2000 and were collected in mid-September. Therefore, the general approach was to model both July of the data year and January of the following year and then take the average of the two results. In effect, this allows capture of the proper fleet turnover but care was taken to ensure that seasonal fuel properties are correctly specified as discussed below. Furthermore, Chicago counties were modeled separately from the remaining counties (including the Metro East St. Louis area) with the former modeled with I/M and the latter, without. A weighted average of the I/M and no I/M results was used for comparison with the RSD data based on the proportion of vehicles (by model year group) captured in the RSD data that were registered in each region. It was assumed that vehicles from the Metro East St. Louis area captured in the RSD set will be too few to warrant a full modeling of that area's I/M program.

I/M

For 1997 and 1998, the I/M program was in transition and according to IL EPA, included a simple idle test (not two-speed idle) performed on all four year old or older gasoline vehicles that are of vintage 1968 or newer (except buses and motorcycles). The program start date is 1986 and test frequency is biennial. IL EPA also provided the compliance rate (96% for all vehicles), the stringency (failure) rate (20% for all vehicles), and waiver rates (3% for both pre-1981 and 1981 and newer vehicles).

For 1999 and 2000, the transition was completed and the I/M program targets three model year fleets using different test types. All 1968-1980 gasoline vehicles (except motorcycles and

buses) face an idle test. 1981-1995 vehicles are subject to an IM240 test while 1996+ model years are required to pass an OBD-based I/M. The same compliance, stringency, and waiver rates were applied. All vehicles less than 4 years old are exempted. IM240 cutpoints are 1.20, 20.0, and 2.50 for HC, CO, and NOx, respectively. These data were all provided by IL EPA.

The cutpoints above were used in MOBILE5 modeling. As such, they apply only to 1981-1993 LDGVs. The previous model assumes that 1994+ LDGVs and 1984+ LDGTs "will be inspected using cutpoints that will result in similar reductions as are estimated for the 1981 thru 1993 model year passenger cars." (EPA, 1996) Since MOBILE6 requires actual separate cutpoints for LDGVs and LDGT1/2/3/4, the final cutpoints available from the IL EPA web site had to be used. These are scheduled for February 1, 2001 but closely resemble the previous (2000) values in many cases.

All programs are centralized (test only). Evaporative I/M was not modeled since the evaporative emission factors are irrelevant for purposes of comparison with the RSD data.

RVP

Chicago has had an RFG program in place since 1995. Under this program, summer RVP is 8.1 psi. MOBILE6 fixes the RVP value of summer RFG at 8.0 psi and 6.7 psi for pre-2000 years and 2000, respectively. However, the user input RVP is used for winter RFG. Because 1997-1999 data were collected after September 15, an (assumed) winter value of 13.9 psi was entered for all scenarios except in 2000. In the latter instance, the summer season is more appropriate since all data were collected before September 15. Thus, both scenarios in 2000 were modeled with 8.1 psi as RVP.

For counties outside the RFG area, a value of 8.5 was assumed for summer (i.e., 2000) and 13.9 psi for winter (all other years). The "assumed" values were taken from the 1993/1994 NIPER reports (NIPER, 1994a,b, 1995) for the region. In any case, it is anticipated that these assumptions have small effects on the running exhaust emissions.

Sulfur

According to 1996 data, sulfur content in the Chicago area is relatively high at 490 to 580 ppm.[2] St. Louis also has very high sulfur content (540 ppm).[3] Since RFG regulations limit the sulfur level to 500 ppm, a value of 490 was assumed for all runs, including those counties outside the Chicago RFG area. MOBILE6 fixes the sulfur content of summer RFG and 2000+ winter RFG at values that are significantly lower. Furthermore, all data aside from 2000 were collected after September 15, the end date for summer RFG. Since sulfur content is one of the most important fuel parameters affecting modeled exhaust emissions, it was deemed best that the observed value be used rather than allowing the model to default to the lower values. In practice, this means the fuel for all calendar years besides 2000 was modeled

[2] http://www.epa.gov/otaq/regs/ld-hwy/tier-2/colucci.pdf
[3] http://waw.wardsauto.com/ar/auto_big_auto_vs/

as RFG (North) but with sulfur explicitly specified at 490 ppm. To ensure that this value was used for the July scenarios as well, these runs were forced to be winter via the SEASON command. (Recall that this does not violate the fact that data were indeed collected after September 15.)

For 2000, the RFG was modeled "manually" by explicitly specifying sulfur and oxygenate. The latter specification values follow those assumed for RFG by MOBILE6 (shown in the MOBILE6 User's Guide; EPA, 2002b).

The 'no I/M' counties' fuel was modeled by declaring SULFUR CONTENT explicitly for calendar years before 2000. For 2000, the declaration was made via the FUEL PROGRAM command.

Denver

RSD data for Denver encompass the years 1999 through 2001 and were collected mostly in January. Therefore, the general approach was to model January of the data year. Although, three county groups were modeled separately depending upon whether enhanced I/M, basic I/M, or no I/M is present, only the enhanced I/M model runs were used for comparison with the Denver RSD data since over 90% of the vehicles captured in the Denver RSD data were registered in the enhanced I/M area.

I/M

The enhanced I/M program targets two model year fleets using different test types. All pre-1982 gasoline vehicles (except motorcycles and buses) face a two-speed idle test. 1982+ vehicles are subject to an IM240 test. The same compliance, stringency, and waiver rates were applied. IM240 cutpoints by vehicle class and model year were provided by the Colorado Department of Public Health (CDPHE). (The final cutpoints were used since they took effect January 1, 1999.) Note that 1982+ HDGVs were subject to 2-speed idle testing rather than IM240. However, since these vehicles are not the subject of this investigation, a distinct I/M scenario was not created for them. All programs are centralized (test only). Evaporative I/M was not modeled since the evaporative emission factors are irrelevant for purposes of comparison with the RSD data.

RVP and Oxygenate

The I/M counties also have an oxygenated fuel program with a wintertime RVP of 12.0 psi (obtained from the CDPHE). According to EPA, ether is the oxygenate – blended at 2.7 weight percent.

Sulfur

According to various sources, sulfur content in the Denver area is about 200 ppm which is lower than the national average.[4] For 1999, sulfur level was explicitly fixed at 200 ppm via the SULFUR CONTENT command. For calendar years 2000 and 2001, the fuel sulfur content was again fixed at 200 ppm via the FUEL PROGRAM command.

Vehicle Classification

A significant drawback of the RSD data is a lack of sufficient information to unequivocally assign each vehicle to a MOBILE6 vehicle type. Some information on vehicle type was generated via plate matching as recorded via the LIC_TYPE field; a breakdown of observation counts by LIC_TYPE code is shown in Table 4-2. Bishop (2003) has done some decoding of the vehicle identification numbers (VINs) in the Denver 2001 and Chicago 1999 datasets sufficient to allow him to separate vehicles subject to the passenger car certification standards from those subject to the LDT standards. Table 4-2 compares these classifications with the LIC_TYPE codes for the Denver 2001 data. These results provide an indication of the car/truck breakdown by LIC_TYPE code likely to be found in the 1999 and 2000 data. Vehicles designated LTK are overwhelmingly light trucks, whereas vehicles designated PAS are a mixture of cars and light trucks. Other LIC_TYPE codes occur much less frequently and are not of interest in the present study (e.g., BUS for buses and M/C for motorcycles) so these vehicles were not included in our analysis.

Table 4-2. Vehicle counts by VIN decoding results and LIC_TYPE field: Denver, 2001.

	VIN Decoding		
LIC_TYPE	**Unknown**	**Car**	**Truck**
27E	0	0	1
BUS	20	0	5
FTK	0	0	31
GVW	7	0	13
LTK	147	12	4269
M/C	4	0	0
MTH	12	0	1
PAS	524	10292	5588
RTK	4	0	107
SME	2	0	0
SMM	1	0	0
SVW	2	0	0

Table 4-3 shows vehicle counts by LIC_TYPE and VIN decoded classification for the 1999 Chicago dataset. The LIC_TYPE field in these data include only two codes (1 and 0) the exact meaning of which are not known although we surmise from the VIN decoding results that 0 designates commercial plates while 1 designates private. As in the Denver data, the Chicago data include a vehicle body style field but for most vehicles this is not sufficient to

[4] http://www.msnbc.com/local/RMN/DRMN_1522463.asp

distinguish between cars and light trucks. For example, there are 497 vehicles with body style identified as STN WAGON (presumably station wagons) but the VIN decoding shows that 50 of these are classified as trucks. There are 51 unique body style classifications in the four years of Chicago data. Classifications based on VIN decoding are likely to be more reliable than the body style designations so we did not rely on the body style information in our analysis.

Table 4-3. Vehicle counts by VIN decoding results and LIC_TYPE field: Chicago, 1999.

	VIN Decoding		
LIC_TYPE	**Unknown**	**Car**	**Truck**
0	74	687 (21%)	2541 (77%)
1	141	15209 (77%)	4436 (22%)

Unlike the Denver data, the LIC_TYPE field in the Chicago data do not provide a category consisting of almost all trucks. We nevertheless chose to segregate the Chicago data by LIC_TYPE to provide some indication of car/truck differences. As with the Denver data, comparisons with MOBILE6 results were made on a weighted average basis as described below.

Since the number of HDTs (over 8,500 lbs GVWR) included in the RSD data are likely very small (if any), vehicles identified via the VIN decoding as trucks were assumed to correspond to the MOBILE6 vehicle categories LDGT1-4 (gas powered) and LDDT12 and LDDT34 (diesel powered). The Denver data for 2000 and 2001 include fuel type and GVW information but the Denver 1999 and the Chicago data (all years) do not. To put all of the data on an equal footing to the greatest possible extent, we assumed that vehicles in the Denver dataset with LIC_TYPE equal to LTK, RTK and FTK with FUEL_TYPE equal to G correspond to MOBILE6 types LDGT1-4, those with FUEL_TYPE equal to D correspond to MOBILE6 types LDDT12 and LDDT34. Vehicles with LIC_TYPE equal to PAS were assumed to be a mix of gas powered cars and light trucks with the car/truck splits by vehicle age group determined from the 2001 VIN decoding results. A similar approach was used for analyzing the Chicago data where separate car/truck splits for each LIC_TYPE and vehicle age group were determined from the 1999 VIN decoding results. Since gas/diesel splits were not available for Chicago, we assumed all cars were gas powered and the LDT gas/diesel splits were the same as in Denver.

RSD Data Processing

Vehicles were grouped by age into four bins of five years each as shown in Table 4-4.

Table 4-4. Model year ranges included in each age group.

	RSD Observation Year				
Age Group (years)	**1997**	**1998**	**1999**	**2000**	**2001**
1 – 5	1992-1996	1993-1997	1994-1998	1995-1999	1996-2000
6 – 10	1987-1991	1988-1992	1989-1993	1990-1994	1991-1995
11 – 15	1982-1986	1983-1987	1984-1988	1985-1989	1986-1990
16 – 20	1976-1981	1978-1982	1979-1983	1980-1984	1981-1985

Mean THC, CO, and NO emission factors in g/gal were computed for each age bin for each RSD observation year for three different vehicle type groupings:

1. Vehicles grouped by LIC_TYPE as described above. Denver data were designated either PAS or TRK; Chicago were designated either type 0 (COMMERCIAL) or type 1 (PRIVATE). These groupings were available for all years of observations in both cities.
2. Vehicles classified as CAR or TRUCK based on VIN decoding performed by Bishop (2003). This classification was available only for the 2001 Denver and 1999 Chicago data.
3. Vehicles classified by fuel type (GAS or DIESEL). This classification was available only for the 2000 and 2001 Denver data.

Each of the vehicle type groupings listed above were mapped to the MOBILE6 vehicle types as described above and summarized in Table 4-5. We assumed all vehicles in the RSD data were LD.

Table 4-5. Mapping of vehicle types in RSD data to MOBILE6 vehicle type categories.

RSD Vehicle Type	MOBILE6 Vehicle Type(s)
PAS, PRIVATE, or COMMERCIAL	Mixture of LDGV and LDDT and LDGT
TRK	Mixture of LDDT and LDGT
CAR	LDGV
TRUCK	Mixture of LDDT and LDGT
GAS	LDGV and LDGT
DIESEL	LDDT

Emission factors from MOBILE6 were averaged over the corresponding vehicle types as indicated in Table 4-5. Weighted averages were computed with combined gas/diesel and car/truck splits by model year used to determine the weights within each of the model year ranges listed in Table 4-4. Weights for the PAS and TRK categories (PRIVATE and COMMERCIAL in Chicago) were determined from the 2001 (for Denver) and 1999 (for Chicago) VIN-decoding results: these were assumed to apply to the other RSD years. No fuel type splits were available for Chicago so the Denver splits were assumed to apply.

Weighting factors for the vehicle subtypes used by MOBILE6 (e.g., LDGT 1,2,3,4) were based on the MOBILE6 default fleet mix. Use of these default weights had little impact on the resulting averages, however, because the major differences in emissions were accounted for by the LDV vs. LDT and gas vs. diesel weights which were determined directly from the RSD data as described above.

I/M program enrollment status for vehicles in the RSD data was determined on the basis of the county of registration included in the RSD data files. In Denver, over 90% of vehicles captured in the RSD data are registered in the enhanced I/M program area so the RSD results were compared with just the enhanced I/M MOBILE6 runs. In Chicago, approximately 80 - 85% of vehicles in the RSD data are registered within the I/M area; a weighted average of the I/M and no I/M MOBILE6 results was used with weights determined separately for each model year group.

RESULTS

Analysis Of CO/NO Ratios

CO/NO and HC/NO mass emission ratios from the RSD data were compared with corresponding ratios from MOBILE6 for the various vehicle type groupings described above. Results are presented for Denver and Chicago in the following subsections.

Denver

Figure 4-1 compares CO/NO mass ratios from the Denver 2001 RSD data for vehicles classified as CARS or TRUCKS based on Bishop's VIN decoding with corresponding MOBILE6 emission factor ratios for LDGV and LDT, respectively. For LDTs, the MOBILE6 ratios exceed the RSD ratios by factors ranging from 1.5:1 to 4:1. The RSD data show a decrease in CO/NO from over 10:1 for the oldest vehicles to just under 4:1 for the newest vehicles; the MOBILE6 ratios decline much more modestly. For cars, MOBILE6 shows an increase in CO/NO ratio with decreasing age that is not seen in the RSD data. This results in MOBILE6 CO/NO ratios for the newest cars that are almost 3.5 times as large as ratios observed by the RSD. Since relatively few diesel vehicles are included in the RSD data, these results are representative of spark ignition vehicles; removing the diesel vehicles from the comparison was found to have little effect on the comparison.

RSD and MOBILE6 CO/NO ratios were also compared by vehicle age for vehicles classified as PAS or TRK. This allows us to take advantage of all years of RSD data, since the PAS/TRK classification is the only classification available in each year. As suggested by an earlier analysis of Denver RSD data (Slott, 2001), emissions are comparable between RSD measurement years for the same vehicle age group. An example of this for CO in Denver is shown in Figure 4-2. Results for NO and HC are similar. In addition, the Denver MOBILE6 results also show very consistent predictions of CO, NO, and HC with vehicle age across the three modeling years. This allows us to compare CO/NO (and HC/NO) ratios averaged over all three modeling years by vehicle age as shown in Figure 4-3. These results are similar to the CAR/TRUCK comparison from the 2001 data shown in Figure 4-1 with the PAS vehicles similar to those classified via the VIN decoding as CAR and the TRK vehicles similar to those classified via the VIN decoding as TRUCK.

Further examination of the results described above shows that MOBILE6 CO/NO ratios exceed the RSD ratios to a much greater degree for the newest vehicles because the RSD results show a greater decrease in CO relative to NO with decreasing age than does MOBILE6. In fact, MOBILE6 shows CO *increasing* relative to NO with decreasing age for PAS vehicles. This discrepancy is due to a smaller difference between older and newer vehicles in the MOBILE6 CO emissions than is seen in the RSD data. The CO emission factor trend is illustrated in Figure 4-4 which compares the change with vehicle age of MOBILE6 CO g/mile emission factors (left y-axis) vs. Denver RSD data g/gal emission

factors (right y-axis). In contrast, the relative rate of change with vehicle age in MOBILE6 NO emission factors is approximately the same as seen in the RSD data (Figure 4-5).[5]

The comparisons of CO and NO changes with vehicle age between MOBILE6 and the RSD data described above ignore trends in fuel economy with model year. Updated average fuel economy estimates by model year for LDV's and LDT's were compiled by EPA during development of MOBILE6.3. Summaries of these estimates (Landman, 2002, Appendix G) show that between 1979 (the oldest model year included in the 16-20 year age bracket) and the 2000 model year, average fuel economy improved by roughly 40% for both LDV's and LDT's. However, LDV's average 25% better fuel economy than LDT's and the LDT sales fraction is higher in more recent model years, offsetting the fuel economy gains. Furthermore, fleet average fuel economy for LDV's and LDT's is nearly unchanged between the 1984 model year (representing 15 year old vehicles in 1999) and 2000. Taking weighted averages of these fuel economy estimates with weights based on the LDV/LDT mix by model year for the PAS and TRK vehicle types in the Denver 2001 RSD data and averaging by model year group produces the results shown in Table 4-6. Since the average fuel economy changes by less than 10% between model year groups, the impact of fuel economy changes on the CO and NO mass emission factor trends shown in Figures 4-4 and 4-5 is negligible.

Table 4-6. Average miles/gallon by model year group based on Denver 2001 PAS and TK fleet mix (MPG figures from Landman, 2002, App. G)

	1980+thru1985	1985+thru1990	1990+thru1995	1995+thru2000
pas.mpg	20.2	22.0	22.4	22.3
trk.mpg	17.8	18.0	17.7	17.5

Chicago

Comparisons of RSD with MOBILE6 CO/NO ratios for PRIVATE and COMMERCIAL license types in Chicago (see Figure 4-6) produced results very similar to that for PAS and TRK vehicle types in Denver. This is not surprising when we recall that the vehicles in the 1999 Chicago dataset with private license type (LIC_TYPE = 1) consisted of 77% cars (based on VIN decoding of the 1999 data) while the vehicles with commercial license type (LIC_TYPE = 0) consisted of 77% trucks (again based on VIN decoding of the 1999 data).

Variations in CO by RSD measurement year are slightly greater in Chicago (see Figure 4-7) than in Denver: the values in 2000 stand somewhat apart from those of the other three years. Results for HC and NO are similar with respect to inter-year differences.

More significantly, changing fuel RVP and temperature parameters in Chicago caused MOBILE6 to predict significantly different CO for the same age vehicles in each of the four modeling years in Chicago (see Figure 4-8). These year-to-year differences are fairly constant with vehicle age, are not supported by the RSD data and confound inter-year comparisons of RSD and MOBILE6 results. This is illustrated in Figure 4-9 by comparing the solid lines (representing RSD data in g/gal) and dashed lines (representing MOBILE6 g/mile results

[5] Error bars shown in these figures represent 95% confidence intervals for the mean assuming the means are normally distributed. From the central limit theorem, we know this will be approximately true for all but the smallest sample sizes.

scaled to the 1997 RSD g/gal values).[6] For each fixed model year group, there is little year-to-year difference in mean CO emission in the RSD data (ignoring 1977 – 1981 model year vehicles for which very few observations are available in the 1999 and 2000 datasets). In contrast, MOBILE6 results (as depicted by the dashed lines) exhibit significant decreases in CO for these vehicles between 1997 and 2001, despite the fact that the vehicles are getting older. MOBILE6 is not predicting that CO emissions decrease with vehicle age but rather that the deterioration effect is overwhelmed by the impact of RVP and temperature adjustments.[7]

A similar analysis for NO (see Figure 4-10) shows generally better agreement in deterioration between the RSD data and MOBILE6 results, especially for the newest vehicles.[8] However, MOBILE6 predicts that 2000 NO emissions should have been less than 1998 emissions by an amount greater than that seen in the RSD data for the two midrange model year groups.

Analysis of HC/NO Ratios

Denver

HC/NO ratios in the RSD data and MOBILE6 results were analyzed in a manner analogous to CO/NO ratios described above. Figure 4-11 compares HC/NO ratios from the RSD data with ratios from MOBILE6 for PAS and TRK vehicles. Ratios for both PAS and TRK vehicles exhibit similar trends with vehicle age in both the RSD and MOBILE6 results. PAS HC/NO ratios are of comparable magnitude while MOBILE6 ratios for TRK are consistently higher than the RSD ratios. Results (not shown) are similar when vehicles from the 2001 data are classified as cars vs. trucks via the VIN decoding, with the PAS vehicles behaving like the cars and the TRK vehicles like the trucks (as we would expect).

Dependence of HC emission factors on vehicle age in the Denver RSD data and in MOBILE6 are shown in Figure 4-12.[8] Trends with vehicle age are similar for TRK vehicles with the exception of a sharper decline in the newest TRK vehicles in the RSD data. The rate of change in HC emissions with vehicle age for PAS vehicles is nearly constant over the full age range in the RSD data whereas the MOBILE6 estimates show a greater rate of increase as the vehicles get older.

[6] Error bars shown in this figure represent 95% confidence intervals for the mean calculated under the assumption of normally distributed means as in Figure 3-4. Confidence intervals are not shown for the oldest (1977-1981) model year group because these intervals are expected to be very wide given the small sample sizes in the later measurement years and the normal theory is not likely to hold.

[7] We also examined the influence of changes in the Chicago area I/M program over the years on the MOBILE6 predictions but found that these did not account for the decreases in CO emissions.

[8] Error bars shown in these figures represent 95% confidence intervals for the mean assuming the means are normally distributed. From the central limit theorem, we know this will be approximately true for all but the smallest sample sizes.

Chicago

Comparisons of HC/NO ratios from the Chicago RSD data (with vehicles classified by PRIVATE vs. COMMERCIAL license types) are very similar to the Denver results (Figure 4-13) although the HC/NO ratio for 11-15 year old commercial vehicles does not follow the expected trend with vehicle age. This may have to do with the fact that there are very few commercial vehicles in the two oldest age groups in the 1997 and 1998 Chicago data. It is also useful to note that the HC/NO ratios within each vehicle age bin are far less consistent from one RSD data year to the next for commercial vehicles in Chicago (and to a lesser extent for TRK vehicles in Denver) than is the case with private/passenger vehicles or with CO/NO ratios for either vehicle class. This is likely the result of smaller sample size along with greater variability in the HC data and adds to uncertainties in the HC/NO comparisons.

The RSD data show little difference between measurement years in HC emissions for the same age vehicles with the exception of 16 – 20 year old vehicles (and to a lesser extent for 11 – 20 year old vehicles). In contrast, MOBILE6 predicts consistent year-to-year differences in HC emissions for the same age vehicles with the arithmetic (and multiplicative) differences being much greater for the older vehicles. Deterioration of HC emissions is shown in Figure 4-14.[9] An increase in HC emissions with vehicle age is evident in the RSD data for the two newest model year groups. The two older groups show more variability, most likely related to limitations in the sample size (especially for the oldest vehicles). In contrast, MOBILE6 predicts decreases in HC emissions between the first and last measurement years for each model year group. As noted above in the case of CO, this is the result of temperature and fuel RVP correction factors applied by MOBILE6 in each year modeled. These corrections do not appear to be consistent with the RSD data and overwhelm the impact of MOBILE6 deterioration factors.

Influence of Vehicle Specific Power

Both g/gal and g/mi emission factors are known to vary as a function of vehicle specific power (Pokharel et al., 2001, 2002). It is unlikely that the VSP distribution varies significantly with vehicle age or measurement year in the RSD data although Pokharel et al. (2002) report increased congestion at the Denver site during 2001 which had some impact on VSP. Of potentially greater significance are differences between the distribution of VSP observed at the RSD sites and the distribution of VSP in the ramp driving cycle used to develop the MOBILE6 basic exhaust emission rates. Figures 4-15 and 4-16 compare these VSP distributions. These comparisons show that the MOBILE6 distribution includes significantly more negative VSP than appears in the RSD data and the RSD data include some high VSP's ($> = 28$ kW/tonne) that are not found in the ramp cycle. The difference in negative VSP frequencies is not surprising since the RSD sites were specifically chosen to capture vehicles during positive VSP operation. For VSP > 0, the RSD data show a sharper peak in the distribution as compared to the MOBILE6 ramp cycle. Again, this is not surprising given that the RSD data are

[9] Error bars shown in this figure represent 95% confidence intervals for the mean calculated under the assumption of normally distributed means as in Figure 3-4. Confidence intervals are not shown for the oldest (1977-1981) model year group because these intervals are expected to be very wide given the small sample sizes in the later measurement years and the normal theory is not likely to hold.

collected at a fixed location along the ramp. These differences in the VSP distributions can be expected to impact CO/NOx and HC/NOx ratios. To better understand the potential impact, we computed mean g/gal emission factors for each VSP bin in the Denver and Chicago 2000 data sets. Since the number of RSD observations in bins 12, 13, and 14 (VSP > 28 kW/tonne) were quite small, these last three bins were lumped together. Results for Denver are illustrated in Figure 4-17; results for Chicago are shown in Figure 4-18. CO and HC emissions decrease with increasing VSP below approximately 15 kW/tonne and increase at higher VSP's. NOx emissions increase for increasing positive VSP's up to 23 kW/tonne at which point they start to decrease, possibly as a result of increased prevalence of commanded enrichment (the expected concomitant increase in CO is seen in the Chicago data but not in Denver).

Mean emission by VSP bin described above were used to estimate mean emission factors under the VSP distribution corresponding to the MOBILE6 ramp cycle by computing a weighted average of the bin mean factors using the ramp cycle VSP distribution for the bin weights. Resulting ramp cycle weighted emissions are compared with the straight data averages in Table 4-7.

Table 4-7. Average emissions under actual VSP distribution observed at RSD data collection sites and averages adjusted to reflect VSP distribution in the MOBILE6 ramp driving cycle.

Emissions (g/gal)	Denver (2000)			Chicago (2000)		
	Data Mean	Ramp Cycle Wtd. Mean	Δ %	Data Mean	Ramp Cycle Wtd. Mean	Δ %
CO	132	172	30%	100	111	11%
HC	5.51	8.60	56%	5.60	8.02	43%
NO	19.0	15.4	-19%	13.2	11.2	-15%
CO/NO	6.94	11.2	61%	7.60	9.97	31%
HC/NO	0.290	0.560	93%	0.423	0.718	70%

As a result of the higher frequency of negative VSP's in the ramp cycle, the ramp cycle weighted mean CO and HC is higher than the observed average while the NO is lower. This results in significantly higher CO/NO and HC/NO ratios based on the ramp VSP adjusted means.

Increasing the RSD CO/NO ratios across the board by 61% in Denver changes the degree to which MOBILE6 ratios exceed the RSD ratios originally shown in Figure 4-3 as illustrated in Figure 4-19. Although there is better agreement for 11 – 15 year old vehicles, MOBILE6 ratios for 1 – 5 year old PAS vehicles still exceed the VSP adjusted RSD ratios by a factor of two. Similarly, in Chicago the MOBILE6 CO/NO ratio for 1 – 5 year old PRIVATE vehicles exceeds the VSP adjusted RSD ratio by a factor of 2.8. We also examined the effect of the VSP adjustment by vehicle age to see if the difference between MOBILE6 and the RSD data in the dependence of CO/NO ratio on vehicle age is impacted by the VSP adjustment. It was necessary to do this for all PAS and TK vehicles in Denver combined to insure sufficient sample sizes in each VSP bin. Results, as shown by the bold dashed line in Figure 4-19, indicate no significant impact of the VSP adjustment on the relationship of CO/NO ratio with vehicle age. Thus, differences between the distribution of VSP in the RSD data and the MOBILE6 ramp cycle do not appear to explain the discrepancy between the observed and

predicted relationship of CO emissions to vehicle age which is responsible for the overprediction of CO/NO ratios in the newer vehicles.

Increasing the RSD HC/NO ratio to account for the VSP adjustment as shown in Table 4-7. results in RSD HC/NO ratios for PAS vehicles in Denver that are much larger than the corresponding MOBILE6 predictions. For PRIVATE vehicles in Chicago, the across the board adjustment results in better agreement for new vehicles but more underprediction for older vehicles.

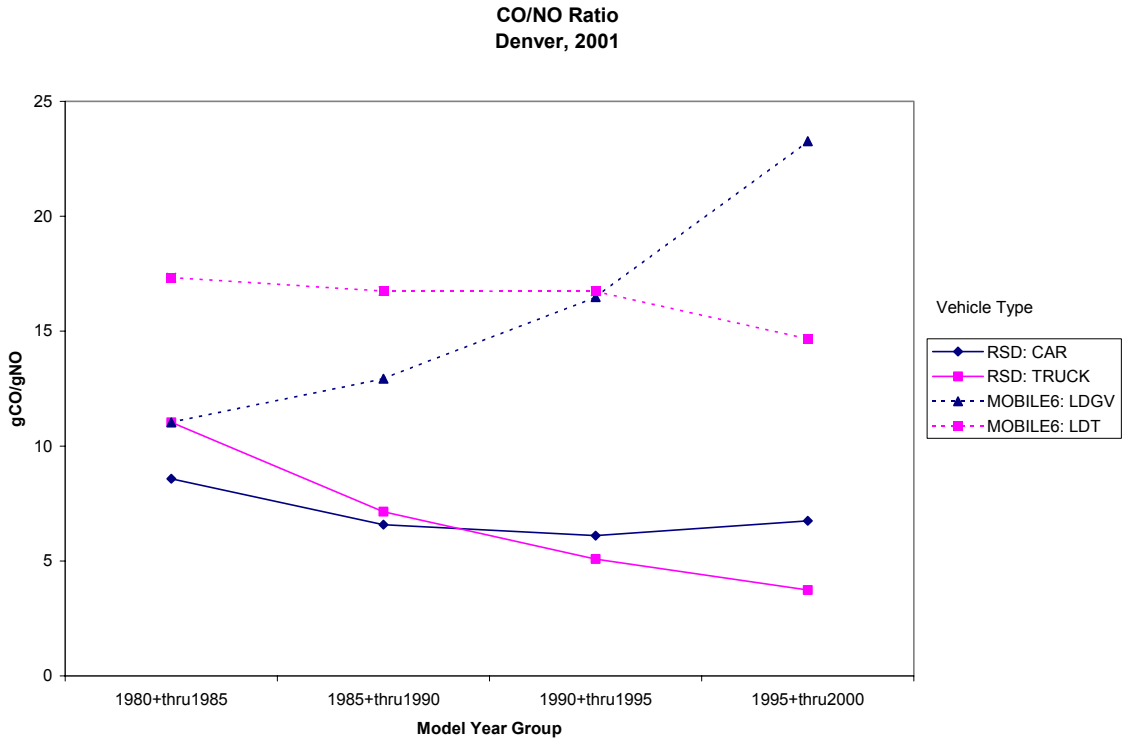


Figure 4-1. CO/NO mean mass emission ratios by model year group from RSD data and MOBILE6 for vehicles classified via VIN decoding as cars or trucks: Denver, 2001.

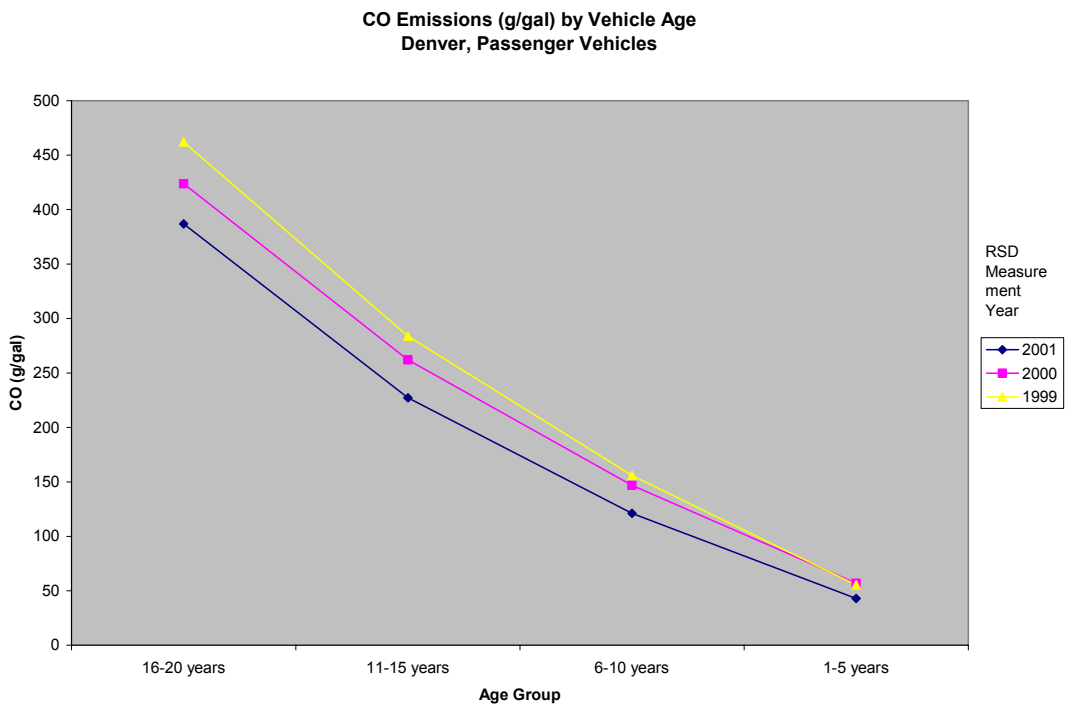


Figure 4-2. Mean CO emissions (g/gal) by vehicle age bin for each year of RSD measurements in Denver (1999 – 2001).

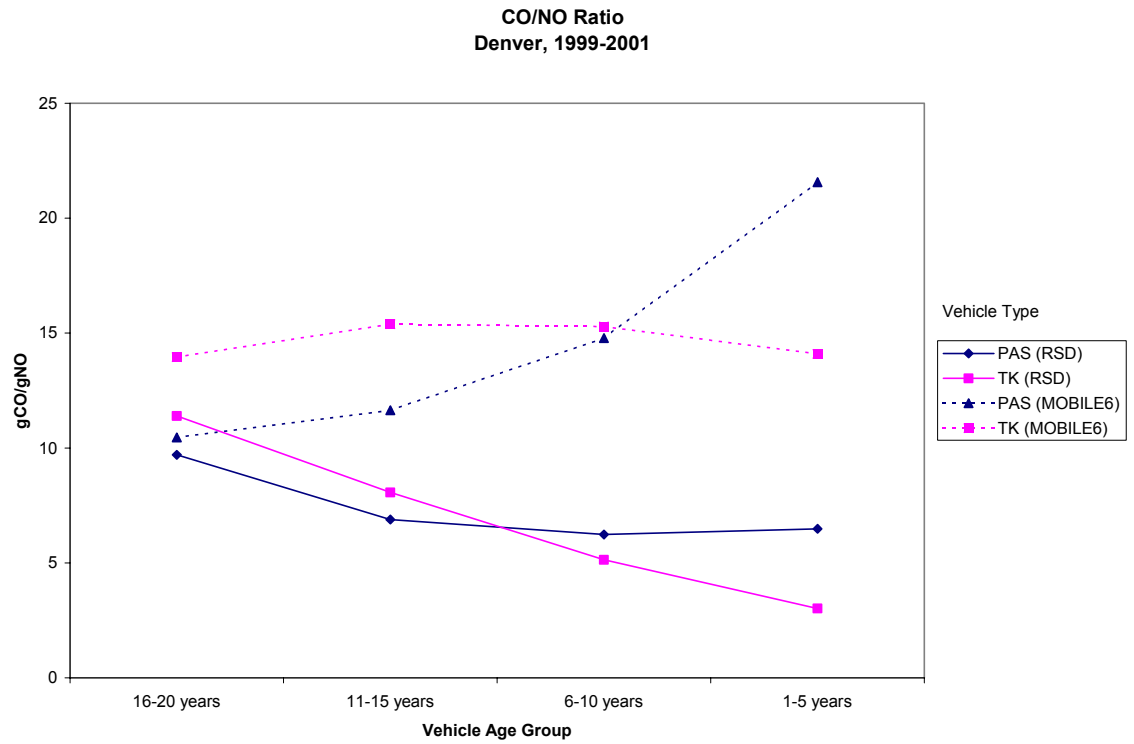


Figure 4-3. CO/NO mean mass emission ratios [(gCO/mile)/(gNO/mile) for MOBILE and (gCO/gal)/(gNO/gal) RSD data] by vehicle age from RSD data and MOBILE6 for vehicles with license types PAS (passenger vehicles) or TRK (trucks): Denver, 1999-2001.

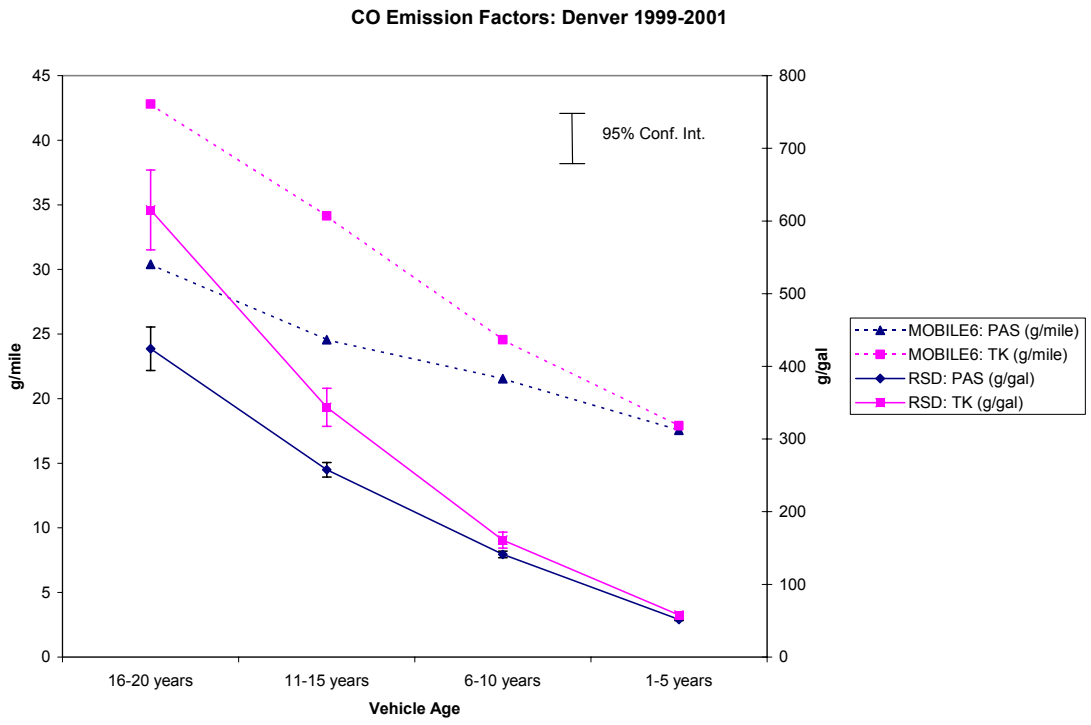


Figure 4-4. Mean CO emissions as a function of vehicle age from RSD data (in g/gal) and from MOBILE6 (in g/mile): Denver, 1999 – 2001.

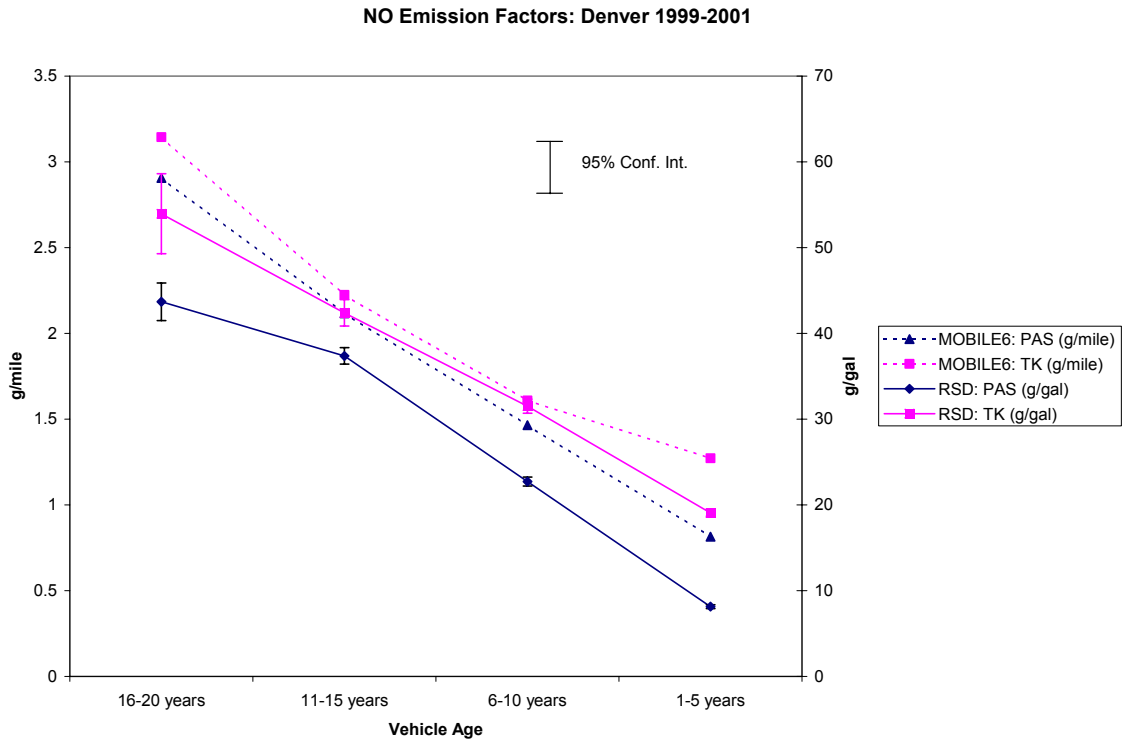


Figure 4-5. Mean NO emissions as a function of vehicle age from RSD data (in g/gal) and from MOBILE6 (in g/mile): Denver, 1999 – 2001.

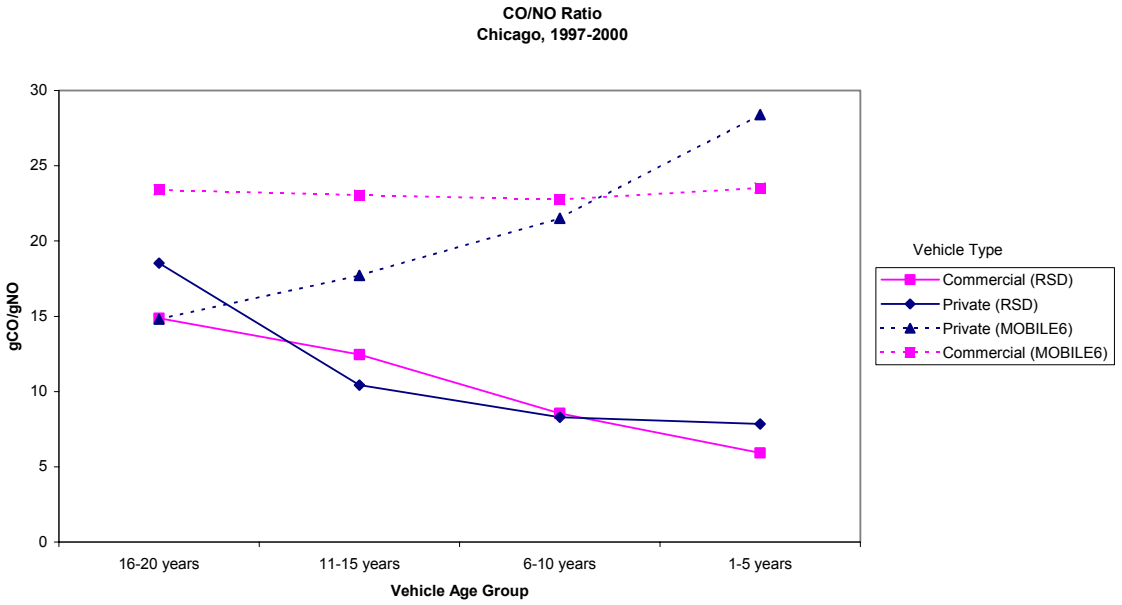


Figure 4-6. CO/NO mean mass emission ratios [(gCO/mile)/(gNO/mile) for MOBILE and (gCO/gal)/(gNO/gal) RSD data] by vehicle age from RSD data and MOBILE6 for vehicles with license types PRIVATE or COMMERCIAL: Chicago, 1997-2000.

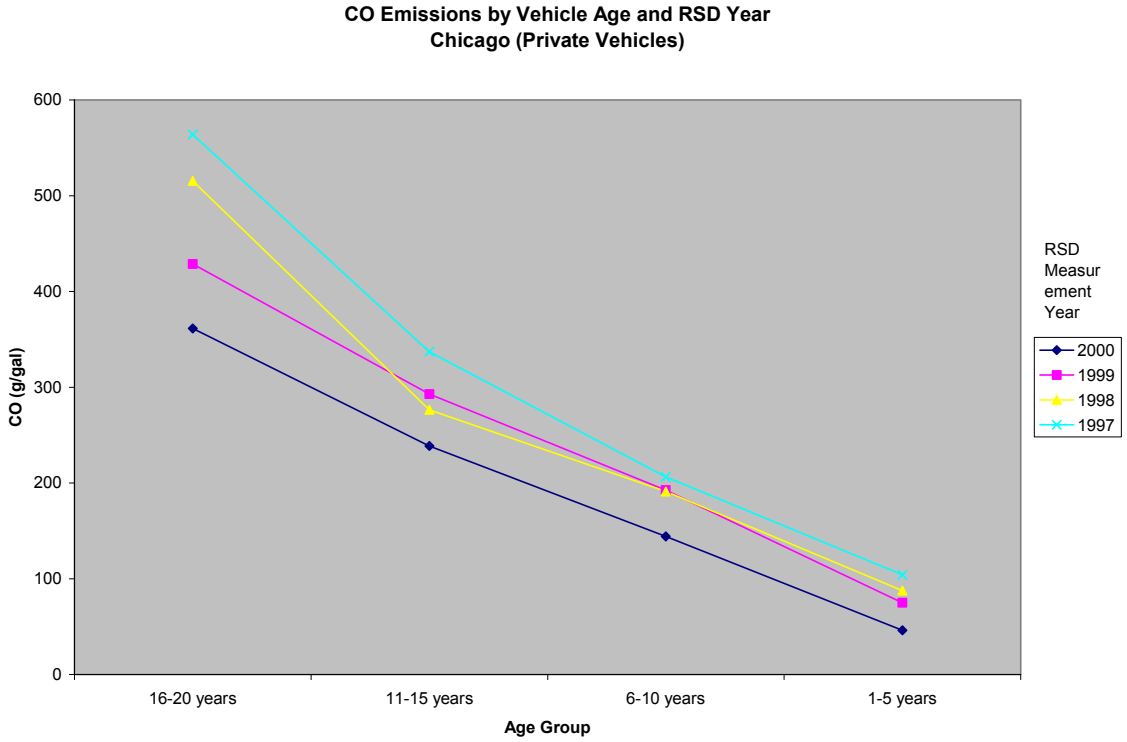


Figure 4-7. Mean CO emissions (g/gal) by vehicle age bin for each year of RSD measurements in Chicago (1997 – 2000).

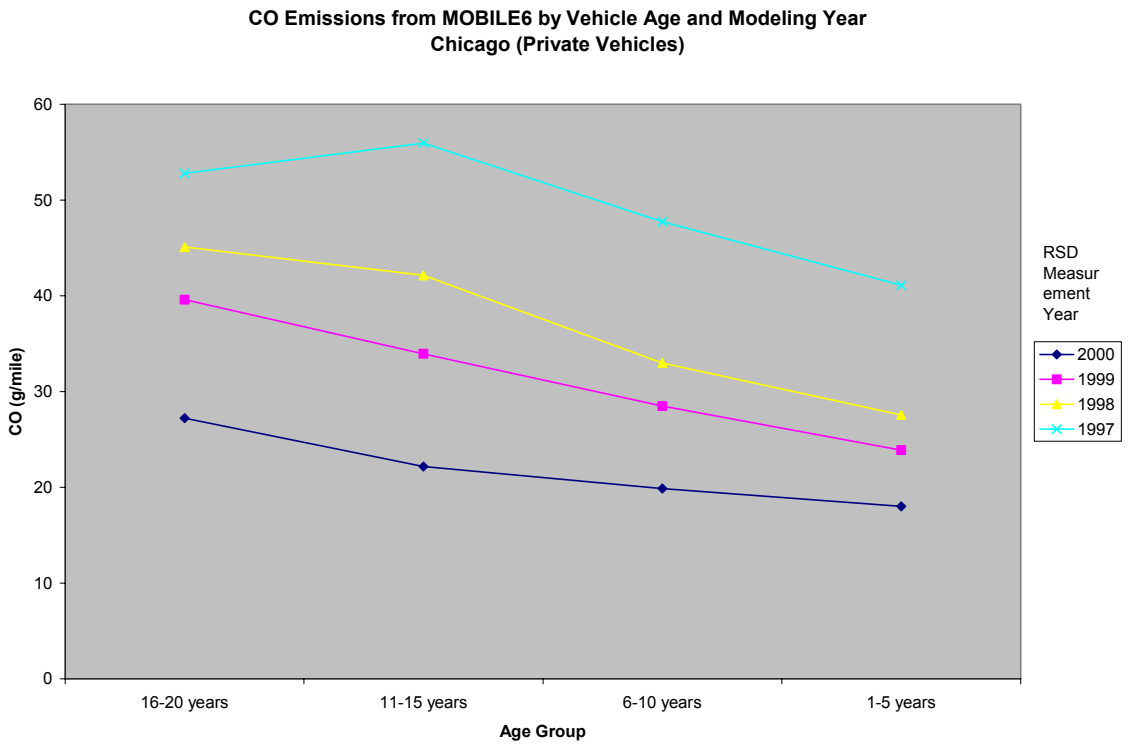


Figure 4-8. Mean CO emissions by vehicle age and modeling year: Chicago, 1997 – 2000.

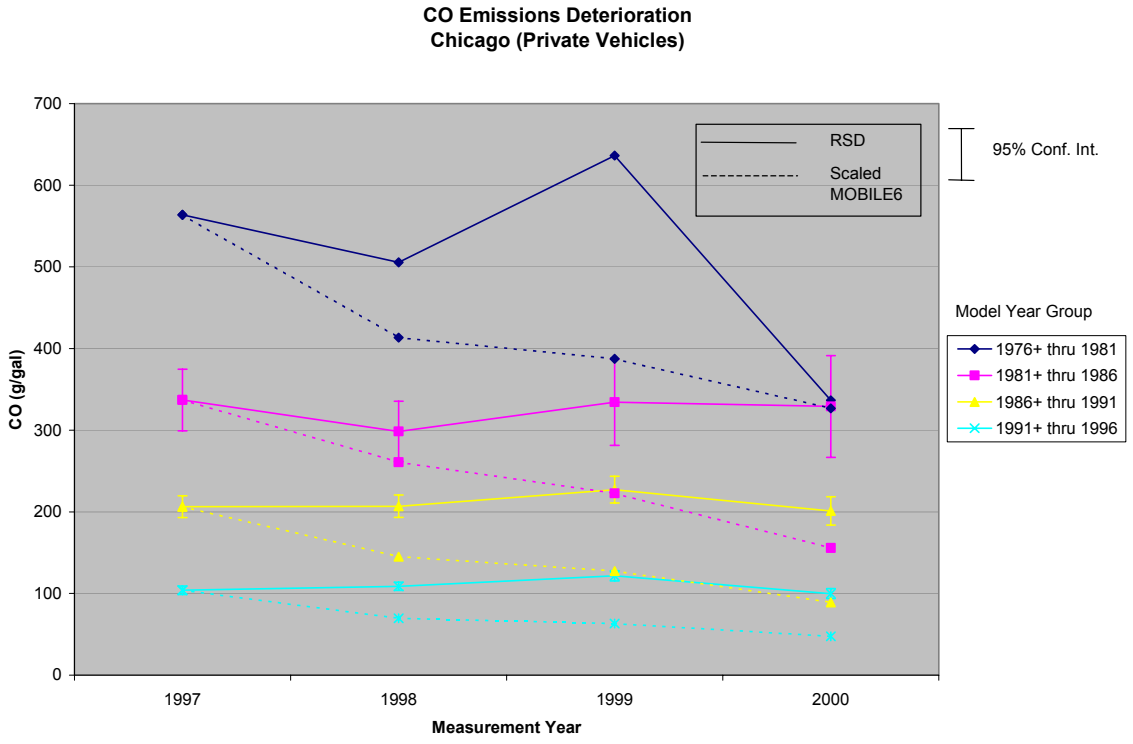


Figure 4-9. Deterioration in mean CO emissions by model year group from RSD data (g/gal) and from MOBILE6 (g/mile scaled to RSD g/gal value in 1997): Chicago, 1997 – 2000.

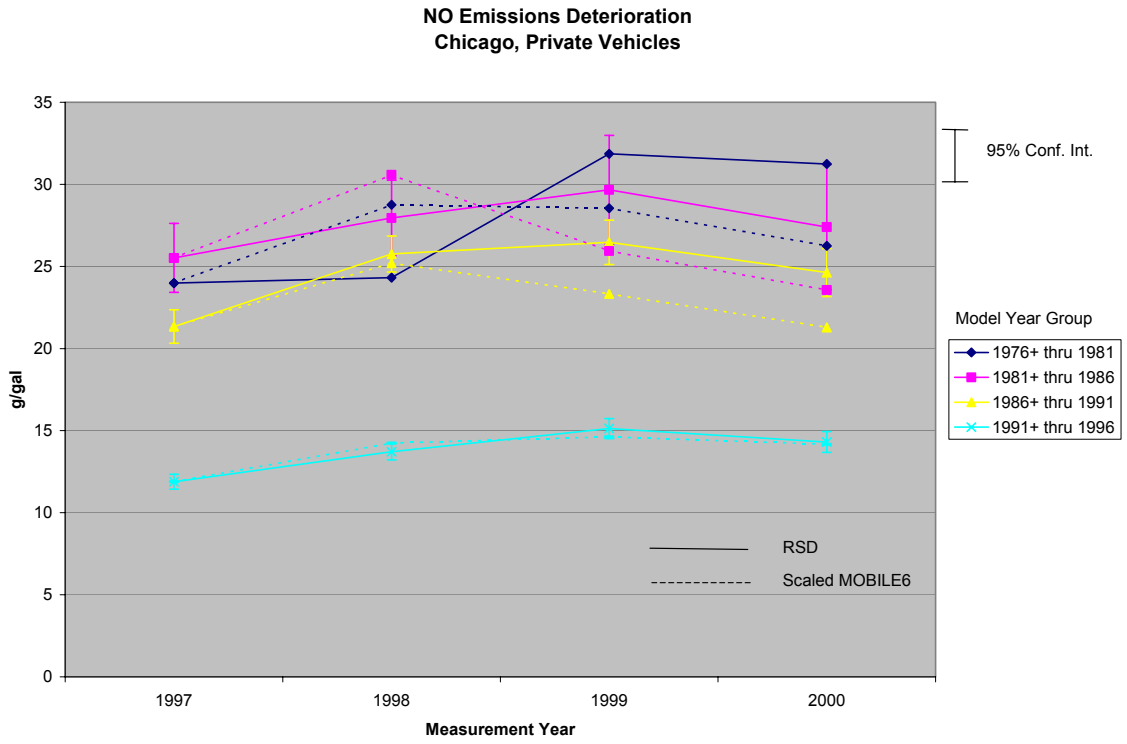


Figure 4-10. Deterioration in mean NO emissions by model year group from RSD data (g/gal) and from MOBILE6 (g/mile scaled to RSD g/gal value in 1997): Chicago, 1997 – 2000.

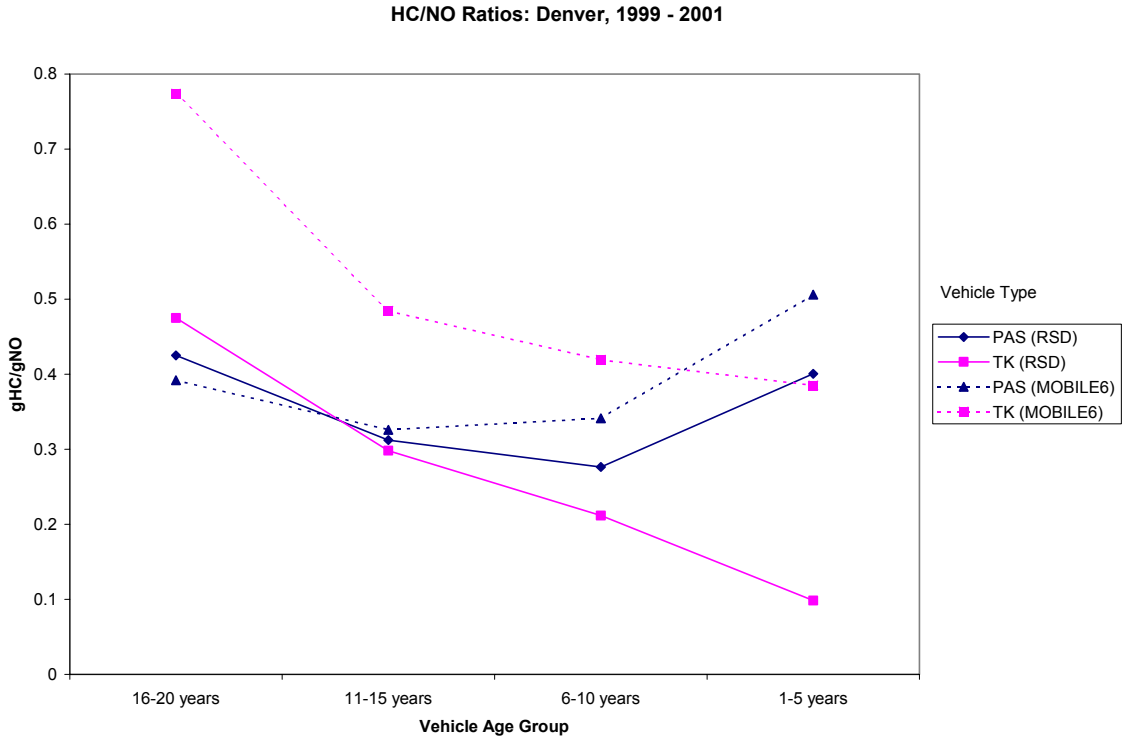


Figure 4-11. HC/NO mean mass emission ratios [(gHC/mile)/(gNO/mile) for MOBILE and (gHC/gal)/(gNO/gal) RSD data] by vehicle age from RSD data and MOBILE6 for vehicles with license types PAS (passenger vehicles) or TRK (trucks): Denver, 1999-2001.

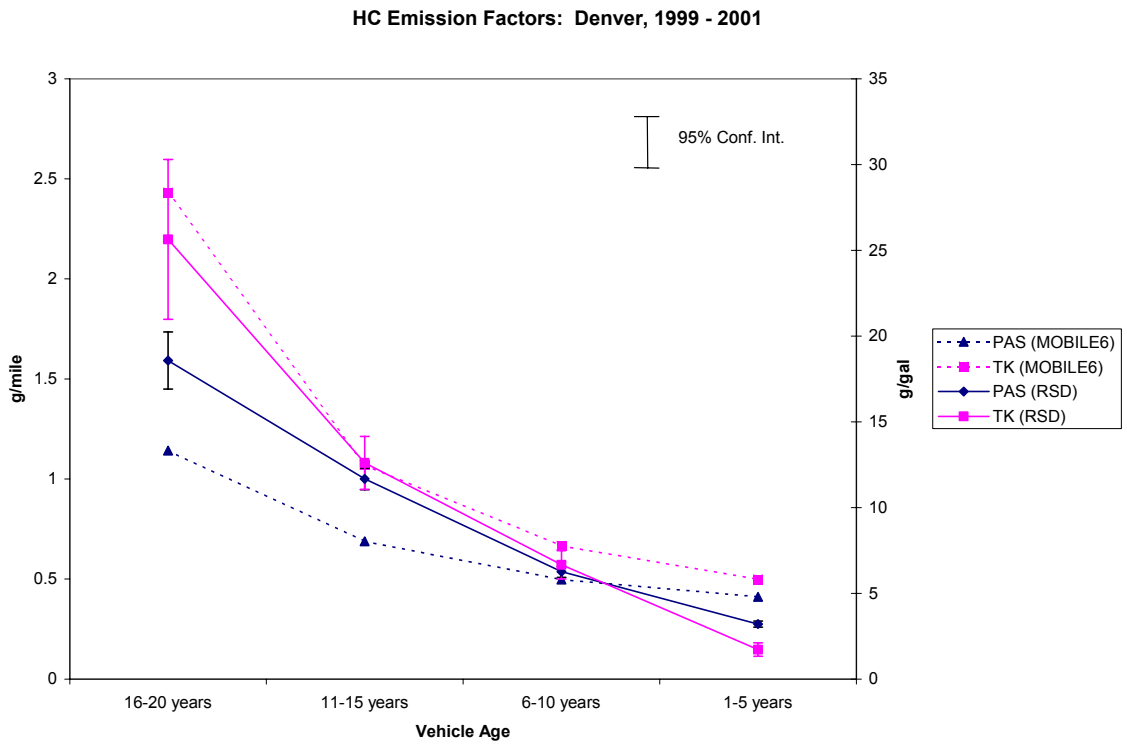


Figure 4-12. Mean HC emissions as a function of vehicle age from RSD data (in g/gal) and from MOBILE6 (in g/mile): Denver, 1999 – 2001.

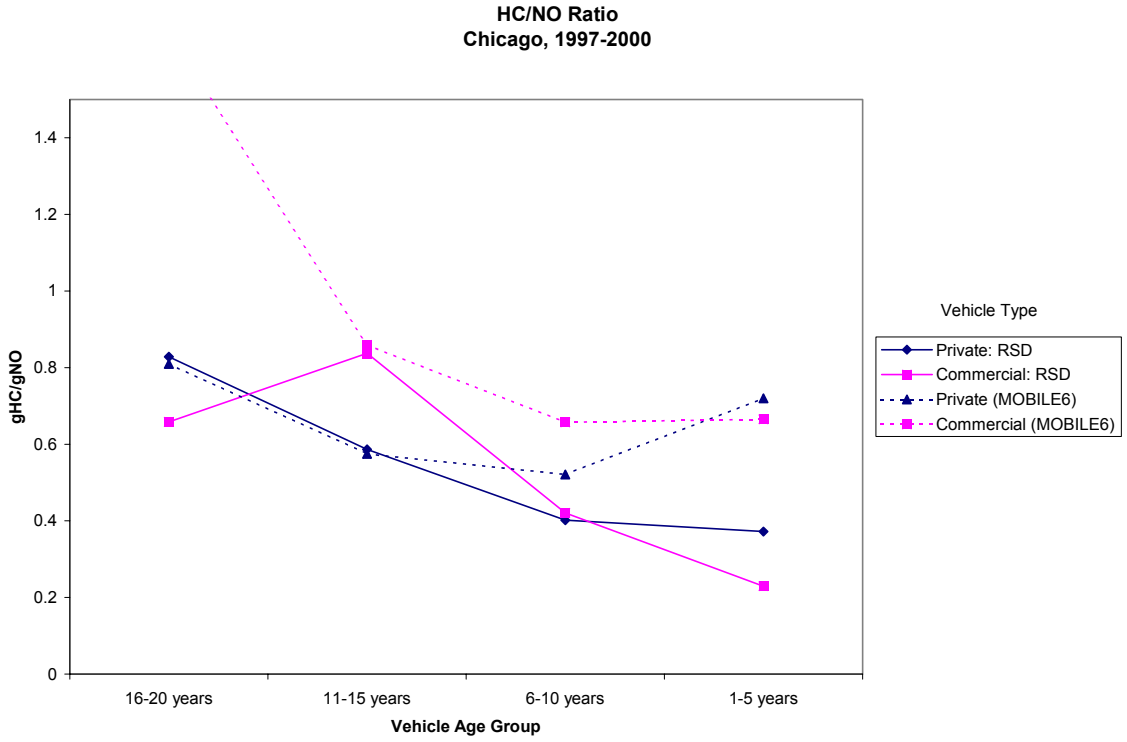


Figure 4-13. HC/NO mass emission ratios by vehicle age from RSD data and MOBILE6 for vehicles with license types PRIVATE or COMMERCIAL: Chicago, 1997-2000.

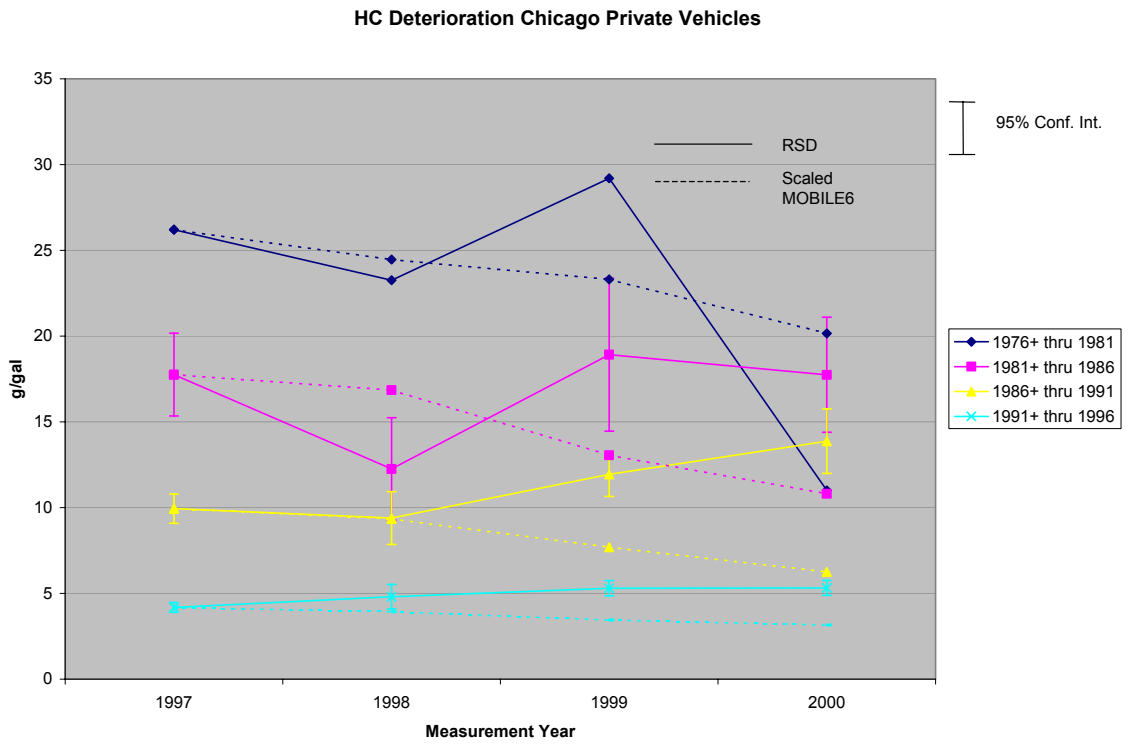


Figure 4-14. HC emissions deterioration by model year group from RSD data (g/gal) and from MOBILE6 (g/mile scaled to RSD g/gal value in 1997): Chicago, 1997 – 2000.

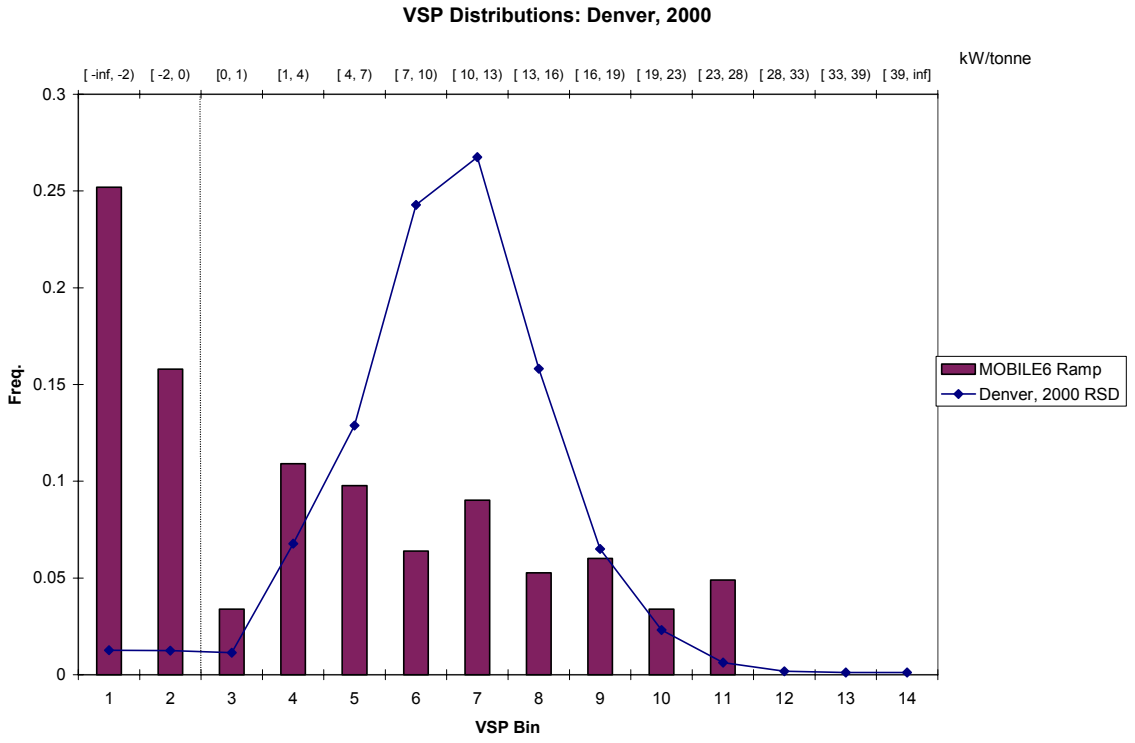


Figure 4-15. Comparison of VSP frequency distribution in Denver, 2000 RSD data with VSP distribution in MOBILE6 ramp driving cycle.

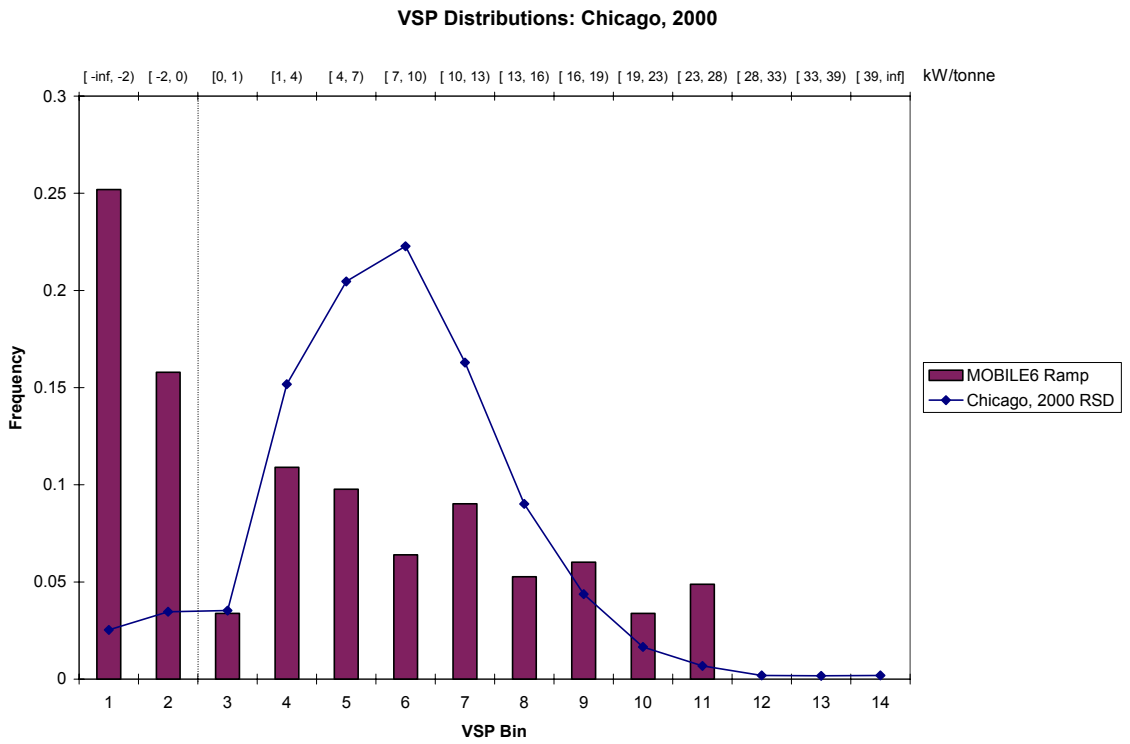


Figure 4-16. Comparison of VSP frequency distribution in Chicago, 2000 RSD data with VSP distribution in MOBILE6 ramp driving cycle.

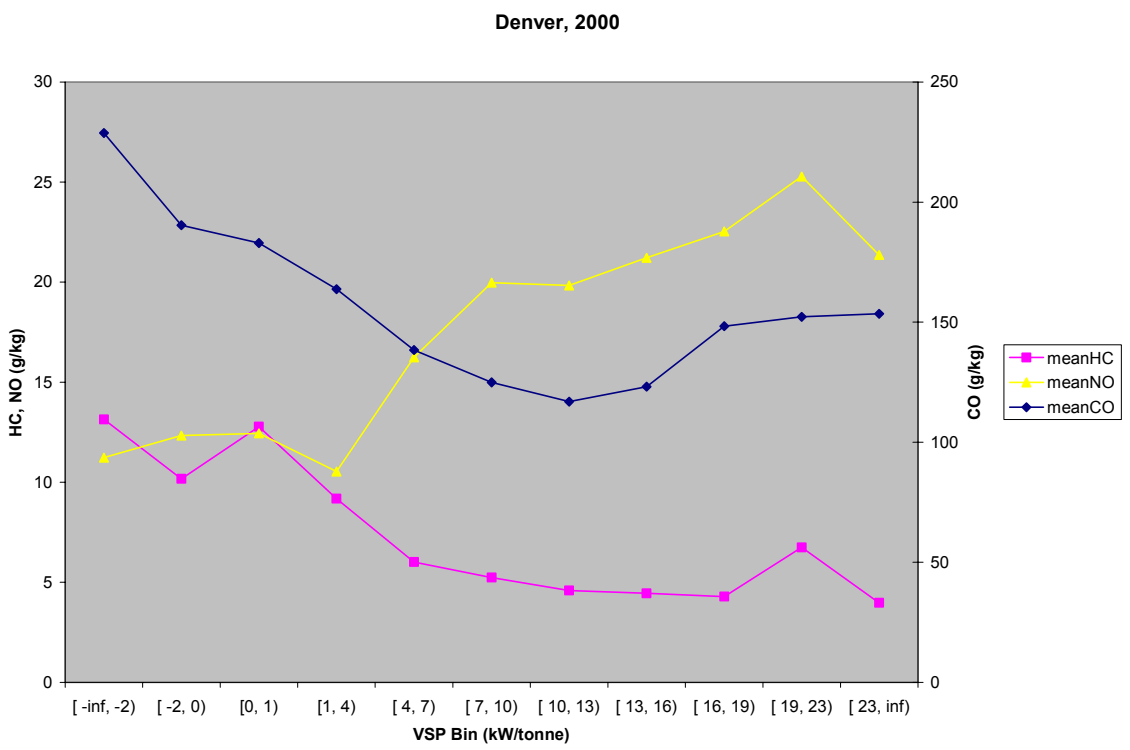


Figure 4-17. Mean HC, NO, and CO emissions (g/gal) as a function of VSP in RSD data: Denver, 2000.

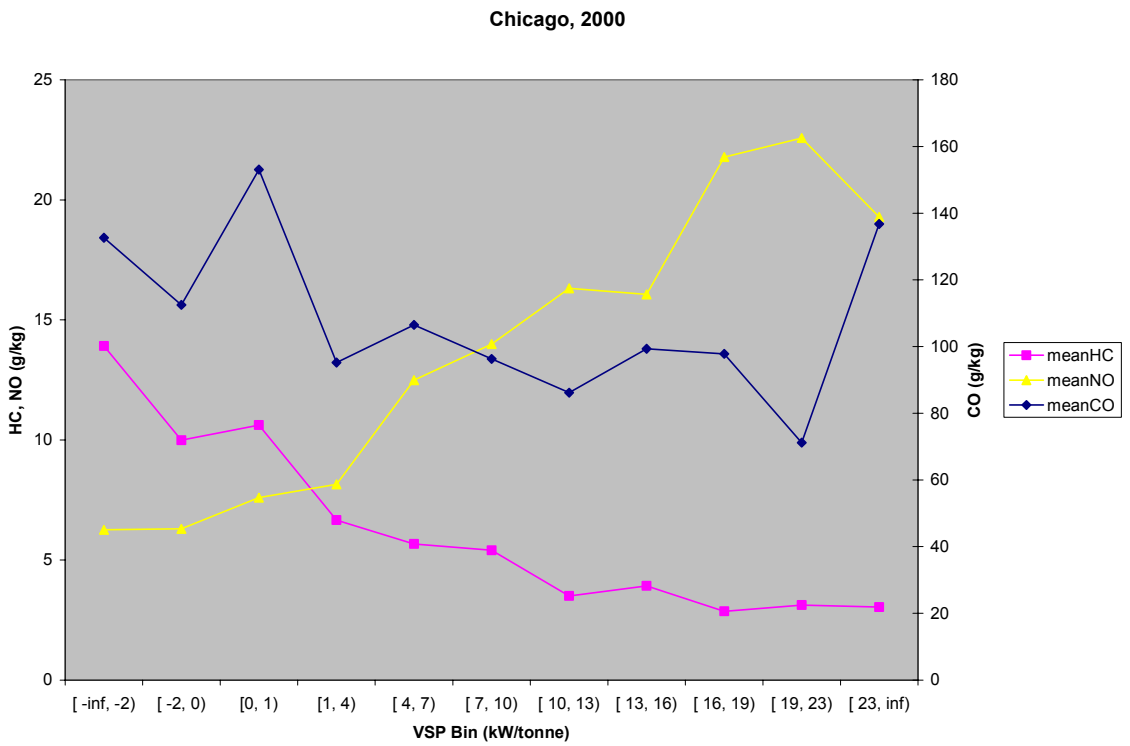


Figure 4-18. Mean HC, NO, and CO emissions (g/gal) as a function of VSP in RSD data: Chicago, 2000.

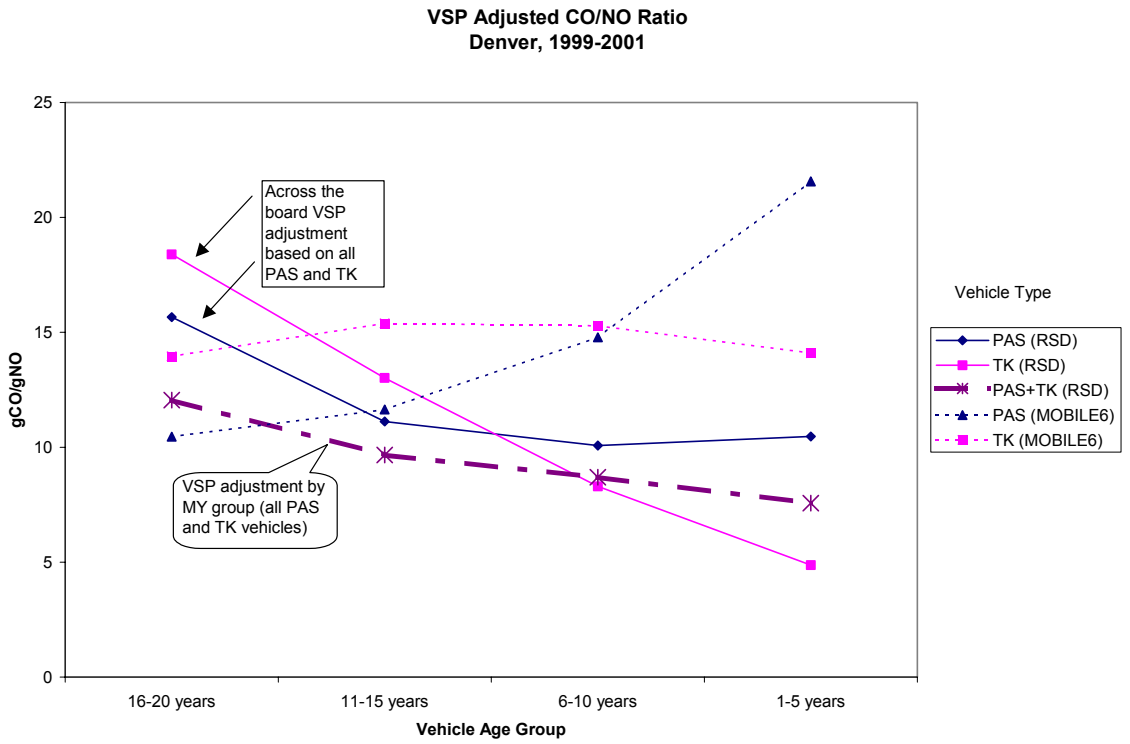


Figure 4-19. CO/NO mass emission ratios by vehicle age from RSD data (adjusted to reflect MOBILE6 ramp cycle VSP distribution) and from MOBILE6 for vehicles with license types PAS (passenger vehicles) or TRK (trucks): Denver, 1999-2001.

SUMMARY AND CONCLUSIONS

In comparison to RSD CO/NO ratios, MOBILE6 overestimates CO relative to NO for newer vehicles by up to a factor of three. This appears to be a result of the fact that MOBILE predicts a much greater increase in CO with vehicle age than is evident in the RSD data; there appears to be much better agreement between MOBILE6 and the RSD data on the dependence of NO emissions on vehicle age.

MOBILE6 HC/NO ratios for vehicle classes composed mostly of LDGV's (PAS vehicles in Denver and PRIVATE vehicles in Chicago) are in much better agreement with the RSD data than is the case for CO/NO ratios. For vehicle classes more heavily weighted towards LDT's (TRK vehicles in Denver and COMMERCIAL vehicles in Chicago), the MOBILE6 HC/NO ratios consistently exceed the RSD ratios (by up to a factor of 4). For both types of vehicle classes, however, the dependence of HC/NO ratios (and of HC emission factors) on vehicle age predicted by MOBILE6 tracks reasonably well with the RSD data, although there is less of a relative difference in HC emission factors between 1-5 year old vehicles and 6-10 year old vehicles in the MOBILE6 predictions than is found in the RSD data.

In Chicago, where temperature and fuel RVP changed more significantly over the course of the four year measurement program than was the case over the three years of measurements in Denver, MOBILE6 predicted significantly lower CO and HC emissions in 2000 as compared to 1997. For a fixed model year group (say, 1986 – 1991 which represents vehicles that were 6 – 10 years old in 1997 and 9 – 13 years old in 2000), the RSD data showed essentially no change in CO emissions between 1997 and 2000, whereas MOBILE6 predicted emissions in 2000 that were less than half of the 1997 prediction. Since MOBILE6 was run with temperature, humidity, and fuel parameters representative of actual conditions during each measurement year, this suggests that the MOBILE6 temperature/RVP correction factors may not be appropriate for the vehicles and driving conditions captured in the RSD data.[10] Similar discrepancies were found for HC emissions and, to a lesser extent, for NO.

Comparison of the distribution of VSP in the Denver and Chicago RSD data with the VSP distribution for the MOBILE6 ramp cycle showed that the ramp driving cycle (which was used to obtain the Basic Exhaust Emission Rates upon which the MOBILE6 estimates are based) includes significantly higher frequencies of negative VSP modes than was observed in the RSD data. This is not unexpected as the ramp cycle is intended to represent driving behavior over the entire length of a ramp, whereas the RSD data collection sites were specifically chosen to capture vehicles during acceleration events. On the other hand, the RSD data included a small fraction of events (less than 0.5% in Denver) with VSP's above 28 kW/tonne, whereas the ramp cycle does not include any VSP's above this level. Adjusting the RSD data according to the ramp cycle VSP distribution produces a 61% increase in the overall mean CO/NO ratio in Denver (31% in Chicago) and a 93% increase in the HC/NO ratio (70% in Chicago). Applying this adjustment decreases the degree to which MOBILE6 overpredicts the CO/NO ratios relative to RSD values for 1 – 5 year old vehicles. For example, for PAS vehicles in Denver, the overpredictions are reduced from a factor of three to a factor of two. However, the ratios for the oldest vehicles are underpredicted when the adjustment is applied.

[10] Inspection of the MOBILE6 results and output of sensitivity runs suggested that the humidity differences did not play a major role.

Differences in VSP distributions between vehicle age bins was found to be fairly minor and making the VSP adjustment on a vehicle age bin basis had little effect on the dependence of CO/NO ratio on vehicle age seen in the RSD data. Increasing the RSD HC/NO ratio to account for the VSP adjustment results in RSD HC/NO ratios for PAS vehicles in Denver that are much larger than the corresponding MOBILE6 predictions.

5. COMPARISON OF HEAVY-DUTY DIESEL CHASSIS EMISSIONS DATA WITH MOBILE6

INTRODUCTION

Unlike the data used to develop LDV emissions estimates, whole vehicle testing of emissions using chassis dynamometers were not used to estimate HDV emissions in MOBILE6. Chassis dynamometers are equipment that allows the entire vehicle to be driven on rollers that can provide the resistance through the wheels that a vehicle experiences when driven on the road including rolling resistance, wind resistance, grade, and inertia. There may be many potential reasons why whole vehicle testing was not used in the development of MOBILE6 HD emission rates including the historic focus on LDVs, lack of representative in-use driving behavior (also called testing cycles which are speed-time traces for a driver to follow), cost of recruitment, and the availability of such testing equipment has been limited to few testing groups\sites because the size and weight of HDVs require additional specifications than those used for LDVs.

Instead emission factor estimates for HDVs in MOBILE6 rely on engine emission testing as a function of work (EPA, 1999a) where work is defined as the mechanical energy developed at the flywheel of the engine. (In operation with a whole vehicle, the engine work would be converted through the transmission to the wheels to propel the vehicle along the road.) An energy conversion factor is an algebraic method to translate engine work to vehicle activity in terms of miles traveled. The emission estimates used in MOBILE6 were developed using the engine emission results converted to emission per vehicle mile traveled as demonstrated in the following equation:

$$\text{MOBILE6 EF (g/mile)} = \text{EF (g/hp-hr)} * D / (FE * BSFC)$$

Where EF = emission factor from engine testing with adjustments
FE = fuel economy (miles/gallon)
BSFC = brake specific fuel consumption (lb./hp-hr)
D = fuel density (lb./gal.)

Engine emission testing data was developed for older, 1978 and earlier, model year engines from testing performed in 1982, and test data for later model year engines was derived from certification test results. Engine emission testing has been performed by removing the engine from the vehicle and mounting it in a test cell where it is loaded through the engine flywheel to simulate in-use behavior. Because of the difficulty and cost of recruitment (not the least of which is petitioning the owner to allow removal of the engine from a vehicle) and testing, it was impractical to perform a similar number of tests on in-use engines that have been performed on in-use LDVs, so engine certification data was used for most emission estimates. The estimates used to develop the conversion factor were taken from in-use surveys for the fuel density and fuel economy with engineering judgment adjusted surveys of the brake-specific fuel consumption as detailed in EPA (1999a). Certification data and survey estimates may not reflect in-use emissions, so verification of these estimates could be useful to justify the current emission rate estimates.

EPA (1999b) included an adjustment for HDDVs in MOBILE6 to address engines that employed a NOx defeat device where the NOx defeat device is an engine management design feature reported to increase NOx during some operation modes. Available short cycle data (as is currently available) may be comparable only to arterial facility types where the defeat device was not expected to have as much of an effect on emissions. Another concern with MOBILE6 is that available data were generated with vehicles not retrofitted with the low NOx rebuild kits, while future year MOBILE6 estimates assumed that nearly all engines will be fitted with such kits eventually.

It is also particularly important to validate HD NOx emissions because HDVs are more significant in MOBILE6 than in MOBILE5, now representing up to half of the total NOx emissions for an urban area.

There is a growing database of emissions results developed by running complete HDVs on chassis dynamometers, and such information was used in the development of the California Air Resources Board EMFAC2000 model (and later model updates). The database included test data developed on a variety of testing cycles (which are by and large speed and time traces for the driver to follow) though not all vehicles were tested on all test cycles. While the chassis data included relatively short cycle driving cycles and may not have been entirely indicative of all facility types or speeds, these data provided a verification method for emission estimates included in MOBILE6 for HDVs. Most data were gathered on diesel-powered vehicles including light, medium, and HHDVs, and transit buses, but there were limited and insufficient (for this work) data available for light HDGVs.

Test cycles for whole vehicle testing were limited to only a few that have not been completely vetted as representative of in-use activity. These test cycles were typically of short duration, lasting less than 20 minutes and are often highly 'idealized' or may not be truly representative of in-use driving. These driving cycles included the EPA's Urban Dynamometer Driving Schedule (UDDS, a modified statistical representation of in-use driving), the West Virginia 5-Mile and 5-Peak cycles (idealized using alternating acceleration and cruise modes), the Central Business District cycle (another idealized acceleration and cruise cycle used primarily for transit buses), and two higher speed cycles developed under Department of Energy and California Air Resources Board contracts. LHDVs can often follow the LDV driving cycles, so emissions for these vehicles were generated on the chassis certification test cycle for LDVs. The dataset for each of the driving cycles was analyzed independently for emissions estimates and compared with the MOBILE6 estimates for those average speeds.

A database of available chassis data was compiled from a variety of sources; however, the number of functioning HD chassis dynamometers was limited to a handful of testing locations and groups. The database was developed from publicly available data and contacting sponsors of and researchers from all groups in North America known to have performed HD chassis testing including CE-CERT (at the University of California Riverside), Southwest Research Institute, West Virginia University, CIFER (at the Colorado School of Mines), and Environment Canada. The database included vehicle testing on many test cycles, though not every vehicle was tested on all test cycles. Therefore, the average emissions calculated for each test cycle may have reflected differences in either average vehicle emissions and/or the effect of the test cycle.

Vehicles were grouped according to similar model years (primarily according to the emissions standards described below), HDV class distinction (generally by gross vehicle weight [GVW], also described below), and by driving cycle type. Average emissions rates were determined and compared with MOBILE6 estimates for those groupings and average speeds.

The basis for vehicle selection was not well described in most studies, but typically the studies relied on voluntary offers of vehicles. Potential selection bias could influence the results. One study where selection bias could be a concern was the McCormick et al. (2001), where the purpose of the study was to investigate the effect of test and repair. For this study, vehicles may have been selected because of potential to fail the opacity test though not all vehicles did fail the test.

This report outlines the HDV class distinctions by vehicle weight and emission standards groupings and compares MOBILE6 estimates against data available as of January 2003 using these groupings. The data used for this comparison was selected from studies where emissions data was generated only using whole vehicle chassis dynamometers to directly compare with the whole vehicle estimates in MOBILE6. Each of these studies used a variety of test cycles and HDVs. The data was sorted and combined by like test cycles and the like vehicle types of Light Heavy-Duty Diesel Vehicles (LHDDV), Medium Heavy-Duty Diesel Vehicles (MHDDV), Heavy Heavy-Duty Diesel Vehicles (HHDDV), and transit buses. In this manner, the summary estimates in this report were compared to the MOBILE6 estimates for similar vehicles and in-use driving behavior.

EMISSION STANDARDS SUMMARY

Table 5-1 provides a summary of the emission standards for HDD engines, those vehicles with gross vehicle weight ratings (GVWR) greater than 8,500 lbs. Because of averaging, banking, and trading provisions in the HD engine regulations, the emission standards in Table 5-1 do not necessarily result in a proportional effect on each model year grouping.

Table 5-1. Federal emission standards for HDD engines.

Model Year	Emission Standard (g/hp-hr)					Smoke* (Opacity)
	HC	CO	NOx	HC + NOx	PM	
1970–1973	---	---	---	---	---	A:40%; L:20%
1974–1978	---	40	---	16	---	A:20%; L:15%; P:50%
1979–1984	1.5	25	---	10	---	A:20%; L:15%; P:50%
1985–1987**	1.3	15.5	10.7	---	---	A:20%; L:15%; P:50%
1988–1989	1.3	15.5	10.7	---	0.6	A:20%; L:15%; P:50%
1990	1.3	15.5	6.0	---	0.6	A:20%; L:15%; P:50%
1991–1992	1.3	15.5	5.0	---	0.25	A:20%; L:15%; P:50%
1993	1.3	15.5	5.0	---	0.25 truck 0.10 urban bus	A:20%; L:15%; P:50%
1994–1995	1.3	15.5	5.0	---	0.10 truck 0.07 urban bus	A:20%; L:15%; P:50%
1996–1997	1.3	15.5	5.0	---	0.10 truck 0.05 urban bus	A:20%; L:15%; P:50%
1998–2003	1.3	15.5	4.0	---	0.10 truck 0.05 urban bus	A:20%; L:15%; P:50%
2004–2006	---	15.5	---	2.5	0.10 truck	A:20%; L:15%; P:50%

Model Year	Emission Standard (g/hp-hr)					Smoke* (Opacity)
	HC	CO	NOx	HC + NOx	PM	
				combined NMHC + NOx***	0.05 urban bus	
2007 and later	0.14 NMHC		0.20	---	0.01	A:20%; L:15%; P:50%

* A = Acceleration; L = Lug; P = Peaks
** Emission test cycle changed from a 13 mode steady-state to a transient
*** Emission test adds a not-to-exceed standards for higher power level groups

EPA (1999a) has assumed in MOBILE that the effect of these standards has been to begin significant engine design changes starting with the 1988 model year with a PM standard on a transient test. The 1991 emission standard precipitated and the 1994 emission standard solidified the need for electronically controlled and turbocharged diesel engines. These model year groupings along with late model vehicles (those produced since the 1998 emission standards came into effect) comprise five general categories -- <1988, 1988-1990, 1991-1993, 1994-1997, and >1997 -- to compare available chassis data with the MOBILE6 estimates. The 1998 model year could be significantly different than other model years because of the defeat device effect, and the 1990 model year may be somewhat different from 1988 and 1989 model years because of the unique NOx standard in effect for that year.

Table 5-2. HDV classifications used in MOBILE6 (EPA, 1999a).

Designation		Description	Gross Vehicle Weight (lbs.)
Gasoline Vehicles			
	HDGV (class 2B)	Light heavy-duty gasoline trucks	8,501-10,000
	HDGV (class 3)	Light heavy-duty gasoline trucks	10,001-14,000
	HDGV (class 4)	Light heavy-duty gasoline trucks	14,001-16,000
	HDGV (class 5)	Light heavy-duty gasoline trucks	16,001-19,500
	HDGV (class 6)	Medium heavy-duty gasoline trucks	19,501-26,000
	HDGV (class 7)	Medium heavy-duty gasoline trucks	26,001-33,000
	HDGV (class 8a)	Heavy heavy-duty gasoline trucks	33,001-60,000
	HDGV (class 8b)*	Heavy heavy-duty gasoline trucks	>60,000
Diesel Vehicles			
	HDDV (class 2B)	Light heavy-duty diesel trucks	8,501-10,000
	HDDV (class 3)	Light heavy-duty diesel trucks	10,001-14,000
	HDDV (classes 4)	Light heavy-duty diesel trucks	14,001-16,000
	HDDV (class 5)	Light heavy-duty diesel trucks	16,001-19,500
	HDDV (class 6)	Medium heavy-duty diesel trucks	19,501-26,000
	HDDV (class 7)	Medium heavy-duty diesel trucks	26,001-33,000
	HDDV (class 8A)	Heavy heavy-duty diesel trucks	33,001-60,000
	HDDV (class 8B)	Heavy heavy-duty diesel trucks	>60,000
Buses			
	HDGB	Heavy-duty gasoline buses (all types)	All
	HDDB (school)	Heavy-duty diesel school buses	All

Designation		Description	Gross Vehicle Weight (lbs.)
	HDDB (transit & urban)	Heavy-duty diesel transit & urban buses	All

*Few HDGV8b exist.

Engine certification data consisted of zero-mile level (ZML) emissions (new engine emissions) typically given in grams of pollutant per brake horsepower-hour (g/bhp-hr), and additional g/bhp-hr deterioration at the end of the vehicle's "useful life." For HD diesel engines, the certification data sets also generally included an intended service class for each engine model (light, medium, heavy, and bus). These intended service classes defined the useful life over which the manufacturer is responsible for emissions certification as shown in Table 5-3.

Table 5-3. Intended service classes and useful life for HD engines (EPA, 1999b).

Engine Class	Useful Life (miles)
All HDG engines	110,000
LHDD engines	110,000
MHDD engines	185,000
HHDD engines and buses	290,000*

* Under the 2004-and-later standards, the useful life for HHDD engines is 435,000 miles.

DATA GENERATION

Several studies investigated the emissions from HDDVs using large chassis dynamometers designed for these HDVs. There were two types of data sets available: studies where true HDVs were tested on cycles intended to represent HDV behavior and LHDVs tested on cycles designed for LDVs.

The data set used was compiled from publicly available information of studies performed by research testing laboratories including the College of Engineering – Center for Environmental Research and Technology (CE-CERT) at the University of California Riverside, Colorado Institute for Fuels and High Altitude Engine Research (CIFER) at the Colorado School of Mines, Environment Canada, Southwest Research Institute (SwRI), and West Virginia University (WVU). A general description of the major research groups, which have conducted laboratory emissions tests on HDDVs, is shown in Table 5-4. All studies from which test data were drawn for this analysis are included in the references list. Some of the data were derived from meta references where data from these research groups had been summarized including those from the NREL and reports to the State of New York.

Table 5-4. Summary of research groups performing chassis emissions testing on HDVs.

Testing Group	Purpose of Study
CE-CERT	Bulk of light heavy-duty vehicle data tested on light-duty dynamometers for the South Coast Air Quality Management District, National Renewable Energy Laboratory (NREL) of the Department of Energy (DOE), and EPA
SwRI	Light and heavy heavy-duty vehicle testing primarily funded by EPA and including other published studies. Includes historical (dating to 1980) and recent (as late as 2001) emission testing studies
CIFER	Medium and heavy-duty vehicle testing for Colorado for the Northern Front Range Study and opacity inspection
Environment Canada	Studies limited to a few vehicles in published studies
WVU	Extensive studies on medium and heavy-duty trucks and buses including those funded by NREL-DOE, State of New York, CRC E-55, and other published work

Below are described the results from the compilation of HDV data derived from chassis testing for light, medium and HHDVs. The medium and HHDV results are presented for three types of vehicles; medium (Class 6 and 7), heavy (Class 8a and 8b), and transit buses. For light and heavier vehicles, the data indicated that an incremental increase in emissions during cold start could be discerned.

LIGHT HEAVY-DUTY DIESEL VEHICLES (LHDDV) COMPARISONS WITH MOBILE6

From a number of studies (Durbin et al., 1999; Norbeck et al., 1998; Durbin et al., 2002; Durbin et al., 2001; CE-CERT, 1999; and SwRI, 2000), 38 measurements were available for light heavy-duty diesel vehicles (LHDDV) using similar fuels and the same test cycle. Of these only four measurements were on LHDDV3, and those four were just above the 10,000 lb GVWR cutoff from LHDDV2b, so the LHDDV3 were included in this data set for comparison with the MOBILE6 data.

The largest set of measurements existed for vehicles tested primarily empty or near empty on the light-duty Federal Test Procedure (FTP) chassis test cycle. This test cycle consists of three segments (Bags 1, 2, and 3), where Bag 1 is a cold start and Bag 3 is a hot start on the same driving cycle. Data for each bag were available, allowing an evaluation of the start emissions. This test cycle has been used as the basis for light-duty emission estimates including MOBILE5 and earlier versions of the MOBILE model. LHDVs tend to be pick-up trucks and passenger vans, distinguished from LDVs only by carrying capacity. Therefore, the light-duty test cycles may be considered reasonable representations of in-use behavior for these types of vehicles.

Table 5-5 provides the emission averages for the data set by model year groupings that reflect the emission standards. Table 5-6 provides the emissions rates that MOBILE6 predicted for the most appropriate vehicle type on the average speed typical for the test cycle. Figures 5-1 and 5-2 show those comparisons graphically.

Table 5-5. Light-duty FTP Composite results for LHDDV (with uncertainty ranges calculated with 90% confidence levels).

Model Year	Counts	THC (g/mile)	CO (g/mile)	NOx (g/mile)	PM (mg/mile)
<1988	10	0.59 " 0.19	1.97 " 0.42	3.46 " 0.45	424 " 126
1988 – 1990	7	0.42 " 0.20	1.67 " 0.42	5.19 " 1.75	248 " 87
1991 - 1993	4*	1.96 " 3.18	3.12 " 3.27	5.68 " 2.66	705 " 1122
1994 - 1997	10	0.50 " 0.13	1.78 " 0.43	6.62 " 0.84	80 " 21
>1997	7	0.30 " 0.06	1.67 " 0.19	6.13 " 1.19	149 " 56

* A significant outlier influences the THC, CO, and PM results.

Table 5-6. MOBILE6 LHDDV2b on an arterial at 19.5 mph typical of the FTP Composite.

Model Year	THC (g/mile)	CO (g/mile)	NOx (g/mile)
<1988	0.64	3.9	7.85
1988 – 1990	0.55	1.5	5.46*
1991 - 1993	0.39	0.4	5.22
1994 - 1997	0.21	1.1	4.68
>1997	0.21	1.0	3.80

* 1990 model year 6.0 g/mile NOx

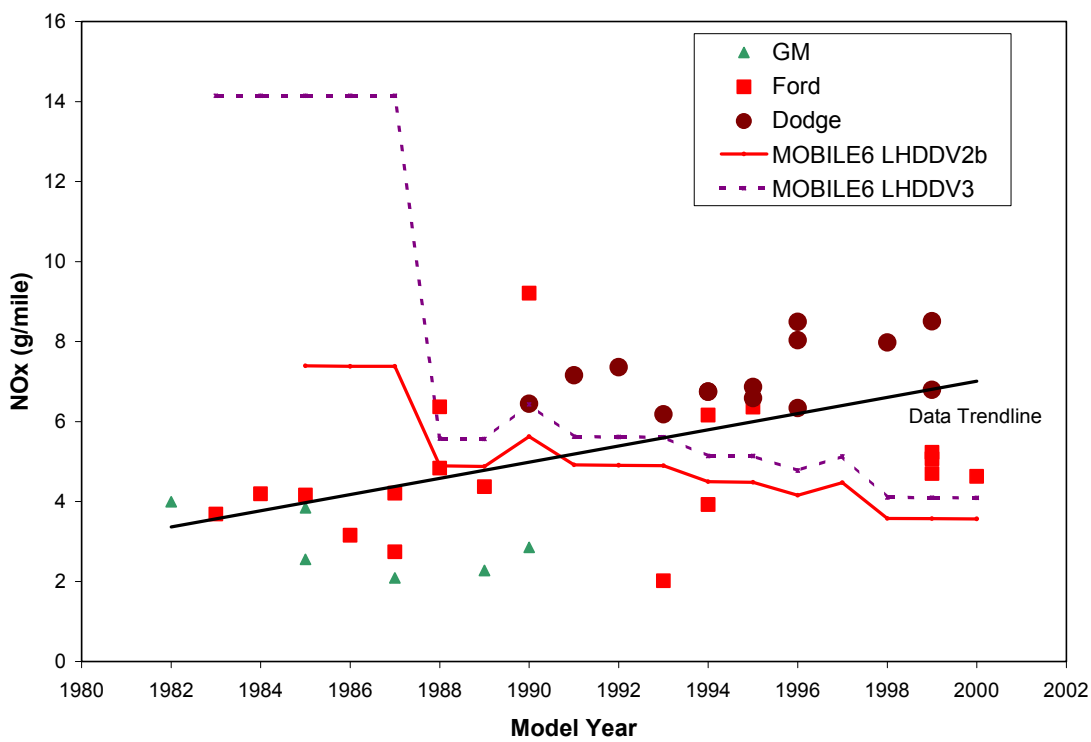


Figure 5-1. LHDV NOx emissions on the light-duty FTP test cycle.

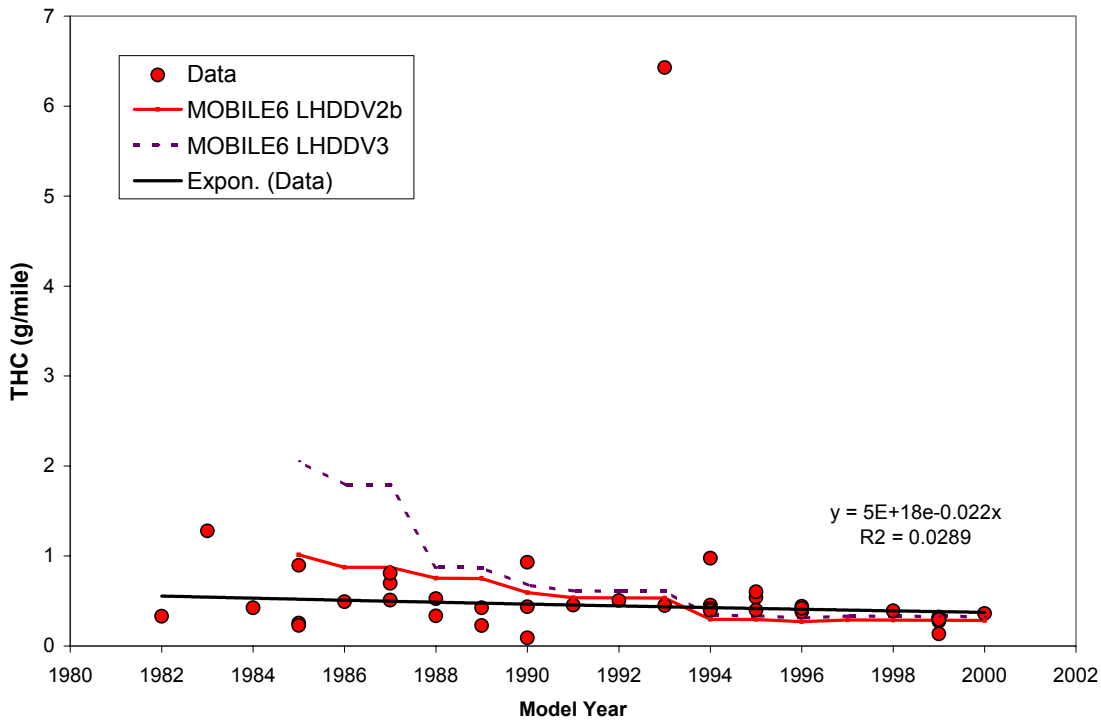


Figure 5-2. LHDV THC emissions on the light-duty FTP test cycle.

The data show that NOx emissions for LHDDV have been trending upward, but this may be explained by design differences from past to future model years and by manufacturer. Figure 5-1 shows that the vehicle make may also influence the trend, with only older GM (GMC and Chevrolet) makes having lower NOx emissions, and only newer Dodge trucks and vans having higher NOx emissions. The MOBILE6 NOx emission estimates have been trending downward, though the predicted average NOx emission levels have been relatively constant for model years of 1988 and later. The MOBILE6 THC and CO estimates are reasonably comparable with the data except for one outlier with 10 times the average emission rate from a vehicle obviously malfunctioning.

There were design differences between older (1993 and earlier) and more recent LHDD engines where manufacturers migrated from indirect injected diesel (IDI) to direct injection (DI) diesels around 1994, and from naturally aspirated to turbocharged engines during 1992 – 1994 model years. IDI was a technology employed likely for drivability, but it also produces lower NOx emissions. Also, engine compression ratios were likely increasing during this period, and with all other design elements held constant, a higher compression ratio will produce higher NOx emissions.

The emission effect of the transition from naturally aspirated to turbocharged engines is uncertain but may influence the start emissions, shown in Table 5-7, where NOx start emissions increased with newer model years. The start emissions indicated that a start increment on all pollutants was apparent, but the start emissions are not explicitly included in MOBILE6. Start emissions were analyzed to provide a basis for determining if the starts should be explicitly included in emissions estimates as they now are in MOBILE6 for LDVs.

Table 5-7. Influence of model year on cold start emissions (Bag 1 – Bag 3) for LHDDV (typically 3.75 miles).

Model Year	Delta THC (g/mile)	90% CL	Delta CO (g/mile)	90% CL	Delta NOx (g/mile)	90% CL	Delta PM (mg/mile)	90% CL
<1988	0.227	0.157	0.491	0.194	0.393	0.197	136	95
1988 – 1990	0.123	0.102	0.422	0.227	0.292	0.508	84	98
1991 – 1993	0.324	0.412	0.887	0.175	1.646	1.797	113	162
1994 – 1997	0.099	0.088	1.169	0.247	1.008	0.547	16	19
>1997	0.047	0.070	0.941	0.412	2.226	0.187	0.1	28
All	0.151	0.058	0.781	0.127	1.006	0.273	67	33
Start Increment (g/start)	0.57	0.22	2.93	0.48	3.77	1.03	253	124

Other test data included two separate EPA-funded studies, which, in addition to the data included in the analysis above, investigate the effect of light-duty cycles and payload. These resulted in a small subset of the available data, but demonstrated that vehicle weight increases emissions, and test cycles with lower average speeds result in higher per-mile emissions rates.

Test data on driving cycles other than the light-duty FTP were minimal for LHDDV (Classes 2b and 3), with only single tests points for some test cycles. Very little data for other LHDDV, Classes 4 and 5 were available. The emissions levels for Class 4 and 5 trucks might presumably be considered to have emission rates between the LHDDV described above and those of Class 6 and 7 trucks shown below.

HEAVY-DUTY VEHICLE TESTING

The test cycles used in studies for heavier vehicles varied greatly, and there was no single test cycle that was used in all studies. These test cycles have been reviewed in early studies, e.g. Clark et al. (2002a, 1998, 1994), and are summarized in Table 5-8. Two cycles, Central Business District (CBD) and WVU 5-peak or 5-mile route, are idealized test cycles where the vehicle repeatedly accelerates to a cruise speed, cruises, and brakes to an idle condition in a clipped 'saw-tooth' speed-time trace. The other test cycles were developed using field measurements to represent various types of driving behavior. The test data included results on the test cycles listed in Table 5-8.

Table 5-8. Summary average speeds over test cycle.

Test Cycle	Average Speed (mph)	Distance* (miles)
NY Garbage Truck	~ 2.3	---
NY Truck	~ 8	---
NY Composite	~ 9	---
Central Business District (CBD)	~ 9	2.0
City Suburban Heavy Vehicle Route (CSHVR)	~ 14	6.7
Urban Dynamometer Driving Schedule (UDDS)	~ 19	5.4
West Virginia Univ. (WVU) 5-peak; 5-mile	~ 21	5.0
WVU Highway	~ 40	---
ARB Cruise	~ 40	---

* Cold start data were available for these test cycles so the mileage was useful to calculate a cold start increment in excess grams per start.

In the disparate data sets, the test cycles were run either with the driver most closely matching a second-by-second test speed (speed-time) trace, or with the driver and truck operating along a route that simulated the second-by-second test speed as described by Clark et al., 1998. The route method can lead to different instantaneous loads over the cycle, but by and large yielded similar average speeds and cycle distances. An example of this is the WVU 5-mile cycle, which is a route driven that allows for maximum accelerations over the 5-peak cycle where acceleration rates may be more or less than the vehicle can meet, but the average speed may be only slightly higher or lower given the relatively long periods of cruise and idle. Data tested using either method were paired with the similar test cycles. This approach may have led to greater variability in the emission results, but vehicle-to-vehicle variability was considered to be a greater source of variability than the difference between the route and speed-time trace methods, so pairing the tests yielded more vehicles for each test cycle group.

The most used test cycle was the Federal Urban Dynamometer Driving Schedule (Test D) developed from on-road test data and available in the Federal Register. The test cycle was developed from field data and was intended to represent in-use driving behavior of HDVs; however, the official use of this test cycle is to prepare HDGVs for evaporative emission testing.

Data on higher speed test cycles, "NREL Highway" and "ARB Cruise" both with average speeds of about 40 mph, were a result of two test programs (Clark et al., 2002b and WVU, 2002). These test cycles had a small number of vehicles tested on both cycles to determine if the data could be combined. Three vehicles were tested over both test cycles allowing for a comparison of the results, shown in Table 5-9. In general, the emissions for THC, CO, and NOx were comparable for these vehicles, though the NREL cycle produced lower NOx and higher CO emissions. Certainly the THC and PM results for vehicle 16 (a 1985 vehicle compared with 1994 for vehicle number 26 and 1995 for vehicle number 33) were much different on a relative basis, with the NREL cycle much higher. Still the data indicated that the test cycle data for THC, CO, and NOx were reasonably similar between the two test cycles for late model engines.

Table 5-9. NREL Highway and ARB Cruise test cycle comparison (~ 40 mph).

Vehicle	Test Cycle	Fuel Economy (mpg)	THC (g/mile)	CO (g/mile)	NOx (g/mile)	PM (g/mile)
PM-Split 16	ARB	6.29	1.01	2.86	11.53	0.47
PM-Split 16	NREL	8.76	2.16	3.20	9.52	1.23
PM-Split 26	ARB	6.86	0.36	1.67	19.08	0.14
PM-Split 26	NREL	5.71	0.45	1.75	18.60	0.14
PM-Split 33	ARB	6.96	0.17	1.43	27.98	0.11
PM-Split 33	NREL	7.55	0.14	2.41	26.00	0.16

Test data were also available at high altitude in Colorado (McCormick et al., 2001 and Graboski et al., 1998) and with California specification fuel. The high altitude results were adjusted to low altitude to be comparable to other data sources according to the MOBILE6 altitude adjustment (EPA, 1999a). Results with California diesel fuel were adjusted upward according to the estimated NOx emission reduction of 6.2% according to EPA (2001) with the use of California fuel. There may have been emissions benefits with the use of California fuel for other pollutants, such as total hydrocarbon, carbon monoxide, or particulate matter, but EPA did not estimate an adjustment, so none was applied in this work.

The test weight for the vehicles varied according to its curb weight and weight of extra loads, and emissions results are related to the test weight. The most common practice used in most of the studies used in this report was to half load (the average of the curb and gross vehicle weight rating) each vehicle prior to emission testing. When the same vehicle was tested with several different weights, the results for the closest to common weight were used for comparison.

MEDIUM HEAVY-DUTY DIESEL VEHICLES (MHDDV) COMPARISONS WITH MOBILE6

Data for Class 6 trucks were available in limited numbers for a few test cycles. The test results are shown in Tables 5-10 and 5-11, with the MOBILE6 estimates for comparison in Table 5-12. THC and CO emissions were much higher for certain vehicles indicating the potential for a skewed average emission level due to high emitters. The data for NOx emissions indicate a slightly higher emission rate for late model vehicles (those made after 1991).

Table 5-10. HDDV Class 6 on UDDS – Test D (~ 19 mph).

Model Year	Counts	THC (g/mile)	CO (g/mile)	NOx (g/mile)	PM (g/mile)
<1988	1	21.9	12.8	12.5	1.11
1988 – 1990	1	0.71	2.34	12.6	0.77
1991 - 1993	3	0.86 " 0.39	6.76" 2.10	14.9" 2.2	1.05" 0.11
1994 - 1997	2	11.7	11.5	14.1	1.91
>1997	0	-	-	-	-

Table 5-11. HDDV Class 6 on WVU 5-Peak/5-Mile (~ 21 mph).

Model Year	Counts	THC (g/mile)	CO (g/mile)	NOx (g/mile)	PM (g/mile)
<1988	0	-	-	-	-
1988 – 1990	1	0.65	1.80	10.0	0.59
1991 - 1993	2	0.71	5.50	11.8	0.59
1994 - 1997	2	0.38	2.31	11.8	0.35
>1997	0	-	-	-	-

Table 5-12. MOBILE6 HDDV on an arterial (19 mph).

Model Year	THC (g/mile)	CO (g/mile)	NOx (g/mile)
<1988*	1.9 – 5.0	12 – 14	15 – 25
1988 – 1990	1.3	4.5	12.4
1991 - 1993	0.86	3.1	9.4
1994 - 1997	0.65	2.0	9.1
>1997	0.63	1.8	7.9

* Low end of range typical of 1979-1988; and high end 1978 and earlier.

There were more data available for Class 7 trucks as shown in Tables 5-13 and 5-14; these data afford the opportunity for a much better comparison between data and MOBILE6 estimates shown in Table 5-15. Here the data demonstrated emissions for THC, CO, and NOx very similar to that predicted by MOBILE6. As with Class 6 trucks, late model vehicles (1991 and later) tended to produce higher NOx than predicted by MOBILE6.

Table 5-13. HDDV Class 7 on UDDS – Test D (~ 19 mph).

Model Year	Counts	THC (g/mile)	CO (g/mile)	NOx (g/mile)	PM (g/mile)
<1988	12	3.68 " 1.87	19.1 " 9.3	18.9 " 2.8	2.92 " 1.58*
1988 – 1990	6	0.72 " 0.22	12.2 " 7.9	19.9 " 5.6	1.50 " 0.29
1991 - 1993	4	1.96 " 1.58	3.94 " 1.05	14.8 " 3.3	0.97 " 0.06
1994 - 1997	8	0.41 " 0.16	4.78 " 2.15	17.0 " 2.7	0.62 " 0.14
>1997	0	-	-	-	-

* Includes one extraordinary PM high emitter of 12 g/mile.

Table 5-14. HDDV Class 7 on WVU 5-Peak/5-Mile (~ 21 mph).

Model Year	Counts	THC (g/mile)	CO (g/mile)	NOx (g/mile)	PM (g/mile)
<1988	3	1.49 " 0.81	6.16 " 4.28	16.2 " 6.2	1.30 " 0.13
1988 – 1990	6	0.72 " 0.25	3.44 " 0.70	16.2 " 4.0	0.74 " 0.24
1991 - 1993	2	0.63	3.02	10.1	0.50
1994 - 1997	7	0.38 " 0.17	2.98 " 1.49	14.9 " 3.1	0.44 " 0.18
>1997	0	-	-	-	-

Table 5-15. MOBILE6 HDDV Class 7 on an arterial (19 mph).

Model Year	THC (g/mile)	CO (g/mile)	NOx (g/mile)
<1988*	2.2 – 6.0	13.4 – 16.2	17 – 29
1988 – 1990	1.6	5.5	15.1
1991 – 1993	1.0	3.8	11.6
1994 – 1997	0.8	2.5	11.3
>1997	0.8	2.3	9.8

* Low end of range typical of 1979-1988; and high end 1978 and earlier.

Two studies (Clark et al., 2002b and WVU, 2002) used two different test cycles with average speeds of about 40 mph. As described above, the emissions on these two cycles were roughly equivalent for the three vehicles tested on both of these cycles. The results for these two cycles are sparse for this vehicle class but are presented in Tables 5-16 and 5-17. Only one model year grouping had significant data, indicating that the average emission estimates corresponded to the low end of the range predicted by MOBILE6 for older Class 7 vehicles, shown in Table 5-18.

Table 5-16. HDDV Class 7 on NREL Highway (~ 40 mph).

Model Year	Counts	THC (g/mile)	CO (g/mile)	NOx (g/mile)	PM (g/mile)
<1988	1	2.16	3.2	9.5	1.23
1988 – 1990	1	1.13	7.9	15.4	1.86
1991 - 1993	0	-	-	-	-
1994 - 1997	0	-	-	-	-
>1997	0	-	-	-	-

Table 5-17. HDDV Class 7 on ARB Cruise (~ 40 mph).

Model Year	Counts	THC (g/mile)	CO (g/mile)	NOx (g/mile)	PM (g/mile)
<1988	7	1.64 " 0.81	5.8 " 1.4	14.2 " 2.7	3.10 " 3.55*
1988 – 1990	0	-	-	-	-
1991 - 1993	2	1.05	1.8	13.1	0.36
1994 - 1997	1	0.51	3.4	30.1	0.23
>1997	0	-	-	-	-

* Includes one extraordinary PM high emitter of 16 g/mile.

Table 5-18. MOBILE6 HDDV Class 7 on a freeway (40 mph).

Model Year	THC (g/mile)	CO (g/mile)	NOx (g/mile)
<1988	1.2 – 3.2	6 - 8	15 – 26
1988 – 1990	0.88	2.6	13.2
1991 – 1993	0.56	1.8	10.1
1994 – 1997	0.43	1.2	9.9
>1997	0.42	1.1	8.7

* Low end of range typical of 1979-1988; and high end 1978 and earlier.

HEAVY HEAVY-DUTY DIESEL VEHICLES (HHDDV) COMPARISONS WITH MOBILE6

Classes 8a and 8b represent an extremely important truck category because these vehicle types represent the largest fraction of the HDD emissions. Class 8b may be more important than Class 8a trucks, but more emissions data were available for Class 8a trucks. Class 8a trucks typically use similar types of engines as Class 8b, though the vehicle weight is less resulting in slightly lower predicted emissions levels.

The results are shown in the Tables 5-19 – 5-21 for each of the test cycles; the results are also shown graphically for NOx emissions in Figure 5-3. The data indicated that MOBILE6 predictions, shown in Table 5-22 and Figure 5-3, are quite close to the test data for all pollutants. Emissions of NOx were more likely to be overpredicted for older model years and underpredicted for newer model years. For very old model year trucks, the low end of the NOx emission prediction range was equivalent to the data available on average.

Table 5-19. HDDV Class 8a on CSHVR (~14 mph).

Model Year	Counts	THC (g/mile)	CO (g/mile)	NOx (g/mile)	PM (g/mile)
<1988	0	-	-	-	-
1988 – 1990	0	-	-	-	-
1991 - 1993	3	0.94 " 0.53	12.3 " 9.1	21.0 " 11.8	1.08 " 0.53
1994 - 1997	5	0.85 " 0.42	11.2 " 4.0	26.3 " 6.0	0.68 " 0.32
1998	2	1.52	16.0	43.2	0.48
1999-2001	3	1.15 " 0.80	10.4 " 2.0	14.4 " 3.9	0.65 " 0.08

Table 5-20. HDDV Class 8a on UDDS – Test D (~19 mph).

Model Year	Counts	THC (g/mile)	CO (g/mile)	NOx (g/mile)	PM (g/mile)
<1988	13	4.28 " 2.35	27.6 " 7.3	25.9 " 4.7	2.83 " 0.71
1988 – 1990	7	4.36 " 4.62	15.8 " 4.4	20.3 " 2.4	2.22 " 0.47
1991 - 1993	7	0.58 " 0.24	6.7 " 3.2	14.3 " 2.2	0.89 " 0.11
1994 - 1997	7	0.90 " 0.65	11.1 " 4.8	19.1 " 5.1	1.08 " 0.49
1998	3	0.62 " 0.31	5.0 " 3.0	21.9 " 7.3	0.66 " 0.32
1999-2001	1	0.85	7.5	22.2	0.52

Table 5-21. HDDV Class 8a on WVU (~21 mph).

Model Year	Counts	THC (g/mile)	CO (g/mile)	NOx (g/mile)	PM (g/mile)
<1988	2	1.37	6.53	17.7	1.24
1988 – 1990	4	0.91 " 0.33	9.8 " 9.3	16.3 " 1.8	1.23 " 0.33
1991 - 1993	4	1.14 " 0.55	4.0 " 2.1	13.7 " 3.2	0.61 " 0.13
1994 - 1997	2	1.35	3.3	17.0	0.57
1998	3	0.55 " 0.03	3.3 " 1.6	19.7 " 11.7	0.37 " 0.22
1999-2001	0	-	-	-	-

Table 5-22. MOBILE6 HDDV Class 8a on an arterial (19 mph).

Model Year	THC (g/mile)	CO (g/mile)	NOx (g/mile)
<1988*	3.2 – 9.4	23 – 25	24 – 40
1988 – 1990	1.6	10.5	21.5
1991 – 1993	0.9	6.0	19.2
1994 – 1997	0.7	3.5	20.1
1998	0.8	3.3	18.7
1999 – 2000	0.7	3.2	12.7

* Low end of range typical of 1979-1988; and high end 1978 and earlier.

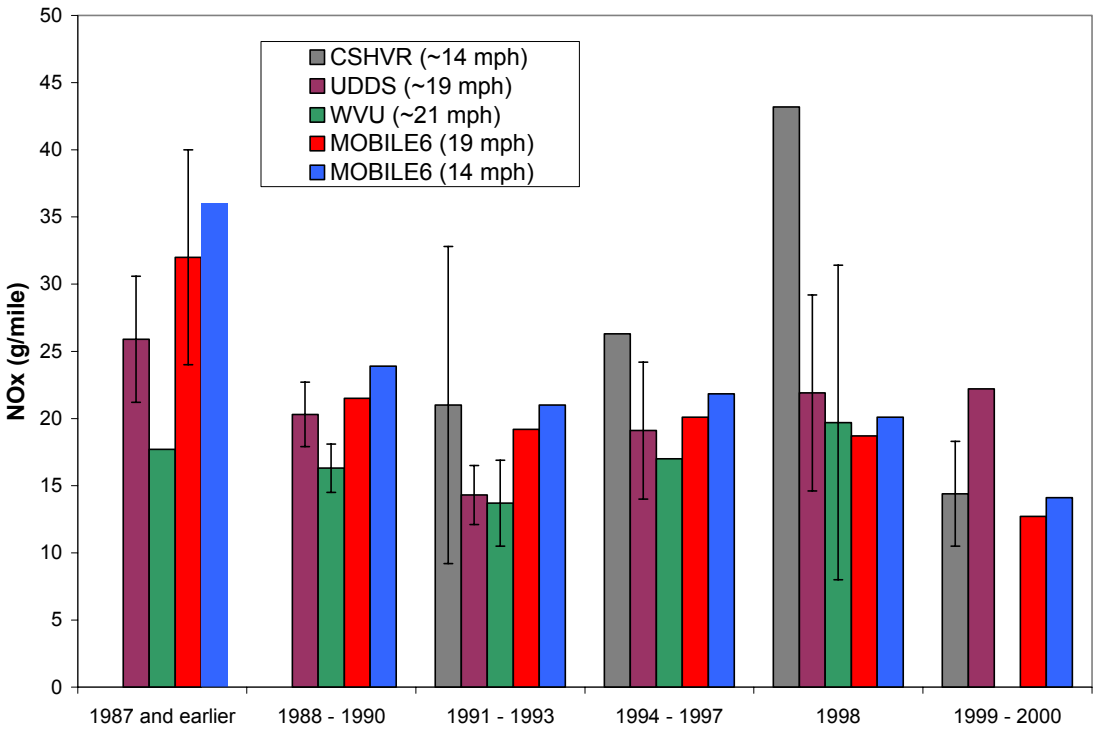


Figure 5-3. Comparison of NOx emission rates by model year for Class 8a trucks on arterials driving at or about 19 mph.

Two studies (Clark et al., 2002b and WVU, 2002) used two different test cycles with average speeds of about 40 mph. As described in the introduction, the emissions on these two cycles were roughly equivalent for the three vehicles tested on both of these cycles. The higher speed cycles allowed a more direct comparison of the effect of defeat device on in-use emissions because the comparison here is with MOBILE6 on a freeway facility type where the defeat device was expected to be in greater use.

The results, shown in Tables 5-23 and 5-24 (test data) and 5-25 (MOBILE6 predictions), indicate close agreement on THC emissions, and MOBILE6 underprediction of CO emissions. The NOx emission results are shown graphically in Figure 5-4, and indicate approximate agreement for model years before 1994 with the MOBILE6 estimates, but the later model years and especially those of the 1998 model year were significantly underpredicted by MOBILE6.

Table 5-23. HDDV Class 8a on NREL Highway (~ 40 mph).

Model Year	Counts	THC (g/mile)	CO (g/mile)	NOx (g/mile)	PM (g/mile)
<1988	0	-	-	-	-
1988 – 1990	0	-	-	-	-
1991 - 1993	3	0.48 " 0.26	4.5 " 1.3	21.1 " 12.9	0.45 " 0.08
1994 - 1997	5	0.39 " 0.15	5.4 " 2.4	22.9 " 3.9	0.30 " 0.14
1998	1	0.68	6.0	48.1	0.22
1999-2001	3	0.44 " 0.35	4.7 " 1.0	12.9 " 2.4	0.27 " 0.04

Table 5-24. HDDV Class 8a on ARB Cruise (~ 40 mph).

Model Year	Counts	THC (g/mile)	CO (g/mile)	NOx (g/mile)	PM (g/mile)
<1988	2	0.94	8.7	23.5	0.71
1988 – 1990	2	0.21	5.6	18.6	0.71
1991 - 1993	2	0.24	11.1	17.0	0.61
1994 - 1997	3	0.37 " 0.19	2.5 " 1.0	23.6 " 4.2	0.16 " 0.06
1998	2	0.28	3.7	36.6	0.29
1999-2001	1	0.44	3.7	18.4	0.22

Table 5-25. MOBILE6 HDDV Class 8a on an arterial (40 mph freeway).

Model Year	THC (g/mile)	CO (g/mile)	NOx (g/mile)
<1988*	1.7 – 5.1	11 – 12	21 – 35
1988 – 1990	0.86	5.0	19.1
1991 – 1993	0.48	2.9	17.5
1994 – 1997	0.39	1.7	18.5
1998	0.36	1.6	17.4
1999	0.35	1.6	11.5
2000	0.35	1.5	11.4

* Low end of range typical of 1979-1988; and high end 1978 and earlier.

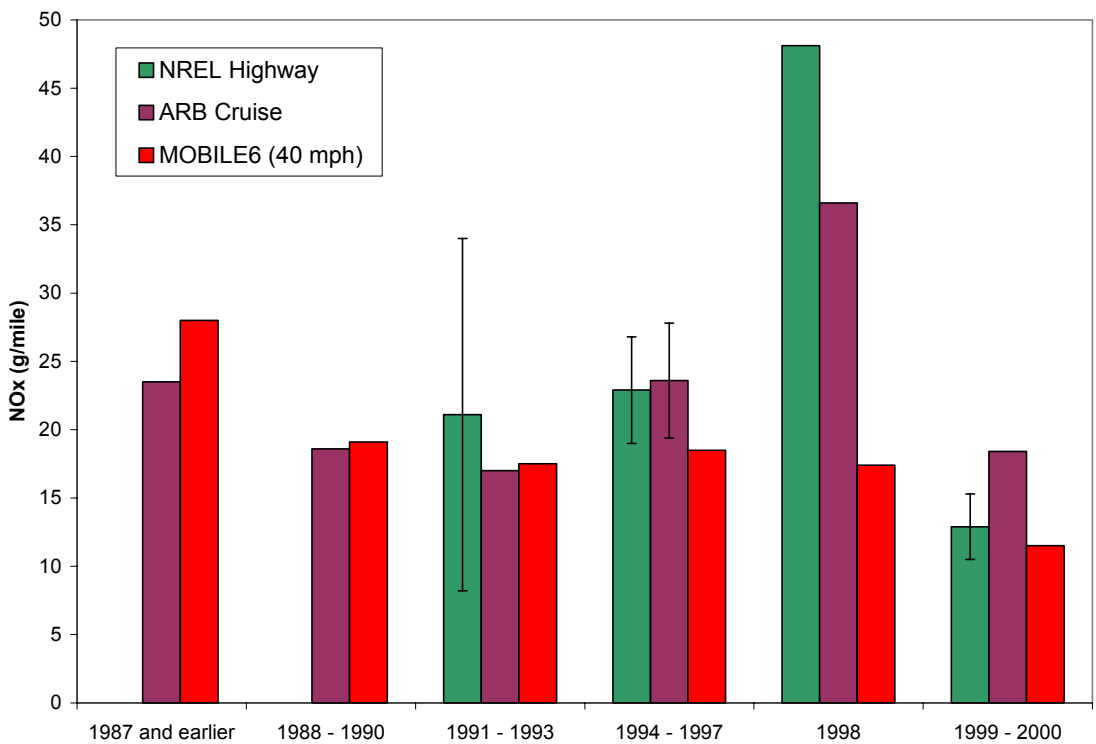


Figure 5-4. Comparison of NOx emission rates by model year for Class 8a truck driving on a freeway with an average speed of 40 mph.

Class 8b trucks typically represent the single most important category of HDVs, but unfortunately less data were available for this category. The available data are shown in Tables 5-26 and 5-27 for comparison with the MOBILE6 estimates in Table 5-28. Except for the high emitters for the two vehicles of 1989 and 1990 model years, MOBILE6 reasonably predicted the results for THC, CO, and NOx, except for the 1998 model year where NOx emissions were measured higher than MOBILE6 predicted.

Table 5-26. HDDV Class 8b on UDDS – Test D.

Model Year	Count	THC (g/mile)	CO (g/mile)	NOx (g/mile)	PM (g/mile)
<1988	8	3.45 " 1.50	30.9 " 8.4	36.2 " 7.1	2.13 " 0.76
1988 – 1990*	2	13.42	47.8	23.6	7.70
1991 - 1993	0	-	-	-	-
1994 - 1997	3	1.04 " 1.46	5.4 " 3.7	22.0 " 6.8	0.79 " 0.26
1998	2	0.50	3.1	37.5	0.52

*Both vehicles might be considered high emitters for THC, CO, and PM

Table 5-27. HDDV Class 8b on WVU (~21 mph).

Model Year	Counts	THC (g/mile)	CO (g/mile)	NOx (g/mile)	PM (g/mile)
<1988	2	3.44	14.2	18.3	2.41
1988 – 1990	1	1.85	8.8	16.1	0.95
1991 - 1993	2	1.36	5.2	15.8	0.70
1994 - 1997	4	0.65 " 0.75	4.5 " 1.9	18.3 " 0.8	0.59 " 0.22
1998	2	0.40	1.8	25.5	0.36

Table 5-28. MOBILE6 HDDV Class 8b on an arterial (19 mph).

Model Year	THC (g/mile)	CO (g/mile)	NOx (g/mile)
<1988*	3.3 – 13.5	29 – 34	25 – 52
1988 – 1990	1.8	14.9	24.5
1991 – 1993	1.0	6.8	24.6
1994 – 1997	0.8	4.0	22.6
1998	0.8	3.7	21.4
1999 – 2000	0.7	3.5	14.2

* Low end of range typical of 1979-1988; and high end 1978 and earlier.

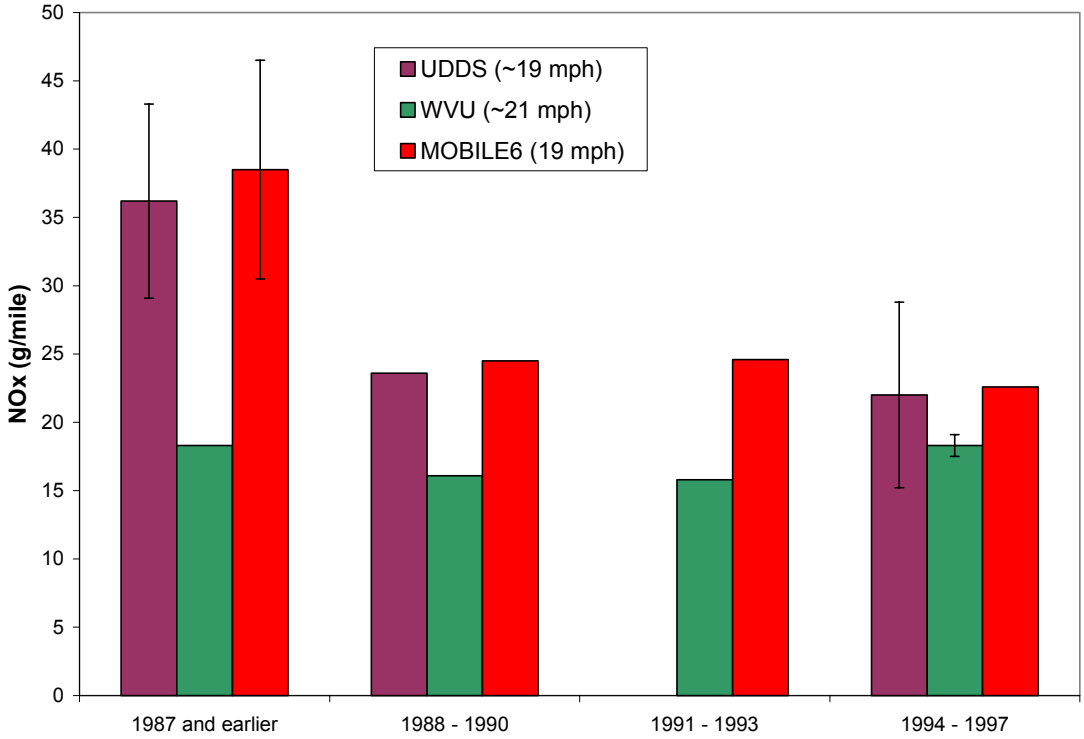


Figure 5-5. Comparison of NOx emission rates by model year for Class 8b trucks on arterials driving at 19 mph.

Transit Buses

The primary database for transit buses was generated using the Central Business District (CBD) test cycle, with additional data on the New York Composite Cycle with a similar average speed of about 9 mph. The test data averages, shown in Tables 5-29 and 5-30, are similar to the MOBILE6 estimates shown in Table 5-31 for NOx and lower for THC and CO emissions.

Table 5-29. Transit Bus on CBD Test Cycle (~ 9 mph).

Model Year	Counts	THC (g/mile)	CO (g/mile)	NOx (g/mile)	PM (g/mile)
<1988	2	1.81	16.5	41.5	1.49
1988 – 1990	3	1.82 " 0.16	15.5 " 6.0	34.6 " 9.2	1.63 " 0.56
1991 - 1993	3	1.83 " 0.78	7.8 " 2.1	26.0 " 5.2	0.94 " 0.49
1994 - 1997	3	0.39 " 0.45	4.1 " 0.8	29.4 " 1.8	0.39 " 0.25
1999	1	0.22	2.0	24.5	0.22

Table 5-30. Transit Bus on NY Composite Test Cycle (~ 9 mph).

Model Year	Counts	THC (g/mile)	CO (g/mile)	NOx (g/mile)	PM (g/mile)
<1988	2	2.30	20.6	24.1	1.61
1988 – 1990	3	2.19 " 1.26	10.9 " 7.0	21.3 " 5.0	2.21 " 0.92
1991 – 1993	1	1.06	2.2	12.7	0.64
1994 – 1997	2	0.59	5.4	26.6	0.35
>1997	0	-	-	-	-

Table 5-31. MOBILE6 Transit Bus on an arterial (9 mph).

Model Year	THC (g/mile)	CO (g/mile)	NOx (g/mile)
<1988*	5 – 14	42 – 44	33 – 51
1988 – 1990	3.2	12.8	31.9
1991 – 1993	3.3	22.2	24.3
1994 – 1997	0.54	8.4	26.2
1998	0.52	8.5	21.2
1999 – 2000	0.52	8.5	21.2

* Low end of range typical of 1979-1988; and high end 1978 and earlier.

Some data were also available for earlier model years on the higher speed UDDS – Test D test cycle demonstrating equivalent emissions for all pollutants between the available data (Table 5-32) and the MOBILE6 predictions (Table 5-33).

Table 5-32. Transit Bus on UDDS – Test D (~ 19 mph).

Model Year	Counts	THC (g/mile)	CO (g/mile)	NOx (g/mile)	PM (g/mile)
<1988	3	2.12 " 0.70	36.2 " 29.7	34.4 " 14.4	1.89 " 1.11
1988 – 1990	3	1.04 " 0.73	11.9 " 7.6	19.7 " 8.7	2.00 " 0.20
1991 - 1993	2	0.24	10.4	23.2	1.17
1994 - 1997	0	-	-	-	-
>1997	0	-	-	-	-

Table 5-33. MOBILE6 Transit Bus on an arterial (19 mph).

Model Year	THC (g/mile)	CO (g/mile)	NOx (g/mile)
<1988*	3.3 – 9.0	22 – 24	25 – 38
1988 – 1990	2.1	6.8	24.0
1991 – 1993	2.1	11.9	18.3
1994 – 1997	0.4	4.5	19.7
1998	0.3	4.5	16.0
1999 – 2000	0.3	4.5	16.0

* Low end of range typical of 1979-1988; and high end 1978 and earlier.

HEAVY HEAVY-DUTY VEHICLE COLD START INFORMATION

Cold start emissions were analyzed to provide a basis for determining if emission rates for starts should be explicitly modeled as they are for LDVs in MOBILE6. Data for cold starts could be important depending upon the number of starts that vehicles have per day and the typical mileage driven per day. As shown for LHDDV, starts were measurable, and given that some heavier vehicles are used as delivery trucks, the number of starts combined with the start emissions may be sufficient to affect overall emissions estimates.

There was additional cold start information for 27 vehicles tested over various test cycles under hot running and cold start conditions. These tests included vehicles with GVWR of 11,000 to 60,000 lbs and model years 1981 through 1999. The test cycles include the UDDS, the WVU 5-mile/5-peak, the CBD, and the CSHVR. The start increment was determined by using the difference between hot running and cold start emission rates multiplied by the distance of each test cycle. All vehicles were averaged together; results are shown in Table 5-34.

One vehicle from WVU (2002) was a significant outlier for cold start NOx emissions, so results are provided with and without that vehicle. With the removal of that vehicle, cold start emissions were found to be significant for NOx as well as all other pollutants with or without the outlier. This vehicle had a 1998 Detroit Diesel engine, so an issue with its engine control system may have influenced the test results.

Table 5-34. Cold start increment.

Model Year	THC (g/start)	CO (g/start)	NOx (g/start)	PM (g/start)
Start Increment	2.55" 1.55	10.16" 4.60	2.13" 10.33	2.44" 1.01
Removing NOx Outlier Vehicle				
Start Increment	2.60" 1.61	9.70" 4.71	7.98" 2.75	2.39" 1.04

To illustrate the effect of model year and GVWR, and to show the NOx outlier value for the 1998 model year vehicle, on the cold start emissions Figures 5-6 and 5-7 were prepared. No trends are readily observable with either model year or GVWR.

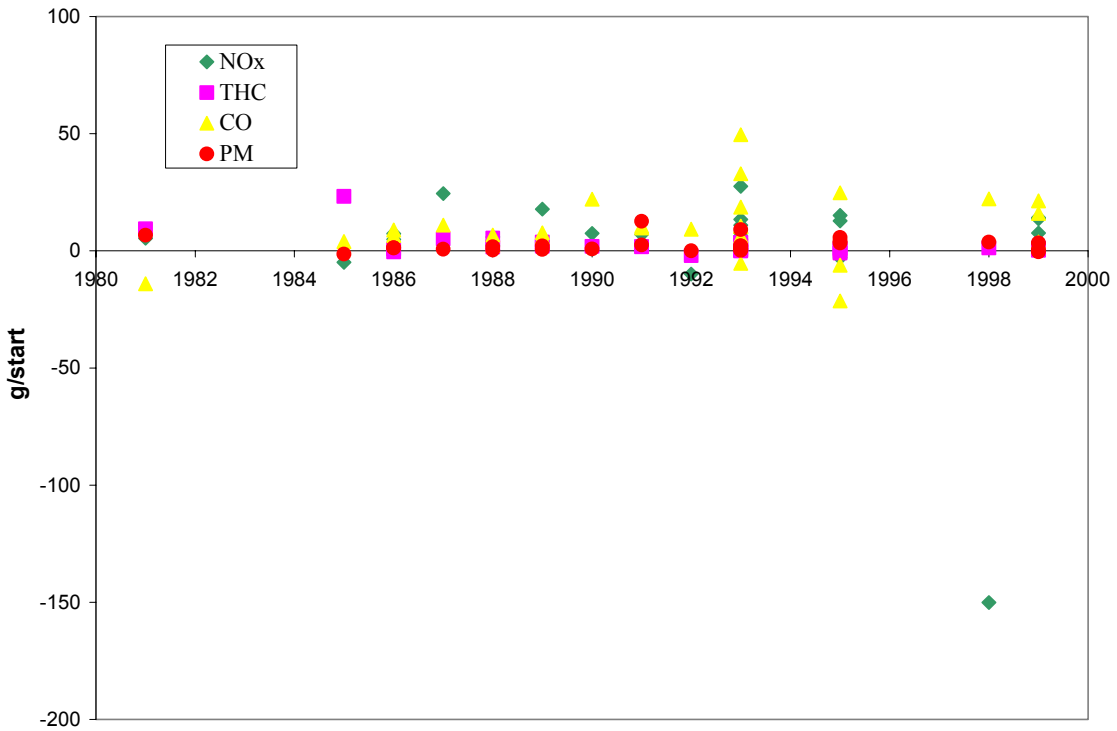


Figure 5-6. HDDV start increment by model year.

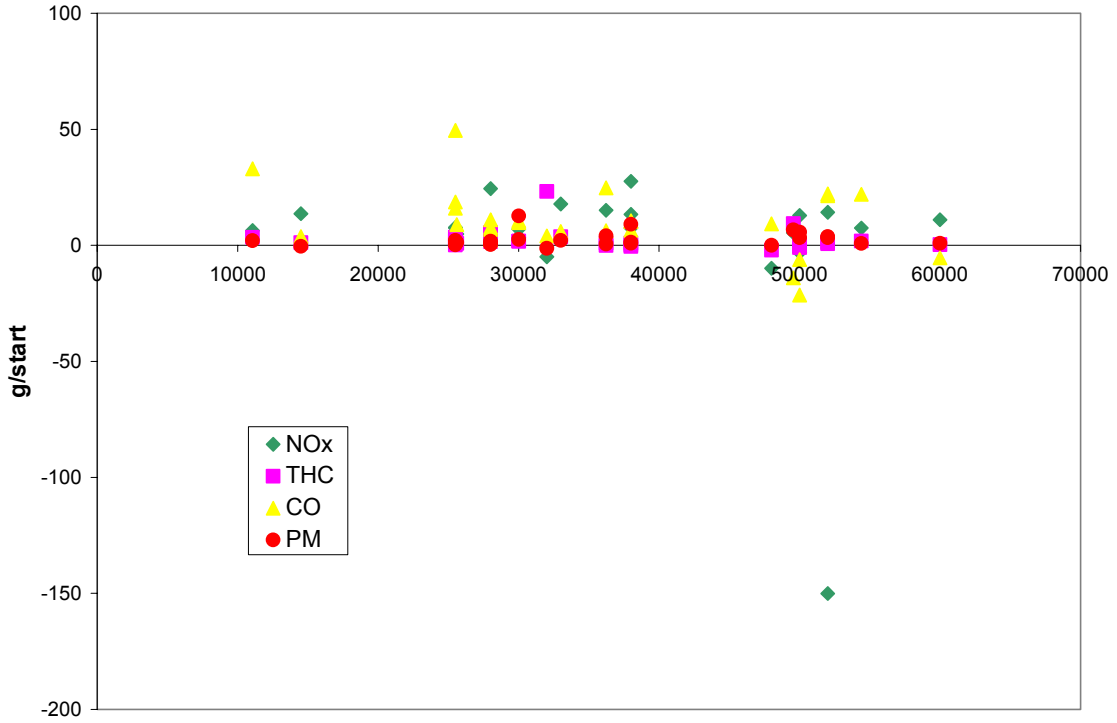


Figure 5-7. HDDV start increment by gross vehicle weight rating.

Other than specific engine controls influencing the results; the effect of a cold start is to increase the emissions of all pollutants. MOBILE6 does not explicitly model cold starts, but these results indicate that a cold start effect should be considered even for diesel engines without catalytic after treatment.

SUMMARY

There was not sufficient chassis data to allow a complete comparison for all vehicle types and all model years, but the database can inform the average estimates and provides verification of the general estimates. The methodology used by MOBILE6 has been questioned because of the use of certification results, energy conversion factors, and other adjustments that represent in large measure, "engineering judgment." Chassis data represent a closer approximation to expected in-use emissions rates.

One vehicle type where there was sufficient data to demonstrate the effect of model year on the emissions level was for Class 2b HDVs. The results indicate that while the emissions levels for HC and CO are similar for most model years, the NOx emissions were overpredicted for earlier model years and underpredicted for late model vehicles. The data also indicate that the make of the vehicle and engine could be important to the emission estimate.

For trucks and buses heavier than Class 2b, the emission rates for the chassis data were quite close to those predicted by MOBILE6 for most vehicle types and model years. There was an indication that THC and CO high emitters exist, and late model (1994 and later) vehicles may have higher NOx than predicted. The THC and CO high emitters could be a concern for estimates for toxic emissions and is likely a concern for future estimates of PM emissions. The NOx emissions were underpredicted for late model vehicles and overpredicted for older model year vehicles. EPA (1999b) had estimated that engine controls were programmed to artificially increase NOx emissions for late model vehicles, yet the chassis data indicate that the increase in NOx emissions was insufficient to model the in-use emission results. Because of the low number of vehicles tested, however, the data may not be entirely representative of the in-use fleet such as by type of engines in-use or other reasons associated with the paucity of data.

The high NOx emissions for late model vehicles highlight a need for further investigation because these vehicles will be used for many years to come. The studies used in this work were typically performed before 2002, and many of the diesel engines in the late model vehicles may be subject to low NOx rebuild programs when they are rebuilt. So the NOx levels measured in these studies may represent pre-rebuild conditions.

Individual high THC and CO emitters were identified and likely indicate that high PM emitters may also exist. The difficulty with high emitters is to determine the fraction of the in-use fleet that exhibits this behavior to determine the overall impact on emissions that these high emitters have.

Interestingly, there were enough data on the effect of the cold starts to demonstrate that cold start is a measurable effect and estimates were quantified. MOBILE6 explicitly estimates cold

start emissions for LDVs, and cold starts could be important for HDVs as well. Future versions of the model should include a cold start effect on HDDVs accompanied with default data about the typical number of starts for each type of vehicle.

6. COMPARISON OF DIESEL FUEL SALES DATA WITH MOBILE6 FUEL CONSUMPTION

INTRODUCTION

Typically for the development of an emissions inventory, MOBILE6 emission factors (in units of grams per mile) are combined with estimates of vehicle miles traveled. It has been suggested that diesel fuel sales could be used as another and perhaps more accurate measure of highway diesel vehicle activity (Dreher and Harley, 1998). The use of fuel sales (an indicator of fuel consumption) as the activity indicator would necessitate the use of fuel-based emission factors that could be derived from MOBILE6 or from field measurements using either tunnel studies or remote sensing.

In order to estimate fuel consumption, activity rates in terms of vehicle miles traveled (VMT) were used and combined with MOBILE6 fuel economy estimates. VMT estimates were derived for each state using the Federal Highway Administration's Highway Statistics 1999 (see NEI, 2002). The MOBILE6 fuel economy estimates were developed to convert brake-specific engine emissions rates (grams per horsepower-hour) to vehicle emission rates (grams per mile) using the following equation:

$$EF\ (g/mile) = EF\ (g/hp\text{-}hr) * D / (FE * BSFC)$$

Where FE = fuel economy (miles/gallon)
BSFC = brake specific fuel consumption (lb./hp-hr)
D = fuel density (lb./gal.)

Therefore, MOBILE6 implicitly uses fuel consumption in the calculation of emission factors, so the accuracy of the emission factors estimates depend in part upon the accuracy of the fuel consumption rate estimates.

An alternative source of fuel sales information is available from the Department of Energy's Energy Information Administration (EIA, 2001). Fuel sales for highway diesel are available for each state to provide a state-by-state comparison of fuel consumption estimates and sales. The data are available for both diesel and gasoline fuel sales. However, highway gasoline is used in engines other than highway vehicles such as recreational marine (Price Waterhouse, 1992) and other off-road use. Because of the highway tax and sulfur restrictions on highway diesel, the cost of highway diesel fuel is higher than for off-road diesel, so highway diesel fuel is expected to be used primarily or exclusively in highway vehicles.

The primary purpose of this work was to attempt to verify highway-diesel fuel consumption estimates using national and state VMT estimates combined with MOBILE6 fuel consumption rates with fuel sales information available from DOE-EIA. This comparison provides a reasonable analysis of the accuracy of the national HDDV activity estimates and whether individual state estimates could be considered accurate for state or regional inventories.

DOE-EIA FUEL SALES INFORMATION

EIA determines fuel sales information from surveys of fuel suppliers, who were asked to supply fuel sales information by end user, so highway fuel could be distinguished from other diesel fuel uses. However, EIA did not survey independent fuel dealers such as truck stops, so EIA used highway diesel fuel sales estimates derived from the Federal Highway Administration.

There was additional information about other uses of distillate fuels including off-road engines and equipment, residential or industrial heating. But there was no independent data indicating that fuel sold for use in highway vehicles was sold as off-road, or whether fuel sold for off-road use was used in highway vehicles. Even though off-road diesel fuel typically costs less than highway diesel fuel because of highway fuel taxes and fuel sulfur limits on highway diesel fuel, it is illegal to sell and consume off-road fuel in highway vehicles.

As a verification of the EIA fuel sales information, Dreher and Harley (1998) reported that California state tax records indicated that 2,100 million gallons of highway diesel fuel were sold in California in 1996, while EIA estimated 2,458 million gallons were sold in California in 1999. Accounting for economic and therefore truck activity growth from 1996 to 1999, the EIA estimates for California diesel fuel sales could be considered equivalent to the tax revenues. Also, John Nordlie (2003) reported that highway diesel fuel sales in Wisconsin as compiled from fuel distributors was 691.6 million gallons for the year ending June 30, 1999, compared with the EIA estimate of 672.2 million gallons. Therefore, the EIA estimates accurately reflected the fuel sales for these two states.

MOBILE6-VMT FUEL CONSUMPTION ESTIMATES

The basis for the MOBILE6 fuel consumption estimates was derived from EPA (1998) estimates used in the preparation of the HDDV emission rates, which in turn had been derived from the Census (1993) Bureau's 1992 Truck Inventory and Use Survey (TIUS, now called the Vehicle Inventory and Use Survey, or VIUS). These fuel consumption rates are shown in Table 6-1. Fuel consumption rates for LDD passenger cars and trucks were derived from EPA (2002c) for completeness though the overall fuel consumption of LDDVs was projected to be only about 1% of all diesel fuel consumed by highway vehicles. Fuel economy estimates were held constant for model years before 1987 and after 1996.

Table 6-1. Fuel economy estimates for HDDV used in MOBILE6 (miles per gallon).

Model Year	Vehicle Weight Class								Diesel Buses	
	2B	3	4	5	6	7	8A	8B	Transit	School
1987	11.69	10.52	9.56	9.12	8.20	7.43	5.96	5.51	3.94	6.29
1988	11.83	10.65	9.63	9.21	8.25	7.44	6.03	5.59	3.99	6.28
1989	11.97	10.77	9.70	9.29	8.31	7.45	6.10	5.68	4.04	6.27
1990	12.11	10.90	9.77	9.38	8.37	7.46	6.17	5.77	4.08	6.25
1991	12.26	11.03	9.85	9.46	8.42	7.47	6.24	5.86	4.13	6.24
1992	12.40	11.15	9.92	9.54	8.48	7.48	6.31	5.95	4.17	6.23
1993	12.54	11.28	9.99	9.63	8.54	7.49	6.38	6.03	4.22	6.22
1994	12.68	11.41	10.06	9.71	8.59	7.51	6.45	6.12	4.26	6.20

Model Year	Vehicle Weight Class								Diesel Buses	
	2B	3	4	5	6	7	8A	8B	Transit	School
1995	12.82	11.53	10.13	9.80	8.65	7.52	6.52	6.21	4.31	6.19
1996	12.96	11.66	10.20	9.88	8.71	7.53	6.59	6.30	4.36	6.18

As described in NEI (2002), EPA used the default MOBILE6 vehicle registrations to estimate national emissions for the 1999 National Emissions Inventory (NEI). The default vehicle registrations were combined with default mileage accumulation rates in MOBILE6 to estimate travel fractions by model year. The travel fractions by model year combined with the fuel economy estimates by model year shown above in Table 6-1 were used to calculate the average fuel consumption rates by vehicle type for all model years in 1999.

The VMT estimates were available for 1999 for each state by the general vehicle types listed in Table 6-2 from the 1999 NEI (NEI, 2002). The NEI documentation describes how the general vehicle types were disaggregated into the individual vehicle types in MOBILE6 using HPMS data. The vehicle types in the VMT database are shown in Table 6-2.

Table 6-2. General vehicle types used in the 1999 NEI (NEI, 2002).

NEI General Vehicle Types	EPA Class Distinction	Gross Vehicle Weight Rating	Class Fraction
Class 2b	Class 2b	8,500 to 10,000	100 % Class 2b
LHDDV	Class 3, 4, and 5	10,000 to 19,500	Class 3 ~ 48% Class 4 ~ 37% Class 5 ~ 15%
MHDDV	Class 6 and 7	19,500 to 33,000	Class 6 ~ 39% Class 7 ~ 61%
HHDDV	Class 8a and 8b	33,000 to 80,000	Class 8a ~ 22% Class 8b ~ 78%
HDDB	School and transit buses		School ~ 43% Transit ~ 57%

California VMT estimates were provided only for all HDVs combined, so MOBILE6 default values were used to disaggregate by individual vehicle type. The estimate of fuel consumption for California was much more uncertain because the assumptions about the vehicle type mix were not explicitly detailed.

RESULTS

The results of applying average fuel consumption estimates to the state VMT estimates are shown in Table 6-3 and in Figures 6-1 and 6-2. National fuel consumption estimates were 6.3% less than the fuel sales estimates excluding California, and 8.5% including California. The results indicate that individual states could have more or less fuel consumption than sales.

In terms of states running large deficits of fuel sales compared with fuel use, New York, Florida, North Carolina, and Michigan rank the most important. On a percentage basis the states with considerably more fuel consumed than sold are many of the northeast states

including New York, Massachusetts, Delaware, Rhode Island, and New Hampshire, though Hawaii ranks highest on a percentage basis. States with larger fuel sales than fuel consumed are led by Indiana (349 million gallons), Ohio (318 million gallons), Texas, Wyoming, Illinois, but on a percentage basis are led by Wyoming (67% less fuel consumed), Nebraska (32% less fuel consumed), Arkansas, Indiana, and Nevada.

Based on the information available, California had 36% more sales than consumption, or over 880 million gallons more sold than consumed. California has a unique fuel specification for diesel fuel that applies to both on-road and off-road sales, so fuel sales estimates may not distinguish well between on-road and off-road sales in that state. Given that it is illegal to use off-road diesel fuel in highway vehicles, off-road fuel sales may be recorded as on-road diesel fuel sales. But, as described above, the California fuel consumption estimate was more uncertain than those for other states. For California especially, it is therefore more difficult to determine if the conclusion that more highway diesel fuel is sold than used in highway vehicles is true.

The reasons for the state-to-state variability could be many fold. The price of the fuel may encourage fuel sales in one state over another, and the price may be influenced by road taxes, proximity to refineries or pipelines, or other reasons. Other reasons could be the proximity of trucking firm depots or other refueling sites. Reasons for the states to have more traffic than sales may seem straightforward; e.g., for Delaware, much of the truck traffic would be expected to be passing through the state along the interstate freeway.

Nationally, the two types of estimates were similar with a slight bias (<10%) toward higher sales than consumption. This verifies that the fuel economy estimates used in MOBILE6 to prepare the emission factors combined with the national VMT estimates are reasonably consistent with national fuel consumption estimates. However, given that the activity data for both fuel consumption and fuel sales were derived from FHWA data, it was surprising that the national estimates were any different at all, but they may have come from two different data sources.

Considering that the two data sources were from the same source, the state-to-state variability was even more unexpected. The state information about fuel sales was not well described by EIA (2001), but based on the information from Dreher and Harley (1998) and the State of Wisconsin, the FHWA (as reported by DOE-EIA) data reflect fuel tax revenues for California and Wisconsin. Therefore, based on the comparison of this work, state or regional fuel sales information is not comparable with state or regional estimates of fuel consumption on the basis of VMT activity estimates. More research is needed to determine which estimate, fuel sales or VMT, is more accurate for a given state or region, but it suggests that the fuel sales data by state does not reflect diesel vehicle activity within that state.

While this analysis was not a clear validation of MOBILE6 diesel vehicle emission rates, the heavy-duty diesel vehicle (the primary consumer of highway diesel fuel) fuel consumption rates are used in estimating per mile emission rates. To the extent that national fuel consumption is closely predicted (within 5 to 10 percent, biased low) using MOBILE6 and VMT estimates with the estimate of national fuel sales, one can be reasonably confident that MOBILE6 is accurately predicting fuel consumption and CO_2 emission rates. The state-by-

state fuel consumption rates, however, do not provide any confidence that fuel sales could be used as an alternative method for predicting emissions within a given state.

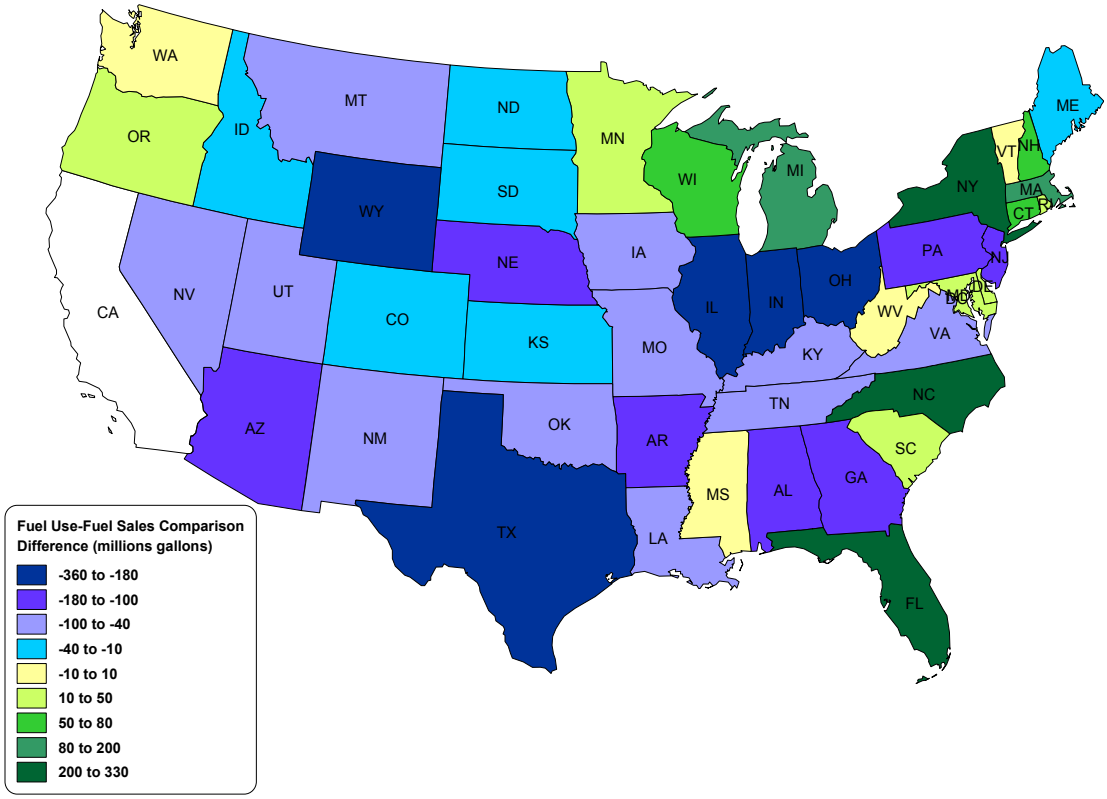


Figure 6-1. 1999 US Statewide difference in diesel fuel consumption and fuel sales.

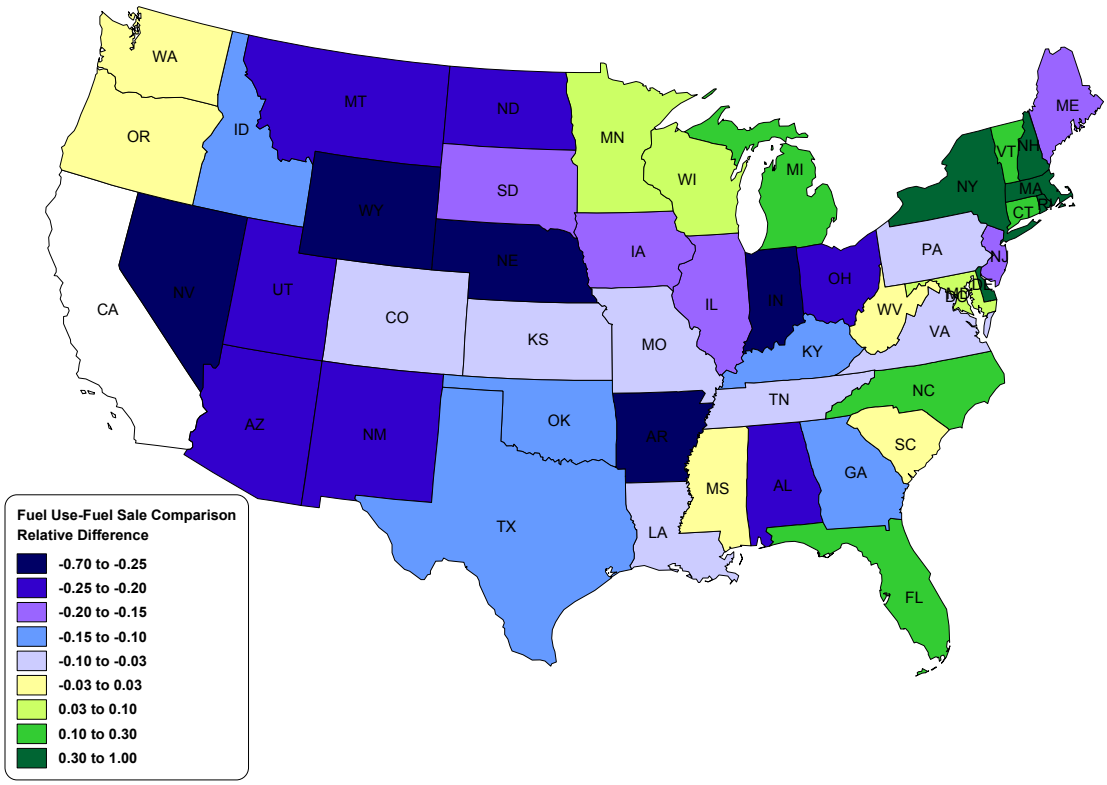


Figure 6-2. 1999 US Statewide relative difference in diesel fuel consumption and fuel sales.

Table 6-3. Fuel consumption estimates compared with fuel sales (millions of gallons).

State	Fuel Consumption	EIA	Difference	Difference in %
Alabama	578.1	742.1	(163.9)	-22%
Alaska	57.7	75.0	(17.4)	-23%
Arizona	524.8	660.6	(135.9)	-21%
Arkansas	400.8	556.2	(155.4)	-28%
California	*1574.9*	*2458.3*	*(883.4)*	*-36%*
Colorado	418.8	437.8	(18.9)	-4%
Connecticut	305.7	235.4	70.3	30%
Delaware	99.9	57.5	42.5	74%
District of Columbia	27.6	22.1	5.5	25%
Florida	1469.9	1239.4	230.5	19%
Georgia	1151.4	1322.7	(171.3)	-13%
Hawaii	86.7	32.7	54.0	165%
Idaho	191.2	214.7	(23.5)	-11%
Illinois	1093.7	1292.3	(198.7)	-15%
Indiana	879.7	1228.8	(349.2)	-28%
Iowa	400.5	494.8	(94.4)	-19%
Kansas	350.6	374.3	(23.7)	-6%
Kentucky	625.0	703.2	(78.3)	-11%
Louisiana	531.6	585.6	(54.0)	-9%
Maine	120.6	148.7	(28.2)	-19%
Maryland	531.8	486.4	45.5	9%
Massachusetts	503.7	377.1	126.6	34%
Michigan	1092.7	905.1	187.6	21%
Minnesota	630.8	604.5	26.2	4%
Mississippi	559.2	555.8	3.4	1%
Missouri	798.4	862.3	(63.9)	-7%
Montana	145.4	187.4	(42.0)	-22%
Nebraska	243.4	359.4	(116.0)	-32%
Nevada	194.3	261.6	(67.3)	-26%
New Hampshire	158.2	103.3	55.0	53%
New Jersey	638.7	769.0	(130.3)	-17%
New Mexico	305.4	398.6	(93.1)	-23%
New York	1339.5	1011.1	328.4	32%
North Carolina	1083.4	879.0	204.4	23%
North Dakota	106.5	141.3	(34.7)	-25%
Ohio	1198.8	1516.5	(317.6)	-21%
Oklahoma	528.5	616.3	(87.8)	-14%
Oregon	439.4	428.6	10.8	3%
Pennsylvania	1200.2	1327.9	(127.6)	-10%
Rhode Island	76.1	55.6	20.5	37%
South Carolina	603.9	590.8	13.1	2%
South Dakota	121.7	145.3	(23.6)	-16%
Tennessee	778.4	860.0	(81.6)	-9%
Texas	2278.0	2538.3	(260.3)	-10%
Utah	225.8	300.0	(74.2)	-25%
Vermont	96.2	87.0	9.2	11%
Virginia	864.2	952.6	(88.4)	-9%
Washington	573.2	568.9	4.3	1%
West Virginia	276.3	274.0	2.3	1%
Wisconsin	730.4	672.2	58.2	9%
Wyoming	114.2	344.6	(230.4)	-67%
TOTAL w/o California	**27,751**	**29,604**	**(1,853)**	**-6.3%**
TOTAL with California	**29,326**	**32,062**	**(2,737)**	**-8.5%**

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APPENDIX A

DRI Tunnel Study Locations And Run Descriptions

DRI TUNNEL STUDY LOCATIONS AND RUN DESCRIPTIONS

(written by Alan Gertler, Desert Research Institute)

During the period 1992 through 1999, DRI performed a series of on-road emissions studies in highway tunnels. These studies were supported by a number of organizations including API, AOAQIRP, CRC, Environment Canada, EPA, FHWA, HEI, NREL, SCAQMD, and SOS. Table A-1 lists the tunnel locations, length of the tunnels, tunnel classification (urban/interstate), and year the studies were performed.

Table A-1. Summary of DRI tunnel locations and year measurements performed.

Tunnel	Location	Length (m)	Fleet	Year
Fort McHenry Tunnel	Baltimore, Maryland	2174	Highway	1992, 1993, 1995
Tuscarora Mountain Tunnel	Pennsylvania Turnpike, Pennsylvania	1623	Highway	1992, 1999
Cassiar Connector	Vancouver, British Columbia	730	Urban	1993
Callahan Connector	Boston, Massachusetts	1545	Urban	1995
Deck Park Tunnel	Phoenix, Arizona	804	Urban	1995
Lincoln Tunnel	New York/New Jersey	2440	Urban	1995
Sepulveda Tunnel	Los Angeles, California	582	Urban	1995, 1996
Van Nuys Tunnel	Los Angeles, California	222	Urban	1995

Data for all the studies listed in Table A-1 may be used for comparing observed emissions with mobile source emission factor model predictions except for the 1995 Fort McHenry study sponsored by API. This project focused on measuring dioxin and furan emissions from the in-use fleet. Emissions of CO, HC, and NO_x were not measured. Results of the 1993 Fort McHenry study, sponsored by FHWA, are of limited use for comparing observed and predicted emissions. The only pollutant quantified in this study was PM_{10}. Descriptions of the tunnels follow.

Fort McHenry Tunnel, Baltimore (1992, 1993, 1995)

The Fort McHenry Tunnel is a four-bore tunnel, two lanes per bore, carrying Interstate 95 east-west under the Baltimore Harbor. The downgrade reaches –3.76% and the upgrade reaches +3.76%, with no significant level portion. Average grade from west portal to bottom is 1.8% and, from bottom to east portal, +3.3%. The four tunnel bores are designated 1 and 2 westbound (toward Washington, DC), and 3 and 4 eastbound (toward Philadelphia). The 1992 study was conducted in Bores 3 and 4, the eastbound bores (length 2174 meters), measuring in the two bores simultaneously (Table A-2). LD vehicles are allowed in both bores; however, trucks are directed into Bore 4, the right-hand bore and all but 3% of them complied in the June 1992 experiment. Posted speed was 50 mi/hr in the tunnel, 55 outside. Traffic flowed freely except for sporadic light braking/slowdown at the exit at rush hour during a few sampling runs. The nearest entrance ramps before the tunnel eastbound, and carrying any significant amount of traffic, range upwards of 2200 meters west of the entrance

portal; all of these ramps connect with arteries, not local streets and DRI concludes that the vehicles were in hot stabilized operation.

The ventilation system of the Fort McHenry Tunnel comprises two sections. Ventilation air from above each end of the tunnel is supplied through ducts beneath the roadway, and tunnel air is removed through overhead exhaust ducts. In addition, there is a dividing plane between the east and west supply ducts 95 meters before the low point of the tunnel. Thus DRI was able to measure emissions for the downhill, uphill, and total tunnel.

Descriptions of the 1993 and 1995 experiments are not presented, since they are of limited use for the current study. The 1993 study measured only PM emissions and gaseous emission rates were not determined. In the 1995 study, DRI focused on dioxin and furan emissions from HD vehicles.

Table A-2. Run description, Fort McHenry Tunnel, 1992.

	Run 1		**Run 2**		**Run 3**		**Run 4**		**Run 5**		**Run 6**		**Run 7**		**Run 8**		**Run 9**		**Run 10**		**Run 11**	
Bore	3	4	3	4	3	4	3	4	3	4	3	4	3	4	3	4	3	4	3	4	3	4
Date	18-Jun		19-Jun		19-Jun		20-Jun		21-Jun		21-Jun		22-Jun		23-Jun		23-Jun		24-Jun		24-Jun	
Day	Thu		Fri		Fri		Sat		Sun		Sun		Mon		Tue		Tue		Wed		Wed	
Start Time	1230		1030		1600		1200		1200		1600		1100		300		1300		400		1600	
T (°C)	24		21		25		24		20		20		17		17.5		22		20		21	
Av. Sp. (mph)	51		46		52		43		48		53		52		45		53		45		38	
Total Vehicles	356	1809	164	279	2519	2451	954	2052	995	2136	1960	1144	1265	938	102	262	1194	1041	125	257	2826	1836
F LD	0.99	0.79	0.98	0.32	0.99	0.90	0.99	0.95	0.99	0.96	1.00	0.92	0.99	0.62	0.98	0.28	0.98	0.66	0.98	0.28	0.96	0.88
F HD	0.01	0.21	0.02	0.69	0.01	0.10	0.01	0.05	0.01	0.04	0.00	0.08	0.01	0.38	0.02	0.73	0.02	0.35	0.02	0.72	0.04	0.12

Tuscarora Mountain Tunnel, Pennsylvania Turnpike (1992, 1999)

The Tuscarora Mountain Tunnel is a two-bore tunnel, two lanes each bore, 1623.2 meters (5325.4 ft) long, carrying the Pennsylvania Turnpike (Interstate 76) east-west through Tuscarora Mountain in south-central Pennsylvania at an altitude of ~305 meters. The tunnel is flat (grades +0.30% towards the middle from either end) and straight. Posted speed is 55 mi/hour both in and outside the tunnel. The nearest interchange west of the tunnel is 10 km west of the tunnel entrance. It is very lightly used. Other accesses from the west are the Sideling Hill service plaza (22 km to the west), the interchange with Interstate 70 (40 km to the west, heavily used), and other interchanges and service plazas farther west. Effectively the minimum trip length before reaching the tunnel is 15 minutes (much of it following hot start) and DRI estimates that trips longer than 50 minutes before reaching the tunnel constitute some 75% of all trips. Accordingly, cold-start and hot-start operations are inconsequential in Tuscarora eastbound. The Tuscarora Mountain Tunnel is ventilated entirely by the traffic piston effect and the prevailing westerly wind; there is a supply ventilation system but it was not operated during either the 1992 or 1999 experiments. Run descriptions for both studies are summarized in Tables A-3 and A-4.

Table A-3. Run description, Tuscarora Mountain Tunnel, 1992.

	Run 1	Run 2	Run 3	Run 4	Run 5	Run 6	Run 7	Run 8	Run 9	Run 10	Run 11
Date	2-Sep	2-Sep	3-Sep	4-Sep	5-Sep	6-Sep	6-Sep	7-Sep	7-Sep	8-Sep	8-Sep
Day	Wed	Wed	Thu	Fri	Sat	Sun	Sun	Mon	Mon	Tue	Tue
Start Time	300	1500	400	1700	1130	1130	1300	200	1300	800	2101
T (°C)	13	20.5	20.5	24	21	19	19	18.5	20.5	21	19.5
Av. Sp. (mph)	56	55	59	57	58	56	58	58	59	60	58
Total Vehicles	186	530	185	928	661	585	659	79	1329	435	351
F LD	0.242	0.736	0.200	0.909	0.920	0.916	0.921	0.734	0.940	0.703	0.590
F HD	0.758	0.264	0.800	0.091	0.080	0.084	0.079	0.266	0.060	0.297	0.410

Table A-4. Run Description, Tuscarora Mountain Tunnel, 1999.

	Run 1	Run 2	Run 3	Run 4	Run 5	Run 6	Run 7	Run 8	Run 9	Run 10	Run 11	Run 12	Run 13	Run 14	Run 15	Run 16	Run 17	Run 18	Run 19	Run 20
Date	18-May	18-May	18-May	19-May	19-May	19-May	19-May	19-May	20-May	20-May	21-May	21-May	21-May	21-May	22-May	22-May	22-May	22-May	23-May	23-May
Day	Tue	Tue	Tue	Wed	Wed	Wed	Wed	Wed	Thur	Thur	Fri	Fri	Fri	Fri	Sat	Sat	Sat	Sat	Sun	Sun
Start Time	1200	2000	2200	0000	200	1900	2100	2300	100	1600	500	700	900	1700	1100	1300	1500	1700	1000	1200
Av Spd. (mph)	54.9	54.8	57	54.9	55.1	57.7	54.4	53.6	55	53.2	58.1	57.5	53.8	56.9	57	56.5	57	59.5	58.1	61.7
Total Vehicles	529	385	293	206	192	454	359	252	201	730	248	402	473	814	554	539	488	442	529	1681
LD	334	177	104	31	26	240	148	70	43	505	88	208	366	706	490	444	406	377	435	1400
HD (4-6)	24	11	10	4	10	14	20	4	6	23	9	27	17	16	11	15	12	14	14	29
HD (7-8)	171	197	179	171	156	200	191	178	152	202	151	167	90	92	53	80	70	51	80	252
F LD	0.631	0.460	0.355	0.150	0.135	0.529	0.412	0.278	0.214	0.692	0.355	0.517	0.774	0.867	0.884	0.824	0.832	0.853	0.822	0.833
F HD (7-8)	0.323	0.512	0.611	0.83	0.813	0.441	0.532	0.706	0.756	0.277	0.609	0.415	0.19	0.113	0.096	0.148	0.143	0.115	0.151	0.15

Cassiar Connector, Vancouver (1993)

The Cassiar Connector is an urban two-bore tunnel 730 meters in length, with two lanes of traffic per bore. It is situated on the Trans-Canadian Highway, Highway 1, in Vancouver, BC. Traffic is generally heavy during the day with an average speed of around 90 km/h. During this study, hourly traffic counts ranged from around 100 vehicles during the early morning hours to almost 3000 vehicles during the afternoon rush hours. The grade varies from +1.66% at the south end of the tunnel to -1.29% at the north end. The nearest entrance ramps before the tunnel are over 1,000 meters to the south and connect with major arteries. Cold-start operation should therefore be minimal in the tunnel. Ventilation for the tunnel is achieved from the piston effect of the vehicles traversing it, and from the fans positioned along the ceiling throughout the tunnel. The fans were used only when high levels of CO were present in the tunnel. They were never activated throughout the course of this study. The area surrounding the tunnel is primarily residential at both the north and south ends of the tunnel. There is one major urban street located approximately over the middle of the tunnel. Descriptions of the sixteen runs are presented in Table A-5.

Table A-5. Run description, Cassiar Connector, Vancouver.

	Run 1	Run 2	Run 3	Run 4	Run 5	Run 6	Run 7	Run 8	Run 9	Run 10	Run 11	Run 12	Run 13	Run 14	Run 15	Run 16
Date	13-Aug	13-Aug	13-Aug	13-Aug	14-Aug	15-Aug	16-Aug	16-Aug	16-Aug	18-Aug	18-Aug	18-Aug	18-Aug	18-Aug	18-Aug	18-Aug
Start Time	200	600	1000	1500	900	900	200	600	800	200	600	800	1000	1200	1400	1600
T (°F)	56.1	55.9	59.7	62.6	59.5	59.5	57.6	58.3	59.5	55	56.1	62.4	66.9	68	70.3	72.5
Av. Sp (mph)	57.9	59.1	56.6	57	57.9	57.4	58.8	59.7	57.2	57	60	56.8	55.7	55.9	56.1	55.6
Std. Dev (mph)	9.1	12.6	12.8	17.2	14	10.1	5.7	22.1	18.8	6.5	9.6	15.4	14.1	16.9	18.3	22
Total Vehicles	125	1678	1821	2502	1470	948	93	1622	1859	100	1650	2074	1769	1850	1977	2975
LDSI	111	1532	1607	2354	1356	897	75	1434	1605	90	1471	1837	1546	1638	1800	2866
HDSI	4	58	79	81	52	39	4	86	121	2	76	108	110	99	67	66
HDD	10	88	135	67	62	12	14	102	133	8	103	129	113	113	110	43
F LD	0.888	0.913	0.882	0.941	0.922	0.946	0.806	0.884	0.863	0.900	0.892	0.886	0.874	0.885	0.910	0.963
F HD	0.112	0.087	0.118	0.059	0.078	0.054	0.194	0.116	0.137	0.100	0.108	0.114	0.126	0.115	0.090	0.037

Callahan Tunnel, Boston (1995)

The Callahan Tunnel, 1545 m in length, is the eastbound tunnel of a pair of tunnels (Sumner and Callahan) carrying traffic between North Boston and East Boston and Logan International Airport. It is a one-bore tunnel with two lanes in the bore. There is no toll plaza on the Callahan Tunnel, which makes the traffic flow slightly smoother; although there was significant variability in the observed average speed for the ten experimental periods (Table A-6). The tunnel ventilation is transverse in design, similar to other underwater tunnels. The Callahan ventilation buildings are placed virtually right at the portals, which greatly simplified the experiment. Both the Sumner and Callahan tunnels are controlled from a single control building in East Boston. Ventilation fans are on virtually all the time although during the experiment DRI observed several times when the supply air was not on. Actual airflow was monitored continuously with anemometers. The ventilation system in the Callahan Tunnel is divided into two sections, each with a separate blower (fresh air) and exhaust duct. With the addition of the inlet portal and exit portal made a total of six samples per run.

Table A-6. Run description, Callahan Tunnel, Boston.

	Run 1	Run 2	Run 3	Run 4	Run 5	Run 6	Run 7	Run 8	Run 9	Run 10
Date	18-Sep	18-Sep	18-Sep	18-Sep	19-Sep	19-Sep	19-Sep	19-Sep	19-Sep	19-Sep
Day	Mon	Mon	Mon	Mon	Tues	Tues	Tues	Tues	Tues	Tues
Start Time	1100	1300	1500	1700	600	800	1000	1200	1400	1600
T (ºC)	20.0	20.6	18.3	17.2	10.0	13.3	16.1	16.7	17.8	17.2
Avg. Speed (mph)	30.2	27.0	14.1	24.3	30.8	35.3	32.1	30.7	24.0	15.2
Std Dev (mph)	5.2	6.7	4.6	7.4	4.3	5.0	5.5	4.9	5.9	2.3
Total Vehicles	2943	3072	3437	3189	3247	1988	2437	2677	3436	3498
Total LD	2824	2934	3350	3116	3151	1858	2332	2553	3334	3414
Total HD	119	138	87	73	96	130	105	124	102	84
F LD	0.960	0.955	0.975	0.977	0.970	0.935	0.957	0.954	0.970	0.976
F HD	0.040	0.045	0.025	0.023	0.030	0.065	0.043	0.046	0.030	0.024

Deck Park Tunnel, Phoenix (1995)

The Deck Park Tunnel is a 3-bore, urban freeway tunnel 804 m in length, running east/west under Deck Park in downtown Phoenix. The center bore is unused and there are plans to complete it for use as a bus station. There are five lanes and two emergency lanes in the south and north bores. The tunnel has complex ventilation, with fans at each end that can provide either supply or exhaust air. The fans were shut down prior to each run. In both experiments, samplers were located in the center bore and samples were collected from the north side of the south bore. One problem with the Deck Park Tunnel was its large cross section (217 m^2 at the narrowest point). This complicated the sampler placement. Sampling proved problematic for two reasons: air flow inhomogeneities and concentration gradients across the tunnel. This was resolved in the summer experiment through the use of a non-reactive tracer (SF_6) to characterize the airflow in the tunnel. While results of the winter could be corrected, they have a higher degree of uncertainty than those obtained in the other tunnel studies. Descriptions of the eight January experimental runs and nine July experimental runs are presented in Tables A-7 and A-8, respectively.

Table A-7. Run description, Deck Park Tunnel, Phoenix, January 1995.

	Run 1	**Run 2**	**Run 3**	**Run 4**	**Run 5**	**Run 6**	**Run 7**	**Run 8**
Date	24-Jan	24-Jan	24-Jan	25-Jan	25-Jan	26-Jan	26-Jan	26-Jan
Day	Tues	Tues	Tues	Wed	Wed	Thur	Thur	Thur
Start Time	600	800	1600	600	800	600	800	1000
T (ºC)	12.6	13.2	21.6	18.7	17.8	13.3	14.4	14.7
Avg. Speed (mph)	59.8	58.2	59.8	59.0	56.3	59.3	57.3	60.1
Std Dev (mph)	3.8	3.8	5.2	4.2	4.5	4.7	4.4	4.8
Total Vehicles	7330	5770	7210	7300	7740	6980	6798	4613
Total LD	7132	5650	7052	7094	7400	6752	6488	4344
Total HD	198	120	158	206	340	228	310	269
F LD	0.973	0.979	0.978	0.972	0.956	0.967	0.954	0.942
F HD	0.027	0.021	0.022	0.028	0.044	0.033	0.046	0.058

Table A-8. Run description, Deck Park Tunnel, Phoenix, July 1995.

	Run 1	**Run 2**	**Run 3**	**Run 4**	**Run 5**	**Run 6**	**Run 7**	**Run 8**	**Run 9**
Date	25-Jul	25-Jul	26-Jul	26-Jul	26-Jul	26-Jul	27-Jul	27-Jul	27-Jul
Day	Tues	Tues	Wed	Wed	Wed	Wed	Thur	Thur	Thur
Start Time	1230	1700	730	1000	1300	1500	600	900	1100
T (°C)	43.8	46.1	31.1	38.3	43.8	45.5	29.4	36.6	41.6
Av. Speed (mph)	58.8	58.7	58.0	60.7	60.2	59.1	60.4	61.9	59.7
Std Dev (mph)	5.4	5.6	5.5	6.4	5.7	5.7	4.8	6.2	5.7
Total Vehicles	4307	6520	8405	5022	5468	5999	7112	4978	5089
Total LD	3992	6375	8062	4668	5101	5762	6626	4648	4752
Total HD	315	145	343	354	367	237	486	330	337
F LD	0.927	0.978	0.959	0.930	0.933	0.960	0.932	0.934	0.934
F HD	0.073	0.022	0.041	0.070	0.067	0.040	0.068	0.066	0.066

Lincoln Tunnel, NY/NJ (1995)

The Lincoln Tunnel is a three-bore tunnel with two lanes per bore running under the Hudson River between Weehawken, New Jersey and Manhattan Island. The tunnel is the world's only three-tube underwater vehicle tunnel and the world's busiest underwater tunnel. The Center tube (2,280 m long) opened December 22, 1937, the North tube (2,504 m long) opened February 1, 1945, and the South tube (2,440 m long) opened May 25, 1957. The average eastbound weekday traffic volume in 1993 was 56,153 vehicles. The tunnel is operated such that under normal circumstances the North tube is for westbound traffic, the Center tube is switched depending on need, and the South tube is for eastbound traffic. The experiment was conducted exclusively in the South tube. The tunnel ventilation is transverse in design, similar to other underwater tunnels. The ventilation system in the Lincoln Tunnel is divided into four sections, each with a separate blower (fresh air) and exhaust duct. The ventilation sections are numbered 1 to 4, with 1 being the first 271 m in from New Jersey, 2 and 3 being the center sections of the tunnel, and 4 being the last 488 m into New York. Due to the complexity of the entrance section, DRI decided to begin sampling 271 m into the tunnel, at the New Jersey ventilation building. A total of eight sampling stations were required to determine the emissions from motor vehicles traveling through the tunnel. Eleven periods were sampled during this study (Table A-9).

Table A-9. Run description, Lincoln Tunnel, New York.

	Run 1	**Run 2**	**Run 3**	**Run 4**	**Run 5**	**Run 6**	**Run 7**	**Run 8**	**Run 9**	**Run 10**	**Run 11**
Date	16-Aug	16-Aug	16-Aug	16-Aug	16-Aug	17-Aug	17-Aug	17-Aug	18-Aug	18-Aug	18-Aug
Day	Wed	Wed	Wed	Wed	Wed	Thur	Thur	Thur	Fri	Fri	Fri
Start Time	700	900	1100	1700	1900	800	1000	1300	730	930	1130
T (ºC)	26.4	27.5	30.6	30.6	28.6	26.7	28.9	31.9	27.8	29.4	32.8
Av. Spd (mph)	26.5	28.7	26.3	20.4	24.9	25.6	29.7	30.0	26.8	29.6	29.1
Std Dev (mph)	4.3	3.6	5.6	3.4	3.7	3.4	4.6	4.5	5.0	4.0	4.8
Total Vehicles	2749	2316	2133	2215	2804	2912	2003	1733	2689	1899	1750
Total LD	2417	2047	1861	1718	2458	2628	1749	1432	2438	1645	1465
Total HD	332	269	272	497	346	284	254	301	251	254	285
F LD	0.879	0.884	0.872	0.776	0.877	0.902	0.873	0.826	0.907	0.866	0.837
F HD	0.121	0.116	0.128	0.224	0.123	0.098	0.127	0.174	0.093	0.134	0.163

Sepulveda Tunnel, Los Angeles (1995, 1996)

The Sepulveda Tunnel was chosen to represent a more affluent and potentially lower emitting fraction of the LA fleet than operates in the Van Nuys Tunnel. The tunnel is a covered roadway with the top portion being part of the airplane runway and taxiway for the Los Angeles International Airport (LAX). The covered portion of the roadway is 582 m long, straight, and approximately flat in the covered portions, although there is a downgrade approaching the tunnel and an upgrade leaving it. There are two bores, three lanes each with a sidewalk on the right side of each bore. A concrete wall running most of the length of the tunnel separates the two bores of the tunnel. There are 17 openings in this wall, each approximately 10 ft wide by 12 to 14 ft tall. In order to obtain mass emission factors in the

tunnel, DRI needed to seal off these openings so there would be no air transfer between the two bores. There is a ventilation system in the tunnel, although it was not in operation when DRI was sampling. The 1995 and 1996 experiments were conducted in the west bore, which carries Sepulveda Boulevard southbound from the LAX terminals. Immediately after the tunnel there is a turn lane to allow access to the on-ramps to highway 105 which connects to the 405. During some time periods, considerable numbers of the vehicles going through the tunnel head toward these freeways and if the freeway metering lights are on, these vehicles occasionally back up into the tunnel. Congestion in the tunnel was more pronounced during the 1996 study and additional sampling runs were performed in order to obtain a sufficient number of runs with an average speed > 40 mph for comparison with the 1995 data (Tables A-10 and A-11).

Table A-10. Run description, Sepulveda Tunnel, 1995.

	Run 1	**Run 2**	**Run 3**	**Run 4**	**Run 5**	**Run 6**	**Run 7**	**Run 8**
Date	3-Oct	3-Oct	3-Oct	3-Oct	3-Oct	4-Oct	4-Oct	4-Oct
Day	Tues	Tues	Tues	Tues	Tues	Wed	Wed	Wed
Start Time	700	900	1200	1500	1700	600	800	1100
T (oC)	19.4	25.6	26.7	25.0	23.3	18.3	20.6	27.8
Av. Spd. (mph)	47.5	47.7	44.2	44.4	39.9	49.2	48.6	44.5
Std Dev (mph)	8.3	7.2	8.0	8.7	9.2	6.5	8.8	7.4
Total Vehicles	2650	1998	2908	3371	4167	1495	2654	2807
Total LD	2596	1935	2853	3304	4096	1454	2589	2724
Total HD	54	63	55	67	71	41	65	83
F LD	0.980	0.968	0.981	0.980	0.983	0.973	0.976	0.970
F HD	0.020	0.032	0.019	0.020	0.017	0.027	0.024	0.030

Table A-11. Run description, Sepulveda Tunnel, 1996.

	Run 1	**Run 2**	**Run 3**	**Run 4**	**Run 5**	**Run 6**	**Run 7**	**Run 8**	**Run 9**	**Run 10**	**Run 11**	**Run 12**	**Run 13**	**Run 14**	**Run 15**	**Run 16**	**Run 17**	**Run 18**
Date	23-Jul	23-Jul	23-Jul	24-Jul	24-Jul	24-Jul	24-Jul	25-Jul	25-Jul	25-Jul	25-Jul	26-Jul	26-Jul	26-Jul	26-Jul	27-Jul	27-Jul	27-Jul
Day	Tues	Tues	Tues	Wed	Wed	Wed	Wed	Thur	Thur	Thur	Thur	Fri	Fri	Fri	Fri	Sat	Sat	Sat
Start Time	1100	1500	1700	600	800	1000	1400	700	900	1900	2100	1400	1600	1800	2000	700	830	1000
T (ºC)	20.0	20.6	20.6	18.3	18.9	20.6	23.3	20.0	22.8	22.8	22.2	27.8	26.7	25.0	22.2	20.0	22.8	22.8
Avg. Spd (mph)	41.7	18.8	21.4	48.0	44.5	41.7	26.9	47.3	45.2	42.4	40.7	24.9	21.4	26.1	41.6	50.2	47.9	45.7
Std Dev (mph)	7.0	8.7	8.3	7.4	7.1	5.9	9.4	8.1	5.6	6.9	8.4	8.5	5.4	8.3	8.2	5.7	5.8	6.4
Model Year	86.7	87.4	86.7	86.9	85.4	87.8	87.1	86.9	87.5	86.6	88.1	87.5	87.3	86.3	86.5	87.1	87.0	86.6
Total Vehicles	2888	3459	4131	1864	3875	2402	3578	3007	2237	3393	2631	3718	4157	4186	2786	1953	1622	2785
Total LD	2781	3369	4060	1813	3799	2315	3504	2933	2140	3329	2579	3617	4074	4093	2739	1881	1571	2737
Total HD	107	90	71	51	76	87	74	74	97	64	52	101	83	93	47	72	51	48
F LD	0.963	0.974	0.983	0.973	0.980	0.964	0.979	0.975	0.957	0.981	0.980	0.973	0.980	0.978	0.983	0.963	0.969	0.983
F HD	0.037	0.026	0.017	0.027	0.020	0.036	0.021	0.025	0.043	0.019	0.020	0.027	0.020	0.022	0.017	0.037	0.031	0.017

Van Nuys Tunnel, Los Angeles (1996)

The Van Nuys Tunnel is a two-bore, urban tunnel, 222 m in length, running east/west under the runway of the Van Nuys Airport. There are three lanes per bore along with a narrow walkway adjacent to the north and south lanes. Vent buildings are located on the southeast and northeast edges of the tunnel and were not in operation during the experiment. There are nine door-size openings between the bores. The openings were covered with plywood prior to the commencement of sampling. Traffic lights are located within a few hundred meters of both the tunnel exit and entrance. Because of the lights, vehicles accelerated upon entering the tunnel and often decelerated at the exit. A total of nine periods were sampled (Table A-12) in the North Bore, the same as in the 1987 experiment.

Table A-12. Run description, Van Nuys Tunnel, 1995.

	Run 1	**Run 2**	**Run 3**	**Run 4**	**Run 5**	**Run 6**	**Run 7**	**Run 8**	**Run 9**
Date	9-Jun	9-Jun	9-Jun	10-Jun	10-Jun	11-Jun	12-Jun	12-Jun	12-Jun
Day	Fri	Fri	Fri	Sat	Sat	Sun	Mon	Mon	Mon
Start Time	700	1000	1800	1100	2100	1900	730	1200	1500
T (ºC)	30.1	32.3	29.0	38.9	31.3	37.1	34.8	42.1	42.7
Av. Spd. (mph)	42.6	42.4	43.3	44.7	43.4	45.4	43.2	43.6	44.2
Std Dev (mph)	6.1	5.3	5.4	5.0	4.7	5.3	5.7	5.3	5.5
Total Vehicles	1558	1624	1554	1603	670	1046	2183	2021	1315
Total LD	1489	1559	1530	1581	665	1040	2092	1973	1259
Total HD	69	65	24	22	5	6	91	48	56
F LD	0.956	0.960	0.985	0.986	0.993	0.994	0.958	0.976	0.957
F HD	0.044	0.040	0.015	0.014	0.007	0.006	0.042	0.024	0.043

APPENDIX B

UC Berkeley Caldecott Tunnel Field Study Description

UC BERKELEY CALDECOTT TUNNEL FIELD STUDY DESCRIPTION

(written by Rob Harley, UC Berkeley)

The Caldecott Tunnel is located in the San Francisco Bay area on state highway 24 between Alameda and Contra Costa Counties. The tunnel comprises 3 two-lane traffic bores, with the direction of traffic in the middle bore switched to accommodate commuter peaks. Light-duty vehicle emissions have been measured in the middle bore of the tunnel in summers 1994-97, 1999, and 2001. Heavy-duty vehicle emission factors for NO_x and $PM_{2.5}$ were inferred from additional pollutant measurements made in the southernmost bore (bore 1) of the tunnel in summer 1997.

For each tunnel sampling period in 1997, traffic was counted in three weight categories: light (cars plus 2-axle/4-tire trucks), medium (2-axle/6-tire), and heavy (3 or more axles). Survey data indicate that about half the medium and almost all the heavy vehicles are diesel-powered. From 1230-1530 h in bore 1, the fraction of diesel trucks ranged from 3 to 5% of total traffic, whereas in the middle bore from 1530-1830 h, the diesel truck fraction was much lower. In all cases, vehicles were traveling uphill on a 4.0% grade. Heavy trucks traveled through bore 1 on the uphill grade more slowly (65 ± 11 km/h, N=13) than light-duty vehicles (89 ± 11 km/h, N=8 for 21 July; 70 ± 9 km/h, N=17 for 22-24 July). A license plate survey indicated an average model year of 1988 for 156 heavy-duty diesel trucks sampled at random in bore 1.

Diesel trucks were estimated to contribute 3-5% of total CO, 15-19% of total CO_2, 38-41% of total NO_x, and 76-79% of total $PM_{2.5}$ concentrations measured in bore 1 from 1230-1530 h. Using a carbon balance, HD diesel emission factors for NO_x and $PM_{2.5}$ were estimated to be 42 ± 5 and 2.5 ± 0.2 grams per kg of diesel fuel burned, respectively. Uncertainties in CO_2 apportionment affect both of these emission factors, and uncertainty in the NO_X apportionment is also important. Uncertainty in $PM_{2.5}$ apportionment is less important because diesel trucks were responsible for such a high fraction (>75%) of total $PM_{2.5}$ emissions in bore 1.