IN-USE EMISSIONS FROM HEAVY-DUTY DIESEL VEHICLES

By

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ABSTRACT

This is a three part thesis regarding the regulated emissions from in-use heavy duty diesel vehicles. The first part of the thesis involves the collection and analysis of emissions from 21 vehicles. Emissions of particulate matter (PM), nitrogen oxides (NO_x), carbon monoxide (CO), total hydrocarbon (THC) and PM sulfate fraction were measured as well as smoke opacity. The vehicles were tested on three different driving cycles. This study found that when emissions were converted to a g/gallon basis, the effect of driving cycle was eliminated for NO_x and reduced for PM. Sulfate comprised less than 1% of the emitted PM. Smoke opacity was not well correlated with mass emissions of any of the regulated pollutants. Multivariate regression analysis indicated that in-use NO_x emissions did not decrease for this fleet during the years 1986 to 1995 while engine certification standards dropped sharply during that time. A review of all in-use emissions data in the scientific literature supported this result. The review also showed that PM emissions were widely variable among vehicles certified under identical standards. The variability was attributed to environmental factors, inertial weight, test cycle, driver variability and vehicle condition, but the relative importance of these factors could not be determined based on previously collected data. The wide variability in PM emissions within model years and the uniformity of NOx results despite the change in engine standards showed that engine certification was not an accurate tool to predict and control emissions from vehicles. To better understand the relationship between engine and chassis test results, a computer model was developed that estimates engine speed and load from vehicle speed. Using this model it was possible to detect the use of electronic controls which operate the engine in a different mode during engine testing and other more typical types of operation. The model also showed that the engine certification test generated consistently less PM than the vehicle tests when the model was used to put the engine and chassis emissions on a consistent basis. A good correlation was found between rate of HP increase integrated over the test cycle and PM emissions for both the chassis and engine tests. The engine test procedure includes engine behaviors that cannot be duplicated in in-use operation, and appears to favor lower rates of acceleration than typical chassis operation. The model also showed how small changes in vehicle speeds (+/-2 mph) due to driver variability can lead to a doubling of load on the engine.

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CHAPTER 1

INTRODUCTION

This work is concerned with emissions from over-the-road heavy-duty diesel fueled vehicles. Heavy-duty vehicles are those that have a maximum loaded weight of greater than 8,500 pounds. Diesel vehicles and engines that are not used for transport on public roads, such as marine vehicles and off-road heavy equipment are not regulated under the same standards and are excluded. The primary focus of the analysis is emissions of particulate matter (PM), oxides of nitrogen (NO_x), carbon monoxide (CO), and total hydrocarbon (HC) from heavy duty engines which are regulated by the federal government.

Since 1985, engine manufacturers have been required to conduct a certification emissions test, known as the federal test procedure (FTP) (1) on a prototype of each new heavy-duty engine model prior to the use of the engine in any on-road vehicle. These emissions measurements are important, first, because they are the primary method that the federal government uses to regulate emissions from diesel vehicles and, secondly, because they have been used to generate the heavy-duty vehicle emissions factors for several generations of EPA’s air emissions models. Estimates of vehicle emissions are necessary to determine the effectiveness of heavy-duty engine regulations, for pollutant inventories needed to understand air pollution problems, and for air quality planning purposes.

Diesel emissions are a significant air pollution issue. The California Air Resources Board estimates that heavy-duty diesel vehicles currently account for 30% of on-road vehicle emissions of NO_x and 65% of particulate matter (PM), even though these vehicles comprise only 2% of the total on-road vehicle population (2). EPA estimates that heavy-duty diesel vehicles produce 27% of on-road NO_x and more than 60% of on-road PM nationwide (3). Another recent study estimates that heavy-duty diesel vehicles are responsible for nearly half of on-road emissions of NO_x and greater than 75% of motor vehicle exhaust emissions of PM (4). Because particulate matter emitted from diesel engines is smaller than 2.5 micron in diameter (5), its PM contribution will be increasingly important as regulations for smaller particles come into effect. Emissions of THC and CO from diesel vehicles are a much smaller proportion of the total motor vehicle contribution and so are considered of lesser importance.

Included in this thesis are three papers, two of which have been previously published (6,7) and the third (8) which has recently been submitted for publication. An error, regarding the rate of decline in the PM regulatory standard in the first paper (Chapter 2) was corrected, and several figures were added to the third paper (Chapter 4) for completeness. A section comparing measured emission factors with the EPA models was added to Chapter 3 and minor updates have been made throughout. The abstracts from each paper have been edited and combined into the thesis abstract. Portions of the three introductions have been included in the thesis introduction, and the remaining introductory material has been edited to improve the flow of this document.

The first paper involves the collection and analysis of regulated emissions from 21 in-use heavy-duty diesel vehicles on the chassis dynamometer. The study was a part of the Northern Front Range Air Quality Study (NFRAQS). One of the objectives of NFRAQS was to determine the fraction of ambient carbonaceous particulate matter originating from various sources, including on-road diesel vehicles. The PM mass emissions data described in this report were used in that analysis (9).

The results of the 21 vehicle study had several interesting results. First, multivariate regression analyses indicated that in-use NO_x emissions for these vehicles did not reflect emissions improvements expected based on the stricter engine certification test standards put into effect since 1985. Secondly, NO_x emissions were relatively constant regardless of test cycle when measured on a g/gallon basis. Since, power generated by an modern fuel efficient engines is close to proportional to fuel usage this suggests that the mass rate of NO_x generation is proportional to Horsepower (HP). Thirdly, the regression analysis also indicated a strong dependence of PM emissions on test inertial weight relative to engine horsepower. Fourth, sulfate comprised less than 1% of the total particulate matter mass for all vehicles. When compared to particulate speciation results reported by others using higher sulfur content fuel, this result suggested that the mandatory reduction in sulfur content of diesel fuel had effectively reduced PM emissions; and finally, smoke opacity, which is the only emissions parameter used in several state run annual inspection and maintenance programs, was shown to not be well correlated with mass emissions of PM, CO, NO_x or THC. This final topic has since been addressed by others at the Colorado Institute for Fuels and High-Altitude Engine Research (10). Their results supported those of the 21 vehicle study, i.e. that opacity measurements are poorly correlated with mass emissions.

As only 21 vehicles were included in this study, it could not be considered representative of the entire heavy-duty fleet, which includes a wide range of vehicles including small and large delivery trucks, school buses, transit buses, garbage trucks and snow plows etc. And while the expense and difficulty of recruiting and testing large numbers of heavy-duty vehicles has precluded individual researchers from collecting emissions data from a representative sample of these types of vehicles, there was a large body of in-use emissions data collected using a variety of methods available in the scientific literature. The next step was to collect and analyze all of the available data on in-use emissions testing of regulated emissions.

This comprehensive review supported many of the results found in the smaller study. Chassis dynamometer results indicated that the average emissions of PM, CO, and THC had been reduced since the implementation of regulations of on-road diesel engines, but the reductions in PM were less than expected based on the regulations. Average emissions of NO_x had not changed at all during this time period despite a reduction of greater than 50% in allowable emissions as measured by the engine test. Chassis dynamometer emissions of PM were widely variable, typically spanning an order of magnitude for any given model year. These differences were attributed to variations in altitude, vehicle weight, test cycle, vehicle condition and driver variability. Different measurement methods were discussed and compared and several improvements in study methodology were suggested.

Overall the critical review highlighted the problems of using engine tests to predict emissions behavior of in-use vehicles. In order to develop a clearer understanding of in-use emissions, the effects of weight, driving cycle, driver variability and the differences between the drive trains in vehicles using identical engines would need to be quantified. These factors, which are also widely variable under real life conditions, are expected to cause significant differences in emissions. Chapter 4 outlines a methodology for relating emissions from engine testing to vehicle speeds and inertial loads, using a computer model of a drivetrain. This model is known as the Colorado School of Mines Transmission Model (CSMTM).

This computer program models the behavior of a heavy-duty vehicle driveline and estimates engine speeds and load from vehicle speed for vehicles with automatic transmissions. Engine behavior under different test cycles and test weights, and even small differences in driver behavior, can be compared and emissions results predicted. The model has been validated by comparing emissions behavior between three engines tested alone and then placed in a chassis and tested on the chassis dynamometer. The results provided several interesting insights into the relationship between NO_x and PM emissions and engine operation.

CHAPTER 2

CHASSIS DYNAMOMETER STUDY OF EMISSIONS FROM 21 IN-USE HEAVY-DUTY DIESEL VEHICLES

Emissions studies of the in-use fleet have been conducted in a limited manner, primarily due to the expense and difficulty of recruiting and testing large numbers of heavy-duty diesel vehicles. Previous studies found that in-use emissions from diesel vehicles, particularly CO and PM, were widely variable for different vehicles under identical test cycles (11-15). Choice of test cycle can result in CO and particulate emissions changes of from 50 to more than 100%, and lesser percentage changes in other pollutants (13,14). A study of Australian diesel vehicles reported that vehicle-to-vehicle variability in emissions was also high and could not be readily attributed to differences in model year, mileage or engine type (12). Additionally, there is wide variability in the in-use fleet as 291 different heavy-duty diesel engines were certified for on-road use by the EPA in 1996 alone (16). Published studies (11-15, 17-21) do not adequately reflect this variability nor do they include late model engines and vehicles. The large change in emissions regulations since 1985 (1) and normal turnover of the fleet make studies of newer vehicles important for evaluating the impact of regulations and for air quality planning purposes. For example, in the study area encompassed by this report, vehicles more than 10 years old comprised only about one fourth of the registered vehicles (22).

Opacity tests are used by many states as part of inspection programs required for diesel vehicles, and are part of the testing required by the federal government for certification of new engine models. Opacity tests are conducted using a light extinction opacity meter or by a trained inspector who conducts visual comparisons to known opacity standards. High opacity may indicate engine malfunction and increased emissions of air pollutants, primarily unburned hydrocarbons (white smoke) or soot particles (black smoke) (23).

The objectives of the study described in this chapter were to: (a) measure mass emissions of PM, CO, NO_x, and THC from a small fleet of vehicles in-use in the study area using a representative fuel; (b) analyze the particulate matter to determine the sulfate fraction; (c) measure smoke opacity via several tests and compare with mass emissions of regulated pollutants; (d) perform an analysis of the data to determine which factors have the most important effect on emissions.

Experimental Section

Fleet Selection. The twenty-one vehicles tested in this study are listed in Table 2.1 along with their important characteristics. The vehicles chosen represent a range of different operating duties (for example, a water truck, school buses, transit buses, food distribution trucks), different types of ownership (public fleet, private fleet, rental fleet), a range of sizes (from 11,500 lb GVWR to 80,000 lb GVWR), and model years (from 1981 to 1995). All but two of the vehicles were equipped with 4-stroke engines and all of the

Table 2.1 Characteristics of vehicles tested.

Vehicle	Use	Curb Wt, lb	GVWR, lb	Engine Make	Engine Model	Engine HP	Emissions Model Year	Odometer Mileage[a]
#1	RTD Bus	28,680	38,000	DDC	6V92TA[b]	300	1981	119,000
#2	Food Delivery	15,800	33,000	Navistar	DT466 E185	185	1990	142,242
#3	Lease	15,540	25,500	Navistar	DT408 A210	210	1993	122,406
#4	RTD Bus	28,680	38,000	DDC	Series 50	275	1993	85,200
#5	Dump Truck	13,233	28,000	Navistar	DT466	215	1987	89,528
#6	Lease	8050	11,050	Isuzu	PSZ0235FAA8	130	1993	82,618
#7	School Bus	17,920	28,000	GMC	V8-8.2T	225	1988	89,054
#8	RTD Bus	28,680	38,000	DDC	Series 50	275	1993	65,234
#9	Food Delivery	16,500	80,000	Cummins	N14 093Q	330	1991	477,969
#10	Dump Truck	30,000	80,000	Cummins	NTC 350	285	1984	595,606
#11	School Bus	17,920	30,000	Cummins	6BTA5.9	195	1991	62,549
#12	Furniture Delivery	12,380	22,000	Isuzu	6BG1XN	172	1993	150,788
#13	Concrete Mixer	24,300	60,000	Cummins	L10 343E	280	1993	96,262
#14	Dump Truck/Plow	21,800	36,220	Navistar	DT466 SNV466	250	1995	5,320
#15	Garbage Hauler	29,800	50,000	Cummins	LT A10	270	1990	72,251
#16	Dump Truck/Plow	16,600	33,000	Navistar	DT466 D210F	210	1989	101,925
#17	Telephone Truck	19,500	80,000	Cummins	NTC400	400	1983	80,876
#18	RTD Truck	8,900	28,000	GMC	V8-8.2T	225	1989	13,518
#19	Water Truck	20,800	49,560	Cummins	NTC400	400	1981	17,867
#20	CDOT Truck	17,700	36,220	Navistar	DT466 E250	250	1993	37,009
#21	RTD Bus	28,680	38,000	DDC	6V92TA[b]	275	1986	66,780

[a] Since rebuild. [b] Two-stroke engines

vehicles were equipped with turbochargers. The vehicles approximately match heavy-duty vehicles registered in Colorado in terms of emissions model year and GVWR distributions. However, it is clear that a twenty-one vehicle test fleet cannot be representative of the in-use fleet as a whole and it is unknown how closely vehicle registration reflects the actual makeup of the in-use fleet. No grossly malfunctioning vehicles or vehicles that had been obviously tampered with were included in this study.

Test Fuel. A Northern Front Range wintertime average fuel was employed in all chassis testing. The major Front Range area refiners/marketers of No. 2 diesel were surveyed to determine their market share and their wintertime diesel strategy (i.e., in terms of adjusting cold flow properties for lower temperatures). All but one refiner blends some percentage of No. 1 diesel with No. 2 to produce wintertime fuel. The market share of each of these refiners was used to calculate the blending volume needed to prepare approximately 1000 gallons of a representative fuel. Each major refiner/marketer provided the appropriate volume of wintertime fuel, or a No. 2 base stock sample, No. 1 fuel, and blending instructions. The fuels were splash blended and drummed. Table 2.2 presents the composition and properties of the resulting NFR average fuel. The most significant differences between NFR fuel and industry average fuel (or certification fuel, supplied by Phillips Petroleum) are that the NFR fuel T_{90}+ fraction boils at a slightly higher temperature, and the NFR fuel has a slightly lower aromatic content. Engine transient test emissions for three different engines were almost identical for NFR and certification fuel (24).

Table 2.2 Results of analysis of Northern Front Range (NFR) Wintertime Average Fuel and Industry Average Certification Fuel.

Test	Chapter 2 Test Method	NFR Fuel	Units
API Gravity	ASTM D-287	37.0	@60/60°F
Cetane Number	ASTM D-613	46.4	
Elemental Analysis			
Carbon	ASTM D-5291	86.17	Wt%
Hydrogen	ASTM D-5291	13.80	Wt%
Nitrogen	ASTM D-4629	1	ppm
Sulfur	ASTM D-2622	0.0343	Wt%
Distillation	ASTM D-86		
I.B.P		336	°F
10%		403	°F
50%		492	°F
90%		599	°F
95%		632	°F
End Point		650	°F
Hydrocarbon Type	ASTM D-1319		
Aromatic		29.0	Vol%
Olefin		1.3	Vol%
Saturate		69.7	Vol%
SFC Aromatics	ASTM D-5186-91		
Non-aromatic		70.8	Wt%
Aromatic		29.2	Wt%
Polynuclear Aromatic		6.14	Wt%

Chassis Dynamometer Simulation. The chassis dynamometer is suitable for operating at vehicle speeds up to 60 mph. The vehicles are driven on twin 40-inch rolls which spin at 500 rpm at a road speed of 60 mph. The DC dynamometer is located 90 degrees to the rolls and shaft power is transmitted through two 5:1 ratio Falk gearboxes. An inline torque meter is located on the dynamometer shaft and reads the dynamometer load. The chassis gearboxes are lubricated with circulating gear oil heat traced and insulated to maintain the surface at a constant temperature. Friction heat is removed from the oil in a small water-cooled heat exchanger. Regulating the oil temperature fixes the oil viscosity and minimizes friction variations due to changeable oil properties.

Inertia is simulated with mechanical flywheels located on the high-speed dynamometer shaft. Up to 55,000 pounds of inertia can be simulated in increments of 2,500 pounds. The inertial weight is set at approximately the average of the curb weight and the Rated Gross Vehicle Weight (GVWR). However Vehicles 12 and 20 were also tested at inertial weights equal to 58% and 97% of GVWR (Vehicle 12) and 47% and 97% of GVWR (Vehicle 20). Load simulation for running friction is accomplished with control circuitry to vary the dynamometer applied load in response to the vehicle speed. Vehicle wind and rolling friction losses are estimated from published studies (25) and the dynamometer controller is operated to provide the appropriate control of torque at the rolls based upon weight and frontal area.

The vehicle speed is managed by the vehicle driver. The cycle is displayed for the driver using a driver's aid prompt that shows the driver current speed and approximately

30 seconds into the future to anticipate shifting. For quality control purposes, ±2 mph error bands are displayed for the driver. A single driver was used for all chassis testing performed under this program. Tests are rejected if the driver misses a shift and must brake the vehicle to repeat the acceleration.

Test Cycles. In this study, three test cycles were employed, the EPA Urban Dynamometer Driving Schedule for Heavy-duty Vehicles (heavy-duty transient or HDT cycle) (26), the Central Business District (CBD) cycle (27) and the West Virginia Truck (WVT) cycle (28). Several vehicles were driven on all three cycles. Each cycle is specified as series of vehicle speeds at one-second increments. These cycles are further described in Chapter 3.

Emissions Measurement. The system for emissions measurement for regulated pollutants (THC, CO, NO_x, and PM) includes supply of conditioned intake and dilution air, an exhaust dilution system, and capability for sampling of particulate and analysis of gaseous emissions. All components of the emissions measurement system meet the requirements for heavy-duty engine emissions certification testing as specified in Code of Federal Regulations 40, Part 86, Subpart N. The intake air conditioning, exhaust dilution, and emissions measurement systems have been described in more detail elsewhere (29). All testing was performed at an intake air temperature of 77±9° F and humidity of 75 grain/lb dry air so that the NO_x correction factor is 1±0.03.

Sulfate Analysis. Sulfate analysis was performed by Hazen Research, Inc., of Golden, Colorado using the procedure outlined by the Coordinating Research Council

(30). The procedure involves washing the particulate filters with a carbonate/ bicarbonate solution to dissolve the sulfate. Any filter material is then removed and the solution is injected into an ion chromatograph. Sulfate is determined by comparison against a four-point calibration curve using potassium sulfate as a standard. The procedure measures the sulfate fraction of total primary PM and does not include other forms of sulfur that are not water-soluble.

Smoke Opacity Testing. Smoke opacity was measured using three different smoke cycles. These were (a) Colorado Regulation 12 Lug Down dynamometer cycle; (b) Colorado snap idle procedure; (c) SAE J1667 snap idle procedure (31). Smoke opacity was measured by a Wager Digital Smoke Meter, Model No. 6500. No correction was made for altitude.

The State of Colorado Regulation 12 diesel inspection and maintenance procedure requires that a gear be chosen at which the engine can achieve its rated speed while the vehicle speed is between 20 and 40 mph. An engine mounted tachometer, or hand-held strobe tachometer, was used to measure engine speed. With the dynamometer operating in “speed mode”, the vehicle speed is adjusted to 90%, 80%, and 70% of engine rated speed at wide-open throttle, and smoke opacity data are logged.

Snap-idle tests are performed with the warmed up vehicle in neutral. The accelerator pedal is quickly pushed to the floor while smoke opacity is measured. For the SAE J1667 test three practice tests are first performed. This is followed by three real tests that are averaged to obtain the reported value. The three tests must meet certain validation criteria, and the percentage smoke opacity is corrected for stack diameter using an extinction coefficient specific to the instrument. J1667 opacity results were not corrected for altitude. The State of Colorado is currently evaluating a snap-idle test, but at this time it is not used to determine inspection and maintenance compliance. For the State of Colorado snap-idle test, the vehicle is warmed up, one snap acceleration is performed, and smoke opacity is measured. No extinction coefficient correction for stack diameter is performed for the State test.

Results and Discussion

Pollutant Mass Emissions. Table 2.3 reports average results for hot and cold runs performed for each vehicle in the study. Emissions of PM, THC, NO_x, and CO are reported in g/mi and fuel economy is reported in mi/gal. Inspection of the table reveals that emissions varied over a wide range for this fleet, including some vehicles that had apparently very low emissions as well as some with very high emissions values. For hot start tests, PM ranged from 0.30 to 7.43 g/mi with a mean of 1.96; NO_x ranged from 4.15 to 54.0 g/mi with a mean of 23.3; CO ranged from 2.09 to 86.2 g/mi with a mean of 19.5; and THC ranged from 0.25 to 8.25 g/mi with a mean of 1.70 g/mi. The CBD cycle, the most aggressive cycle (i.e., has the most accelerations), produced the highest emissions with mean PM of 2.85 g/mi, NO_x of 30.4 g/mi, CO of 30.4 g/mi, and THC of 1.98 g/mi. The second highest emissions were produced by the HDT cycle with a mean PM rate of 1.68 g/mi, NO_x of 21.0 g/mi, CO of 16.8 g/mi, and THC of 1.31 g/mi. The lowest

Table 2.3 Regulated emissions results for average cold and hot start test. Vehicles tested at the average of curb weight and GVWR, except for vehicles 12 and 20 tested as noted.

Vehicle	Cycle	Hot/Cold	PM g/mi	HC g/mi	NO_x g/mi	CO g/mi	Fuel mi/gal
1	CBD	Ave Hot	6.03	2.34	49.1	86.2	3.34
2	WVT	Ave Hot	0.89	0.39	13	3.41	7.33
	CBD	Ave Hot	1.69	0.42	20	8.21	4.87
	HDT	Ave Hot	1.46	0.26	15.4	4.93	N/A
3	HDT	Cold	1.38	1.24	15	18.4	5.5
	HDT	Ave Hot	0.98	0.59	13.6	9.22	6.41
	CBD	Ave Hot	1.27	1.14	17.1	17.2	4.78
	WVT	Cold	0.72	1.23	12.6	12.1	7.27
	WVT	Hot	0.68	0.78	11.6	8.36	8.14
4	CBD	Cold	1.28	0.36	50.1	25	2.7
	CBD	Ave Hot	0.82	0.26	43.4	19.7	2.97
	HDT	Ave Hot	0.73	0.14	32.6	14.1	4.13
5	HDT	Ave Hot	2.31	2.07	9.93	14.2	6.65
	WVT	Cold	1.39	3.47	13.7	13.5	6.78
	WVT	Ave Hot	1.25	2.44	8.81	11.3	8.26
6	HDT	Ave Cold	1.95	1.4	5.43	19.6	6.99
	HDT	Ave Hot	1.58	0.76	4.24	13.5	8.24
	CBD	Ave Hot	2.82	1.3	4.15	22.9	8.29
7	HDT	Cold	2.13	1.69	19	7.17	5.57
	HDT	Ave Hot	1.8	0.67	17.9	6.74	6.04
	CBD	Cold	2.11	1.86	25	15.4	4.11
	CBD	Ave Hot	1.92	1.04	23.9	12	4.57
8	HDT	Cold	2.42	0.36	40.2	14.1	3.82
	HDT	Ave Hot	0.73	0.17	35.1	12.2	4.41
	CBD	Ave Hot	0.71	0.29	46.5	17.4	3.38
9	WVT	Ave Hot	0.75	1.52	16.6	7.91	5.39
10	WVT	Ave Hot	2.85	3.45	17.7	9.94	4.25
11	CBD	Cold	3.96	1.59	18.4	23.3	4.32
	CBD	Ave Hot	2.7	0.69	16.8	18	4.54
	HDT	Cold	3.95	0.66	13.8	9.54	5.04
	HDT	Ave Hot	1.61	0.33	12.5	7.73	5.28
12							
Wt=12759 lbs	HDT	Cold	1.12	1.76	13.1	10	6.41
Wt=12759 lbs	HDT	Ave Hot	0.94	1.26	17.1	5.8	7.39
Wt=17120 lbs	HDT	Ave Hot	1.18	1.33	17.6	6.09	6.62
Wt=21319 lbs	HDT	Ave Hot	2.67	1.27	18.1	5.3	6.98

Vehicle	Cycle	Hot/Cold	PM g/mi	HC g/mi	NO_x g/mi	CO g/mi	Fuel mi/gal
12 (cont'd)							
Wt=12759 lbs	CBD	Ave Hot	1.27	3.04	20.6	13.3	5.59
13	WVT	Cold	1.29	2.12	23.2	9.33	4.68
	WVT	Ave Hot	1.11	2.07	21	10.4	5.1
14	HDT	Cold	1.54	0.73	20.8	12.6	5.75
	HDT	Ave Hot	0.77	0.55	18	8.01	5.72
	CBD	Ave Hot	0.82	0.9	24.1	12.3	4.35
15	WVT	Ave Hot	1.73	0.63	18.3	26.7	5.63
	CBD	Ave Hot	6.07	1.14	33.9	66.1	3.14
	HDT	Ave Hot	3.89	0.92	27.9	45.2	4.15
16	CBD	Ave Hot	2.52	2.1	44.1	40.2	4.18
	HDT	Cold	2.56	1.9	39.1	30.5	4.81
	HDT	Ave Hot	2.17	1.21	35.8	29.4	5.41
	WVT	Ave Hot	0.61	1.44	27.8	5.32	7.38
17	WVT	Ave Hot	1.96	3.42	18.8	18.4	4.56
	HDT	Ave Hot	3.5	4.31	24.7	50.7	3.75
	CBD	Ave Hot	5.99	7.08	31.2	62.4	2.39
18	CBD	Ave Hot	1.21	1.03	19	9.96	5.4
	CBD	Cold	1.51	2.02	20.8	13.8	4.43
	HDT	Ave Hot	1.17	0.52	13.4	25.2	6.74
	WVT	Ave Hot	0.3	0.62	12.8	3.41	9.56
19	HDT	Ave Cold	4.48	8.57	20.9	25.7	4.09
	HDT	Ave Hot	3.25	6.83	19.9	28.3	4.47
	CBD	Ave Hot	7.43	8.25	28.8	54.3	2.51
20							
Wt=17000 lbs	HDT	Cold	0.55	0.51	13.3	5.93	
Wt=17000 lbs	HDT	Ave Hot	0.53	0.28	11.5	3.78	
Wt=25000 lbs	HDT	Cold	0.82	0.25	12.6	5.87	5.3
Wt=25000 lbs	HDT	Ave Hot	0.72	0.26	12.3	4.7	5.34
Wt=35000 lbs	HDT	Cold	3.15	0.72	15.93	7.3	
Wt=35000 lbs	HDT	Ave Hot	1.14	0.24	13.43	5.16	
21	HDT	Ave Hot	0.65	1.33	46.5	5.77	4.24
	WVT	Ave Hot	0.41	1.22	31.8	2.09	5.92
	CBD	Cold	1.36	1.48	57.7	13	2.8
	CBD	Ave Hot	0.68	1.72	54	9.76	3.41

emissions were produced by the WVT cycle with a mean PM rate of 1.24 g/mi, NO_x of 17.8 g/mi, CO of 9.75 g/mi, and THC of 1.90 g/mi. Inertial weight was varied for vehicles 12 and 20. Emissions of PM and NO_x were found to increase with increasing test weight. The impact of cold start on emissions is discussed in a later section.

Sulfate. Particulate samples from a subset of the tests run on the first 16 vehicles were analyzed for sulfate and the results are reported in Table 2.4. Sulfate is not well correlated with total PM emissions. Nor does the sulfate comprise a significant portion of the particulate matter emitted, (less than 1% in all cases). Emissions tests done in 1992 found that sulfate comprised on average of 11% of the particulate (19), at a time when the allowable sulfur content in diesel fuel was 5000 ppm. In comparison, the current allowable level of 500 ppm has resulted in significantly lower emissions of sulfate suggesting that reducing the sulfur content in the fuel to 500 ppm has been effective at reducing PM emissions.

Correlation Between Regulated Pollutants. Correlations between types of emissions may prove valuable for inspection and maintenance programs, remote sensing, air quality planning, as well as in analysis of chassis testing data. The only significant correlation observed in this study is between CO and PM emissions and is reported in Figure 2.1. The regression line is for all test cycles and indicates an excellent correlation given the number of data points and confounding factors.

Table 2.4 Sulfate emission results.

Vehicle	Cycle	Hot/Cold	Sulfate (g/mi)	Sulfate (g/gal)
1	CBD	H	0.020	0.064
2	CBD	H	0.011	0.052
3	HDT	H	0.008	--[a]
4	CBD	C	0.041	0.111
5	HDT	H	0.008	0.051
6	HDT	H	0.012	0.100
7	HDT	H	0.008	0.047
8	CBD	H	0.021	0.073
9	WVT	H	0.013	0.075
10	WVT	H	0.012	0.053
11	HDT	H	0.008	0.044
12	HDT	H	0.004	0.028
13	WVT	H	0.013	0.064
14	HDT	H	0.007	0.042
15	WVT	H	0.005	0.025
16	HDT	C	0.012	0.059
Average			0.013	0.059
Std. Dev.			0.008	0.024

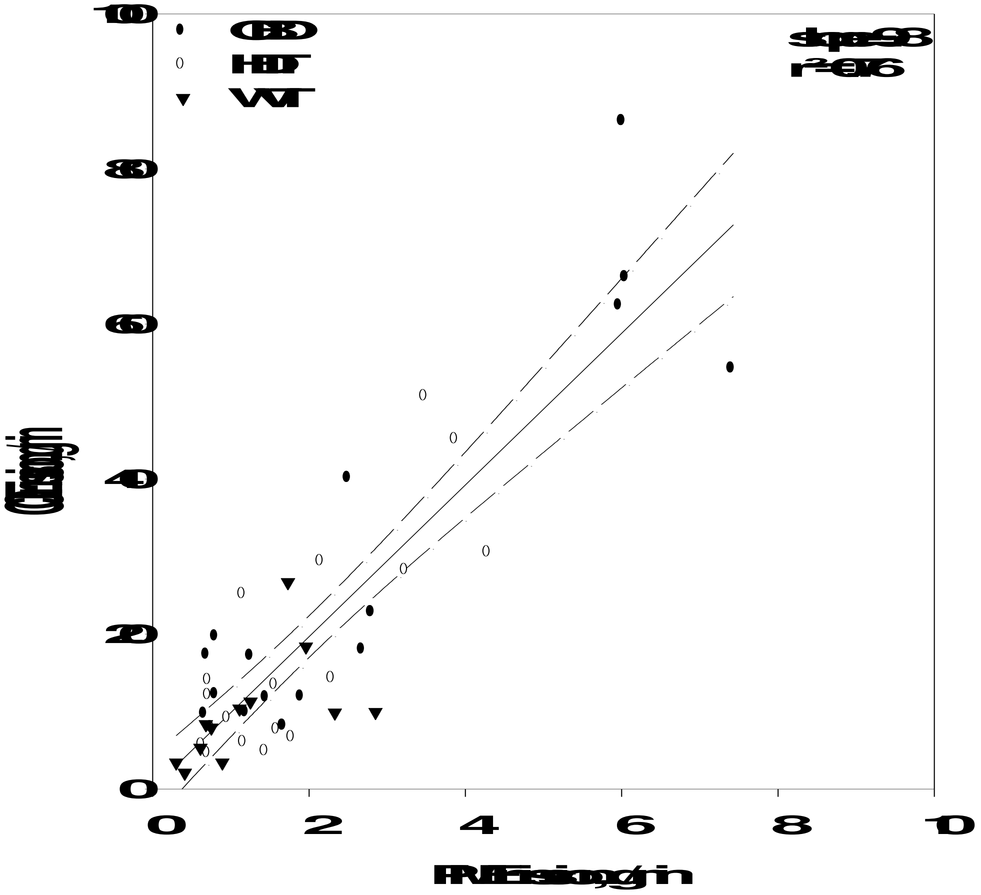


Figure 2.1 Correlation between CO and PM emissions over all test cycles using hot start averages from Table 2.3, 95% confidence interval shown.

Impact of Cold Start. Starting temperature can have a significant impact on emissions from heavy-duty vehicles. Most vehicles used in this study were tested via cold start for at least one test cycle. Cold start is at test cell temperature, typically 77±9°F. The vehicle was allowed to sit overnight and was not warmed up prior to cold start testing. Figure 2.2 compares cold and hot start PM emissions for these vehicles. PM emissions increase from an average of 1.96 g/mi for hot starts to 2.18 g/mi for cold starts, an 11% change. The amount of increase varies considerably between vehicles. The effect of cold starting on CO, NO_x and THC emissions is minimal for the vehicles tested.

Effect of Driving Cycle. As noted above, driving cycle has a significant impact on emissions with g/mi emissions on average following the sequence CBD>HDT>WVT. Chassis NO_x and PM emissions are compared utilizing a parity plot in Figure 2.3. On a g/mi basis it is clear that vehicles which produce high emissions on the CBD also produce high emissions on the HDT and WVT cycles, however since CBD emissions are higher the correspondence is less than 1:1. The g/mi emissions can be converted to a g/gal of fuel consumed basis by multiplying by the fuel economy in mi/gal (reported with the emissions in Table 2.3). The fuel economy measurement is calculated based on the amount of fuel used during the test cycle and is also used to make a carbon balance for each test run for quality control purposes. Emissions of NO_x and PM for the various cycles on a g/gal basis are also compared in Figure 2.3. For NO_x, comparison on a g/gal

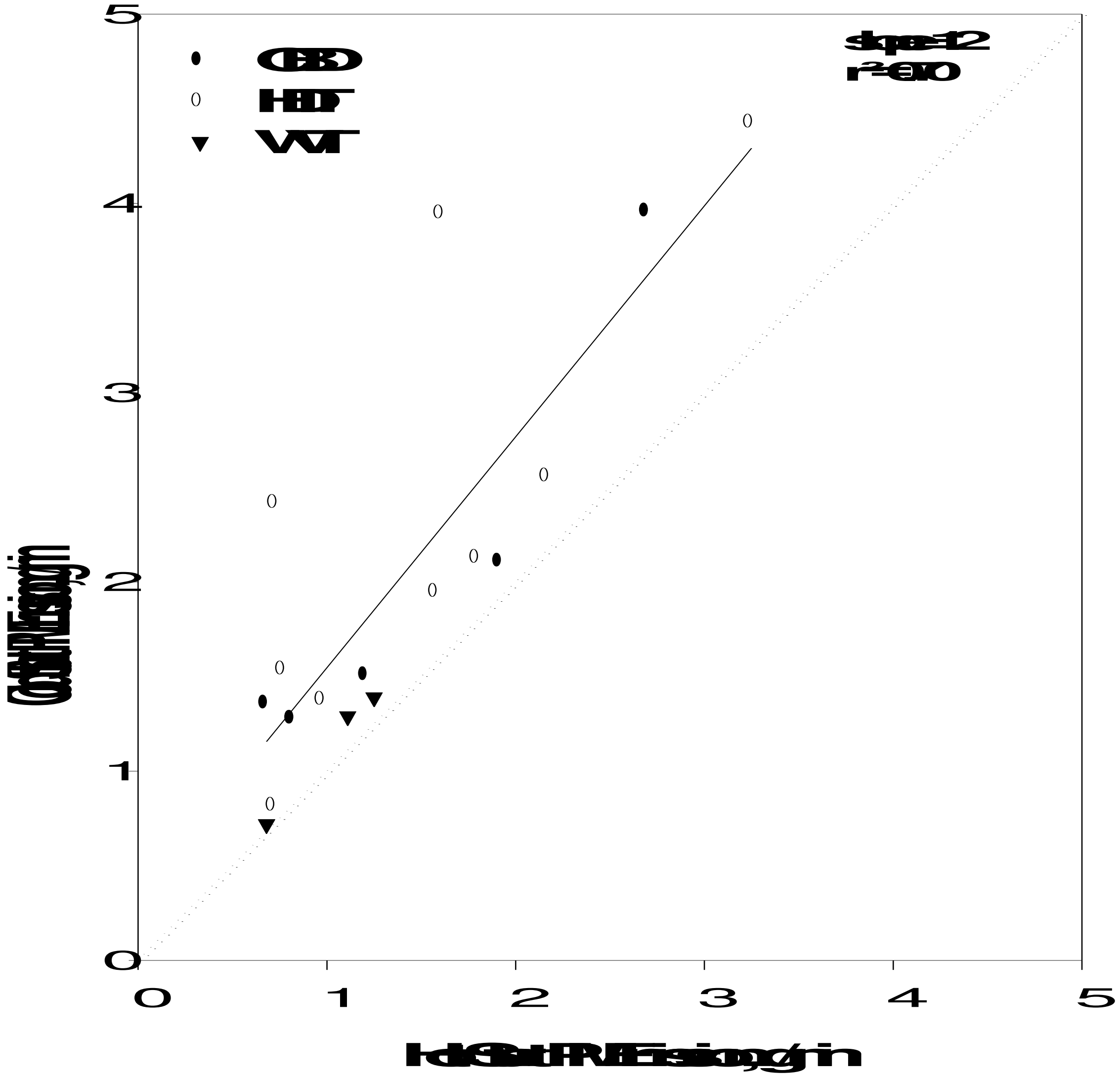


Figure 2.2 Effect of cold starting on PM emissions.

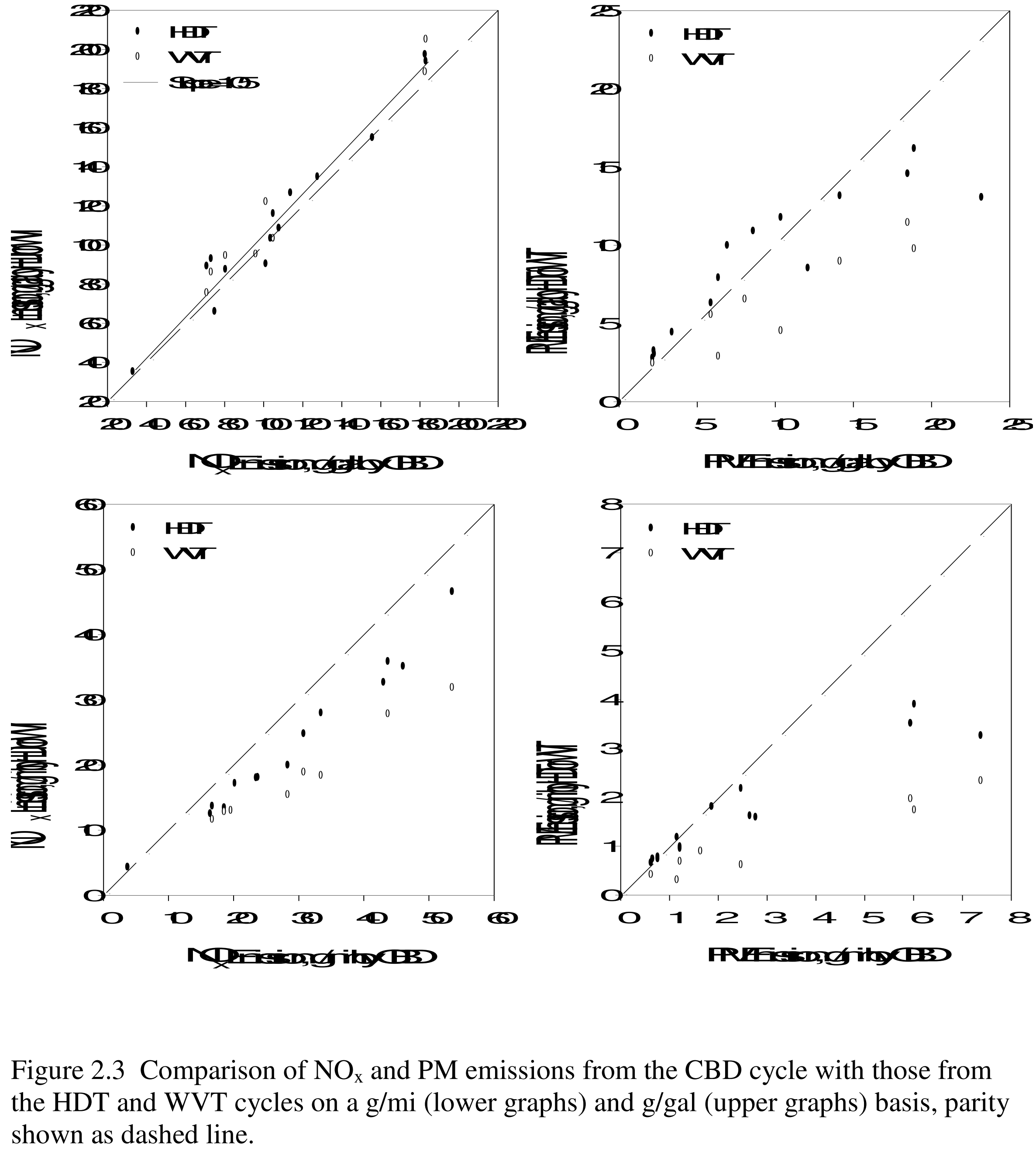


Figure 2.3 Comparison of NO_x and PM emissions from the CBD cycle with those from the HDT and WVT cycles on a g/mi (lower graphs) and g/gal (upper graphs) basis, parity shown as dashed line.

basis eliminates the effect of driving cycle and a linear fit of the data is not statistically different from the parity line. For emissions of PM, driving cycle still has an effect on a g/gal basis but much less than for g/mi. Factors other than fuel economy are clearly important in explaining differences in PM emissions between cycles, especially when comparing the CBD and WVT cycles. It has been proposed that fuel-based emissions factors be employed in emissions inventory calculations because these are much less affected by vehicle load and speed *(32-34)*. The g/gal comparison presented in Figure 2.3 clearly supports this contention for NO_x emissions from heavy-duty vehicles.

Multivariate Regression Analysis. Multivariate regression analyses were performed for the fuel consumption based (g/gal) NO_x and PM emissions to identify factors responsible for variation between vehicles. The analysis was performed on a g/gal basis rather than a g/mile basis to minimize the effect of vehicle weight and driving cycle, as noted above. This focuses the analysis on emissions model year, mileage, and engine size relative to vehicle weight. The chi-square goodness-of-fit test revealed that the data fell more closely into a log-normal distribution than a normal distribution so the regression analyses were performed on the log of the emissions value. This is consistent with the fact that light-duty vehicle emissions distributions are skewed from a normal distribution. A small fraction of light-duty vehicles produced a large fraction of the emissions (35-37) and a similar situation may exist for heavy-duty vehicles. The following model equation was used in the regression analysis:Log Emission (g/gal) = α_1 $_+\alpha_2$ (Inertial Wt/GVWR) + α_3 (Odometer Mileage) (2-1)

$$+ \alpha_4 \text{ (Year)} + \alpha_5 \text{ (GMC =1, not GMC = 0)}$$

$$+ \alpha_6 \text{ (Cummins =1, not Cummins= 0)}$$

$$+ \alpha_7 \text{(Isuzu =1, not Isuzu = 0)}$$

$$+ \alpha_8 \text{(DDC 4-stroke =1, not DDC 4-stroke = 0)}$$

$$+ \alpha_9 \text{(DDC 2-stroke =1, not DDC 2-stroke = 0)}$$

The analyses were performed on the CBD and HDT cycle test results separately and then compared. The results of the WVT cycle were not included because of the limited number of vehicles tested using this cycle. The proposed model and descriptive statistics are included as Tables 2.5 and 2.6 for PM and NO_x, respectively.

As shown in Table 2.5, for PM emissions significant correlation is found for inertial weight/HP for the HDT cycle (coefficient 0.0045; t-statistic 4.9), model year (coefficient for CBD: -.05, HDT: -.05, t-statistic for CBD: -3.9, HDT: -6.6), as well as for engine make. Significant correlation at the 95% confidence level is defined as $|t|>t_{crit}$, which is approximately 2.0 in all cases. Significant correlation with inertial weight/HP for the HDT cycle indicates that emissions increase with increasing vehicle load, and that

Table 2.5 Multivariate Regression Analysis Results for PM

		All Vehicles				Model Year 1988 or Later, Not Including Urban Buses			
		CBD R^2 =.74 Number of observations = 42		HDT R^2=.84 number of observations = 52		CBD R^2=.75 number of observations = 24		HDT R^2=.80 number of observations = 36	
		Coefficient	t-statistic	Coefficient	t-statistic	Coefficient	t-statistic	Coefficient	t-statistic
Intercept	α_1	96	3.9	95	6.6	193	3.8	164	3.8
Inertial Wt/HP	α_2	0.0036	1.6	0.0045	4.9	-0.0008	-0.37	0.0042	4.7
Odometer Miles	α_3	1.1×10^{-6}	1.1	5.3×10^{-7}	0.8	-1.1×10^{-6}	-1.2	-5.5×10^{-7}	-0.69
Year	α_4	-0.05	-3.9	-0.05	-6.6	-0.10	-3.8	-0.08	-3.8
GMC	α_5	0.01	0.11	0.09	1.5	-0.31	-2.0	-0.03	-0.34
Cummins	α_6	0.09	0.7	0.04	-0.6	0.23	2.4	0.03	0.4
Isuzu	α_7	0.45	3.0	0.31	4.0	0.41	3.7	0.44	4.9
DDC-4 stroke	α_8	-0.44	-4.0	-0.32	-5.2	NA	NA	NA	NA
DDC-2 stroke	α_9	-0.45	-3.0	-0.69	-8.0	NA	NA	NA	NA
Critical t-value		t(33,.05)=2.0		t(43,.05)=2.0		t(17,.05)=2.1		t(29,.05)=2.0	

Table 2.6 Multivariate Regression Analysis Results for NO_x

		All Vehicles				Vehicles Model Year 1986 or Later			
		CBD R^2 =.67 Number of observations = 42		HDT R^2=.84 number of observations=52		CBD R^2=.64 number of observations = 33		HDT R^2=.59 number of observations = 47	
		Coefficient	t-statistic	Coefficient	t-statistic	Coefficient	t-statistic	Coefficient	t-statistic
Intercept	α_1	14	0.9	95	6.6	60	1.3	-63	-2.2
Inertial Wt/HP	α_2	0.003	2.2	0.005	4.9	0.003	1.3	0.002	2.0
Odometer Miles	α_3	2.1×10^{-7}	0.4	5.3×10^{-7}	0.8	2.0×10^{-8}	0.02	3.4×10^{-6}	3.9
Year	α_4	-0.006	-0.8	-0.05	-6.6	-0.029	-1.3	0.03	2.2
GMC	α_5	0.018	0.09	0.09	1.5	-0.08	-0.6	0.22	2.7
Cummins	α_6	-0.21	-2.7	-0.04	-0.6	-0.23	-2.6	-0.07	-0.9
Isuzu	α_7	-0.15	-1.6	0.31	4.0	-0.13	-1.2	-0.23	-2.3
DDC-4 stroke	α_8	-0.09	1.3	-0.32	-5.2	NA	NA	NA	NA
DDC 2-stroke	α_9	-0.10	1.2	-0.69	-8.0	NA	NA	NA	NA
Critical t-value		t(33,.05)=2.0		t(43,.05)=2.0		t(24,.05)=2.1		t(38,.05)=2.0	

vehicles which are in some sense under powered can emit higher levels of PM. As shown in Table 2.6, NO_x emissions also correlate with inertial weight/HP (coefficient for CBD: 0.003, HDT:0.005; t-statistic for CBD: 2.2, HDT: 4.9), and with model year for the HDT cycle (coefficient -0.05; t-statistic -6.6).

Although vehicle emissions also appear to be correlated with some engine makes, it is inappropriate to draw conclusions as to the effect of engine make based on these results. Inclusion of engine make in the regression in effect shifts the intercept to account for differences in engine technology.

In order to refine our understanding of how federal engine emission standards have impacted in-use emissions, a regression analysis was performed using only those vehicles affected by the regulations. For PM, the analysis included vehicles of model year 1988 and later. Additionally, the PM analysis did not include the urban bus data (vehicles 1, 4, 8, and 21) because the PM emission standard for urban buses is different from that for trucks beginning in 1993. The NO_x analysis includes all vehicles from 1985 and later. The results of these analyses are also reported in Tables 5 and 6. Under this scenario, PM emissions correlate with inertial weight/HP and model year for the HDT cycle, but only with model year for the CBD. For NO_x, emissions correlate with inertial weight/HP, odometer mileage, and model year for the HDT but with none of the factors considered for the CBD.

The expected coefficient for model year in the PM analysis was -0.11, corresponding to the engine certification emission standard reduction of 23% per year, on average, from

1988 to 1995 (the latest model vehicle tested). For the CBD and HDT cycles, the coefficients are –0.10 and –0.08, with similar error bars. This indicates a PM reduction of 17 to 20% per year, ±6% per year at the 95% confidence level, suggesting that reductions in in-use emissions are slightly less or equal to improvements measured by engine certification testing. Surprisingly, NO_x emissions actually increased at an average rate of 7% ± 4% per year for the vehicles tested here on the HDT cycle (95% confidence interval reported), while model year had no significant effect for testing on the CBD. The emissions standard for NO_x in the engine certification test decreased an average of 8% per year during the 1986-1995 period. Thus, NO_x emissions benefits anticipated from engine regulation and certification testing may not have been realized.

Smoke Opacity. Smoke opacity was measured by the SAE J1667, Colorado snap-idle, and Colorado lug-down tests for vehicles 1 through 16 and by SAE J1667 and the Colorado snap-idle test for vehicles 17 through 20. Results are not corrected for altitude. The J1667 test and the Colorado snap-idle test correlate well (r^2=0.98), as expected based on the similarity of these procedures. The Colorado lug-down test does not correlate with the other smoke tests.

Smoke opacity on various tests is commonly believed to correlate with mass emissions of particulate and to be indicative of engine malfunctions. Comparison of PM emissions with J1667 opacity is reported in Figure 2.4. Opacity and mass emissions of PM are not strongly correlated for any test cycle. PM mass emissions on the CBD and HDT weakly correlate with smoke opacity (r^2 values of 0.22 and 0.25, respectively).

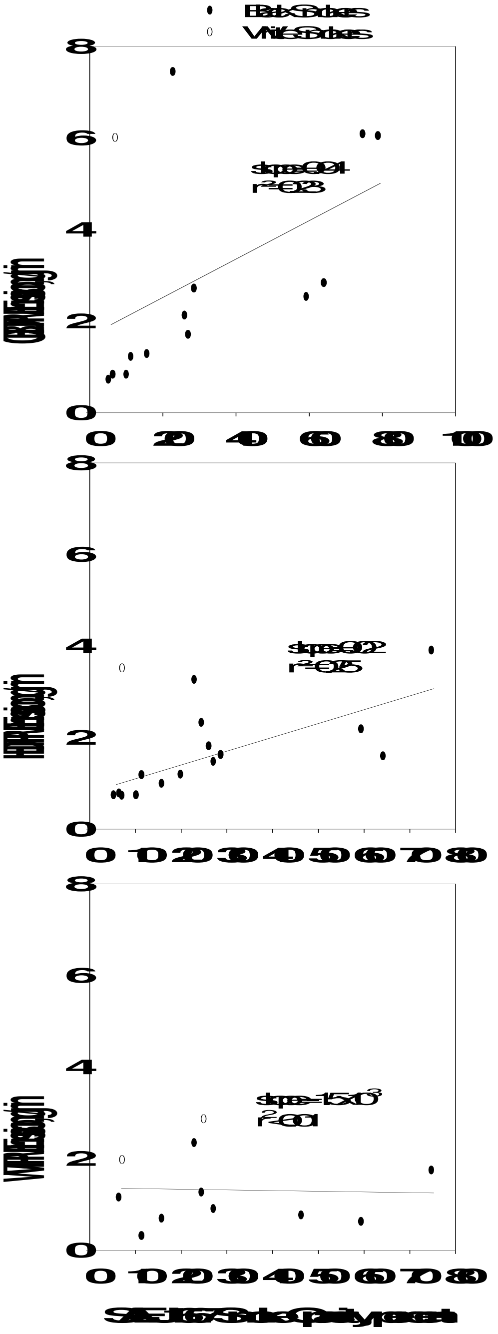


Figure 2.4. Comparison of PM mass emissions with smoke opacity by SAE J1667.

Both correlation coefficient and slope are well below 0.01 for the WVT cycle. Two of the older vehicles (Nos. 10 and 17) were noted to be white smokers and both exhibited opacity values much lower than expected from the correlation based on their PM emissions. CO emissions, which as noted are strongly correlated with PM, are best correlated by J1667 in the CBD cycle. Even in this case the correlation is weak ($r^2=0.34$, slope=0.56), although better than that for opacity and PM. As noted for emissions of PM, CO emissions for white smokers are higher than anticipated based on a smoke opacity measurement. Correlation coefficients (r^2) for the other cycles, and other pollutants (HC and NO_x) are below 0.20 in all cases. Nor is any significant correlation found between regulated pollutant mass emissions and the Colorado Lug-Down Opacity test.

Data Quality. For most vehicles at least one test cycle was replicated 3 or more times allowing statistical estimates of the repeatability of PM, NO_x, CO and THC measurements to be obtained. For PM, most cycles were repeatable with a coefficient of variation (CV, standard deviation as a percentage of the mean) of less than 10% and, in many cases, repeatability is much better than this. For NO_x, all but one of the sets was repeatable to within 1 to 4% based on CV. Variability for hydrocarbon emissions is higher than for the other emissions, ranging from 2% to 21% in all but one case. A large fraction of the THC emissions occur upon startup and this part of the test cycle is extremely difficult to reproduce. Similarly, the CV for CO ranged from 2% to 22%.

Summary. Regulated pollutant emissions were measured from 21 heavy-duty diesel vehicles currently in-use in the Northern Front Range area of Colorado. Hot start emissions average 1.96 g/mi for PM, 23.3 g/mi for NO_x, 19.5 g/mi for CO, and 1.70 g/mi for total hydrocarbon. Sulfate is less than 1% of the emitted PM for all vehicles tested suggesting that the reduction of diesel fuel sulfur content to 500 ppm has been effective in reducing PM emissions. A strong correlation is observed between emissions of CO and PM. Cold starting at 77° F produced a slight increase in PM emissions. Emissions of NO_x, CO, and THC were not significantly different for cold and hot starting. Comparison of emissions from the three driving cycles indicates that on average g/mi emissions follow the trend CBD>HDT>WVT. However, when emissions are converted to a g/gal basis the effect of driving cycle is eliminated for NO_x and largely eliminated for PM. Multivariate regression analyses on these results indicates that in-use NO_x emissions for these vehicles do not reflect emissions improvements expected based on the stricter engine certification test standards put into effect since 1985. The regression analysis also indicates a strong dependence of emissions on test inertial weight relative to engine horsepower. Smoke opacity measurements were not well correlated with mass emissions of PM, CO, NO_x or THC.

CHAPTER 3

CRITICAL REVIEW: IN-USE EMISSIONS FROM HEAVY-DUTY DIESEL VEHICLES

This review is concerned with the regulated emissions from over-the-road heavy-duty diesel fueled vehicles and includes an analysis of various types of emissions measurement methodologies: chassis dynamometer tests, tunnel tests, and remote sensing. The regulated pollutants considered are particulate matter (PM), oxides of nitrogen (NO_x), carbon monoxide (CO), and total hydrocarbon (HC).

Background

Diesel Emissions. The fuel injection process is one of the most important factors in pollutant formation in diesel engines (5). During operation, fuel is injected at high velocity into the cylinder. The liquid atomizes into small drops, which vaporize and mix with air under pressure, and burn. Fuel distribution is non-uniform and the generation of unwanted emissions is highly dependent on the degree of non-uniformity. Carbonaceous soot is formed in the center of the fuel spray where air/fuel ratio is low. As the soot cools, organic compounds derived from the fuel and the lubricating oil adsorb on to the surface or may form organic aerosol by homogeneous nucleation (38,39). Sulfate aerosol (mainly sulfuric acid) also contributes to PM. NO_x is created where the air/fuel ratio is more nearly stoichiometric and high temperatures are generated. Retardation of injection timing, relative to the optimum timing for fuel economy, can decrease NO_x emissions (40). In general, the greater the efficiency of the combustion the greater the NO_x formation, if other factors are held constant. Hydrocarbon, CO, and PM formation are expected to increase under conditions that cause incomplete combustion such as lower combustion temperature or poor mixing. The inverse correlation between NO_x and PM emissions is the main barrier to lowering these diesel emissions simultaneously.

The chemistry and properties of diesel fuel have a direct effect on engine emissions. Relevant fuel properties include sulfur content, addition of oxygenates, aromatic content, and cetane number. Oxygenates can lower PM emissions but may increase NO_x (41). Lowering aromatic content or increasing cetane number can lower emissions of both NO_x and PM (42). Some recent studies suggest that prototype engines, equipped with exhaust gas recirculation and designed to meet future emissions standards, are much less sensitive to aromatic content and cetane number than older engines (43). A recent review of fuel quality impacts on the emissions from heavy-duty diesels also concludes that recent model-year engines show much smaller changes in emissions than older engines for changes in fuel cetane number, density and polyaromatic content (44).

EPA currently regulates diesel fuel to have less than 500 ppm sulfur. The cetane index (a surrogate for actual measurements of cetane number) must be greater than or equal to 40, or the maximum aromatic content must be 35% or less (45). California has placed additional restrictions on the aromatic content of diesel fuel (46). In wintertime in colder climates, on-road No. 2 diesel is well known to contain some percentage (on the order of 15%) of No. 1 diesel to improve cold flow properties. Blending of No. 1 may also lower the aromatic content, resulting in improved emissions performance.

Regulations. Standards for exhaust emissions of PM, NO_x, CO and THC from heavy-duty engines have been set in units of mass emitted per unit work as shown in Table 3.1. Engine manufacturers must comply with standards by testing at least one engine of a given engine model on an engine dynamometer using the Federal Test Procedure (FTP) (1). The FTP (since 1985) requires that the engine be run in a transient manner over a range of load and speed set points, while emissions are measured using specified procedures. The emissions of regulated pollutants are reported in g/bhp-h (or g/kW-h). The test consists of one cold start and at least one hot start run, and results are weighted at a ratio of one cold test to six hot tests to produce a composite emission value. Both cold and hot start runs are conducted at ambient air temperatures of 25+/-5°C.

There is considerable uncertainty as to how to convert engine certification test results into g/mile emissions estimates for the in-use fleet. The EPA has two vehicle emission factor models that provide average in-use fleet emission factors for the four criteria pollutants using conversions from engine certification data. MOBILE5B calculates emission factors for THC, CO and NO_x. PART5 is used to generate PM emission factors. The models include conversion factors (CF) which are multiplied by

Table 3.1 Diesel Emission Standards (g/bhp-h)[81] (0.7457 g/bhp-h=1 g/kW-h).

Year	THC	CO	NO_x	PM (trucks)	PM (urban buses)
1974-1978	16	40	--	--	--
1979-1984	1.5	25	10[a]	--	--
1985[b]-1987	1.3	15.5	10.7	--	--
1988-1989	1.3	15.5	10.7	0.6	0.6
1990	1.3	15.5	6.0	0.6	0.6
1991-1992	1.3	15.5	5.0	0.25	0.25
1993	1.3	15.5	5.0	0.25	0.1
1994-1995	1.3	15.5	5.0	0.1	0.07
1996-1997	1.3	15.5	5.0	0.1	0.05
1998-2003	1.3	15.5	4.0	0.1	0.05

[a] THC+NO_x. [b] Test cycle changed from steady-state to transient operation.

certification engine test results (in g/bhp-h) to get a base emissions rate (BER) unique to the vehicle class and year (47):

$$\text{BER (g/mi)} = \text{g/bhp-h} \times \text{CF (bhp-h/mi)} \tag{3-1}$$

Where:

$$\text{CF} = \rho \text{ (lbs/gal)} / \text{BSFC (lb/bhp-h)} \times \text{FE (mi/gal)} \tag{3-2}$$

ρ = density of diesel fuel

BSFC = fleet wide brake specific fuel consumption for the vehicle class and model year

FE = average fuel economy for a vehicle class and model year

The brake specific emissions are a weighted average based on sales data for a given model year and vehicle class. In-use FE estimates are based on the U.S. Census Bureau's Vehicle Inventory and Use Survey (48), which includes model year and class specific fuel economies. BSFC is calculated based on proprietary values reported by manufacturers, which are then sales-weighted to determine an average for the model year and vehicle class (47).

There are indications that certification test results for at least some of the last decade's engines are not predictive of in-use emissions, as shown below. Some engine manufacturers have used electronic controls to operate engines in a low emissions mode during the certification test. Under conditions that are not characteristic of the FTP, the electronic controls operate the engine in a higher fuel economy mode resulting in considerably higher NO_x emissions for in-use vehicles (49). The EPA has reached a negotiated settlement with engine manufacturers, which requires them to eliminate so-called dual engine maps when the engines are rebuilt, as well as in future engines (50).

The regulations also require engine manufacturers to use "good engineering practice" to estimate the deterioration in emissions over the lifetime of the engine (51) (defined in the regulations as 290,000 miles for heavy heavy-duty diesel engines) (52) Emissions standards in place during the year of manufacture must be met for the lifetime of the engine. Engine manufacturers have demonstrated compliance with this requirement by running the engine for one thousand hours on an engine test stand and extrapolating to the end of its useful life. The actual number of miles simulated in 1000 hours is clearly far less than 290,000. For example, assuming an average speed of 50 mph (a high value), the durability test simulates 50,000 miles. Using this procedure, they find little or no deterioration in emissions. Furthermore, the actual useful life of a heavy-duty engine is can be far above 290,000 miles, with some manufacturers advertising "million mile" engines. There has been minimal in-use testing of engines to determine if deterioration is more significant under in-use conditions.

Environmental Effects. *Altitude.* Altitude effects have historically been an important factor in motor vehicle emissions in high altitude cities such as Denver, Albuquerque, Salt Lake, and Mexico City because of the lower partial pressure of oxygen. This leads to less efficient combustion and higher emissions of partial combustion products such as THC, CO and PM. Diesel engines produced since the early 1980's are turbocharged, i.e., the inlet air to the engine is compressed at higher engine speeds. Therefore, the effect of lower oxygen partial pressure is expected to be most significant at lower engine speeds and idle when the turbocharger is not capable of providing any altitude compensation. Altitude effects will be more severe in cases where turbocharger boost is limited at altitude to prevent turbocharger over speed caused by low intake air density.

Human , Ullman and Baines (53) developed a method for simulating altitude. This system provides reduced pressure to both the intake and exhaust of the test engine. Transient emissions of HC, CO, CO_2, and particulate matter increased with increasing simulated altitude for both transient and steady state operation in a naturally aspirated Cummins NTCC-350 engine. No significant change in NO_x emissions with altitude was observed. They propose a linear correlation between PM emissions and barometric pressure. Chaffin and Ullman (54) used a similar system on a turbocharged DDC Series 60 engine and reported a 35% increase in PM for simulation of Denver's altitude (1600 m) but no effect of altitude on NO_x. The effect of cetane number and aromatic content on emissions at altitude is reported to be the same as at sea level (55).

Humidity. Intake air humidity is well known to affect engine NO_x emissions. Tsunemoto and Ishitani (56) suggest that for diesel engines the reduction in NO_x emissions with increasing humidity is explained by the decrease in oxygen content of the intake air. The Code of Federal Regulations (57) defines a universal humidity correction factor for NO_x emissions:

$$\text{CF for } NO_x = 1/\{1-0.0026\ (H-75)\} \qquad (3\text{-}3)$$

Where H=humidity in grains per pound of dry air. Multiplication of the measured emission by the CF corrects the NO_x emission to a reference humidity of 75 grains of moisture per pound of dry air (multiply by 0.1429 to obtain gm moisture/kg dry air). The correction factor for NO_x will increase and the measured emissions will decrease by about 20% as humidity decreases from 105 to 35 grains per pound of dry air. Additionally, the Engine Manufacturers Association has also recommended a similar correction factor for PM. Humidity can impact the hydration state, and hence the mass, of the sulfate fraction of PM and so a correction of this type will only be accurate at constant fuel sulfur content. McCormick et al. (55) tested the effect of humidity on emissions and found the correction factors to also be accurate at altitude (1600 m).

Temperature. Ambient temperature can be expected to impact emissions from diesel vehicles. However, to date no engine or chassis testing studies at cold temperatures have been reported. The University of West Virginia mobile chassis dynamometer tests vehicles outdoors under ambient conditions, however these researchers have not reported an analysis of their data to determine the impact of ambient temperature on emissions.

Methodology of the Studies Reviewed

Several methods of measuring emissions from heavy-duty vehicles have been used including the chassis dynamometer, tunnel studies, and remote sensing. Also reported has been the development of a trailer to be towed behind the test vehicle while measuring emissions during normal on-road operation (58). However, only limited data are currently available using this approach and it is not discussed further. The following sections review the methodology of these types of studies.

Chassis Dynamometer Studies. A chassis dynamometer system consists of a dynamometer, coupled via gearboxes to drive lines that are connected directly to the vehicle hub, or to rollers, on which the vehicle is parked. The load experienced by a vehicle is the sum of the inertial load, and load due to air and road friction. Researchers have developed a variety of approaches for accomplishing load simulation, typically involving a combination of electric dynamometers and flywheels. Frequently the rollers along with variable weight flywheels are used to simulate the inertial load (59). Other components of the load are estimated from published studies (25) and applied electrically using the dynamometer. Different approaches are also used for emissions collection and analysis. However, the most accurate approach may be a double dilution method, as used for diesel engine certification testing (57).

In chassis dynamometer tests, a vehicle is driven over a cycle or route. A cycle is a speed versus time trace, usually displayed on a computer monitor, which the driver attempts to follow. A route consists of a fixed set of accelerations, decelerations, and cruise sections that are held to a fixed distance by adjusting the length of the cruise sections in real time. The main difference between a cycle and a route is that acceleration at wide-open throttle is used during routes, rather than a specific acceleration rate as in a cycle. In many cases, routes are believed to simulate how real drivers drive heavy-duty vehicles more accurately than cycles.

There is no chassis dynamometer test procedure used for regulatory purposes with heavy-duty diesel vehicles. However, the Code of Federal Regulations presents a heavy-duty chassis test intended for use in measuring evaporative emissions from gasoline and alternative fueled vehicles (26). This test procedure is referred to as the heavy-duty transient truck cycle (HDT or HDTT), the urban dynamometer driving schedule (UDDS), the “schedule d cycle”, and various other names. The cycle is specified as a series of vehicle speeds at one-second intervals, and is shown in Figure 3.1. Procedures for dynamometer calibration and determining the proper inertial load are also outlined. Other chassis dynamometer cycles have also been developed. For example, the Central Business District cycle (60) is an attempt to model inner city driving conditions through repeated accelerations, decelerations and idle periods. This cycle is also shown in Figure 3.1. A commonly used cycle for testing of heavy trucks is the West Virginia 5-Peak or Truck cycle, also shown in Figure 3.1. The cycles shown in Figure 3.1 represent the most widely used cycles to date. However, several other cycles and routes have been used in chassis dynamometer testing (61-65).

Different approaches have been used for the selection of vehicle test inertial weight. For trucks, Clark and coworkers (28) use an inertial weight of 70% of the vehicle’s gross rated weight (GVWR) up to 60,000 lbs, and 42,000 lbs inertial

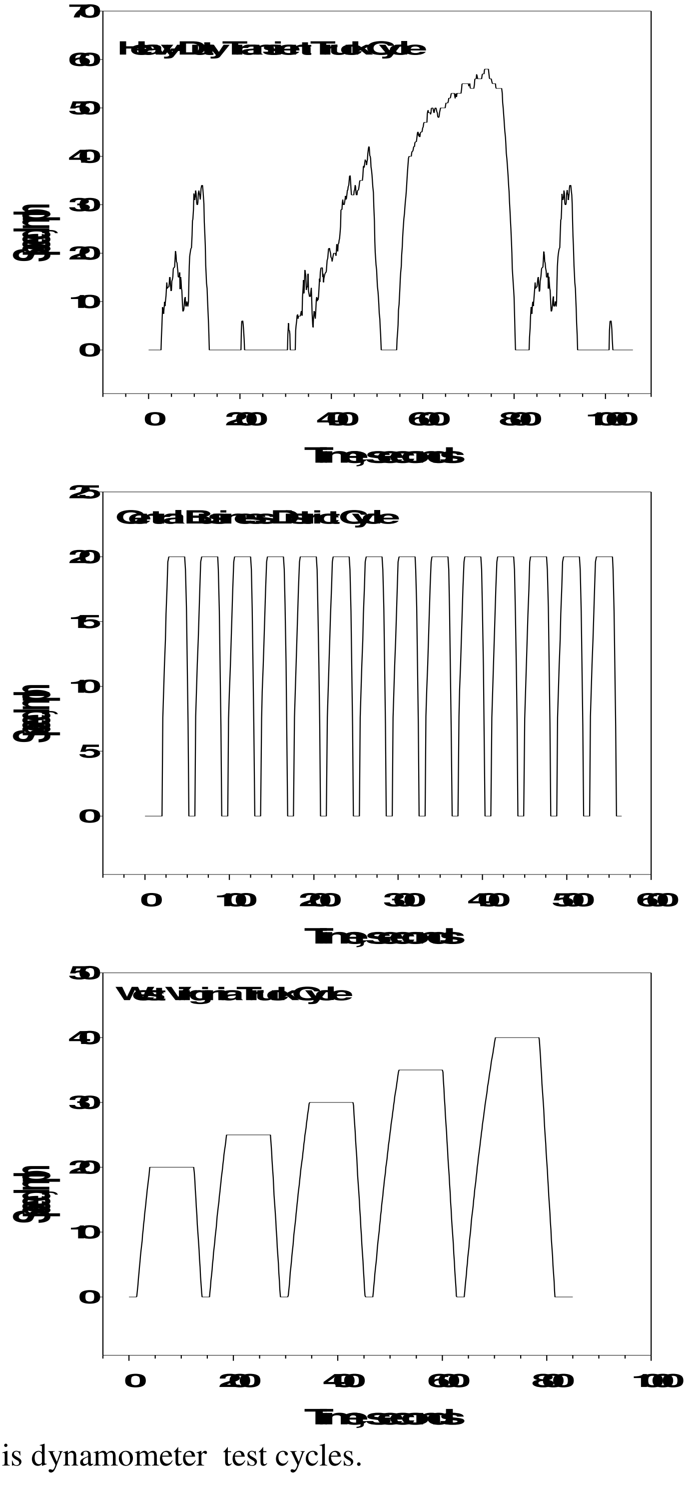


Figure 3.1 Chassis dynamometer test cycles.

weight for heavier vehicles. Buses are tested at an inertial weight equal to their empty weight plus half the passenger load and the bus driver (average weight of each person set at 150 lbs) (66). In the Code of Federal Regulations procedure for the HDT (67), vehicles with empty weight less than one half the rated gross vehicle weight (GVWR) are tested at one half GVWR. Vehicles with an empty weight greater than one half GVWR are tested at their empty weight. In testing at the Colorado School of Mines, the normal approach is to set the inertial weight as the average of the empty weight and the GVWR to simulate the average of a loaded trip and empty return. None of these procedures are based on actual data on load factors typical of truck or bus use. It is common for the inertial loading to be omitted in literature reports of chassis dynamometer tests. This oversight represents a major flaw in these reports because, as shown below, inertial load can have a substantial effect on emissions.

Tunnel Studies. In a tunnel test, the total emissions from all vehicles that enter a tunnel during the test period are measured. Pollutant concentrations are measured in the air at the inlet to the tunnel, in any incoming ventilation air and at the outlet. By multiplying the change in concentration by the estimated airflow through the tunnel, a rate of pollutant emissions is determined. Vehicles traveling through the tunnel are counted and divided into the total emission rate, to generate a per vehicle emissions rate. Taking into account the length of the tunnel puts this on a per mile basis. Other information regarding the measured fleet may be ascertained by observing the license plate numbers of the vehicles through the tunnel, and then comparing with vehicle registration data. Tunnel test results may also measure PM from sources other than tail-pipe emission: road dust, tire wear, etc. However, Weingartner et al. (68) show that very little of the smaller sized particle mass (PM_3) measured in a tunnel in Switzerland is due to tire wear. Early tunnel studies conducted in the 1970's are not included in this review as the vehicles examined would now be more than 25 years old. These data have been reviewed by Pierson and Brachazek (69).

In order to differentiate between heavy-duty and light-duty vehicles, researchers have taken advantage of the fact that vehicles in these two categories may have different daily and weekly cycles. For example, in Switzerland truck travel is not permitted on Sundays, except with a special permit. Some tunnels have more than one bore in each direction (for example, the Fort McHenry Tunnel) (70) and heavy-duty vehicles are only permitted to travel through one of the bores. Thus, it is possible to differentiate the emissions of different types of vehicles by first quantifying the variation in composition of the measured fleet using direct counting. Emissions at different times of the day or week, or in the different bores of a tunnel, are then compared using a regression analysis to quantify the difference (4, 68-71). The regression analysis assumes that average light and heavy-duty vehicle emissions are not affected by daily or weekly cycles, that change the ratio of light to heavy-duty vehicles. Temperature, ambient humidity and traffic speed changes can also bias regressions. If averaging times for emissions measurements are short, individual high or low emitting vehicles may also influence the results. Additionally, there are usually far more light-duty vehicles than heavy-duty vehicles.

Thus, determination of heavy-duty vehicle emissions (especially in the case of CO or THC) may be dependent on a small difference between two large total emission values leading to a large experimental uncertainty. Development of new approaches to quantify diesel emissions in tunnels is necessary to reduce this uncertainty.

Rogak et al. (72) have measured true tunnel dilution rates with a vehicle emitting a known quantity of sulfur hexafluoride (SF_6) and concluded that the very simple aerodynamic models typically used to interpret tunnel test data may lead to errors of as much as a factor of 2. In their study of the Cassiar tunnel in Vancouver, BC they attribute these errors to inlet (background) pollutant concentrations, which are significantly higher near the roadway than at the top of the tunnel. Additionally, outlet air flow velocity profiles are not accurately predicted by standard pipe flow models. Aerodynamic correction is not necessary when emissions are normalized to the amount of CO_2 emitted (i.e., on a fuel burned basis), if the assumption is made that all vehicle emissions are diluted to the same extent over the tunnel test volume.

Remote Sensing Studies. In remote sensing, emissions are measured as vehicles pass by a measurement station. UV and IR light of specific wavelengths is passed through the exhaust plume of the vehicle to a detector. The light absorbed is proportional to the amount of CO, CO_2, or THC (via IR) and NO (via UV) between the light source and the detector. Because the fraction of the exhaust plume analyzed by the system will depend on turbulence, wind direction, and other factors, remote sensing can only be used to determine the ratios of the pollutants to each other, not absolute values. However, it is possible to generate emissions estimates on a fuel-used basis because CO_2 emissions are proportional to the amount of fuel burned. For diesel vehicles, CO and THC emissions are generally too small to significantly effect the carbon balance.

Bishop and Stedman have detailed operation, limitations, and operational problems in remote sensing systems in a recent paper (73). NO_x and CO are specific compounds to which the system can be unambiguously calibrated. Others have found that the wavelength chosen by Bishop and Stedman for non-dispersive infrared (NDIR) analysis of THC accurately measures alkanes, but may yield substantially different results than flame ionization detection methods in the analysis of aromatics and olefins (74). The accuracy and precision of this system has been evaluated by several investigators (75-77). They found that CO_2 and CO readings are accurate to within 5%, THC to within 15%, and NO (tested on one light duty vehicle and two diesel vehicles) to within 20% of standard analytical methods. Because the system measures only NO, and not NO_x, some correction factor must be applied to compare remote sensing measurements with chassis dynamometer or tunnel test methods. Jimenez and coworkers (78), using another type of remote sensing system able to measure NO and NO_2, found that 8% of the total NO_x in diesel exhaust is NO_2. Not addressed by these remote sensing projects, is the variability of NO/NO_2 ratio in diesel engine exhaust with engine load and other factors (79,80). Another potential effect on the NO/NO_2 ratio measured through remote sensing is the catalytic effect of diesel PM on conversion of NO to NO_2 (79).

Operational difficulties arise on rainy or snowy days, and in very cold weather, but remote sensing systems are capable of measuring the emissions of as many as 10,000 vehicles per day under good conditions (73). However, the testing stations must be located so that there is only one lane of vehicles between the light source and detector. The variability of exhaust pipe placement on heavy-duty diesel vehicles (some at curb level, others placed above the cab) means that two systems would be necessary to obtain unbiased results for all the vehicles passing a single location. Additionally, in a recent study conducted at a truck weigh station, remote sensing obtained acceptable signal/noise ratios on only about two-thirds of the tested vehicles (77,81).

Results

In the following, NO_x emissions are reported as equivalent NO_2. That is, the total molar emissions of NO_x have been multiplied by the molecular weight of NO_2 to obtain mass emissions. This formalism was chosen because this is how mass emissions are calculated for compliance with heavy-duty emissions standards.

Chassis Dynamometer Studies. Chassis dynamometer data from more than 250 different vehicles reported in 20 different studies are included in this review (61-65, 82-96). Also included are data collected by West Virginia University and available at www.afdc.nrel.gov, and unpublished results from the Colorado School of Mines. Chassis dynamometer emissions results are listed in Table 3.2 (included on computer disk). Data for vehicles tested on No. 2 diesel fuel and winter blend diesel (where No. 1 diesel is added at about 10-15 volume % in order to improve cold flow properties) are reported. Tests conducted using neat No. 1 diesel, jet fuel, or experimental fuels such as biodiesel or Fischer-Tropsch diesel have been excluded, as the intent is to provide data for typical in-use conditions.

Also excluded from this analysis was a study done in the early 1970's in which over 60 vehicles were tested on a chassis dynamometer (97). These vehicles were excluded because the test methods employed produced emissions in g/bhp-h rather than g/mile and thus are not comparable to the other results presented here. In addition, it is unlikely that many of these 1970 model year vehicles are still on the road. However, it is of interest that the NO emissions (converted to a fuel consumption basis, corrected for humidity, and reported as equivalent NO_2) were considerably higher (averaging about 200 g/gal) and more variable than for more recent studies. PM was not measured.

There are insufficient data available for examination of the impact of lowering fuel sulfur content to 500 ppm that was instituted on October 1, 1993 because there were few tests conducted using in-use fuels prior to 1993. However, engine dynamometer studies (98-100) show a significant decrease in PM emissions from lowering of fuel sulfur content. Several studies included testing of vehicles equipped with experimental emission control devices. These data are not included in the analysis presented here.

Fleet Analysis. More than 4.6 million heavy-duty trucks are presently in-use in the United States (48). More than one-third of the heavy-duty vehicles included in the chassis studies are transit buses. In contrast, EPA's PART5 default VMT projections assume that transit buses comprise less than 5% of total heavy-duty VMT in 1998. Table 3.3 compares the make up of the fleet of trucks that has been tested with the in-use truck fleet according to the 1997 Vehicle Inventory and Use Survey (48). The tested fleet is heavily weighted towards vehicles in the 33,000-60,000 lb range. Testing of more both heavier and lighter vehicles is necessary for a good understanding of pollutant emissions from the in-use truck fleet. Analysis of the tested fleet also shows that the model year distribution is skewed toward newer vehicles. The 1997 Vehicle Inventory and Use Survey[16] indicates a flat distribution with roughly the same number of in-use vehicles for each of the model years in the decade preceding 1997. The 1992 Truck Inventory and Use Survey shows the same trend (101). Analysis of odometer mileage for the tested fleet shows that 45% of the vehicles had less than 50,000 miles at the time of testing. Only 10% of the vehicles had more than 250,000 miles. While the mileage distribution of the in-use fleet is unknown, it seems unlikely to be as heavily weighted to low mileage vehicles. Because of the relatively low mileage of most of the vehicles tested, deterioration of emissions may not be reflected in the results.

Emission Factors. Table 3.4 reports mean g/mile emission values for vehicles tested in the reported studies. Results are reported for the tested fleet in total, as well as the subclasses of buses, all trucks, and different classes of trucks. Similar data in g/gallon are listed in Table 3.5. While these broad averages do not provide a complete picture of in-use diesel emissions (see below), the striking feature is the lack of trends or great difference between vehicles of various classes.

Table 3.3 Comparison of in-use truck fleet with truck fleet tested on chassis dynamometer, percent of total vehicles.

Class	**In-Use Trucks, 1995 Census[16]**	**Tested Trucks**
3	17.7	1
4 & 5	13.3	0
6 & 7	25.0	17
8A	20.9	52
8B	23.1	30

Table 3.4 Mean emission factors obtained from chassis dynamometer testing in g/mile, 95% confidence interval shown.[a]

	THC	*NO$_x$*	*CO*	*PM*
Total Fleet	1.98±0.19	27.3±1.0	11.3±1.0	1.36±0.10
Buses only	1.85±0.20	29.6±1.3	11.5±1.4	1.19±0.11
Trucks only	2.15±0.40	24.4±1.4	11.0±1.3	1.56±0.16
Class 3 Trucks[b]	1.15	4.61	18.7	2.11
Class 4-5 Trucks[c]	--	--	--	--
Class 6-7 Trucks	1.69±0.66	18.1±2.5	11.6±3.7	1.50±0.43
Class 8a Trucks	2.22±0.56	18.1±1.2	9.36±2.69	1.29±0.26
Class 8b Trucks	2.32±0.57	29.8±1.9	11.3±1.7	1.53±0.19

[a] Classes are defined by gross vehicle weight: Class 3=10,001-14,000 lb, Class4-5=14,0001-19500 lb, Class 6-7=19,501-33,000 lb, Class 8A=33,001-60,000 lb, Class 8B>60,000 lb.

[b] Data for class 3 trucks are mean values for one truck tested on three chassis cycles.

[c] No chassis data for class 4-5 trucks was included in the studies reviewed

Table 3.5 Mean emission factors obtained from chassis dynamometer testing in g/gallon, 95% confidence interval shown.

	THC	*NO_x*	*CO*	*PM*
Total Fleet	7.40±0.62	110±3	44.7±3.6	5.71±0.46
Buses only	6.02±0.54	110±5	41.0±4.8	4.39±0.43
Trucks only	9.09±1.2	110±5	49.2±5.4	7.31±0.82
Class 3 Trucks[a]	8.94	35.8	146	16.7
Class 4-5 Trucks[b]	--	--	--	--
Class 6-7 Trucks	9.36±2.7	106±15	62.6±15.1	7.73±1.61
Class 8a Trucks	12.6±3.2	90.9±4.7	43.2±11.0	6.33±1.11
Class 8b Trucks	7.60±1.41	119±6	46.5±6.6	6.30±0.74

[a] Data for class 3 trucks are mean values for one truck tested on three chassis cycles.
[b] No chassis data for class 4-5 trucks was included in the studies reviewed.

Effect of Driving Cycle. It is well known that the choice of driving cycle can impact emissions. The study detailed in Chapter 2 found that in general, g/mile emissions levels for regulated pollutants were highest for the CBD cycle, followed by the HDT cycle and the lowest emissions were generated during the WVT cycle (82). For NO_x this can be seen in Figure 3.2, top. Here g/mile NO_x from the HDT and WVT driving cycles is plotted against NO_x on the CBD cycle for all of the vehicles included in this review that were tested on more than one of these driving cycles. Clark et al. have also studied the effect on emissions of different cycles, including the CBD cycle, the New York Bus Cycle, the WVU 5-Peak Cycle and the WVU 5-Mile Route. They found that variations in cycle could increase PM emissions on a g/mile basis by as much as a factor of six and NO_x emissions to a lesser extent (102). In the previous chapter it was shown that much of the cycle to cycle variability in NO_x, for a given vehicle, could be eliminated by converting the data to a gram per gallon of fuel consumed basis. For the larger data set reviewed here, NO_x emissions from vehicles driven on the CBD cycle, and other cycles are compared in Figure 3.2, bottom. On this basis, CBD emissions are not higher, and the results fall more randomly about the parity line. A linear regression fit of these plots yields r^2 values of 0.34 for the g/mile data and 0.78 for the g/gallon data. A similar analysis for PM and CO using this data set is not as successful. Viewing emissions of these pollutants on a fuel consumption basis does not remove cycle variability. However, a number of recent studies have suggested development of pollutant inventories on a fuel consumption basis (33,34). Furthermore, remote sensing

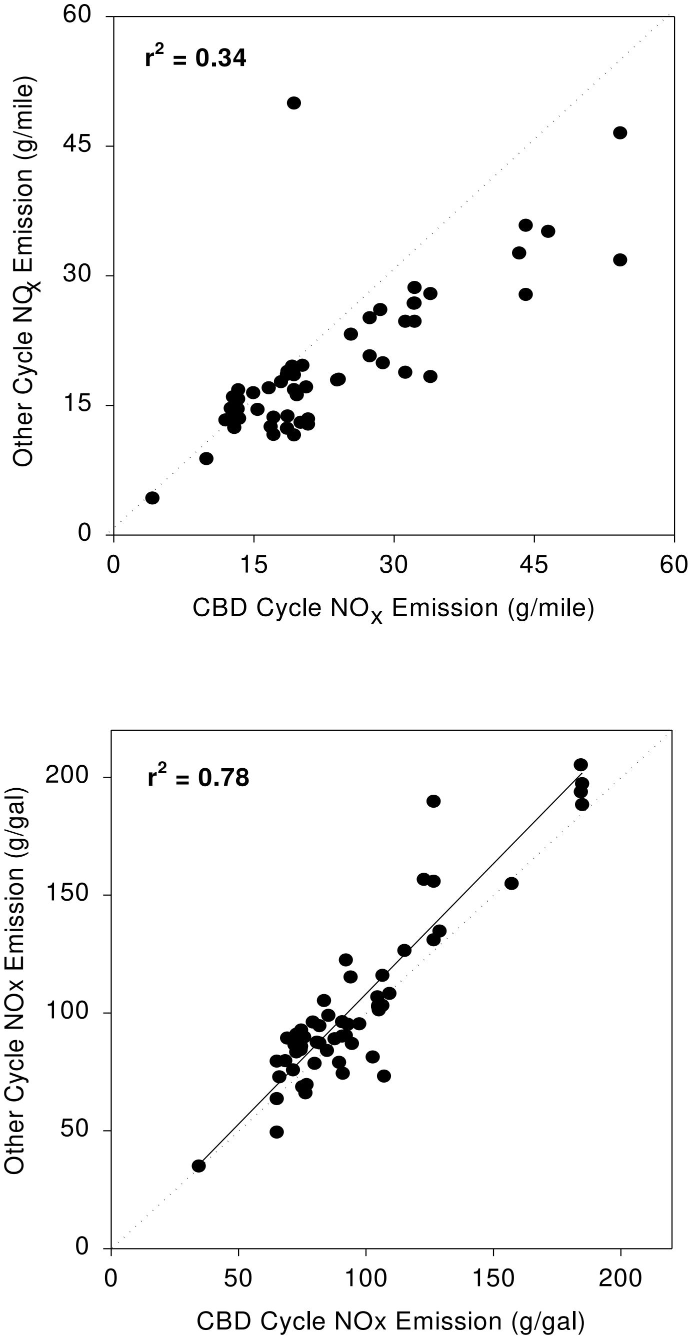


Figure 3.2 Comparison of measured emissions from various cycles to the CBD cycle. Use of g/gallon emissions factors reduces variability between cycles.

studies produce pollutant emission factors that, on a fuel consumption basis, can be compared with chassis and tunnel testing results. For these reasons, most analyses in this paper will be performed using emissions results on a g/gallon of fuel consumed basis.

Effect of Inertial Weight. Recent studies (103, and Chapter 2) on a limited number of vehicles have found the impact of inertial weight on emissions to be significant for PM. An increase in inertial weight from about 50% to 100% of GVWR lead to an increase of PM emissions by a factor of about 2.5 on a g/gallon basis, and 2 on a g/mile basis. A slight increase in NO_x emissions with increasing inertial weight was also observed. Others (104) have shown that increasing the inertial weight by about 50% doubled the CO emissions, and increased NO_x emissions by 30% on a g/mile basis. PM was not measured in this study, but considering all of the chassis dynamometer data together, CO and PM emissions are somewhat correlated ($r^2 = 0.45$) according to the following equation:

$$\text{PM (g/gal)} = \text{CO (g/gal)} \times 0.060 + 0.71 \tag{3-4}$$

The correlation between CO and PM suggests that CO emissions would also have increased significantly with the increase in inertial weight.

Tunnel Studies. There are literature reports of several tunnel tests where the appropriate data were collected to estimate emissions from heavy-duty diesel vehicles. The average model year for heavy-duty vehicles was reported in only one case (4) and was found to be 1988, in a test conducted in 1997. Pierson et al. (70) report that 73 to 99% of the vehicles classified as heavy-duty were diesel, with the remaining spark-ignition, and only 2% were buses. Additionally, speeds were higher and steadier than those normally considered in chassis dynamometer and remote sensing studies. Emission factors from these studies are listed in Table 3.6. In order to allow comparison between observed values, all results have been reported on a common basis. Thus, emission factors reported in the original references in g/gallon and g/mile may vary somewhat from what is reported here, due to different assumptions regarding fuel density, weight percent carbon in the fuel, and average fuel efficiency. See the footnotes to Table 3.6 for the assumptions made. It would be useful if future tunnel studies included ambient humidity measurements, as well as ambient temperature and barometric pressure measurements, to facilitate comparison.

Gram per mile emission factors vary substantially for the various tunnels, with NO_x ranging from 9 to 24 g/mile and PM ranging from 0.6 to nearly 1.7 g/mile. Note that only three of the five studies report emission factors for PM. On a g/gallon of fuel consumed basis agreement between the studies for NO_x emissions is reasonably good, suggesting an average level of 130 g/gal. As noted above, consideration of NO_x emissions on this basis in chassis testing minimizes differences between cycles for a given vehicle, and apparently also between different modes of driving in different tunnels. A similar result was obtained in the Ft. McHenry tunnel study where uphill and downhill emissions were approximately equal on a g/gallon basis but not on a g/mile basis (70).

Table 3.6. Emissions results from tunnel tests.

Reference		Fuel Efficiency (mi/gal)	NO_x[d] (g/mi)	NMHC (g/mi)	CO (g/mi)	PM (g/mi)	CO_2 (g/mi)	NO_x[d] (g/gal)	NMHC (g/gal)	CO (g/gal)	PM (g/gal)
71	Cassiar Tunnel, 1995, Vancouver	8.03[a]	19.50+/-4.22	-0.16+/-0.88	6.79+/-11.8		1280+/-40	157+/-34	-1+/-7	55+/-95	
68	Gubrist Tunnel, 1993, Zurich	5.60[c]				0.62+/-0.02[e]					3.5+/-0.1[e]
70	Fort McHenry Tunnel, Downhill, 1992, Baltimore	11.46[a]	9.66+/-0.32	0.92+/-0.21	6.8+/-1.5		897+/-48	111+/-4	11+/-2	78+/-17	
70	Fort McHenry Tunnel, Uphill, 1992, Baltimore	5.42[a]	22.50+/-1.00	2.55+/-1.05	14.3+/-5.5		1897+/-168	122+/-5	14+/-6	78+/-30	
70	Tuscarora Tunnel 1992 Pennsylvania	6.44[a]	19.46+/-0.85	0.68+/-0.20	6.03+/-1.61		1596+/-78	125+/-5	4+/-1	39+/-10	
4	Caldecott Tunnel, 1997, San Fransisco	5.42[b]	23.82+/-2.98			1.43+/-0.12[f]		129+/-16			7.7+/-0.6[f]

[a] Calculated from observed CO2 emissions assuming fuel density 7.1 lbs/gal and C is 87% of diesel fuel by weight. [b] Since CO2 emissions not available fuel efficiency assumed to be the same as in slightly uphill tunnel (Fort McHenry). [c] slope of tunnel unknown so used average fuel efficiency for the United States. [d] NO_x reported as NO_2. [e] PM3. [f] PM 2.5. [g]Uncertainties reported as +/- 1.0 standard deviation, except where literature report did not specify uncertainty level; in those cases uncertainty listed identically to literature result

Remote Sensing Studies. Five remote sensing studies of in-use diesel vehicles have been reviewed. The results are included in Table 3.7. Bishop et al.(105), in a study conducted at the Tuscarora Mountain Tunnel in Pennsylvania, report only average CO and THC emissions for a small number of heavy-duty vehicles. The same measurement system used by Bishop was upgraded with an NO detector and used in 1997 (77,81) to measure the emissions of several thousand heavy-duty commercial trucks at a truck weigh station. Analysis of the study's results included an evaluation of the distribution of emission results with an aim towards determining if repair of a few high emitters could significantly reduce emissions. CO and THC emissions were found to be distributed such that repairing a few percent of the high emitters could reduce overall heavy-duty diesel emissions significantly. However, NO emissions were more normally distributed, such that the worst 5% of the fleet was similar to the remainder of the fleet.

Similar to the tunnel test results, literature reports make varying assumptions as to fuel density and weight percent carbon in fuel in order to estimate fuel economy. The results in Table 3.7 have been recalculated on a common basis using the reported pollutant/CO_2 mole ratios. As with the tunnel studies, measurements of humidity, temperature and barometric pressure should be included in future remote sensing studies. Additionally, to accurately quantify NO_x remote sensing researchers need to address the question of how NO/NO_x ratios change under different driving conditions.

Table 3.7. Remote Sensing Results for Heavy-Duty Vehicles

	Reference	Year Study Conducted	Emissions (g/gal)
NO_x (reported as NO_2)	Jimenez et al., 1998 (78)	1997	150 [a,b,c]
	Countess et al., 1998 (77,81)	1997	108 [a,b,c]
	Countess et al.,1999 (110)	1998	187 [a,b,c]
	Morris et al., 1999 (111)	1998	81 [a,b,c]
CO	Bishop et al., 1996 (105)	1992[e]	59 [b]
	Countess et al., 1998 (77,81)	1997	54 [b]
	Countess et al., 1999 (110)	1998	85 [b]
	Morris et al., 1999 (111)	1998	76 [b]
THC	Bishop et al., 1996 (105)	1992	0.002 HC/CO_2 mole ratio [d]
	Countess et al., 1998 (77,81)	1997	0.00073 HC/CO_2 mole ratio [d]
	Morris et al., 1999 (111)	1998	0.000375 HC/CO_2 mole ratio [d]

[a] Remote sensing measures NO. The reported value was corrected to a NO_x (as NO_2) value by assuming 90% (mole fraction) of NO_x is NO. [b] Emissions in g/gal calculated by assuming that fuel density is 7.1 lb/gal and C is 87% by weight of fuel. [c] No humidity correction factor is included. [d] In order to calculate emissions in g/gal, an average molecular weight is needed. [e]This study was done in conjunction with the tunnel study described by Pierson et al.[40]

Analysis

Impact of Model Year. Only the chassis dynamometer data set provides information on specific vehicle properties such as model year. Graphing all of the chassis dynamometer data in comparison to year, as in Figure 3.3, suggests that in-use emissions of THC, CO and PM have fallen steadily during the last ten years. This is not surprising given that PM emissions standards have tightened considerably during that time (by a factor of 6 for trucks and a factor of 12 for urban buses). A regression line for these data is shown in Equation 5:

$$\log \mathrm{PM} = -.04\,(\text{model year}) + 81.3 \qquad (3\text{-}5)$$

The slope of the regression line suggests that in-use emissions have decreased by 10% for each year during the period studied. For the period 1988 to 1997 a 22% per year change would be expected based on changes in the emission standards. Generally engine changes that reduce PM increase the efficiency of combustion and therefore, will reduce CO and THC emissions.

There is no change in average NO_x emissions during the time frame considered, despite a reduction in the permitted engine emissions from 10.7 g/bhp-h in 1985 to 4.0 g/bhp-h in 1998. This observation is supported by chassis, tunnel, and remote sensing measurements. The constancy of NO_x emissions, may be due to vehicle effects not measured by the engine test or, the use of engine computer programs that EPA has determined to be illegal emissions control defeat devices, an issue addressed by EPA in

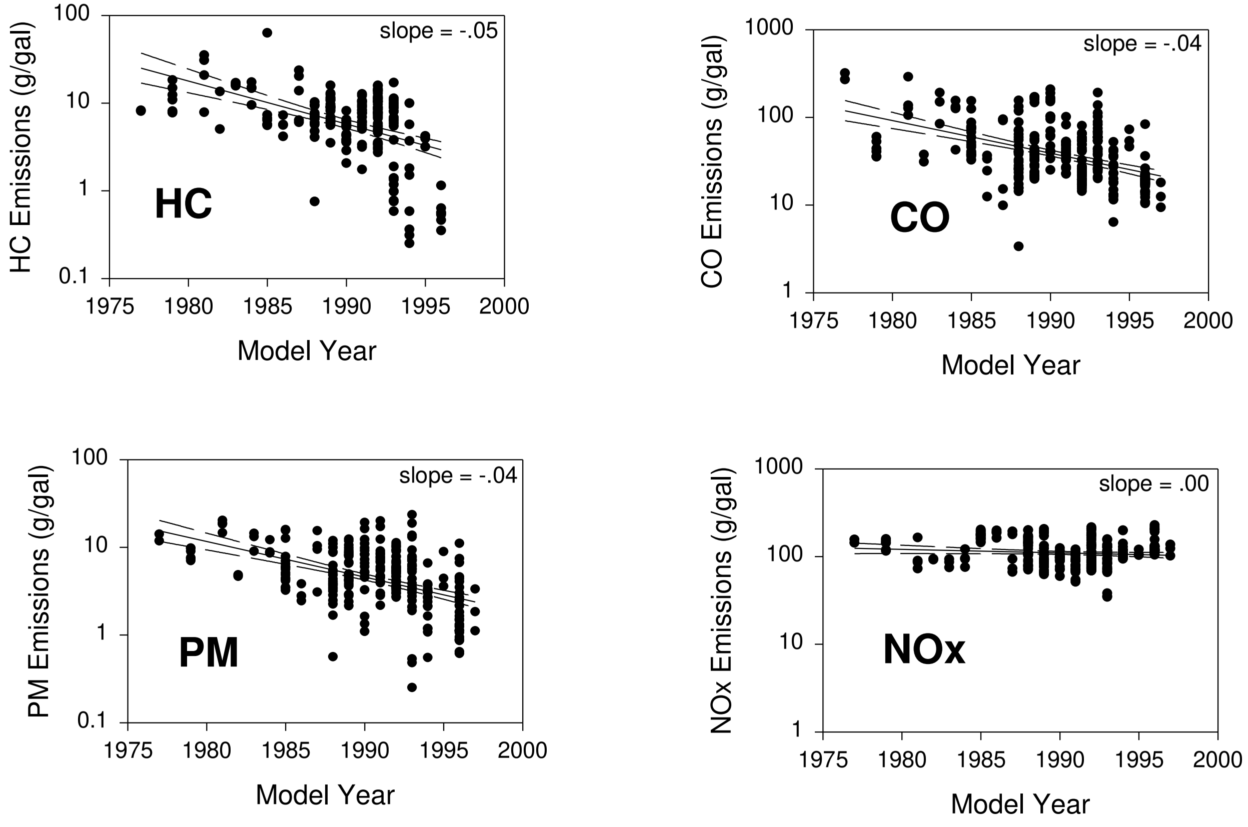


Figure 3.3 Model year trends in heavy-duty diesel emissions as measured on the chassis dynamometer, shown with 95% confidence intervals around the linear regression slope (regression is calculated on the log of pollutant emission).

its recent settlement with the diesel engine manufacturers. Note that the data reviewed here showed only a slight improvement in fuel efficiency (less than 1% per year) over the years studied. Therefore, the trends shown in Figure 3.3 are the same whether shown on a g/gal or g/mile basis. The NO_x data have been plotted in Figure 3.4 with the results from electronic engines (i.e., electronically controlled fuel injection) separated from those for vehicles equipped with mechanical engines. Unfortunately, there were many engines, particularly in the model years 1990-1992 where inadequate information is provided in the literature to determine if the engines have mechanical or electronic fuel injection. When NO_x emissions from mechanical and electronic engines are plotted separately, it can be seen that the highest NO_x emitters over the last ten model years are from the electronically controlled engines. A recent chassis dynamometer study which attempted to minimize confounding factors by conducting all tests in one laboratory and selecting eleven same-sized (80,000 lbs Gross Vehicle Weight) trucks equipped with similar DDC Series 60 engines and varying only the model year between 1991 and1998 found a large and consistent increase in NOx emissions with later model year engines. PM emissions decreased over the same time span (106).

Deterioration. Weaver and coworkers have examined potential causes and frequency of PM emissions deterioration for in-use heavy-duty diesel vehicles (107). Potential causes include manufacturing defects and malfunctions such as retarded timing, fuel injector malfunction, smoke limiting mechanism problems, clogged air filter, wrong or worn turbocharger, clogged intercooler, engine mechanical failure, excess oil

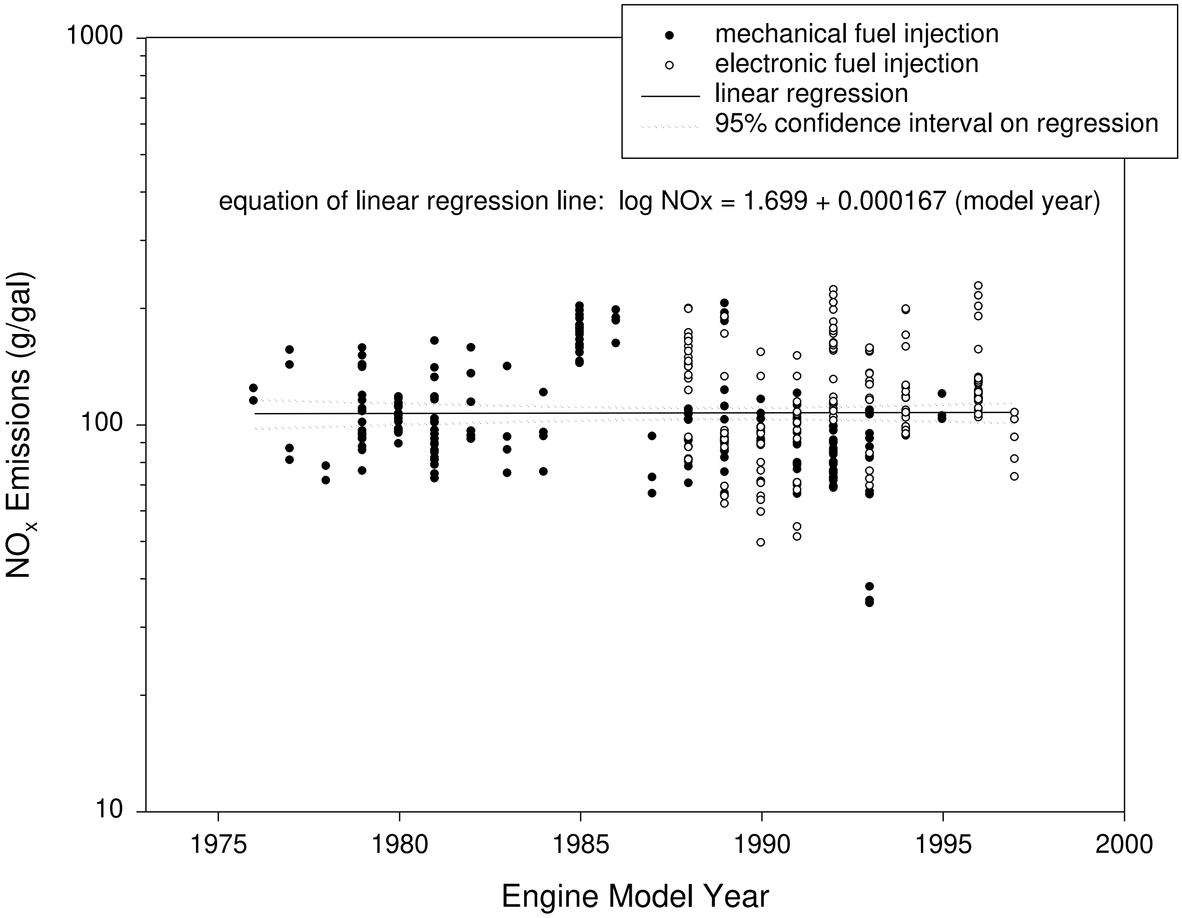


Figure 3.4. NO_x emissions trends by model year for electronic and mechanical engines.

consumption, and electronics that have been tampered with or have failed. The effect of these defects on other pollutants was not considered in Weaver et al.'s study, but generally defects that reduce the efficiency of combustion will also increase HC and CO emissions. NO_x emissions, in contrast, may actually be reduced. Depending on the type of electronic failure or tampering there can be either a positive or negative effect on CO, HC, PM and NO_x emissions. This study concluded that the overall effect of emissions deterioration could be incorporated into the PART5 model by including a deterioration factor. An inspection and maintenance program was predicted to reduce emissions deterioration significantly. Others have attempted to determine if the effects of deterioration could be detected in in-use vehicles. In Chapter 2, a linear multivariate regression analysis found that PM emissions were positively correlated with odometer mileage (several other correlation factors were also identified, including model year). A similar analysis performed on the chassis dynamometer results included in this review found that PM emissions could not be correlated with odometer mileage. For this much larger data set, factors such as differences in measurement methods and test conditions at the various testing facilities may confound the results.

More recently McCormick et al. (108) attempted to quantify the emission benefit of opacity testing and repair on mass emissions. In their study of twenty vehicles which were identified as visible smokers, they found that PM emissions decreased 44% and NOx increased 32% for pre-1991 vehicles upon repair. For the 1991 and newer vehicles repairs improved PM by 43% and increased NOx emissions by 19%. If one can assume that repairs return engines to approximately new condition, it could be concluded that deterioration effects result in PM emissions to increase and NO_x emissions to decrease.

Impact of Environmental Factors. Because of the lower barometric pressure at high altitude, the effective air/fuel ratio in the engine is lower at idle and low engine speeds. As noted in the Background section, this produces higher PM emissions from diesel engines. A comparison of results acquired at various low altitude locations with results obtained in Denver, Colorado at 1600 m above sea level is shown in Figure 3.5. The linear regression models of the low and high altitude data show that low and high altitude results are different at the 95% confidence level. The analysis suggests an approximate 2 g/gallon increase in PM at altitude in the model years studied. This results in an approximately 0.43 g/mile difference for a typical vehicle with a fuel economy of 4.65 mpg.

Two-Stroke versus Four-Stroke Engines. In recent years two-stroke diesel engines have been phased out in favor of four-stroke engines as the latter can produce lower PM emissions on the heavy-duty engine FTP. Note that most two-stroke engines are used in urban buses, and thus almost all available data is for the CBD cycle. However, even when confounding factors are eliminated to the extent possible there is no statistical difference detectable at the 95% confidence level between measured PM and NO_x emissions from the two types of engines. In retrospect, this result is not surprising, as engines manufactured in the same year were required to meet the same regulatory

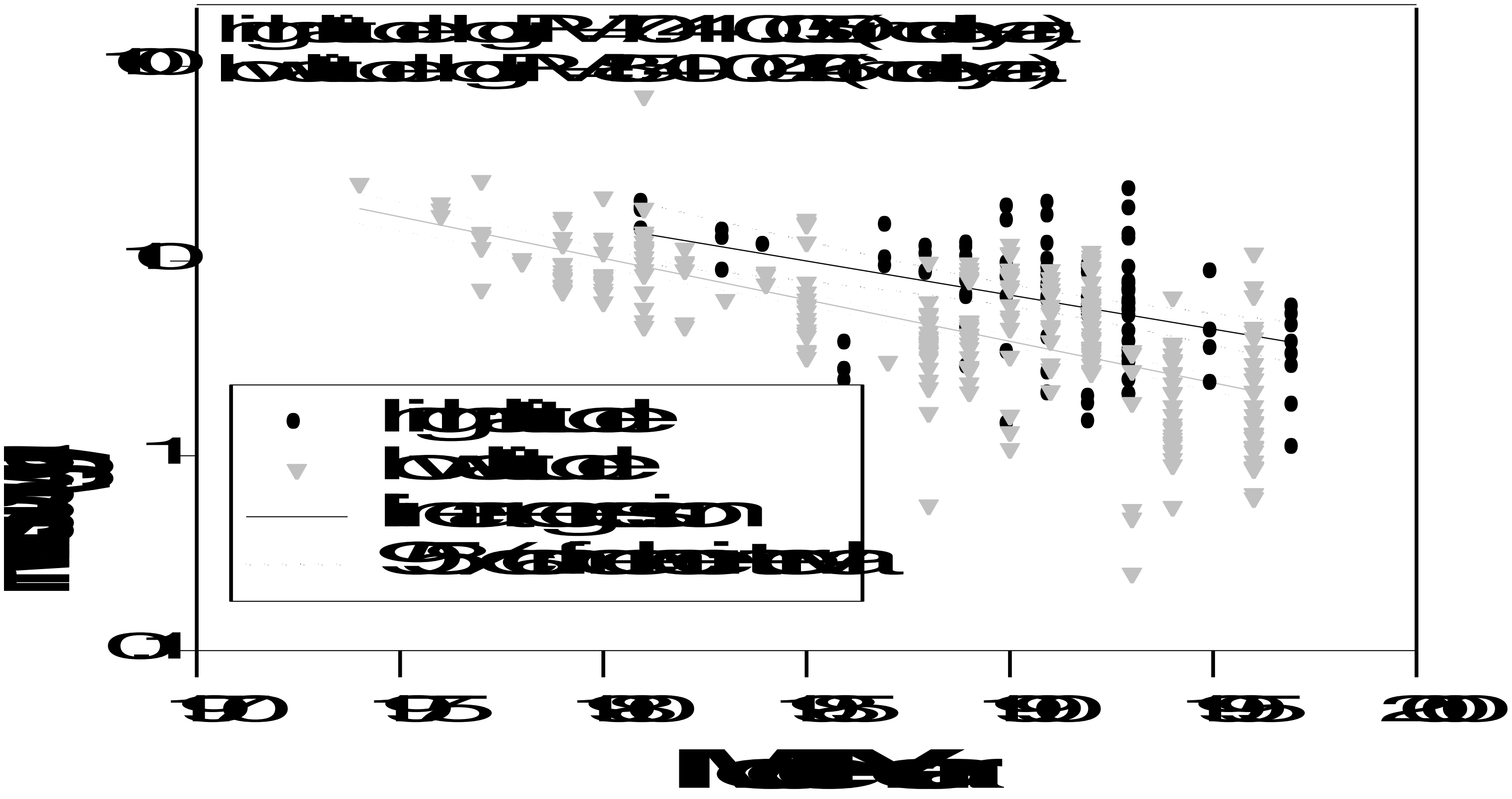


Figure 3.5 Effect of altitude on PM emissions.

standard, regardless of the type of technology used. Four-stroke engines are being used to meet the stricter PM standards in effect today.

Comparison of Emission Factors from Various Studies. Table 3.8 compares g/gallon emission factors from chassis, tunnel, and remote sensing tests by assuming that the on-road fleet measured in tunnels and by remote sensing is made up primarily of vehicles from 0 to 10 years old. There is reasonable, although not quantitative, comparability between chassis results and the average results from tunnel and remote sensing tests. Not surprisingly, the largest difference between chassis and tunnel tests appears for the pollutants PM and THC. Since THC emissions for diesel vehicles are very low in comparison to gasoline vehicles, tunnel test results for THC have a high degree of uncertainty. PM measurements in tunnels are for vehicles traveling at a relatively constant rate of speed while the highest diesel PM emissions occur during acceleration.

For comparison purposes NO remote sensing data used in Table 3.8 has been corrected by assuming that 90% of the NO_x emitted by the vehicle is NO (the balance being NO_2). No correction was made to either tunnel or remote sensing results for humidity, as none of the studies reported humidity measurements. While the average of the NO_x values measured by each method are in reasonable agreement, remote sensing results for CO and NO_x appear to be more variable than tunnel tests. This may be because each remote sensing test location examines engines operating in a unique driving mode, in comparison to the relatively similar highway cruising conditions measured in

Table 3.8 Comparison Between Chassis Dynamometer, Tunnel, and Remote Sensing Results

Pollutant	Chassis Dynamometer Tests[a]		Tunnel Tests[b]		Remote Sensing Tests[b]	
	Model Years	(g/gal)	Year	(g/gal)	Year	(g/gal)
NO_x (as NO_2)	all years[c]	113	1992	111	1997	150
			1992	122	1997	108
			1992	125	1998	187
			1995	157	1998	81
			1996	129		
CO	1982-1992	56	1992	78		
			1992	78	1992	59
			1992	39		
	1985-1995	51	1995	55		
	1987-1997	46			1997	54
	1988-1998	44			1998	85
					1998	76
THC	1982-1992	8	1992	11		
			1992	14		
			1992	4		
	1985-1995	7	1995	-1		
PM	1983-1993	6.2	1993	3.5		
	1986-1996	5.1	1996	9.0		
	1987-1997	5.1	1997	7.7		

[a] Emissions values are average of all values for listed model years. [b] For references and calculation methods used to calculate these values see notes with Tables 6 and 7. [c] NO_x emissions as measured on the chassis dynamometer do not appear to change with model year.

tunnel tests. CO measurements by remote sensing have been consistently higher than measurements made on similar model year vehicles by chassis dynamometer. Remote sensing of heavy-duty diesels carried out in conjunction with a tunnel test showed only fair comparability (i.e., about ±50%) for CO/CO_2 and $NMHC/CO_2$ ratios (70).

Comparison to Model Emission Factors. The EPA has two vehicle emission factor models which provide average in-use fleet emission factors for four criteria pollutants. MOBILE5B calculates emission factors for VOCs, CO and NO_x. PART5 is used to generated particulate emission factors. Estimates can be generated for both historic and future years under various conditions which affect in-use emissions levels. These models are used by the EPA in evaluating mobile source emission control strategies, by states and local and regional planning agencies to develop emission inventories and control strategies for State Implementation Plans under the Clean Air Act, and in the development of environmental impact statements.

The EPA released a new version of the MOBILE model, MOBILE6 in March 2001. Among other changes, MOBILE6 includes updated emission factors for heavy duty diesel vehicles that take into account the higher NO_x emissions due to the dual engine maps in many electronic engines of the 1990's as well as reductions in future emissions due to the newly promulgated PM and NO_x standards (109).

In the case of MOBILE5B, the predicted emissions are dependent on altitude, model year and vehicle speed. MOBILE5B does not differentiate between different vehicle sub-classes within the heavy-duty diesel class. In order to determine the appropriate vehicle speed for input to MOBILE5B an average speed for each of the chassis test cycles was determined.

PART5 outputs separate emissions factors for five different subclasses within the heavy-duty diesel classification (buses, 2B, light-heavy duty, medium-heavy duty and heavy-heavy duty), but the emissions values are not affected by altitude or average speed. Particulate emission factors for diesel vehicles are then adjusted for sulfur concentration in the fuel, depending upon whether the year of interest is before or after 1993. In 1993, the EPA lowered the allowable sulfur content in diesel fuel. It was assumed that all chassis dynamometer tests were conducted using low sulfur fuel. The vast majority of the chassis emissions tests, even those on older vehicles, have been conducted using in-use fuel since 1993 when the fuel regulations for low sulfur fuel came into effect. Figures 3.6 and 3.7 show that both models underestimate emissions as measured on the chassis dynamometer. Note that emissions on both of these figures are shown on a log scale. Ideally, the predicted emissions would be the arithmetic average of the measured emissions. However, these plots indicate that the predicted emissions are generally, well below even the geometric average of the measured emissions. The figures also indicate that emissions from newer vehicles, which are predicted by the models to be lower (i.e., are on the left side of the graphs) appear to be less well predicted by the models than emissions from older vehicles.

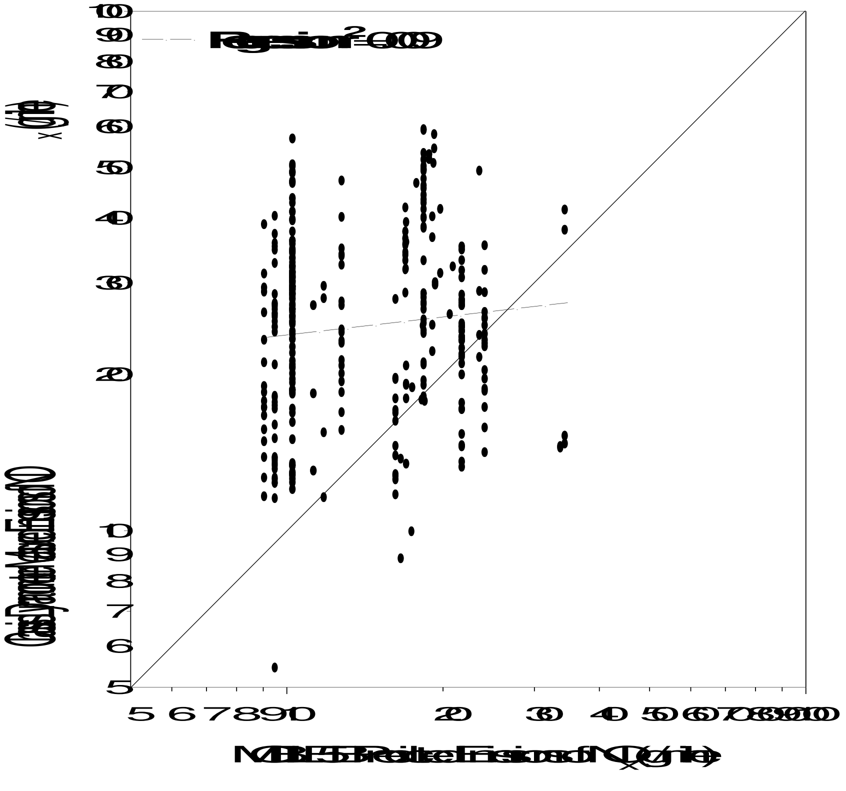


Figure 3.6. Comparison of MOBILE5B predicted emissions of NO_x (g/mile) and emissions measured on the chassis dynamometer.

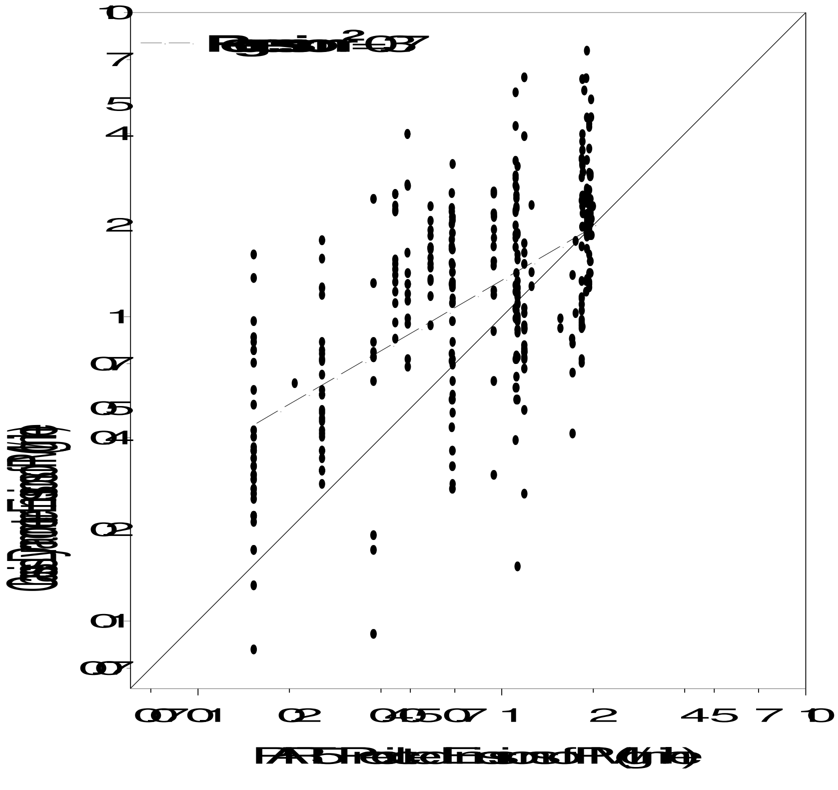


Figure 3.7. Comparison of PART5 predicted emissions of PM (g/mile) and emissions measured on the chassis dynamometer.

Much of the difference between NO_x emissions predicted by the models and the NO_x emissions measured on the chassis dynamometer may be attributable to the dual engine mapping of recent model year engines. Dual mapping of the electronic engine controls allowed for higher emissions from vehicles under operating conditions that were different than the certification testing conditions. This would explain why the recent model year vehicles were more poorly predicted than emissions from older vehicles. Since deterioration would be expected to decrease NO_x, this trend occurs despite deterioration. An effective inspection and maintenance program which limits increases in PM emissions would likely increase the disparity between the NO_x emissions predicted by the model and true NO_x emissions by even more.

The next chapter presents a new explanation as to why PM emissions may be consistently underpredicted by a model based on engine certification test results. Another possible explanation for the underestimation of particulate emissions is deterioration effects which are not included in the PART5 model.

Discussion of Research Needs

Several important observations regarding research needs stand out from this review of heavy-duty emissions studies. The fleet of vehicles tested to date in chassis studies is not representative of the in-use fleet. Future studies should begin to address this issue and in particular should include fewer transit buses. In addition, to be useful for air quality planning and policy purposes, future dynamometer studies must provide more detailed engine descriptions (model year, horsepower rating, and full engine family number at a minimum). Few researchers appear to be aware that manufacturers may certify several different engine models, at different emissions levels, each year under the same model name, e.g. certification results are reported for four different 1995 Navistar 210 HP DT466 engines.

Hundreds of different diesel vehicles would need to be tested every year to obtain a representative sample of the in-use fleet, it is unlikely that a truly representative sample will ever be tested by chassis dynamometer. Realistically, given the high cost of chassis dynamometer testing, remote sensing and tunnel tests must be pursued in parallel with chassis test to develop representative estimates for in-use emissions.

There is wide variability of emissions from in-use vehicles built under identical regulatory standards, and even among identical engines. For example, West Virginia University's laboratory reports (data from www.afdc.nrel.gov) PM emissions from two 1990 Cummins L10 engines in buses, tested under the CBD cycle. One emitted 0.10 g/gal, the other emitted 12.07 g/gallon of PM. Figure 3.3 shows that PM, THC and CO emissions typically vary an order of magnitude in the same model year, NO_x variability is less pronounced. However, in a correlation study performed by West Virginia University and the Colorado School of Mines, emissions from identical engines, in identical vehicles (i.e. identical transmission and chassis), are generally quite comparable (59,84). However, the results indicated that driver technique could have a large effect on CO and PM emissions, as much as a factor of 6 for CO and a factor of 5 for PM, for the same driving cycle. The cause of this variation in driver technique was traced to different driver aggressiveness resulting from differences in laboratory procedures and equipment. This suggests that engine certification testing alone may be missing critical factors that affect in-use emissions. It also suggests that test routes (as opposed to test cycles) which are driven with the throttle either full open or closed may produce results which are more repeatable when driven by more than one driver.

Another possible explanation for emissions variability is intake air and test cell temperature. Approximately half of the chassis data reviewed here was acquired with the University of West Virginia mobile chassis dynamometer. The vehicle testing equipment is set up outdoors and while a correction is made for ambient air humidity, the vehicles are tested at ambient temperature. Other testing reported here was obtained on indoor, fixed base test cells at temperatures in the 20-25°C range. The impact of operating temperature on diesel vehicle emissions has not been extensively investigated, but is not expected to be a major factor, because the engine air intake temperature is primarily determined by the turbocharger after start-up

More studies of emissions deterioration in-use are needed. Based on testing conducted on an engine stand, the engine manufacturers typically report that emissions deterioration for all pollutants is negligible. The frequency and severity of emissions deterioration should be measured for a random set of vehicles brought into the chassis dynamometer laboratory at set intervals. If the cause and extent of emissions deterioration is known, it may be possible to improve air quality with relatively inexpensive inspection and maintenance procedures. Additionally, deterioration is not included in the PART5 inventory model, and may not be accurately accounted for in MOBILE5b. An understanding of deterioration is needed for accurate inventories.

Although remote sensing of diesel emissions shows great promise for testing large numbers of vehicles, most studies are limited to a single location in which all of the tested vehicles are operated at similar speeds and accelerations. Thus, the results cannot be used to determine average emissions from typical vehicle operation. Moreover, it is critical that remote sensing measurements be correlated with chassis dynamometer results prior to using remote sensing for policy purposes. Unfortunately, there is no field-tested method for remote sensing of particulate matter emissions, although a method has been proposed in the literature (109). In addition to pursuing methods for remote sensing of PM emissions from diesel vehicles, it is recommended that the relationship between CO, as measured by remote sensing, and PM measurements be considered as an alternative. Future remote sensing studies could be improved by addressing the uncertainty in the NO/NO_2 ratio and by employing methods for hydrocarbon detection that are responsive to aromatic and olefinic hydrocarbons as well as alkanes.

Tunnel tests will continue to be useful for providing fleet average emissions data, but cannot provide insight into the degree and cause of variability among heavy-duty diesel vehicles. Moreover, they are limited to specific locations and traffic conditions, which may not be representative of all types of driving and types of vehicles. Data analysis methods used to extract diesel emissions from tunnel data can, in some cases, lead to a large experimental uncertainty. New approaches are needed to reduce this error. More quantitative agreement might be obtained if tunnel and remote sensing studies were to report and correct for ambient conditions such as humidity, barometric pressure, and temperature.

While the impact of humidity on diesel emissions seems well understood, and altitude has been shown to have a significant effect on PM, a complete understanding of the impact of ambient conditions on diesel emissions is lacking. In particular, there appear to be no data available on the impact of cold temperatures on diesel emissions.

Summary

Chassis dynamometer studies indicate a significant decline in emissions of PM, CO, and THC over the past two decades, although the decline in PM emissions is less than expected. Emissions of NO_x do not appear to have changed significantly over this time. Surprisingly little difference in average emissions was observed between vehicles of different sizes. There is qualitative agreement between chassis, tunnel, and remote sensing studies on the average emission levels from diesel vehicles for HC, CO, and PM. There is good agreement between the different types of studies for emissions of NO_x. Remote sensing studies indicate that emissions of NO are normally distributed, while emissions of CO and HC are skewed to a few high emitting diesel vehicles.

CHAPTER 4

ON THE PREDICTION OF IN-USE EMISSIONS OF HEAVY-DUTY DIESEL VEHICLES FROM ENGINE TESTING

Since 1985 engine manufacturers have been required to conduct a certification emissions test, known as the federal test procedure (FTP), on a prototype of each heavy-duty engine model prior to use of the engine in an on-road vehicle. These emissions measurements and engine sales data have been used to generate the heavy-duty vehicle emissions factors for several generations of EPA's MOBILE and PART air emissions models. However, as shown in the previous chapter measured emissions trends from in-use heavy-duty vehicles differ considerably from those predicted from the results of the certification engine tests. Estimates of vehicle emissions are necessary to determine the effectiveness of heavy-duty engine regulations, for pollutant inventories needed to understand air pollution problems, and for air quality planning purposes. A clearer understanding of in-use emissions has been hampered by inconsistencies in testing conditions, most significantly inertial weight and driving cycle, and differences between the drive trains in vehicles using identical engines. These three factors, which are also widely variable under real life conditions, cause significant differences in emissions.

Others have approached the problem of predicting in-use emissions in different ways. Ramamurthy et al. (112) found that it was possible to develop reasonably good (non-linear) correlations between instantaneous power at the wheels and NO_x emissions, while ignoring the variability of driveline efficiency. Since driveline efficiency was not known, it was not possible to compare their results directly to engine test results, although their approach could be used to make rough estimates as to the expected changes in NOx emissions due to changes in inertial weight and driving cycle. They found that HP at the wheels could not be correlated with CO.

McKain and others (113,114) used a similar design for a manual transmission vehicle which would mimic the engine FTP by forcing the vehicle to remain in a single gear while controlling the engine speed and load through the throttle pedal and power absorbers at the wheels. Although they had some difficulty in matching FTP engine speed and load simultaneously, it was possible to linearly correlate NO_x in g/BHP-h from the engine with NO_x in g/HP at the wheel from the various chassis tests. Assuming that NO_x is proportional to engine HP, these results from two different chassis over several different gears show that the two tested vehicles had the same driveline efficiency, and that the driveline efficiency was relatively constant for the various gears tested. If driveline efficiency is constant for all vehicles with a manual transmission then this approach may be applicable to other vehicles. However, the authors found that the emissions of CO, HC and PM were not linearly correlated between the engine and chassis tests, suggesting that their single gear approach was not adequately matching transient operation between the engine and chassis testing.

Outlined here is a methodology for relating emissions from engine testing to various chassis driving cycles and inertial loads, using a computer model of a driveline. This model is known as the Colorado School of Mines Transmission Model (CSMTM). The CSMTM was developed to provide a closer link between engine speed and load and those parameters that could be measured during a chassis test, and to better understand the effect of automatic transmissions on engine operation. Automatic transmissions are installed in approximately 55% of all new heavy-duty vehicles (115).

The test vehicles were two buses and a snowplow truck. The testing methods and results have previously been reported by McCormick et al. (116), and are only briefly summarized here. The two buses were driven over two different driving cycles, the Heavy-Duty Truck (HDT) cycle and the Central Business District (CBD) cycle. The snowplow truck was tested using only the HDT cycle but over a range of simulated inertial weights. The engines were then removed from the vehicles and tested on the engine dynamometer, using the engine certification transient test cycle.

Ryan (117) used the results of reference 8 along with the Allison Transmission's proprietary transmission model, SCAAN, to estimate the equivalent vehicle mass for a small portion of the engine transient test. Their analysis was limited by the capabilities of the SCAAN model to a short section of the engine certification test. The SCAAN model was intended for open throttle conditions only, and these conditions occur only during short sections of the engine cycle. Moreover, they were required to select an acceleration section that began from idle, so that they could correctly determine the starting speed for the acceleration. For the short acceleration section they were able to model, they found that only if the engine and transmission were installed in an extremely light truck (on the order of 12,000 lbs) could the rate of increase in engine rotational speed be maintained under the low loads specified by the engine certification test. The work reported here was intended to build upon this previous investigation and determine whether the disparity between the engine test and in-use driving was representative of the entire engine test, and to understand the emissions implications.

Methods

Description of Vehicles and Engines. Properties of the three vehicles and engines tested are listed in Table 1. The first vehicle was a plow/dump truck owned by the Colorado Department of Transportation (CDOT), with a Navistar DTA-466 engine with mechanical control. The transmission was an Allison MT-643, and the torque convertor was an Allison TC-378. This vehicle and engine will be referred to as Mechanical #1. The other two vehicles were identical Neoplan transit buses operated by the Denver Regional Transportation District (RTD). The engines for these vehicles were electronic 1993 DDC Series 50 diesel engines, and the transmissions and torque convertors were Allison HTB-748s and Allison TC-495s, respectively. The second set of

Table 4.1 Vehicle and engine descriptions

Vehicle Make	Engine Model	Model Year	Engine HP	Fuel Injection	GVWR	Odometer Mileage
Navistar	Navistar DTA-466 SN 862790 Family DTA-466-E250	1993	250	Mechanical (#1)	36,220	37,009
Neoplan	DDC Series 50 SN 04R0001731 Model 6047GB28DD2 Family PDD08.SFZK7	1993	275	Electronic (#1)	38,000	85,200 since rebuild
Neoplan	DDC Series 50 SN 04R0002933 Model 6047GK28DD3 Family PDD08.SFZK7	1993	275	Electronic (#2)	38,000	65,234

vehicles and engines are referred to as Electronic #1 and Electronic #2. All engines were certified at 1993 emissions levels and do not employ a diesel oxidation catalyst.

Fuel Analysis. The testing reported here was carried out using a fuel representative of wintertime fuel in the Denver region (NFRAQS fuel). The fuel characteristics are described elsewhere (116).

Engine Testing. Emissions testing for the engines was performed by the heavy-duty transient test as outlined in the Code of Federal Regulations (1).

Chassis Testing. Chassis dynamometer testing methods are reported elsewhere (116). An important point is that engine starting is included in the test. Of interest also is that wind and rolling friction losses are estimated from published studies (25) and simulated electronically, and thus can be modeled exactly. For all three vehicles, inertial weight was set at approximately the average of the Rated Gross Vehicle Weight (GVWR) and the curb weight. For the Electronic Vehicles (transit buses), this was 90% of GVWR while for the Mechanical Vehicle (snowplow truck), it was 70% of GVWR. For the Mechanical Vehicle, inertial settings of 97% and 47% of GVWR were also examined. The vehicle speed is managed by the vehicle driver. The cycle is displayed for the driver using a prompt that shows the driver the current speed and the cycle speed required 30 seconds into the future. A single driver was used for all chassis testing performed under this program.

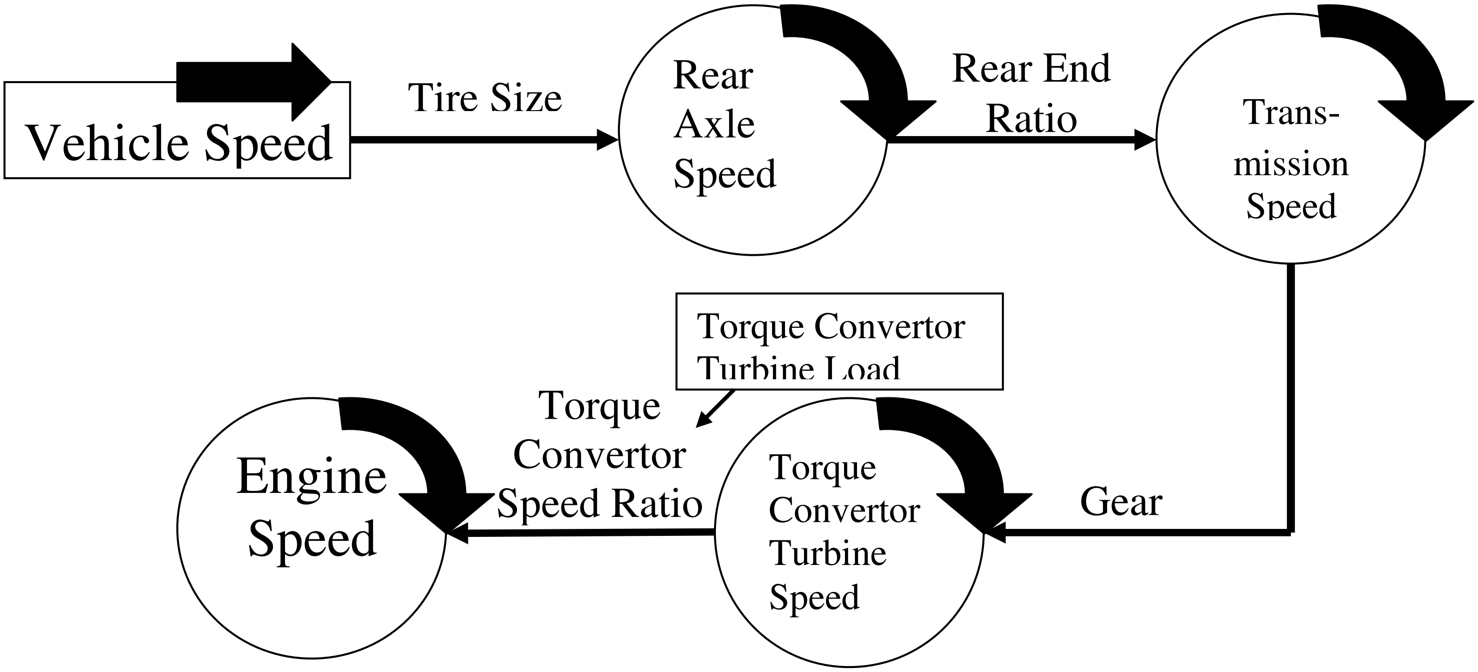
In this study two chassis test cycles were used. The Urban Dynamometer Driving Schedule for heavy-duty vehicles or HDT cycle and the Central Business District (CBD) cycle.

CSM Transmission Model. Rotational speeds at various points in the driveline are calculated using the algorithm shown schematically in Figure 4.1. Engine horsepower is calculated as the horsepower needed at the wheel, plus horsepower losses through the driveline, and horsepower required to operate various auxiliary systems including the fan, alternator and air conditioner. Horsepower at the wheel is a sum of the power needed to accelerate the vehicle and the power needed to overcome road friction and air drag. The driveline losses are a function of rotational speed and load at various points in the driveline and can be obtained as a series of empirical relationships specific to the components in each vehicle. The HP used by the auxiliary systems is dependent on engine rotational speed. These calculations are presented in greater detail in the Appendix.

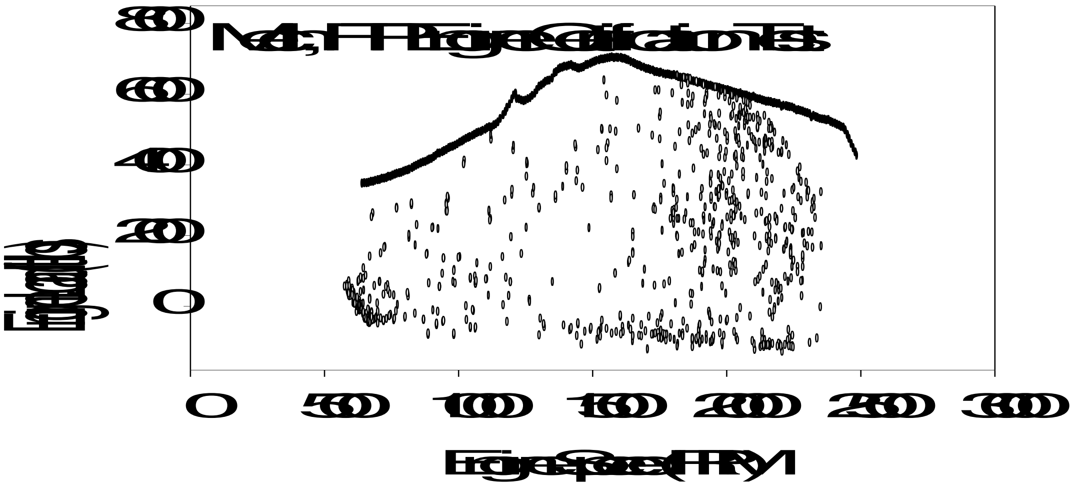
Results and Discussion

Engine Speeds and Loads. The CSMTM estimates engine speeds and loads from a series of second-by-second vehicle speeds. For example, Figure 4.2b shows engine speed and loads for the Mechanical Vehicle driving the HDT at an inertial weight

Figure 4.1. Algorithm for calculating engine speed from vehicle speed.



a)


Mode 4, FTP Engine Calibration Test
800
600
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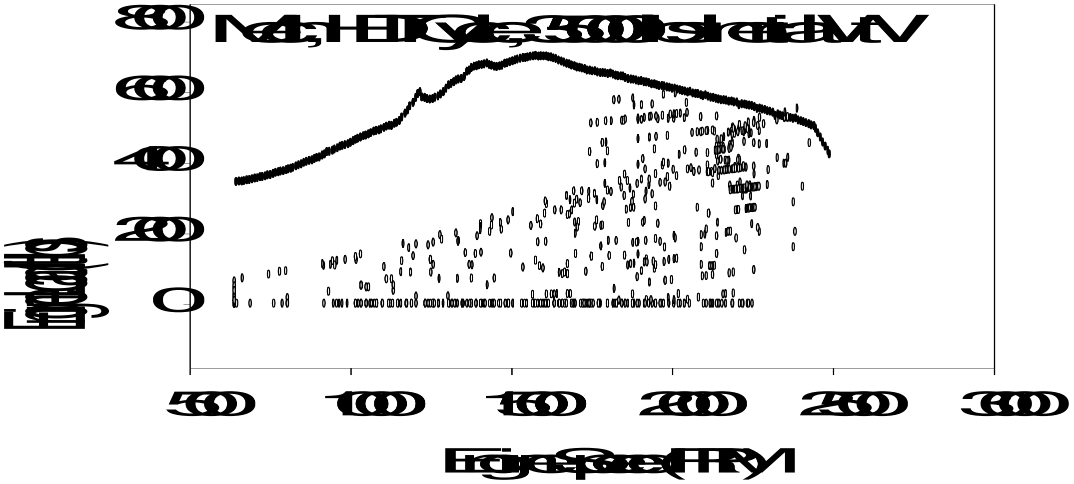
0
0
500
1000
1500
2000
2500
3000
Engine Speed (RPM)

b)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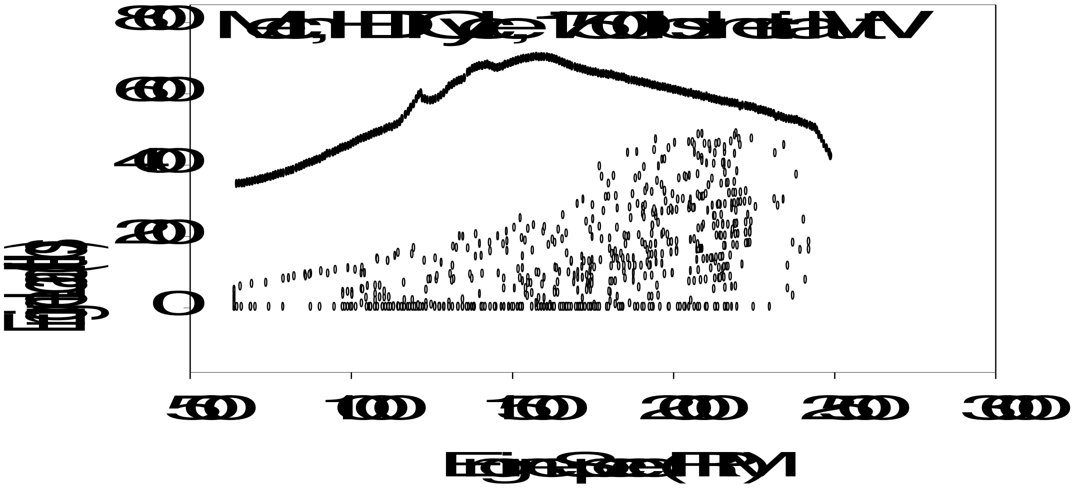
Engine Speed (RPM)

c)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Engine Speed (RPM)

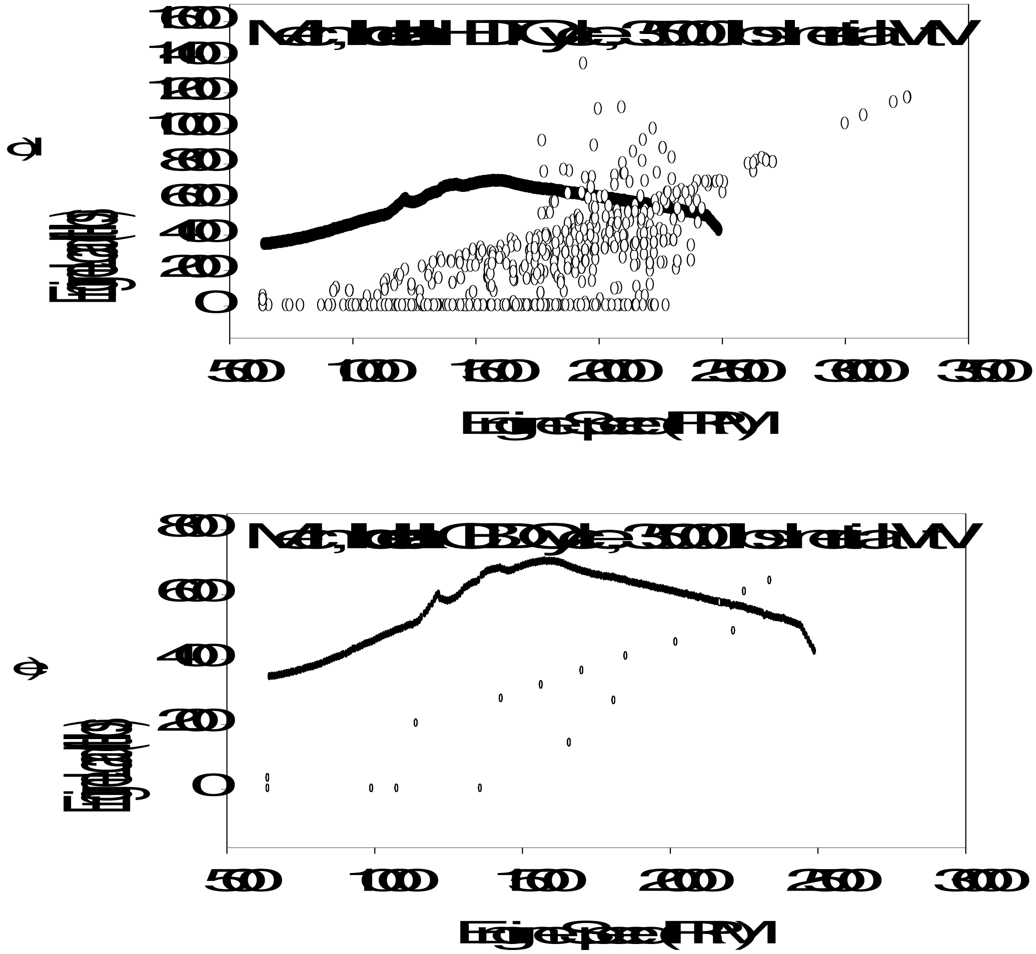


Figure 4.2. Speeds and loads for the Electronic Engine #2/Vehicle #2 for several test cycles. The black line is the engine map.

of 35,000 lbs. There are several points that lie outside the engine “map”, i.e. the projected load exceeds the maximum load that the engine is capable of at the given rotational speed. The engine map is generated on the engine dynamometer for the fully warmed engine at essentially steady state conditions. The throttle is set at full open, and the engine speed is gradually increased, allowing the engine to exert the maximum load of which it is capable at each rotational speed point. It should not be possible for the engine to operate at points outside of the engine map. These outliers could be due to transient operation, an engine warmed differently than for mapping, or due to inaccuracies in the measurement of vehicle speed, which is only measured to the nearest one-tenth of a mile per hour (mph). A difference of one-tenth of a mph at certain points would bring most of these data back within the engine map. As can be seen in Figure 4.2c, at lower inertial weights, the speeds and loads on the engine are well within the engine map.

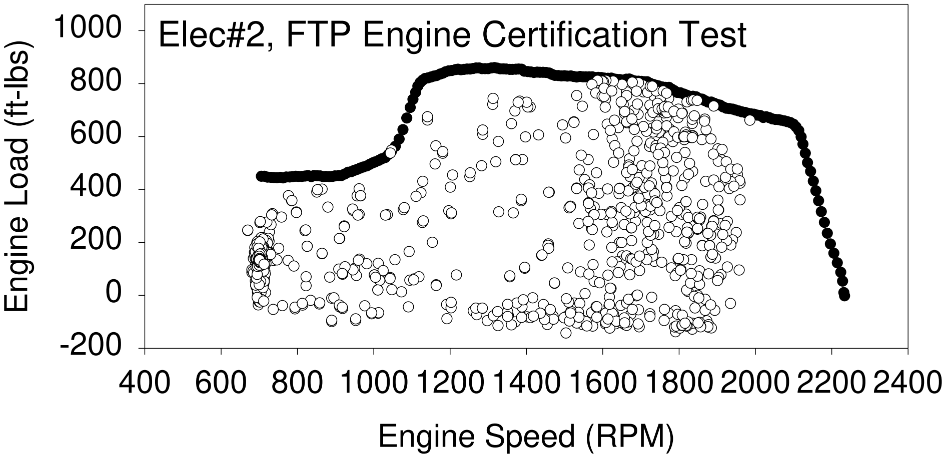
For comparison with the chassis cycle, the engine test for the Mechanical Engine is also shown in Figure 4.2a. The most obvious difference between the engine and vehicle tests is in the low RPM range. In the low engine RPM range, the torque convertor slips to prevent high torque operation during the chassis test. High torque operation at low RPMs is included in the engine test. Many of the speed/load points in the engine test occur in the zone prevented by the torque convertor during chassis testing. Therefore it is not possible to develop a vehicle speed cycle which would mimic the engine operation of the engine FTP for these vehicles, and possibly for any vehicle with an automatic transmission.

If it were possible to match the vehicle speeds required in an HDT cycle perfectly with 35,000 lbs inertial weight, then the engine speeds and loads would be those shown in Figure 4.2d. As that figure makes apparent, the ideal HDT cycle would require engine loads that are above the capacity of this engine. It might be possible to drive these cycles at lower inertial weights with the Mechanical Vehicle. The ideal CBD cycle is shown in Figure 4.2e. Since the CBD cycle is composed of fourteen identical accelerations, cruise periods and decelerations it tests only a small part of the total engine range of operation

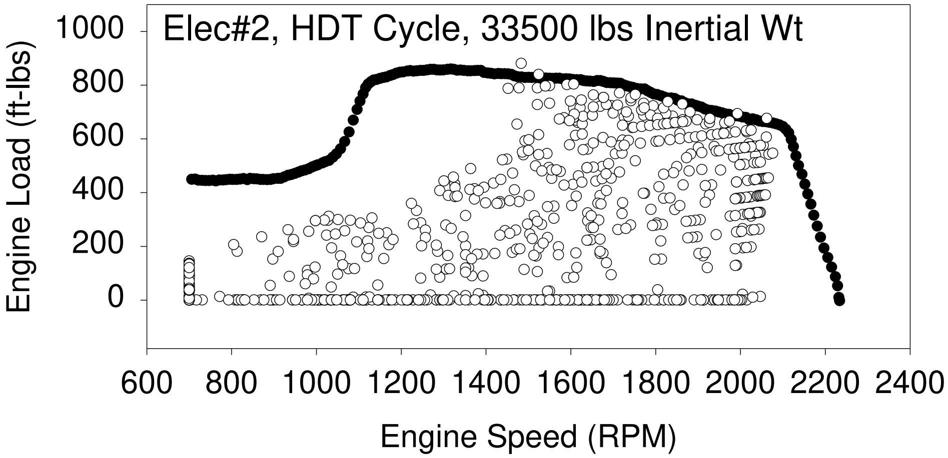
Similar results were found for the Electronic Vehicles as shown for Electronic Engine #2 and Electronic Vehicle #2 in Figure 4.3.

Small changes in vehicle speeds during accelerations can result in large differences in the load on the engine. For example, Seconds 72 - 100 of the HDT test cycle are shown in Figure 4.4. Figure 4.4a shows how the Mechanical Vehicle was actually driven in comparison to the ideal vehicle speeds set in the HDT cycle. Figure 4.4b shows ideal and actual engine load and HP calculated from the CSMTM. At Second 90 vehicle actual speed is only 2 mph below the ideal cycle speed. However, this small speed difference translates into a lowering of required HP and torque by nearly 50%.

NO_x Emissions vs. Horsepower. Figure 4.5 compares NO_x vs Engine HP for all three engines on both engine and chassis tests. The results of both tests show that for all three engines NO_x emissions are proportional to HP on a second by second basis. For the a)


Elec#2, FTP Engine Certification Test
Engine Load (ft-lbs)
1000
800
600
400
200
0
-200
400
600
800
1000
1200
1400
1600
1800
2000
2200
2400
Engine Speed (RPM)

b)


Elec#2, HDT Cycle, 33500 lbs Inertial Wt
Engine Load (ft-lbs)
1000
800
600
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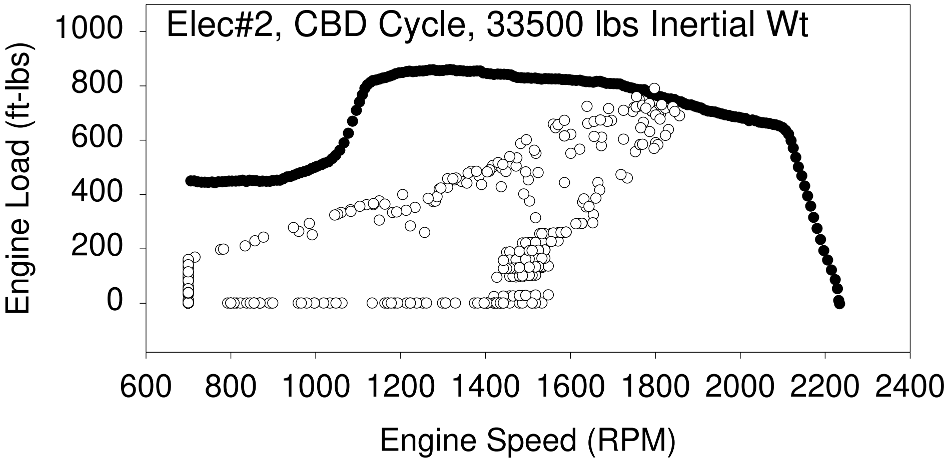
0
600
800
1000
1200
1400
1600
1800
2000
2200
2400
Engine Speed (RPM)

c)


Elec#2, CBD Cycle, 33500 lbs Inertial Wt
Engine Load (ft-lbs)
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1000
1200
1400
1600
1800
2000
2200
2400
Engine Speed (RPM)

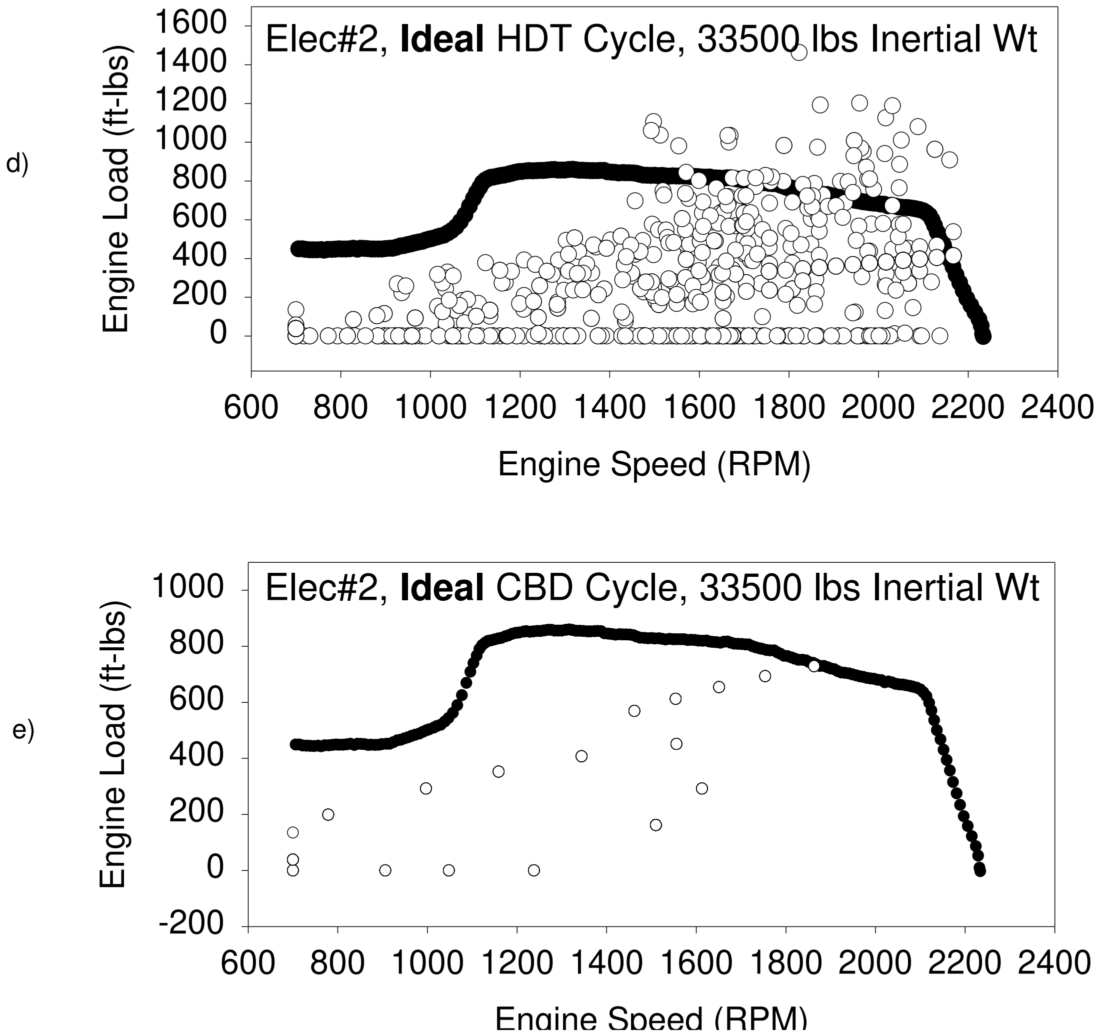

Figure 4.3. Engine speeds and loads for the Electronic Engine #2/Vehicle #2 for several test cycles. The black line is the engine map.

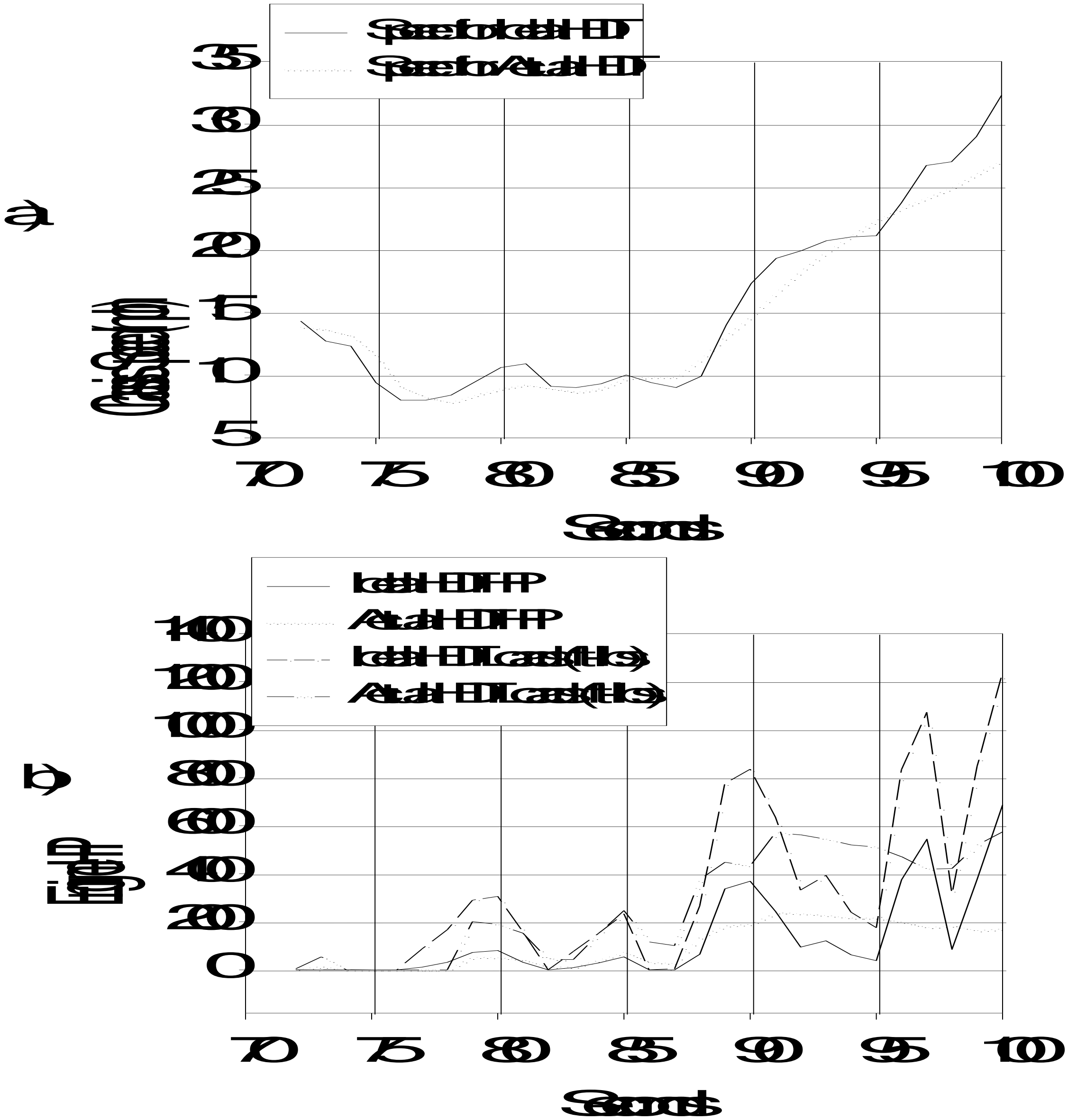

Figure 4.4. The effect of small changes in speed (top) on the engine load and HP (bottom) for the Mechanical Vehicle at an inertial weight of 35000 lbs (97% GVWR), HDT cycle

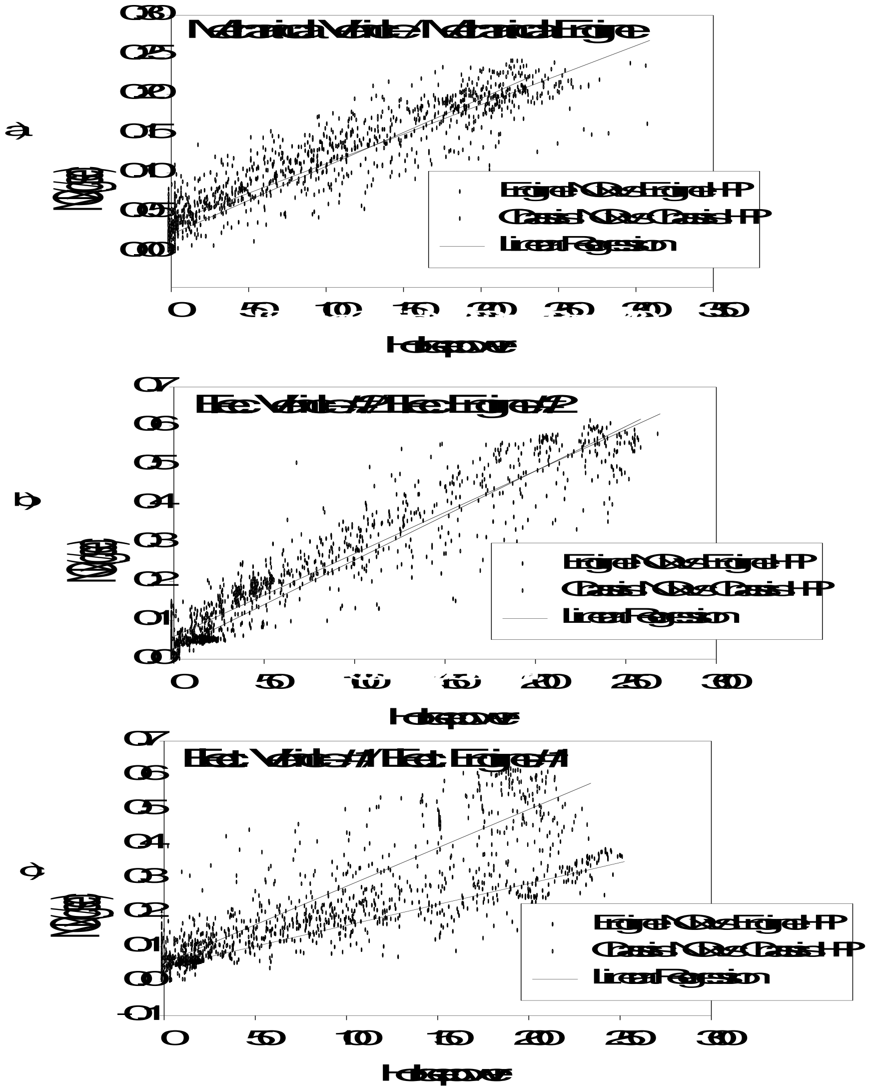

Figure 4.5. NO_x vs engine HP for a) the Mechanical Vehicle, HDT Chassis Cycle, 35,000 lbs inertial weight and for the Mechanical Engine run on the engine certification test. Chassis: NO_x (g/sec) = 7.48E-4 * HP + 0.037 (r^2 = 0.86), Engine: NO_x (g/sec) = 8.62E-4 * HP + 0.022 (r^2 = 0.93). b) Electronic Vehicle #2, CBD Chassis Cycle, 33,500 lbs inertial weight and for (cont'd) the Electronic Engine #2 run on the engine certification test. Chassis: NO_x (g/sec) = 2.13E-3 * HP + 0.059 (r^2 = 0.88), Engine: NO_x (g/sec) = 2.29E-3 * HP + 0.026 (r^2 = 0.93). c) Electronic Vehicle #1, HDT Chassis Cycle, 33,500 lbs inertial weight and for the Electronic Engine #1 run on the engine certification test. Chassis: NO_x (g/sec) = 2.24E-3 * HP + 0.0123 (r^2 = 0.86), Engine: NO_x (g/sec) = 1.22E-3 * HP + 0.041 (r^2 = 0.85).

Vehicle #2 suggests that the model accurately predicts engine HP from vehicle speed. However, for Electronic Engine #1 the chassis and engine test NO_x vs. HP lines are of different slope. The engines and other driveline components in Electronic Vehicles #1 and #2 have identical model names and numbers. The difference between the engine and chassis NO_x vs HP regressions cannot be attributed to errors in the model, since there are numerous NO_x g/sec emission points from the chassis test which are significantly higher than the highest NO_x emissions that occur during the engine test. It is concluded that this disparity in engine and vehicle emissions is due to dual mapping, i.e. the practice of having different fuel injection strategies for in-use operation and engine testing conditions. Use of this type of engine computer program has been determined by EPA to be an emissions control defeat device. It should be noted that negative HP measurements (where the engine is absorbing power) in both the engine test and the calculated chassis cycles were set to 0. It would be expected that during these periods there is no combustion in the engine, and thus no NO_x would be generated.

The results of Clark and coworkers (118), who graphically reported second-by-second NO_x emissions and engine HP for the engine certification test, seem to indicate a similar relationship. Ramamurthy and coworkers (112) compared NOx emissions from several chassis cycles to HP at the wheels, and found that NO_x was correlated with HP at the wheel, although the relationship was not linear. This is not in conflict with the findings of this study since their analysis did not consider losses in the driveline. They found that the slope of their correlation line was in all cases less than 1.

Other Pollutants. Based on the NO_x vs HP discussion above, it is reasonable to conclude that chassis testing in conjunction with a transmission model can be used to determine if in-use NO_x emissions differ significantly from engine certification test emissions (whether through dual mapping or deterioration). It is also apparent that, in the absence of dual mapping and deterioration, the engine test accurately predicts in-use emissions of NO_x on a g/BHP-h basis for these vehicles. It is then of interest to determine how accurately the engine test can predict in-use emissions of other pollutants on a g/BHP-h basis, i.e., how representative the engine test is of in-use operating conditions for other pollutants.

A comparison of engine test emissions with brake specific chassis test emissions estimated using CSMTM is shown in Figure 4.6. For completeness, Electronic Engine #1 has been included in this graph although, as noted above, this engine may use different fuel injection strategies during engine and chassis testing, and those strategies may affect the emissions of pollutants other than NO_x. Figure 4.6 shows that chassis HC emissions are not well predicted by the engine test although there is no consistent bias. HC emissions from diesel vehicles are often very small in comparison to HC from spark ignition vehicles, and are difficult to measure accurately. Carbon monoxide emissions

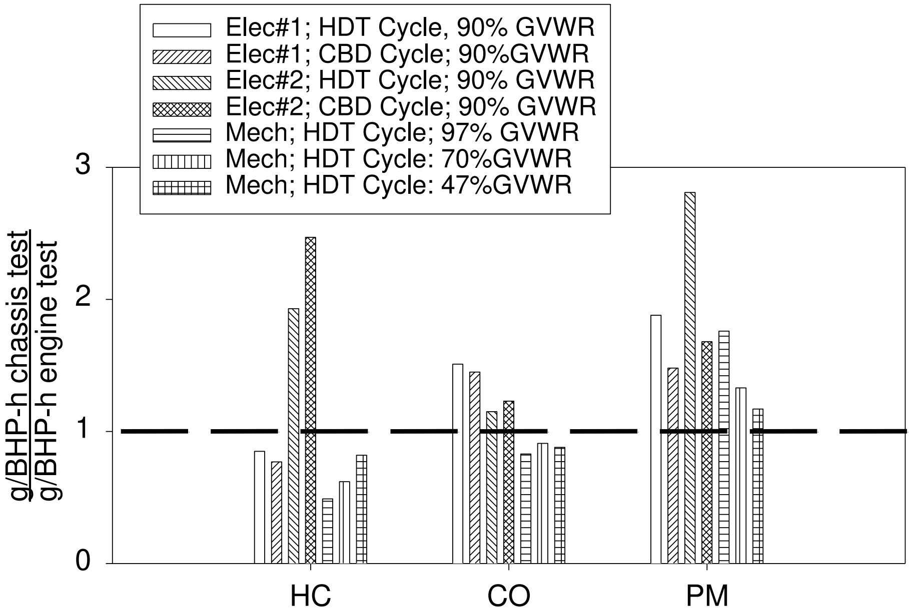

Figure 4.6. Comparison of emissions in g/BHP-h between chassis tests and engine tests.

(on a g/BHP-h basis) are reasonably consistent between the engine and chassis tests, and the engine certification test consistently underestimates particulate matter (PM) emissions on a g/BHP-h basis. PM emissions are also very sensitive to inertial weight.

One possible explanation of the PM bias is suggested by results collected by Hofeldt and Chen (119) who measured transient carbon (not total PM) emissions from diesel buses during the CBD cycle. They found that acceleration transients accounted for roughly 80% of the particulate mass emitted over the cycle but only 45% of the fuel consumption, although the peak carbon emissions were correlated with steep transients in fueling rates. Assuming fuel consumption is roughly proportional to cycle work (or HP-h), the acceleration transients are responsible for a disproportionate fraction of the brake specific PM emitted.

Hofeldt and Chen conclude that these increases in carbon emissions are due to the rate of increase of fuel injected into the engine. If their conclusion is true then one would expect that faster rates of increase in engine HP would generate more carbon (on a g/BHP-h basis), than slower rates of increase. Additionally, more time spent during a cycle increasing the HP (as opposed to steady-state or deceleration) would also be expected to increase PM emissions. It is proposed that the quantity $._{\text{cycle}}\,(dHP_{accel}/dt)\,dt$ is an appropriate parameter to determine test severity in terms of acceleration and to predict PM emissions. The integrand, dHP_{accel}/dt, is the rate of HP increase, and periods of HP decrease (where $dHP/dt < 0$) and stability ($dHP/dt = 0$) are ignored because they do not require fueling. As shown in Figure 4.7, there is a linear relationship between total cycle PM emissions in grams and test severity $(\,._{\text{cycle}}\,(dHP_{\text{accel}}/dt)\,dt\,)$ for both Electronic Engine/Vehicle #1 and Electronic Engine/Vehicle #2. The engine certification test is less severe than the HDT cycle but more severe than the CBD cycle at the tested inertial weight, and consequently yields total cycle PM emissions in between the two tests.

It is important not to confuse the fact that total PM emissions are shown in this case to be proportional to test severity, while PM on a g/BHP-h basis is dependent on both test severity and total work done during the test. Thus PM in g/BHP-h can be higher for test cycles with a lower Test Severity rating. The good linear correlation found between Test Severity and PM emissions for Electronic Vehicle #1, which is believed to employ different injection timing strategies for engine and chassis testing, supports the contention that fueling rates rather than injection timing is the controlling factor in PM emissions under the tested conditions. For the mechanical engine, the emissions data are more scattered. This may reflect a more variable response of the mechanically controlled engine to the exact details of driving. As with the two electronic engines, the PM emission appears to be a function of the cycle severity.

In order to support the thesis that PM emissions are dependent on the rate of HP increase, continuous particulate matter emissions data collected by others during an engine FTP using a tapered element oscillating microbalance (TEOM) (120) were examined. This test was conducted on a Cummins M11 370 HP engine using typical in-use diesel fuel. The TEOM measures the weight of the filter collecting PM from the

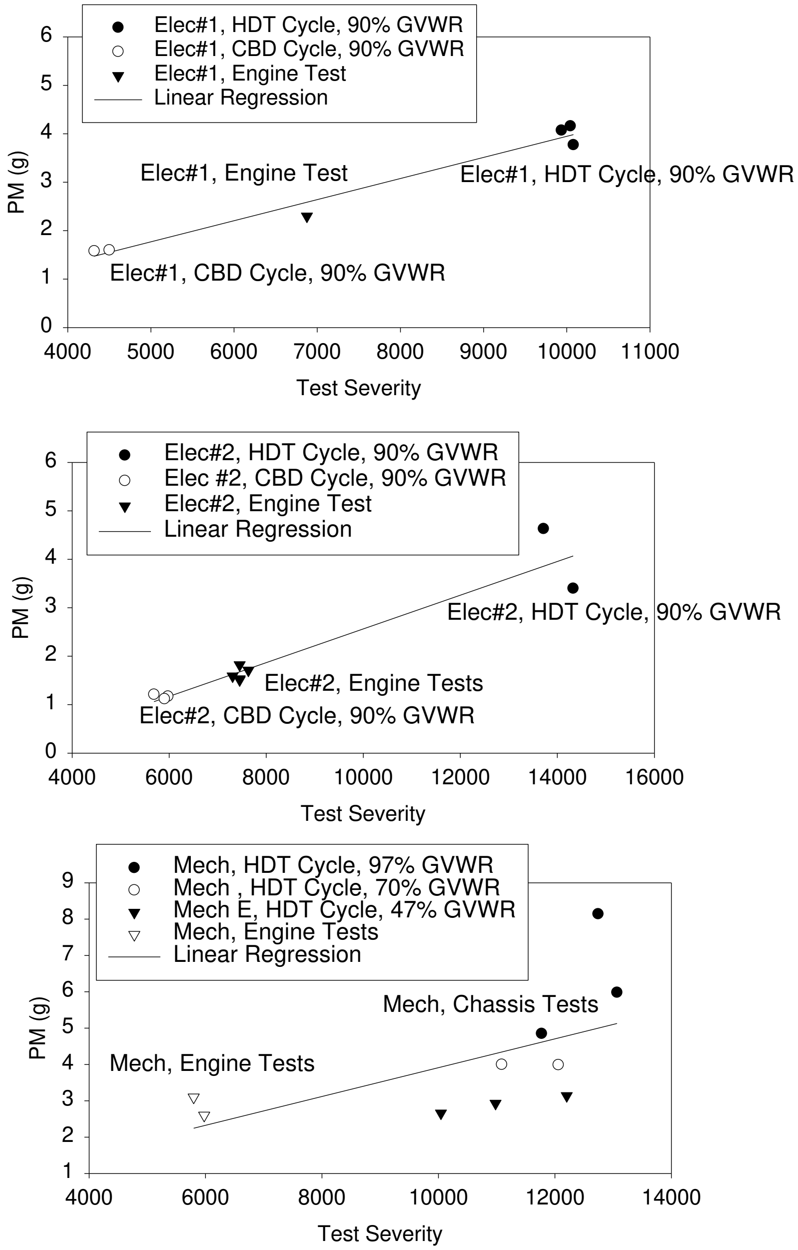

Figure 4.7. Relationship between severity of test ($\int_{cycle}$ (dHP$_{accel}$/dt)dt) and total PM emissions. Electronic Vehicle and Engine #1: PM (g) = (4.36E-4 * (test severity))-0.41; r^2=0.97. Electronic Vehicle and Engine #2: PM (g) = (3.48E-4 * (test severity))-0.92; r^2=0.91. Mechanical Vehicle and Engine: PM (g) = (3.96E-4 * (test severity)) - .052; r^2=0.35.

engine exhaust multiple times throughout the emissions test. In this experiment measurements of PM were made at a rate of 5 Hz. There is considerable oscillation in the weight measured, perhaps due to the intermittent condensation and evaporation of water (121) so it was necessary to smooth the data over ten second intervals. The smoothed data showed a good correlation ($r^2=0.77$) between dHP_{accel}/dt and PM, with an approximately zero intercept as shown in Figure 4.8. Interestingly, the slope of this line as well as all of the regression lines for the other engines (Figure 4.7) is very similar (PM $\approx$ 4E-4 dHP/sec + Constant), suggesting that this slope may be common to all modern engines.

Summary

The Colorado School of Mines Transmission Model represents the driveline of a heavy-duty truck in order to estimate engine speeds and loads from vehicle speed data. The CSMTM has been used to calculate engine HP from vehicle speed measurements. The model shows that vehicles exercise the engine differently than in the engine certification test. Engine testing results have shown that NO_x is proportional to engine HP for the vehicles tested here. The similarity between engine and chassis NO_x vs HP plots in the case of two of the vehicles verified the accuracy of the model and indicated in the third case that one of the vehicles appeared to be using different fuel injection strategies during engine testing and chassis testing. The engine certification test was shown to provide an approximate prediction of CO emissions from the HDT and CBD

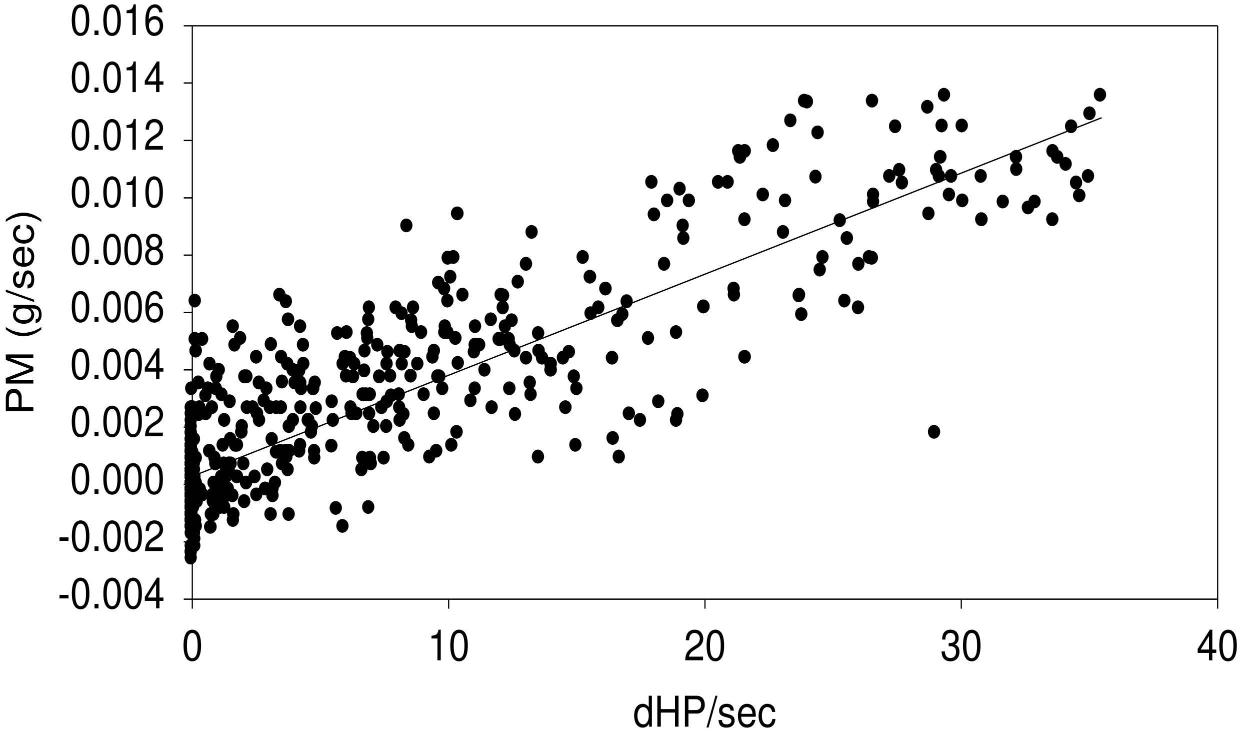

Figure 4.8. Continuous PM emissions versus rate of HP increase for Cummins ISM running the engine FTP (120). Data was smoothed over 10 second averages. Equation of regression line is PM (g/sec) = 3.52E-4*dHP/sec +0.00029; r^2=0.77.

chassis cycles on a g/BHP-h basis. However, the engine certification test consistently underestimates chassis test particulate matter (PM) emissions on a g/BHP-h basis. Chassis hydrocarbon (HC) emissions are also not well predicted by the engine test on a g/BHP-h basis, although there is no consistent bias.

Chassis and engine cycle total PM emissions were found to correlate with a test severity parameter defined as the integral of the rate of HP increase over the test cycle. It was also shown that real-time PM emissions (via TEOM) are linearly correlated with the rate of HP increase. The CSMTM was also used to demonstrate that small changes in vehicle speeds during accelerations (+/- 2 mph for a few seconds) can lead to large changes in load on the engine and possibly large changes in emissions. This suggests that heavy-duty chassis testing laboratories need to develop approaches to carefully and repeatably control vehicle speed during accelerations.

CHAPTER 5

SUMMARY

Major Results

Emissions measurements of PM, NO_x, CO and THC as well as smoke opacity were made on 21 heavy-duty diesel vehicles. Analysis of the results showed that when emissions were converted to a g/gal basis, the effect of driving cycle was eliminated for NO_x and reduced for PM. Multivariate regression analysis was used to show that for the vehicles tested in-use NO_x emissions have not decreased at all since 1988, while engine certification standards have dropped sharply during that time period.

A review of all in-use emissions data in the scientific literature supported these results. Many of the published studies showed methodological problems, and improvements for future studies were proposed. The review also showed that PM emissions were widely variable among vehicles certified under identical standards. The variability was attributed environmental factors, inertial weight, test cycle, driver variability and vehicle condition, but the relative importance of these factors could not be determined. Overall, the review found that there were serious problems associated with using engine certification to predict and control emissions from vehicles.

In order to better understand the relationship between emissions measured during engine tests and in-use emissions, a computer model was developed that estimates engine speed and load from vehicle speed. The model has been validated and has shown that for the three vehicles considered NO_x is proportional to engine HP. Since HP is known to be approximately proportional to fuel usage this result is supported by the earlier result that NO_x/gal is relatively constant regardless of driving cycle. The model showed that the engine test exercised the engine in a different manner than was possible when the engine was placed in the vehicle. Using this model it was possible to detect the use of electronic controls which operate the engine in a different mode during engine testing and other more typical types of operation by changing the proportionality constant between NO_x and HP (this dual engine mapping was declared an illegal emissions control defeat devices by the EPA in 1998) . The engine certification test generated consistently less PM than the vehicle tests. A good linear correlation was found between rates of HP increase integrated over the test cycle and PM emissions for both the chassis and engine tests. This correlation would explain the low PM generated during the engine test in comparison to chassis test cycles and leads one to conclude that the engine certification test is a poor predictor of in-use emissions. The model also showed how small changes in vehicle speeds (+/-2 mph) due to driver variability can lead to significant changes in engine load and HP.

Conclusions and Recommendations for Further Research

This research shows there are significant problems associated with the engine certification test as the primary method for regulating and predicting emissions from diesel vehicles. Since it tests emissions from the engine rather than a vehicle it allowed the presence of dual engine maps (which produced higher NO_x emissions during vehicle operation) to go undetected for years. The emissions test makes no allowances for variations in vehicle weight, variations in drive trains, and driver variability and thus is a poor predictor of in-use emissions. Additionally, the engine test is biased toward lower PM emissions than the typical driving test cycles. Finally, there are important differences between the engine behavior tested during the engine test and in-use engine behavior, a discrepancy that engine manufacturers may be able to exploit with sophisticated electronic engine controls. Thus, this research points to several directions for regulatory action and research: more engine testing to confirm the engine emissions behavior shown in Chapter 4 and to better understand whether the most recent model year engines are exploiting differences between engine certification testing and in-use operation to emit more pollutants, improvements to the vehicle emissions prediction models, and a more effective approach to regulating emissions.

Relationship Between Engine Emissions, Vehicle Emissions and Engine Operating Parameters. In Chapter 4, the relationship between the HP generated by the engine and emissions of NOx and PM were demonstrated for several engines. However, a much wider range of engines, (particularly newer engines) would need to be tested to determine if these relationships are universal and if not, how they vary between different engines inside and outside of the chassis. Moreover, it would be useful to develop a similar understanding of the relationship between engine operating parameters and the other regulated pollutants, CO and THC.

Ideally the engine and vehicle tests should be conducted while measuring several key operating parameters some of which could potentially be varied electronically during vehicle operation and thereby change emissions behavior. Such parameters would include injection timing, rate, shape and pressure, turbocharger boost and the operating characteristics of emissions aftertreatment devices. This type of testing is necessary to determine if engine makers are modulating these parameters in order to decrease emissions during the engine certification test and then optimize other vehicle operating characteristics (at the expense of worsened emissions) during in-use operation.

Emissions Prediction Models. Certain straightforward improvements can be made to the emissions prediction models immediately based on the results of Chapter 3, specifically, improved correction factors for altitude and inspection and maintenance programs, as well as the addition of a humidity correction factor. More difficult is how to address the disparity between engine test results and chassis emissions. The simplest approach would be to use the in-use chassis emissions data in place of the engine test results. It would be necessary to overlook the lack of representativeness of the in-use testing, or arrange to have hundreds of additional in-use vehicles tested to obtain a reasonably representative sample. Additionally, these result alone would not be useful in predicting the effects of changes in driving patterns, for example increased congestion, for which these models are often used. Eventually, the EPA models need significant structural changes to show how NO_x and PM emissions are dependent on HP and dHP/dt, respectively. The current model allows the input of average speed, but in order to make an approximation of HP and dHP/dt, an average rate of acceleration and an estimate of the proportion of time spent accelerating would also be needed. Engine horsepower at different vehicles speeds and acceleration rates and for vehicles with differing inertial weights could then be approximated by using the equation for estimating HP requirements at the wheel (Equation A-1) and an estimated average driveline efficiency. Preliminary analysis suggests that the driveline efficiency would probably be about 80% for standard transmission vehicles, with automatic transmission vehicles modeled to have a similar efficiency at high speeds and a declining efficiency at lower speeds, when the torque convertor is engaged.

Regulation of Emissions. *Inspection and Maintenance Programs.* The second issue which needs to be addressed is how the emissions from heavy-duty diesels can best be regulated. The relationship of opacity testing to mass emissions and the cost and efficacy of repairs based on opacity testing has been considered by others (9). Although based on decade old data, there is a study which estimates the frequency of vehicle malfunctions which affect emissions (107). Combined, these two studies could be used to conduct a cost-benefit analysis of various possible inspection and maintenance programs. The cost-benefit analysis should consider various cutpoints (tests levels at which repairs would be required), testing methods (opacity testing, CO and CO and HC as surrogates for PM, as well as PM analysis), and inspection schedules (annual, biannual, etc.).

Regulation of New Engines and Vehicles. Truck and bus engines are commonly manufactured by one company and then sold to a second company which assembles the engine along with other components into a truck or bus or other heavy-duty vehicle. Because of this industry structure, the federal government decided in the 1980's to base emissions regulations on engine testing rather than vehicle testing. It was deemed too onerous to require engine makers to ensure that emissions standards be met in every possible engine-vehicle combination, and the engine makers could not control into what vehicles their engine would be placed. At the same time, the companies that built the vehicles which incorporated the engines were seen as too distant from the engine design process to be able to control emissions behavior and thus could not be made responsible for emissions regulations.

However, since that time, engine makers and transmission companies have worked together to develop computer models of drivelines similar to the CSMTM, for the purpose of guaranteeing vehicle acceleration rates and fuel economy. (Navistar-International's TCAPE model and Allison Transmission's SCAAN model are discussed and contrasted with the CSMTM briefly in Chapter 4). These models are used to select the best engine for a vehicle, to ensure the engine is adequately powered for the vehicle based on the weight and expected use, and to compare engines from competing companies on the basis of fuel economy and performance. In order to make these models useful, empirical efficiency data and gear ratios on many of the possible drivelines and vehicle configurations have already been collected. Given the relationships between PM and dHP/dt, and NOx and HP demonstrated in Chapter 4 (or more complicated relationships should these relationships prove to not be universal), it would not be difficult for engine manufacturers to predict emissions from various vehicle configurations and numerous different driving cycles before the engines are placed in the vehicle. Thus, it is now reasonable for the EPA to require that vehicle makers be responsible for ensuring emissions standards are met at the tailpipe of a vehicle. The EPA will need to choose a representative driving test cycle (or perhaps more than one, for example garbage trucks might be tested with a different cycle than large cargo trucks which are generally used for long-distance highway transport). The vehicle manufacturers, in turn, could obtain guarantees from engine makers regarding emissions behavior of the engine.

Engine manufacturers are already developing more sophisticated engine controls which vary fuel injection timing and fuel injection rates at various points in the drive cycle. Additionally, add-on emissions control devices are expected to play an important role in meeting proposed future emissions standards. These technological developments will make the relationship between emissions and engine behavior much more complicated than what was demonstrated in Chapter 4. Nonetheless, the type of regulation proposed will effectively control in-use emissions, while allowing engine manufacturers the flexibility to optimize tradeoffs between emissions, and engine performance and fuel economy at different points in the driving cycles as it becomes technologically possible.

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APPENDIX A

COLORADO SCHOOL OF MINES TRANSMISSION MODEL

Engine horsepower is calculated from vehicle speed as follows:

$$HP_{wheel} = HP_{accel} + HP_{roadload} + HP_{airdrag} \qquad \text{(A-1)}$$

$$HP_{accel} = Mv(dv/dt)$$

$$HP_{roadload} = \mu Mgv$$

$$HP_{airdrag} = 1/2C_D\rho Av^3$$

M = mass of the vehicle

μ = rolling resistance coefficient of tires

g = gravity acceleration, 32 ft/sec^2

ρ = density of air, lb$_m$/ft^3

C_D = air drag coefficient

A = frontal projected area of vehicle ft^2

v = vehicle speed, ft/sec

$$HP_{engine} = HP_{wheel} - \text{driveline losses} - \text{auxiliary HP} \qquad \text{(A-2)}$$

The driveline losses are a function of rotational speed and load at various points in the driveline and can be obtained as a series of empirical relationships specific to the components in each vehicle. Driveline losses at the axle, transmission and front pump are small in comparison to the total HP requirements.

For the purposes of estimating these values at various operational conditions, the following relationships were developed based on data provided by Navistar-International and Allison Transmission for the Mechanical Vehicle were considered adequate for estimating values for the other two vehicles. Under low speed conditions the torque convertor is the source of the majority of the driveline losses. The efficiency of the torque convertor can be calculated from a graph of the torque convertor absorption characteristics (Allison Transmission specification sheets), which provide efficiency at various turbine K factors. The turbine K factor is the speed of the torque convertor turbine divided by the square root of the load of the torque convertor.

For all three of these vehicles axle losses were estimated from HP at the wheel and the rotational speed of the axle as follows:

HP_{axle} (that is HP applied to the axle)= (A-3)

$$(HP_{wheel}/0.96)+((0.000006613*(RPM_{axle*}RPM_{axle}))+0.002794*RPM_{axle})$$

Horsepower applied to the transmission was estimated as follows:

$HP_{trans} =$ (A-4)

1st gear: $(HP_{axle}/.9892)+0.0000007914*(RPM_{trans*}RPM_{trans})+0.0005223*(RPM_{trans})\ -0.1471$

2nd gear: $(HP_{axle}/0.9871)+0.0000009414*(RPM_{trans*}RPM_{trans})\ -0.0007477\ *RPM_{trans}+0.1329$

3rd gear, convertor mode (Mechanical Vehicle only):

$(HP_{axle}/0.9958)+0.0000003129*(RPM_{trans*}RPM_{trans})+0.00004657*\quad (RPM_{trans})+0.0757$

2nd gear lockup (Electronic Vehicles #1 and #2 only): $HP_{axle}/0.9958$

3rd or 4th gear lockup: $HP_{axle}/0.9958$

Efficiency of the torque convertor was calculated as follows

Mechanical Vehicle:

$Efficiency_{conv} = (2.19-(0.0215*K_{turb}) + (0.00009294*K_{turb}*K_{turb}))*Speed\ Ratio_{conv}$ (A-5)

$Speed\ Ratio_{conv} = 0.02493+(0.01386*K_{turb})\ -(0.0000537*K_{turb}*K_{turb})$ (A-6)

(The speed ratio of the torque convertor is the ratio of the rotational speeds of the convertor turbine and the engine.)

Electronic Vehicles #1 and #2:

$Efficiency_{conv} = (2.329-(0.0402*K_{turb}) + (0.00004013*K_{turb}*K_{turb}))*Speed\ Ratio_{conv}$ (A-7)

$Speed\ Ratio_{conv} = 0.009675+(0.02413*K_{turb})\ -(0.0002237*K_{turb}*K_{turb})$ (A-8)

$HP_{conv} = HP_{trans}/Efficiency_{conv}$ (A-9)

There are some additional losses due to the front pump:

HP_{engine} - HP_{para}= (A- 10)

$HP_{conv}+0.0000004857*(RPM_{engine}*RPM_{engine})+(0.001327*RPM_{engine})-0.024$

Variable parasitic loads are placed on the engine to power the auxiliaries including the alternator, fan and air conditioning. Allison Transmission provided estimated HP deductions for these loads at various engine RPM from 1100 RPM to 2100 RPM for Electronic Vehicles #1 and #2. Since all runs were made with the air conditioning off, the HP required to run the air conditioner was deducted and the following equation was used to estimate parasitic loads for the two Electronic Vehicles:

Electronic Vehicles #1 and #2:

HP_{para}= (A-11)

<u>Clutch In</u>: $3.307-.009061*RPM_{engine}+.00001409*RPM_{engine}*RPM_{engine}$

<u>Clutch Out</u>: $1.390-.003805*RPM_{engine}+.000005918*RPM_{engine}*RPM_{engine}$

Estimates of the Mechanical Vehicle parasitic HP losses were estimated based upon the Electronic Vehicles' data by shifting the curve to align peak RPM and by prorating the magnitude of the curve by maximum engine HP to get the following equation:

Mechanical Vehicle:

HP_{para}= (A-12)

Clutch In: $4.216-.00870*RPM_{engine}+.00001022*RPM_{engine}*RPM_{engine}$

Clutch Out: $1.771-.0037*RPM_{engine}+.000004912*RPM_{engine}*RPM_{engine}$

Rotational speed at various points in the driveline is calculated as shown in Figure 4.1. In the cases considered here the inputs necessary to complete the calculation are listed below in Table A-1, with the exception of the speed ratio. That value is derived as described above.

At any point in the driveline, load can be calculated as a function of HP and rotational speed as follows:

$$\text{Load (ft-lbs)} = (HP/RPM) * (60\ s/min) * (550\ ft\text{-}lb/s) / (2*\Pi) \quad \text{(A-13)}$$

Table A-1. Constants for Calculation of Rotational Speed and Gear of Driveline. The source of the data is in parentheses.

	Mechanical Vehicle		Electronic Vehicles #1 and #2	
Tire Diameter	40.2 inches (Nav.-Intl.)		42.016 inches (Allison Transmission)	
Rear End Ratio	4.78 (Nav.-Intl.)		4.67 (Allison Transmission)	
1st Gear Ratio	3.58 (Allison Transmission)		3.692 (Allison Transmission)	
2nd Gear Ratio	2.09 (Allison Transmission)		2.021 (Allison Transmission)	
3rd Gear Ratio	1.39 (Allison Transmission)		1.383 (Allison Transmission)	
4th Gear Ratio	1 (Allison Transmission)		1 (Allison Transmission)	
Vehicle Speed (mph) for Gear Change (Allison Transmission)	1st to 2nd	14.82	1st to 2nd	11.24
	2nd to 3rd	26.26	2nd convertor to 2nd lockup	20.88
	3rd convertor to 3rd lockup	34.06	2nd lockup to 3rd lockup	27.54
	3rd lockup to 4th lockup	43.81	3rd lockup to 4th lockup	40.26
	4th lockup to 3rd lockup	39.52	4th lockup to 3rd lockup	39.45
	3rd lockup to 3rd convertor	27.95	3rd lockup to 2nd lockup	26.74
	3rd to 2nd	21.45	2nd convertor to 2nd lockup	18.60
	2nd to 1st	11.57	2nd to 1st	10.44