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Chassis Dynamometer Study of Emissions from 21 In-Use Heavy-Duty Diesel Vehicles

JANET YANOWITZ,[†,§]
MICHAEL S. GRABOSKI,[†,‡]
LISA B. A. RYAN,[†,‡]
TERESA L. ALLEMAN,[†] AND
ROBERT L. MCCORMICK*,[†,‡]

Colorado Institute for Fuels and High Altitude Engine Research, Department of Chemical Engineering and Petroleum Refining, and Department of Environmental Science and Engineering, Colorado School of Mines, Golden, Colorado 80401-1887

Regulated emissions from 21 in-use heavy-duty diesel vehicles were measured on a heavy-duty chassis dynamometer via three driving cycles using a low-sulfur diesel fuel. Emissions of particulate matter (PM), nitrogen oxides (NO_x), carbon monoxide (CO), total hydrocarbon (THC), and PM sulfate fraction were measured. For hot start tests, emissions ranged from 0.30 to 7.43 g/mi (mean 1.96) for PM; 4.15−54.0 g/mi (mean 23.3) for NO_x; 2.09−86.2 g/mi (mean 19.5) for CO; and 0.25−8.25 g/mi (mean 1.70) for THC. When emissions are converted to a g/gal basis, the effect of driving cycle is eliminated for NO_x and largely eliminated for PM. Sulfate comprised less than 1% of the emitted PM for all vehicles and test cycles. A strong correlation is observed between emissions of CO and PM. Cold starting at 77 °F produced an 11% increase in PM emissions. Multivariate regression analyses indicate that in-use PM emissions have decreased at a slower rate than anticipated based on the stricter engine certification test standards put into effect since 1985. NO_x emissions do not decrease with model year for the vehicles tested here. Smoke opacity measurements are not well correlated with mass emissions of regulated pollutants.

Introduction

EPA emission standards for diesel engines regulate total hydrocarbon (THC), CO, NO_x, and total particulate matter (PM). Although emissions from new diesel engines are routinely measured under the EPA engine certification test (*1*), emissions from the in-use diesel fleet have been poorly characterized. The results of diesel engine emissions testing conducted on a mass emitted/bhp-h basis are not easily translated into the mass emitted/mile basis normally used to estimate total emissions from the on-the-road diesel fleet. This is because mobile source emissions may be dependent on vehicle characteristics such as frontal area, weight and engine size, age, maintenance, and usage as well as environmental conditions including temperature, humidity, barometric pressure (*2*), and altitude that are not reflected in the engine test. Driver characteristics may also significantly impact emissions (*3*).

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 (*2, 4−8*). 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 (*6, 7*). 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 (*5*). 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 (*9*). Published studies (*2−8, 10−13*) 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 (*14*).

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) (*15*).

The objectives of the study described in the present contribution 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; and (d) perform an analysis of the data to determine which factors have the most important effect on emissions.

Experimental Section

Fleet Selection. The 21 vehicles tested in this study are listed in Table 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 four-stroke engines, and all of the 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 21 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.

* Corresponding author phone: (303)273-3967; fax: (303)273-3730; e-mail: rlmccorm@mines.edu.
† Colorado Institute for Fuels and High Altitude Engine Research.
‡ Department of Chemical Engineering and Petroleum Refining.
§ Department of Environmental Science and Engineering.

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TABLE 1. Characteristics of Vehicles Tested

vehicle no.	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	8 050	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.

TABLE 2. Results of Analysis of Northern Front Range (NFR) Wintertime Average Fuel and Industry Average Certification Fuel

test	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		
nonaromatic		70.8	wt %
aromatic		29.2	wt %
polynuclear aromatic		6.14	wt %

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 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) 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 (*16*).

Chassis Dynamometer Simulation. The chassis dynamometer is suitable for operating at vehicle speeds up to 60 mph. The vehicles are driven on twin 40-in. 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 (*17*), 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 s 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. Several vehicles were driven on all three cycles. The EPA Urban Dynamometer Driving Schedule for heavy-duty vehicles (heavy-duty transient or HDT cycle) is intended to represent heavy-duty driving in all U.S. urban areas (*18*). The cycle lasts over 18 min and requires the vehicle to drive at various speeds, including highway speeds and with changing acceleration rates. The average speed for this cycle is 18.8 mph, and the maximum speed attained is 58 mph. Inner city transit bus and delivery driving can be represented with the Central Business District (CBD) cycle (*19*). This 9-min cycle consists of 14 replicate accelerations, steady cruise at 20 mph, and braking steps to represent how a delivery truck or bus would perform during an inner city trip. The average speed over this cycle is 15.1 mph. West Virginia University (*20*) studied shifting and acceleration patterns of large trucks and developed the WVT cycle for the Department of Energy. The test lasts 14 min and consists of five steps of increasing speed, steady run, and braking. The maximum speed attained is 40 mph, and the average speed is 21.2 mph.

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 (*21*). All testing was performed at an intake air temperature of 77 $\pm$ 9 °F and humidity of 75 grain/lb dry air so that the NO_x correction factor is 1 $\pm$ 0.03.

Sulfate Analysis. Sulfate analysis was performed by Hazen Research, Inc., of Golden, CO using the procedure outlined by the Coordinating Research Council (*22*). 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; and (c) SAE J1667 snap idle procedure (*23*). 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 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 testing 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, 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 3 (included as Supporting Information) reports average results for hot and cold runs performed for each vehicle in the study. Emissions of PM, HC, 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 HC 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 HC 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 HC of 1.31 g/mi. The lowest 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 HC 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 4 (included as Supporting Information). 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 (*11*), 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, and 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 1. The regression line is for all test cycles and indicates an excellent correlation given the number of data points and confounding factors.

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 $\pm$ 9 °F. The vehicle was allowed to sit overnight and was not warmed prior to cold start testing. Figure 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 HC 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 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.

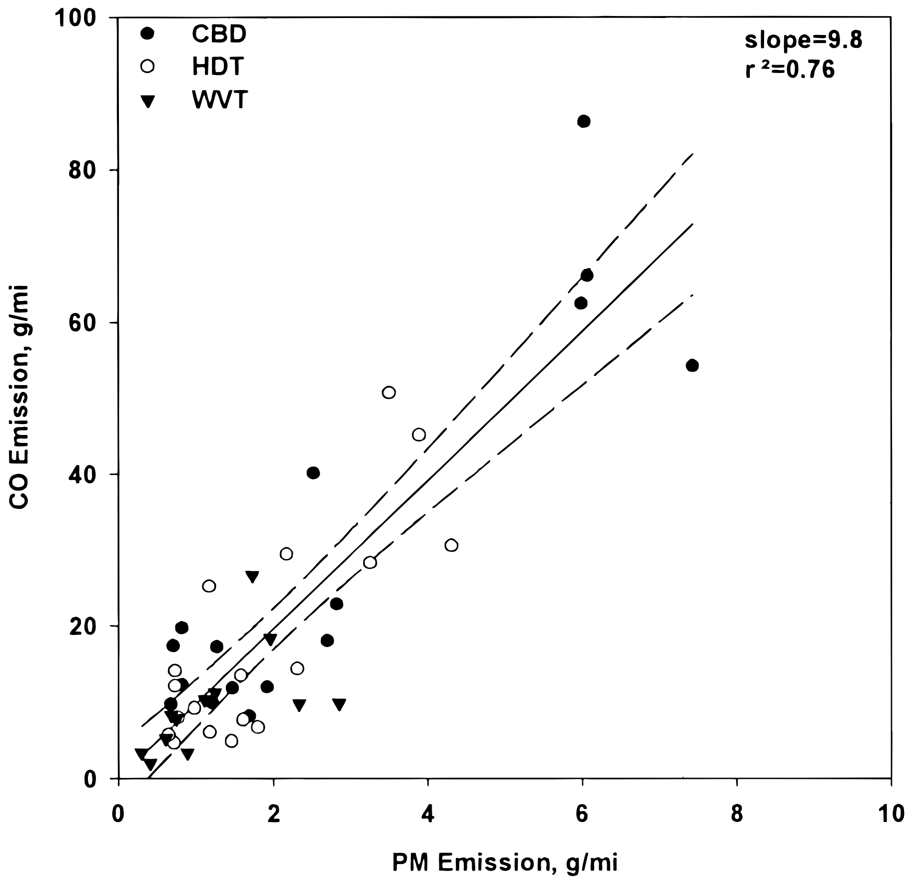

FIGURE 1. Correlation between CO and PM emissions over all test cycles using hot start averages from Table 3 (Supporting Information), 95% confidence interval shown.

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 3, Supporting Information). Emissions of NO_x and PM for the various cycles on a g/gal basis are also compared in Figure 3. For NO_x, comparison on a g/gal 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 (*24*−*26*). The g/gal comparison presented in Figure 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/mi 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 χ^2 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 (*27*−*29*) and a similar situation may exist for heavy-duty vehicles.

The following model equation was used in the regression analysis: log emission (g/gal) = $\alpha_1 + \alpha_2$(inertial Wt/GVWR) + α_3(odometer mileage) + α_4(year) + α_5(GMC = 1, not GMC = 0) + α_6(Cummins = 1, not Cummins = 0) + α_7(Isuzu = 1, not Isuzu = 0) + α_8(DDC four-stroke = 1, not DDC four-stroke = 0) + α_9(DDC two-stroke = 1, not DDC two-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 5 and 6 for PM and NO_x, respectively.

As shown in Table 5, for PM emissions significant correlation is found for inertial weight/HP for the HDT cycle (coefficient 0.0045; *t*-statistic 4.9) and model year (coefficient for CBD: −0.05, HDT: −0.05, *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 vehicles which are in some sense underpowered can emit higher levels of PM. As shown in Table 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.

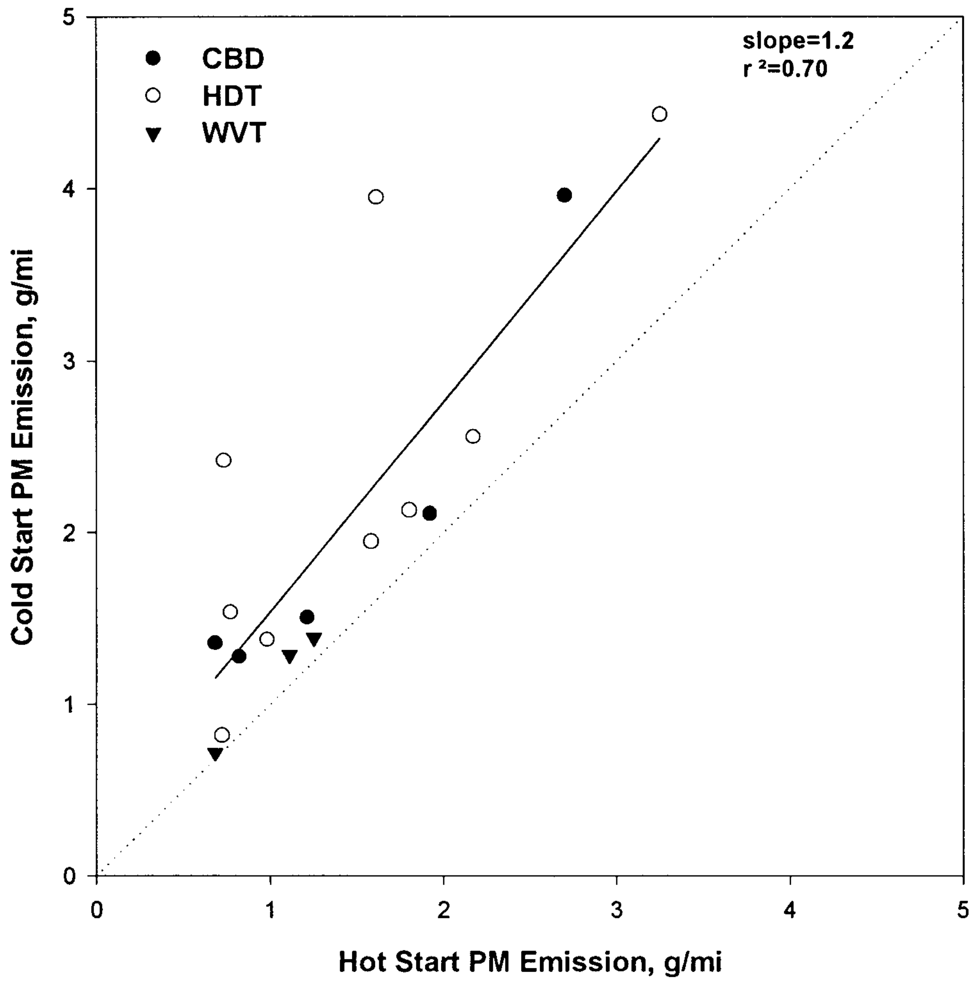

FIGURE 2. Effect of cold starting on PM emissions.

TABLE 5. Multivariate Regression Analysis Results for PM

		all vehicles				model year 1988 or later, not including urban buses			
		CBD $R^2 = 0.74$ no. of observations = 42		HDT $R^2 = 0.84$ no. of observations = 52		CBD $R^2 = 0.75$ no. of observations = 24		HDT $R^2 = 0.80$ no. 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		$t(33, 0.05) = 2.0$		$t(43, 0.05) = 2.0$		$t(17, 0.05) = 2.1$		$t(29, 0.05) = 2.0$	

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 −0.16, corresponding to the engine certification emission standard reduction of 16% 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. This indicates a PM reduction of 8−10% per year, ±3% per year at the 95% confidence level, suggesting that reductions in in-use emissions are slightly less than improvements measured by engine certification testing. Surprisingly, NO_x emissions actually increased at an average rate of 3% ± 2%

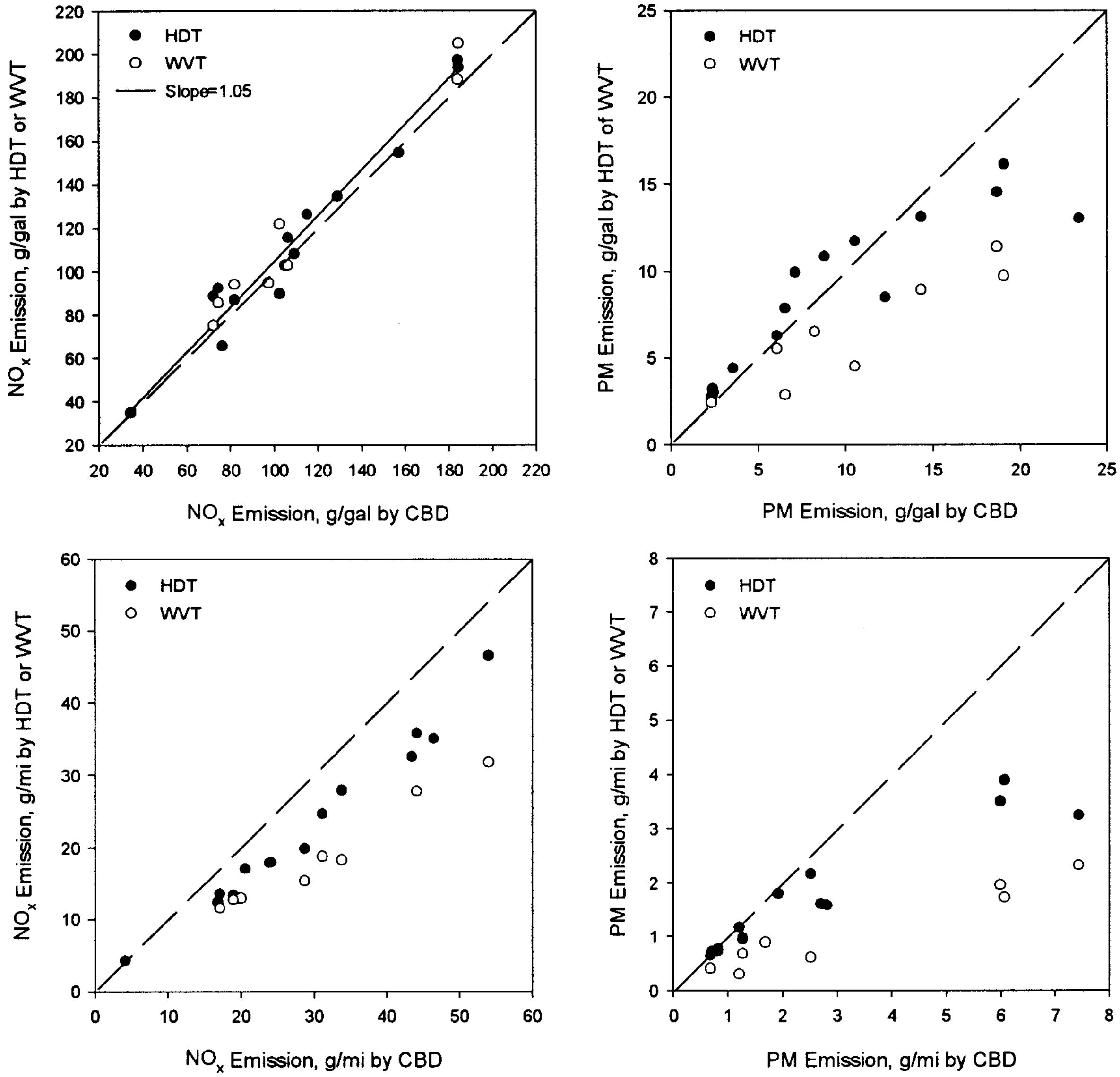

FIGURE 3. Comparison of NO_x and PM emissions from the CBD cycle with those from the HDT and WVT cycles on a g/mi (lower graphs) and g/gal (upper graphs) basis, parity shown as dashed line.

TABLE 6. Multivariate Regression Analysis Results for NO_x[a]

		all vehicles				vehicles model year 1986 or later			
		CBD $R^2 = 0.67$ no. of observations = 42		HDT $R^2 = 0.84$ no. of observations = 2		CBD $R^2 = 0.64$ no. of observations = 33		HDT $R^2 = 0.59$ no. 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		$t(33, 0.05) = 2.0$		$t(43, 0.05) = 2.0$		$t(24, 0.05) = 2.1$		$t(38, 0.05) = 2.0$	

[a] These low values for the coefficient and the *t*-statistic indicate that year does not significantly impact the NO_x emissions for this group of vehicles; however, it is included for the reader's information.

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 4.5% per year during the 1986−1995 period. Thus, NO_x emissions benefits anticipated from engine regulation and certification testing may not have been realized.

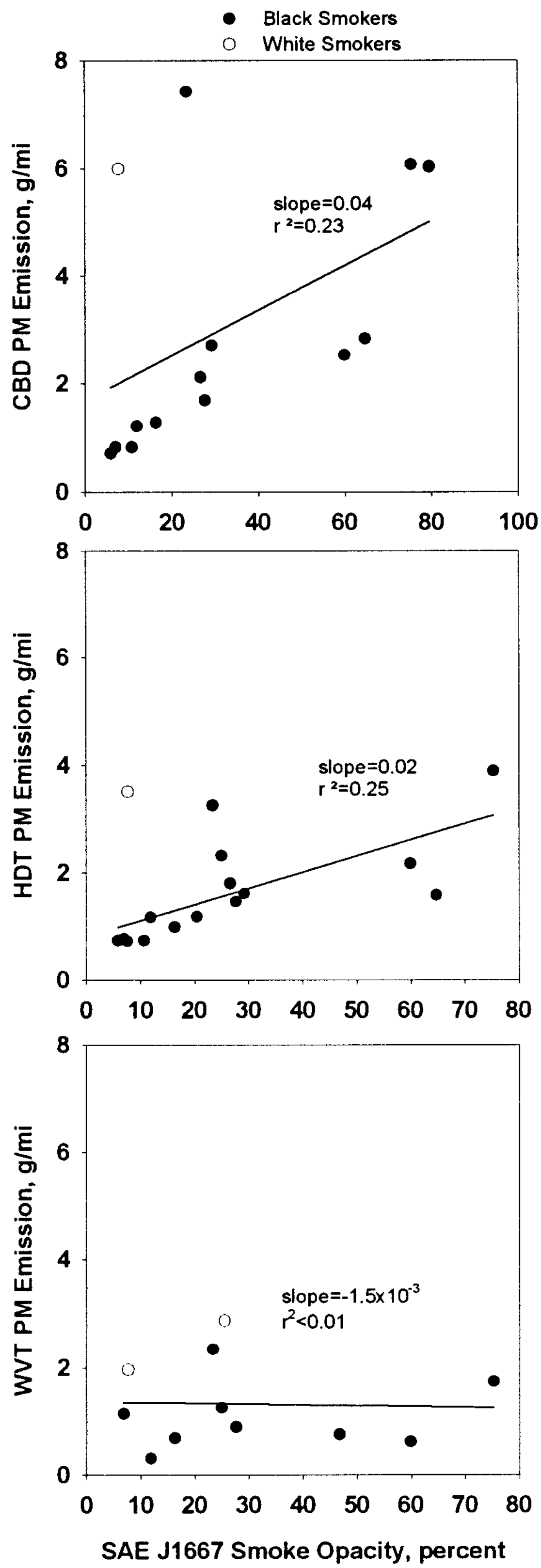

FIGURE 4. Comparison of PM mass emissions with smoke opacity by SAE J1667.

Smoke Opacity. Smoke opacity was measured by the SAE J1667, Colorado snap-idle, and Colorado lug-down tests for vehicles 1–16 and by SAE J1667 and the Colorado snap-idle test for vehicles 17–20. 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 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). 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 (not shown). 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 three or more times allowing statistical estimates of the repeatability of PM, NO_x, CO, and HC 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 1–4% 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 HC emissions occurs 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 HC 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 PM and 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 HC.

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Supporting Information Available

Table 3, regulated emissions results for average cold and hot start tests, and Table 4, sulfate emissions results (4 pages). Ordering information is given on any current masthead page.

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