On-Road Measurement of Emissions from Heavy-Duty Diesel Trucks: Impacts of Fleet Turnover and ARB's Drayage Truck Regulation

Final Report
Contract No. 09-340

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California Air Resources Board

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November 2014

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ACKNOWLEDGMENTS

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

ABSTRACT

The effects of heavy-duty drayage truck fleet modernization and diesel particle filter (DPF) retrofits were examined through measurements of truck emissions near the Port of Oakland. Pollutant concentrations were measured at high time resolution in the exhaust plumes of more than a thousand drayage trucks as they drove toward the Port on a major access road. Emission factors were matched to data from a statewide drayage truck registry, including engine make, model year, and installed emission control equipment, using recorded license plates for each truck. Between 2009 and 2013, Phase 1 of California's Drayage Truck Regulation led to an increase in Port trucks equipped with DPFs (the fraction so equipped rose from 2 to 99%), and a decrease in median engine age from 11 to 6 years. Over the same period, fleet-average emission factors decreased by $76 \pm 22\%$ and $53 \pm 8\%$ for black carbon (BC) and oxides of nitrogen (NO_x), respectively. However, direct emissions of nitrogen dioxide (NO_2) increased, and consequently the NO_2/NO_x emission ratio increased from 0.03 ± 0.02 to 0.18 ± 0.03. DPF-equipped trucks had substantially lower BC and higher NO_2 emission factors than trucks without DPFs. The newest trucks equipped with both DPFs and selective catalytic reduction (SCR) systems for NO_x control had the lowest average emission factors for BC and ultrafine particles (UFP), and an average NO_2 emission factor that was about equal to that of the Port truck fleet that was on the road in 2009 before recent emission regulations took effect. Phase 2 requirements have since led to replacement of nearly one third of the 2013 Port truck fleet (so far mainly pre-2007 engines have been replaced with newer engines equipped with SCR). Additional measurements of drayage truck emissions at the Port of Oakland are recommended in 2015 to quantify the emission impacts of increasingly widespread use of SCR for controlling NO_x emissions from on-road diesel engines, and to continue tracking trends in drayage truck emission factors over time.

EXECUTIVE SUMMARY

Heavy-duty diesel trucks are a major source of nitrogen oxides (NO_x) and exhaust particulate matter (PM) emissions in California. The Statewide Drayage Truck Regulation aims to reduce emissions of these pollutants by requiring universal use of diesel particle filter (DPF) emission control systems and replacement of pre-1994 model year engines on drayage trucks operating at major freight-handling facilities such as ports and rail yards. The DPF requirements were phased in over a period of three years — between Jan 1, 2010 and Dec 31, 2012. In this study, the effects of fleet modernization and DPF retrofits on in-use trucks were examined through field measurements of truck emissions at the Port of Oakland. Truck emissions were measured during 2011 and 2013, using high time resolution (1 Hz or faster) instrumentation that could distinguish the exhaust plumes of individual trucks. This study adds to an existing record of on-road emissions data for drayage trucks measured at the same site in 2009 and 2010. Emission factors quantified in this study include NO_x, nitrogen dioxide (NO_2), black carbon (BC), and ultrafine particle (UFP) number, in addition to particle size distributions. License plates of passing trucks were recorded and matched to entries in the California Air Resources Board's (CARB) Drayage Truck Registry, which provided information such as truck engine model year and DPF retrofit status. In this manner, emission factors could be matched to corresponding truck attributes for 378 trucks in 2011 and 1005 trucks in 2013.

The drayage truck fleet operating at the Port of Oakland was markedly transformed in a short period of time. In 2008, 2% of Port trucks were equipped with DPFs, and the median age of truck engines was 11 years. In 2013, 99% of trucks were DPF-equipped, and 9% also had selective catalytic reduction (SCR) systems for enhanced control of NO_x emissions. The median

age of Port truck engines had decreased to 6 years by 2013. Between 2009 and 2013, fleet-average emission factors decreased by $76 \pm 22\%$ and $53 \pm 8\%$ for black carbon (BC) and NO_x, respectively. The use of DPFs led to increases in fleet-average emission factors for NO_2 and therefore an increase in the NO_2/NO_x emission ratio (from 0.03 ± 0.02 to 0.18 ± 0.03) over the same time period.

The distributions of emission factors for Port trucks have become increasingly skewed over time, with a minority of trucks now responsible for the majority of emissions of all pollutants—except carbon dioxide (CO_2)—measured in this study. The highest emitting 10% of trucks observed in 2013 were respectively responsible for 30, 65, and 80% of total NO_x, BC, and ultrafine particles (UFP) emissions from all Port trucks. The highest emitters of UFP generally were among the lowest emitters of BC, and vice versa. Likewise, the highest emitters of NO_2 generally were among the lowest emitters of BC. Emission factors for UFP were variable, whereas measured BC and especially NO_x emission factors were more reproducible when the same trucks were observed driving by on different days.

In the first set of field measurements performed in 2011 for this study, some trucks (engine model years 2004-2006) were exempt from the requirement to have DPF systems installed. This sub-group of trucks provided a point of reference for comparison with other sub-fleets of trucks equipped with more advanced emission control equipment. These comparisons showed that, on average, DPF-equipped trucks had substantially lower BC and higher NO_2 emission factors compared to trucks without DPFs. Trucks emitted particles ranging in size between 6 and 300 nm, and DPFs were most effective at reducing emissions of UFP with diameters larger than 15 nm. The newest trucks (2010-2013 engines) that were equipped with both DPF and SCR systems had the lowest average emission factors for BC (0.09 ± 0.04 g kg^{-1}),

NO_x (5 ± 1 g kg^{-1}), and UFP ((1.6 ± 1.2) $\times$ 10^{15} particles kg^{-1}). NO_2 emissions from these newest trucks (1.1 ± 0.3 g kg^{-1}) were greater than those from trucks without DPFs (0.6 ± 0.3 g kg^{-1}) but were nearly equal to the pre-regulation fleet-average emissions (1.0 ± 0.7 g kg^{-1}).

Phase 2 of the Drayage Truck Regulation required replacement of all pre-2007 engines, on a schedule that extends beyond the second field study component of this research. Therefore, the fraction of drayage trucks equipped with SCR is expected to increase further. The increases in NO_2 emission factors reported here, resulting from widespread use of DPFs to control diesel PM emissions, may have been partly mitigated as use of SCR systems to control NO_x emissions has become more widespread.

1. INTRODUCTION

Heavy-duty diesel trucks emit the majority of nitrogen oxides (NO_x) and diesel particulate matter (PM) in urban areas in California (Dallmann and Harley, 2010; McDonald et al., 2012). These emissions contribute to exceedances of fine particulate matter ($PM_{2.5}$) and ozone air quality standards (Lloyd and Cackette, 2001; Morawska et al., 2008), and may lead to adverse health effects for exposed individuals (McClellan, 1987; Lloyd and Cackette, 2001; Brugge et al., 2007; IARC, 2012). Black carbon (BC), which is a short-lived yet potent absorber of solar radiation, comprises the majority of diesel PM (Ban-Weiss et al., 2008). Recent studies have suggested control of diesel BC emissions as a near-term strategy to help mitigate climate change due to anthropogenic emissions of longer-lived greenhouse gases such as carbon dioxide (CO_2) (Bond et al., 2013; Ramanathan et al., 2013).

The U.S. Environmental Protection Agency has set increasingly stringent exhaust PM and NO_x standards to reduce emissions from on-road heavy-duty diesel trucks, as shown in Figure 1. The most recent standards correspond to emissions that are 98% lower than emissions from trucks meeting the earliest standards that were in force approximately twenty-five years ago. The most recent standards require the use of exhaust after-treatment control technologies. Trucks with 2007 and newer engines are equipped at the time of manufacture with diesel particle filter (DPF) systems for PM emission control, while trucks with 2010 and newer engines are also equipped selective catalytic reduction (SCR) systems for NO_x control. DPFs can be installed as retrofits on older engines that are already in use (van Setten et al., 2001). Measures to accelerate reductions of in-use diesel truck emissions have been adopted in California, given the long in-service lifetimes of heavy-duty engines.
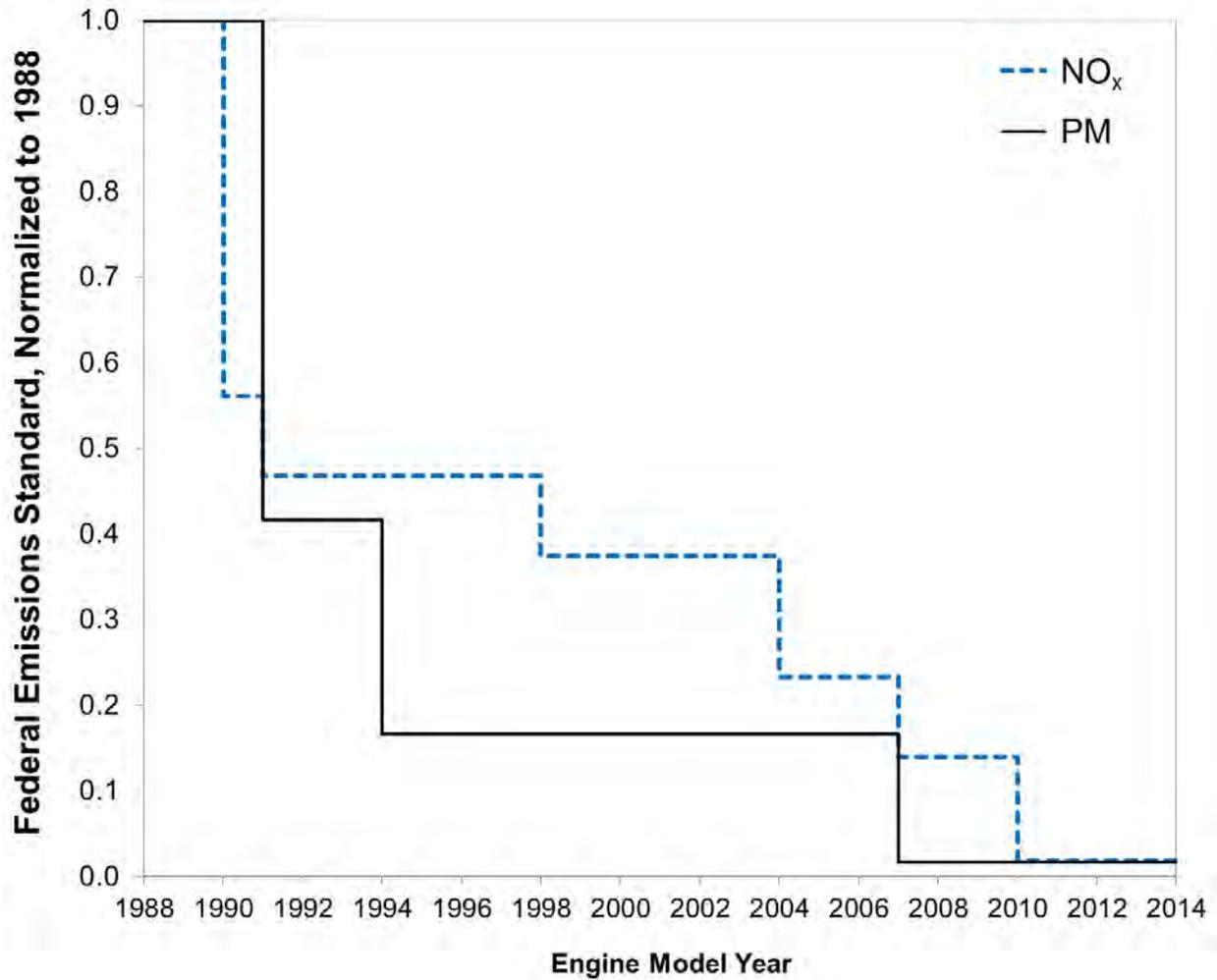
Figure 1. Federal exhaust emissions standards of NO_x and PM for new heavy-duty highway engines (EPA, 2013), normalized to the earliest standards in effect as of 1988.

Previous studies have shown that DPFs effectively remove more than 90% of PM mass from heavy-duty diesel engine exhaust (Biswas et al., 2008; Herner et al., 2009; Barone et al., 2010). Trapped soot particles are oxidized to regenerate the filter either passively, by continuous reaction with NO_2 that is formed by catalytic oxidation of exhaust NO, or actively, for example by episodic injection of fuel to oxidize particles trapped on the filter. The intentional conversion of exhaust NO to NO_2 in passively regenerated DPF systems leads to increased NO_2 emissions

and higher NO_2/NO_x emission ratios (Carslaw, 2005; Herner et al., 2009). This effect is likely to increase with higher exhaust temperature and increased catalytic loading of the DPF system (Herner et al., 2009). Dynamometer testing of DPF-equipped trucks has also shown that the reduction of total particle mass—and thus the availability of condensation surfaces for volatile exhaust gases—may encourage homogeneous nucleation events and thereby increase ultrafine particle (UFP) emissions (Biswas et al., 2008; Herner et al., 2011).

In California, the introduction of DPFs has been accelerated by the Statewide Drayage Truck Regulation. Table 1 shows the implementation schedule for this rule. Over a three year period—from Jan 1, 2010 to Dec 31, 2012—all drayage trucks were required to be equipped with DPFs, either via retrofit or engine replacement (ARB, 2013). Trucks with 2007-2009 model year engines are compliant through 2022, while trucks with 2010 and newer engines are fully compliant. At the Ports of Los Angeles and Long Beach in Southern California, the drayage truck fleet was almost entirely replaced with new equipment. The mean age of trucks there decreased from 12.7 years to 2.5 years over a two-year period, and large associated emissions reductions were observed for carbon monoxide, NO_x, and infrared opacity: 30, 48, and 54%, respectively (Bishop et al., 2012). In contrast to the Southern California ports where truck replacement predominated, there was a significant element of DPF retrofitting on drayage trucks at the Port of Oakland. Dallmann et al. (2011) measured a 41% reduction in NO_x and a 54% reduction in BC emissions between November 2009 and June 2010, associated with the first round of emission control requirements on drayage trucks. The BC reductions resulted primarily from increased use of DPFs, whereas NO_x reductions were attributed to replacement of the oldest trucks with newer equipment that met more stringent NO_x emission standards. The initial round of changes to the drayage truck fleet at the Port of Oakland led to an increased proportion of

trucks with 2004 and newer engines, as well as a reduction in mean engine age from 11.0 to 8.3 years (Dallmann et al., 2011).

Table 1. Phased implementation schedule for the Statewide Drayage Truck Regulation.

Phase of Regulation	Deadline	Engine Model Year	Requirement
Phase 1	January 2010	1993 & Older	Banned
		1994 - 2006	Retrofit with DPF or replace with newer engine
	January 2012	2004	
	January 2013	2005 - 2006	
Phase 2	January 2014	1994 - 2006	Replace with 2007 or newer engine

The current study builds on previous work by Dallmann et al., and features two new field measurement campaigns that include additional pollutants not previously measured, namely NO_2, UFP number, and particle size distributions. Also, emission factors for individual trucks in the current study were linked through transcribed license plate data to truck attributes such as engine model year and DPF retrofit status. This made it possible to compare emission factors for trucks with different technology and engine model year categories in this study, in addition to quantifying changes to fleet-average emission factors measured in current and previous work.

2. METHODS

Following the approach of Dallmann et al. (2011), truck emissions were measured in November 2011 and March 2013 near the Port of Oakland. Pollutant concentrations in the exhaust plumes of individual trucks were measured using an instrumented van that was positioned on an overpass (Figure A1 of the Appendix). Westbound trucks heading toward the Port drove underneath the overpass on a major access road. Exhaust/ambient air mixtures sampled above the roadway were delivered to the instrumented van via a flexible aluminum duct, as shown in Figure 3. In 2011 and 2013, concentrations of CO_2, NO_x, nitric oxide (NO), BC, UFP, and particle size distributions were measured at 1 Hz or faster with the instruments listed in Table A1 of the Appendix. A video camera at roadway level recorded truck license plates, which were later transcribed and matched to entries in the California Air Resources Board's (CARB) Drayage Truck Registry. This provided data for each truck including engine model year, engine manufacturer, and, if applicable, DPF retrofit status and installer.

A sample pollutant concentration time series is presented in Figure 3, and shows peaks associated with three trucks that passed by in rapid succession over a time period of about one minute. Pollutant concentration peaks were integrated to calculate fuel-based emission factors, expressed in units of amount of pollutant emitted per kg of fuel burned, using a carbon balance method:

$$\text{EF}_P = \frac{\int_{t_1}^{t_2}([P]_t-[P]_{t_1})dt}{\int_{t_1}^{t_2}([CO_2]_t-[CO_2]_{t_1})dt} \frac{44}{12} w_c \qquad (1)$$

The emission factor for pollutant P (EF_p) is calculated over the time interval $t_1 \leq t \leq t_2$, with t_1 and t_2 determined independently by the inflection points of each peak to account for the fact that instruments measuring different pollutants have different response times. The numerator and denominator, respectively, represent the baseline-subtracted peak areas for pollutant P and CO_2. When [P] and [CO_2] have mass concentration units (e.g., $\mu g\ m^{-3}$), the ratio represents the relative abundances of pollutant P and CO_2 present in the exhaust. The weight fraction of carbon in diesel fuel ($w_c = 0.87$) is used to convert emission factors from per mass of carbon to mass of fuel burned (Ban-Weiss et al., 2009), and the ratio of 44/12 represents the total/carbon mass ratio in CO_2. NO_2 emission factors were computed by difference for each truck using simultaneously measured emission factors for NO_x and NO.

Emission factors were computed for trucks for which the measured CO_2 concentration peak rose more than 7% above baseline roadway CO_2 concentrations, following Dallmann et al. (2011). The baseline was taken as the concentration measured just prior to the passage of a truck; the timing was determined from recorded video images of passing trucks. Emission factors were computed only when the CO_2 concentration peak could be definitively attributed to a single truck. Thus, emission factors were not computed when multiple trucks passed the sampling location at the same time or in close succession. In 2011, 23% of passing trucks were sufficiently spaced to resolve individual exhaust plumes, and license plates for 35% of those trucks were clearly visible and could be transcribed. In 2013, sampling efforts were focused on the right-hand lane (closer to the curbside video camera). This led to higher plume and license plate capture rates: plumes for 37% of passing trucks were individually distinguishable, and plates for 63% of those trucks were visible and transcribed. In cases where there was successful CO_2 plume

capture without other corresponding pollutant peaks, pollutant emission factors were still computed, as illustrated in Figure 3.

As shown in Figure 4, an in-line dilution system was used to avoid exceeding the concentration limits of the condensation particle counters (CPCs) used to measure UFP number concentrations. The dilutor consisted of a filtered recirculating flow in a closed loop that drew from the outlet side of a standard finned-tubular flow mixer and injected the particle-free flow back into the upstream side of the mixing tube to be combined with the incoming aerosol flow. A matched pair of water-based CPCs (WCPC) was placed on the upstream and downstream sides of the dilutor to actively monitor the dilution rate. All downstream particle number concentrations were post-corrected to account for the dilution factor, which was typically ~6. The dilution factor was determined from the regressed slope of the line forced through zero, with the upstream CPC concentration data as the dependent variable and the downstream CPC concentration as the independent variable. Slight mismatches in data acquisition timing between the independent, asynchronous data streams was handled by averaging down time series with successively larger averaging windows until an asymptotic value was obtained for the regressed slope.

The particle sampling configuration shown in Figure 4 was designed to minimize wall losses by inertial separation and diffusion. Laminar flow was maintained in all sampling lines to minimize inertial losses of accumulation mode particles that could bias particle mass measurements. Diffusional losses of UFP were minimized by avoiding long runs of slow moving sample flows to the instruments measuring particle number and size. Using a standard laminar diffusion model, line losses of 10 nm particles were estimated as ~10% for the CPCs, 2% for the fast mobility particle sizer (FMPS), and 1% for the dilution system's mixing tube.

Prior to field measurements, the full setup of instruments and data logging were staged in the laboratory and exposed to the exhaust of an inverted methane-air diffusion flame (Kirchstetter and Novakov, 2007). The measurement of exhaust plumes of passing trucks was simulated by episodically sampling flame exhaust instead of filtered room air. The analysis of multiple peaks under constant flame exhaust conditions established the precision of the plume capture sampling and carbon balance analysis methods that were used in this study. The NO_2 conversion efficiency of both NO_x analyzers was also evaluated using ozone titration tests, to ensure accuracy of total NO_x ($NO + NO_2$) measurements.

Figure 2. Instrumented van positioned on an overpass sampling the exhaust from a truck en route to the Port of Oakland.
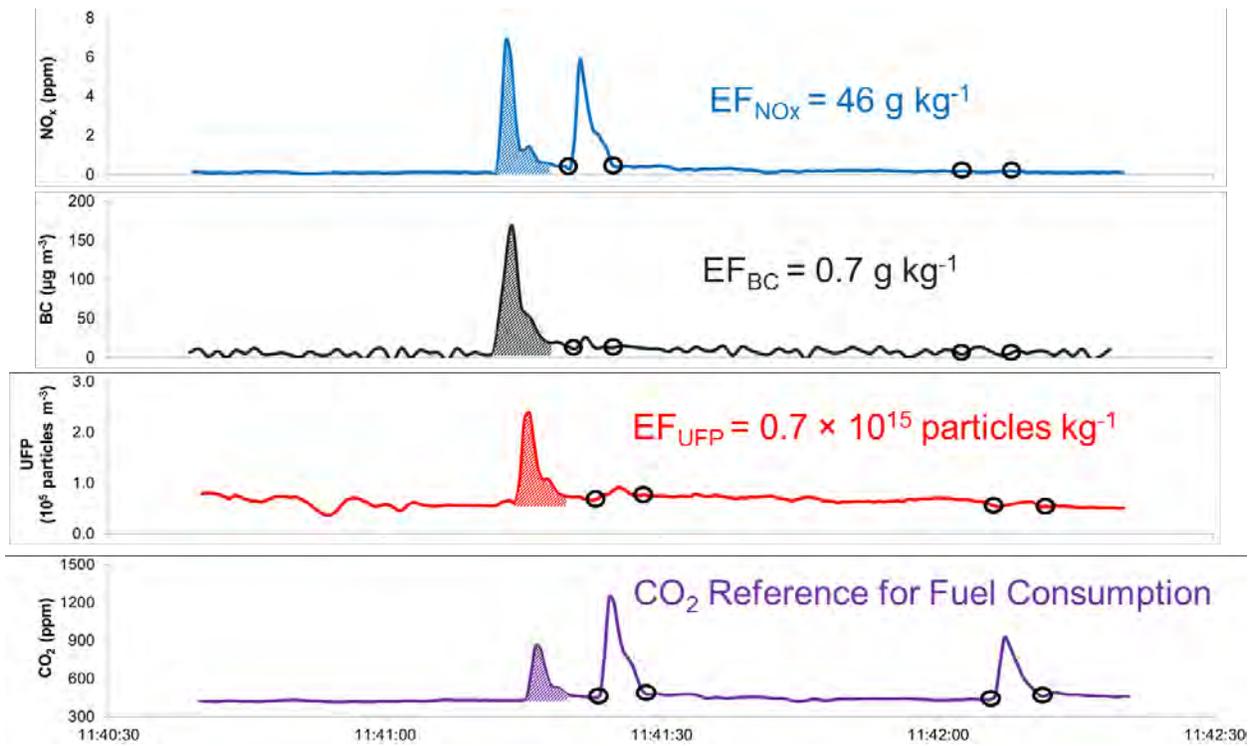
Figure 3. Pollutant concentration time series showing peaks that correspond to the exhaust plumes of three trucks. The first truck emitted appreciable amounts of NO_x, BC, and UFP. The shaded peaks correspond to the integrated areas used to compute the emission factors shown in the figure. The second and third trucks emitted much smaller amounts of BC and UFP and the third truck emitted essentially no NO_x. The integration boundaries are indicated with open circles for the second and third trucks.

9

Figure 4. Schematic of particle sampling instruments and their flow rates, and the in-line dilution system.

Laboratory testing identified a significant time response issue for the LI-COR model 820 CO_2 analyzer, which tends to overs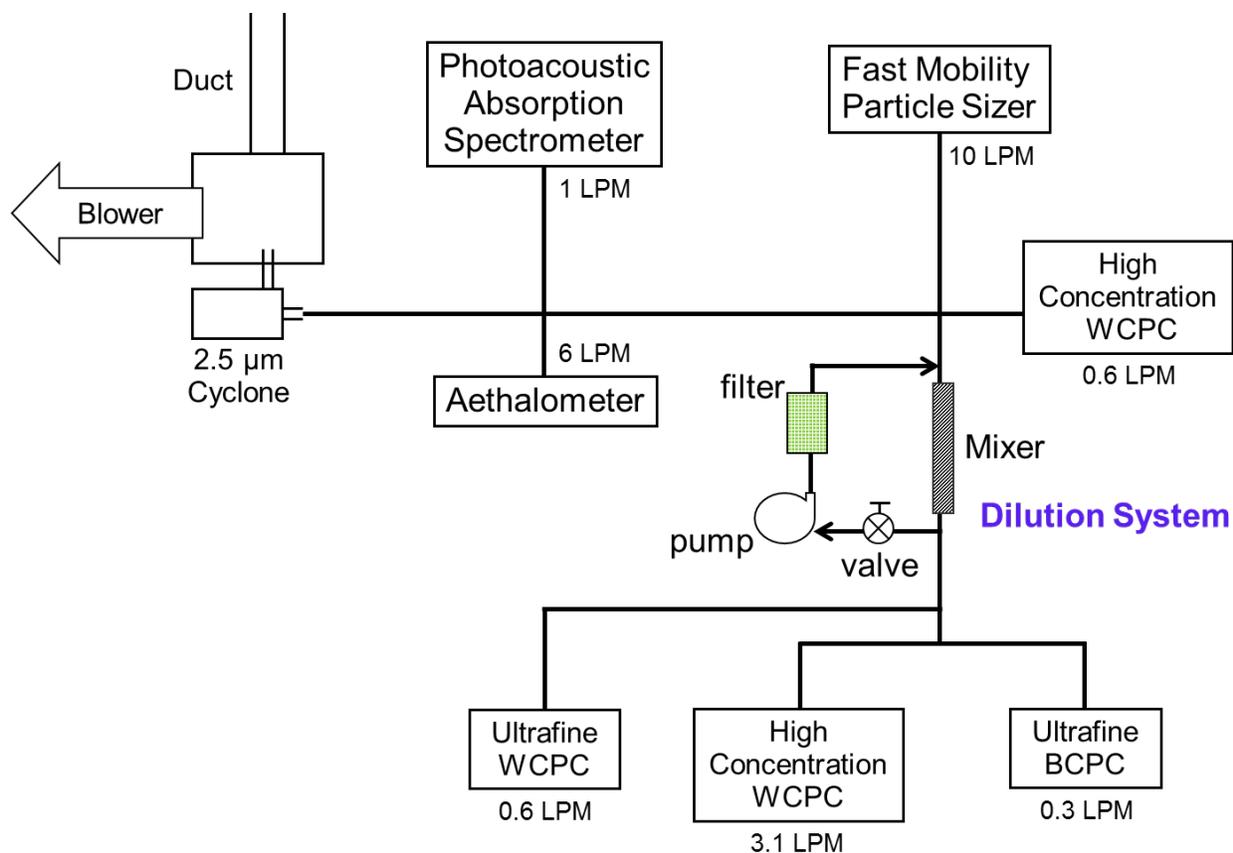hoot in reporting peak concentrations when rapid transitions occur. Therefore, a faster response LI-7000 instrument was used in the present study, as it did not suffer from the same problem (Figure 5). The LI-820 CO_2 analyzer was used in parallel during the 2011 field study to assess the magnitude of possible biases in earlier measurements. Figure 6 shows a frequency distribution of the ratio of integrated CO_2 peaks measured using LI-820 and LI-7000 analyzers. As shown in Figure 6, use of LI-820 data led to overestimates in the magnitude of CO_2 peak areas by $26 \pm 2\%$ (95% confidence interval) for a sample of 389 trucks. Therefore, baseline pollutant emission factors measured in an earlier field measurement

campaign at the Port of Oakland in 2009 (Dallmann et al., 2011) were multiplied by 1.26 in making comparisons with emission factors measured using the more accurate LI-7000 CO_2 analyzer in the 2011 and 2013 campaigns at the Port.
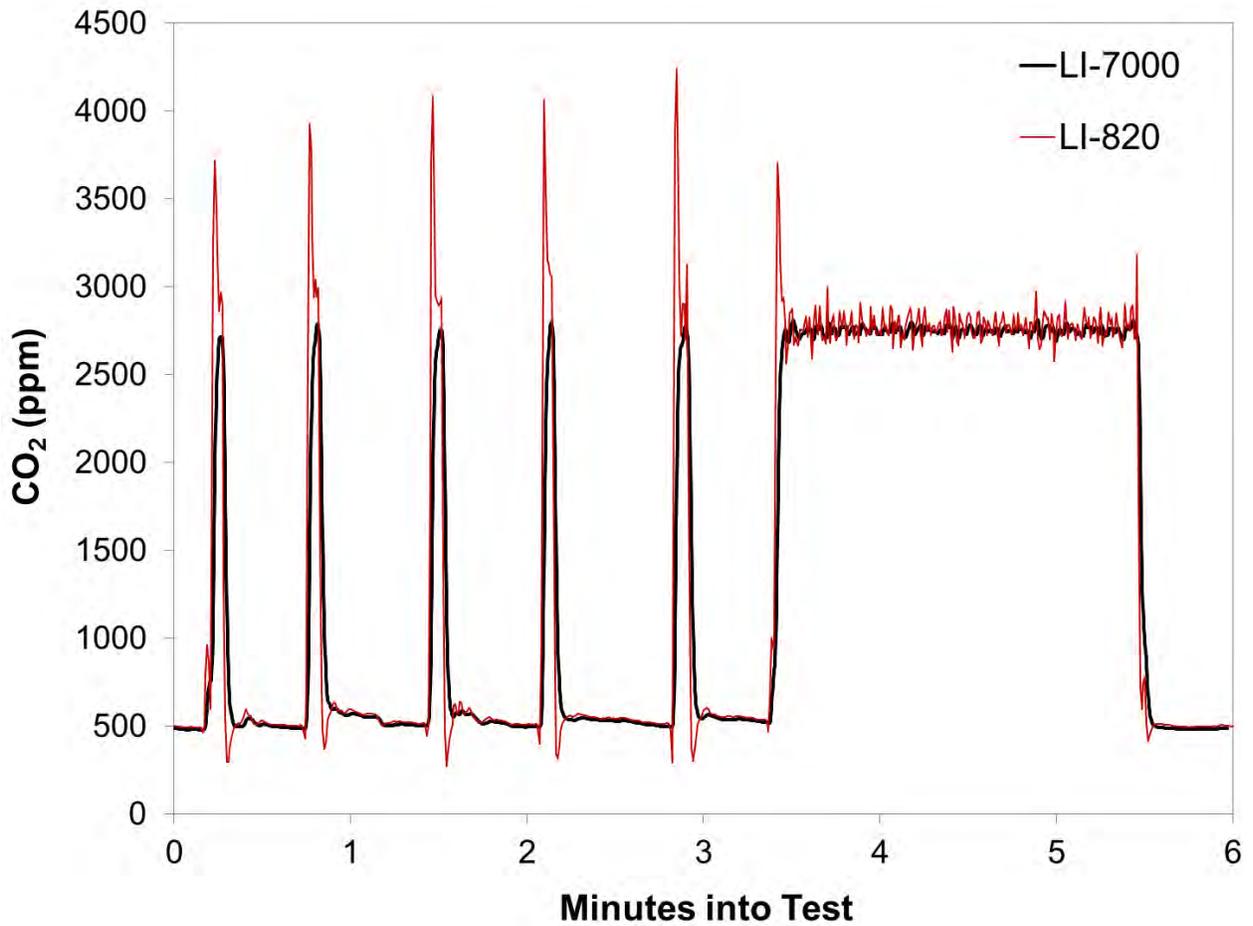

Figure 5. Comparison of time series of CO_2 concentrations measured in laboratory evaluation of two LI-COR CO_2 analyzers: LI-820 and LI-7000. The LI-820 overshoots when concentrations change rapidly, whereas the high performance LI-7000 transitions smoothly to match new peak concentrations.
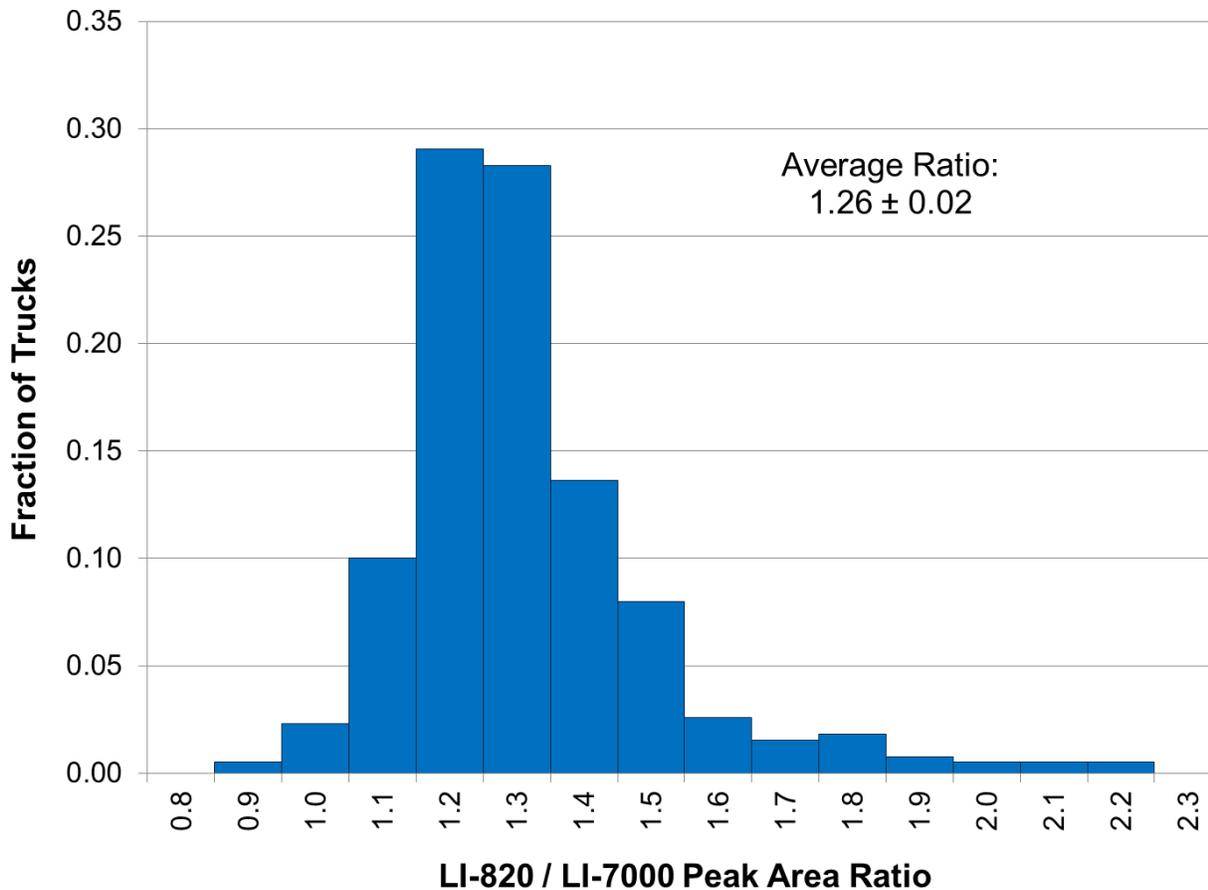
Figure 6. Distribution of average ratio of CO_2 peak areas measured by LI-820 and LI-7000 analyzers, as determined from concurrent measurements during the 2011 study at the Port of Oakland.

The laboratory tests also identified a concern associated with particle size distributions measured using the FMPS. Jeong and Evans (2009) previously reported an unexpected peak near the lower size limit (below 10 nm) of the FMPS, when sampling ambient air in both urban and rural settings. This was posited to be due to either the data inversion algorithm or the calibration of the FMPS electrometers used for particle detection. In our tests, the FMPS overstated the concentration of UFP with diameters less than 10 nm on the trailing side of peaks when particle number concentrations were rapidly decreasing. Figure 7 shows an example of this effect.

Accordingly, when analyzing particle size distributions measured at the Port of Oakland, representative particle size distributions for each truck were chosen from the leading side of the particle number concentration peak. A baseline-subtracted, normalized particle size distribution was then determined for each truck:

$$\text{Emission rate of UFP in each size bin } = \frac{\left(\frac{dN}{d\log dp}\right)_i}{N}\, EF_{UFP} \tag{2}$$

Here, the particle number concentration measured in each size bin i is normalized to the total particle number concentration, N. The product of this normalized size distribution function and the corresponding emission factor gives the emission rate of UFP in each size bin in units of number of particles emitted per kg of fuel burned.

Aethalometer BC data were post-corrected for time-dependent behavior that causes the instrument response to decline with increasing filter darkening (Kirchstetter and Novakov, 2007). A modified version of the correction equation developed by Kirchstetter and Novakov was used to adjust the BC concentration reported by the aethalometer:

$$BC = \left[\frac{BC_0}{a\,\exp\left(\frac{-ATN}{100}\right) + (1 - a)}\right] \tag{3}$$

where BC and BC_0 are the adjusted and unadjusted BC concentrations, respectively, and ATN is the attenuation of light by the filter. The correction parameter, a, adjusts BC_0 such that BC concentrations are independent of filter loading. This correction is validated by plotting the ratio

13

of the light absorption coefficient measured with the photoacoustic absorption spectrometer (PAS) and the BC concentration measured with the aethalometer, as shown in Figure 8. Both instruments respond to light-absorbing PM, but the *in-situ* PAS measurement is not affected by filter loading effects. As shown, the ratio of absorption coefficient to BC_0 increased by a factor of two as the aethalometer filter progressed from pristine (ATN = 0) to heavily loaded (ATN = 150). In contrast, the ratio of absorption coefficient to BC is approximately constant and independent of ATN. This indicates that adjusted BC concentrations can be up to two times higher than unadjusted BC concentrations, assuming aethalometer measurements as the basis.

Our measurements of diesel truck exhaust at the Port of Oakland indicated $a = 0.66$, whereas BC emission factors reported by Dallmann et al. (2011) were corrected with Kirchstetter and Novakov's value of $a = 0.88$, which was derived from testing using laboratory-generated soot. We calculated BC emission factors (n = 1000) for each truck using both values and found that emission factors calculated with $a = 0.66$ were $15 \pm 2\%$ lower on average than those calculated using $a = 0.88$. Therefore, BC emission factors measured in the 2009 campaign (Dallmann et al., 2011) were multiplied by 0.85 before making comparisons to emission factors derived from the 2011 and 2013 field campaigns.
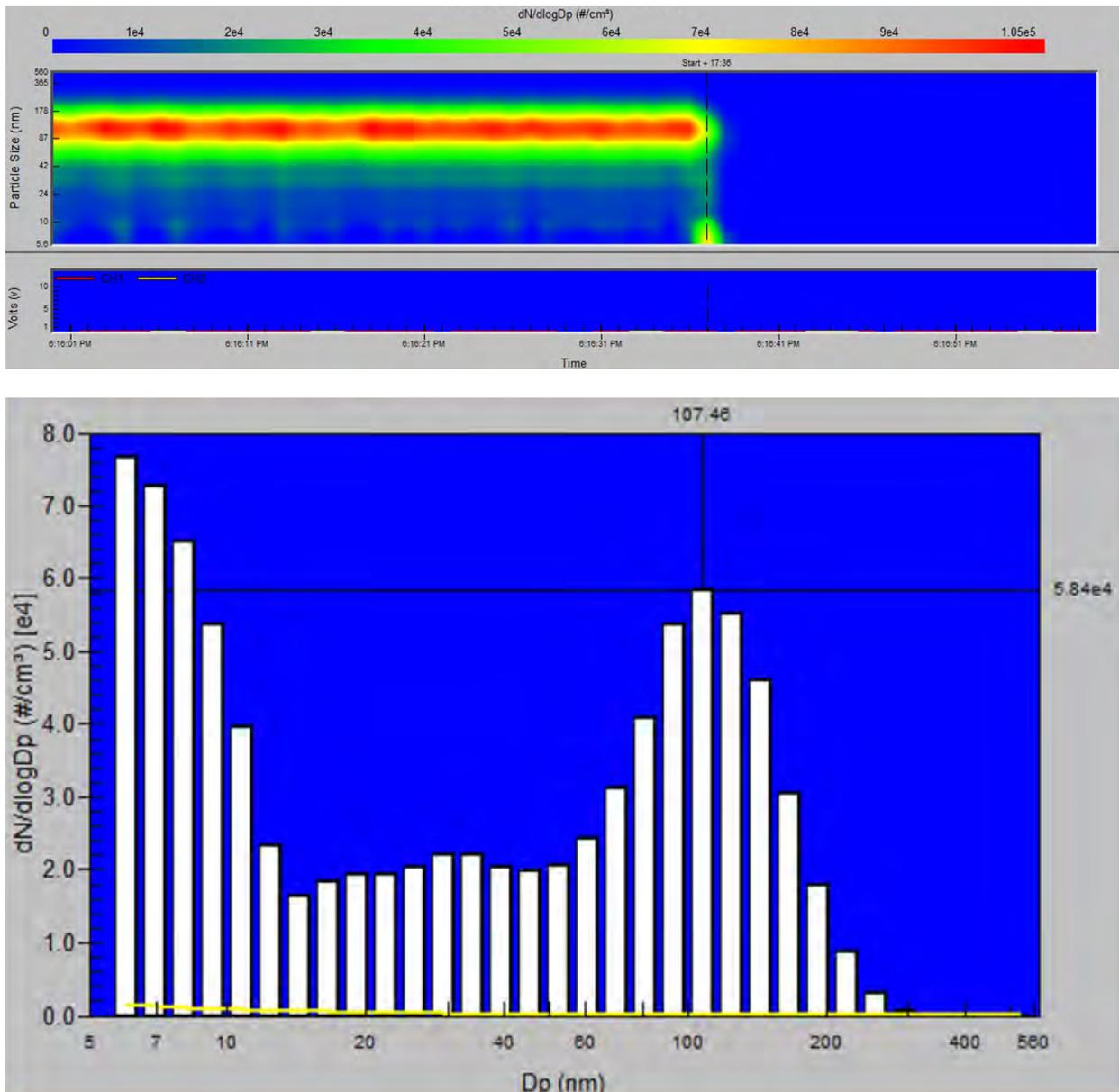
Figure 7. FMPS data showing an example of the artificial increase in ultrafine particle (UFP) number concentration of particles below 10 nm in diameter when particle number concentrations were rapidly decreasing. The top panel shows the measured particle number concentration, sampled from a constant source in the laboratory. The vertical dashed line shows the time at which the number concentration decreased and the artificial pulse of particles < 10 nm was reported. The bottom panel shows the corresponding particle size distribution at that time, including an anomalous peak in particle sizes below 10 nm.

Figure 8. BC concentration measured using an aethalometer and absorption coefficient measured using a photoacoustic absorption spectrometer (PAS). Unadjusted BC concentrations (BC_0) depend on aethalometer filter loading (i.e., ATN) whereas the adjusted BC concentrations (BC) are independent of ATN.
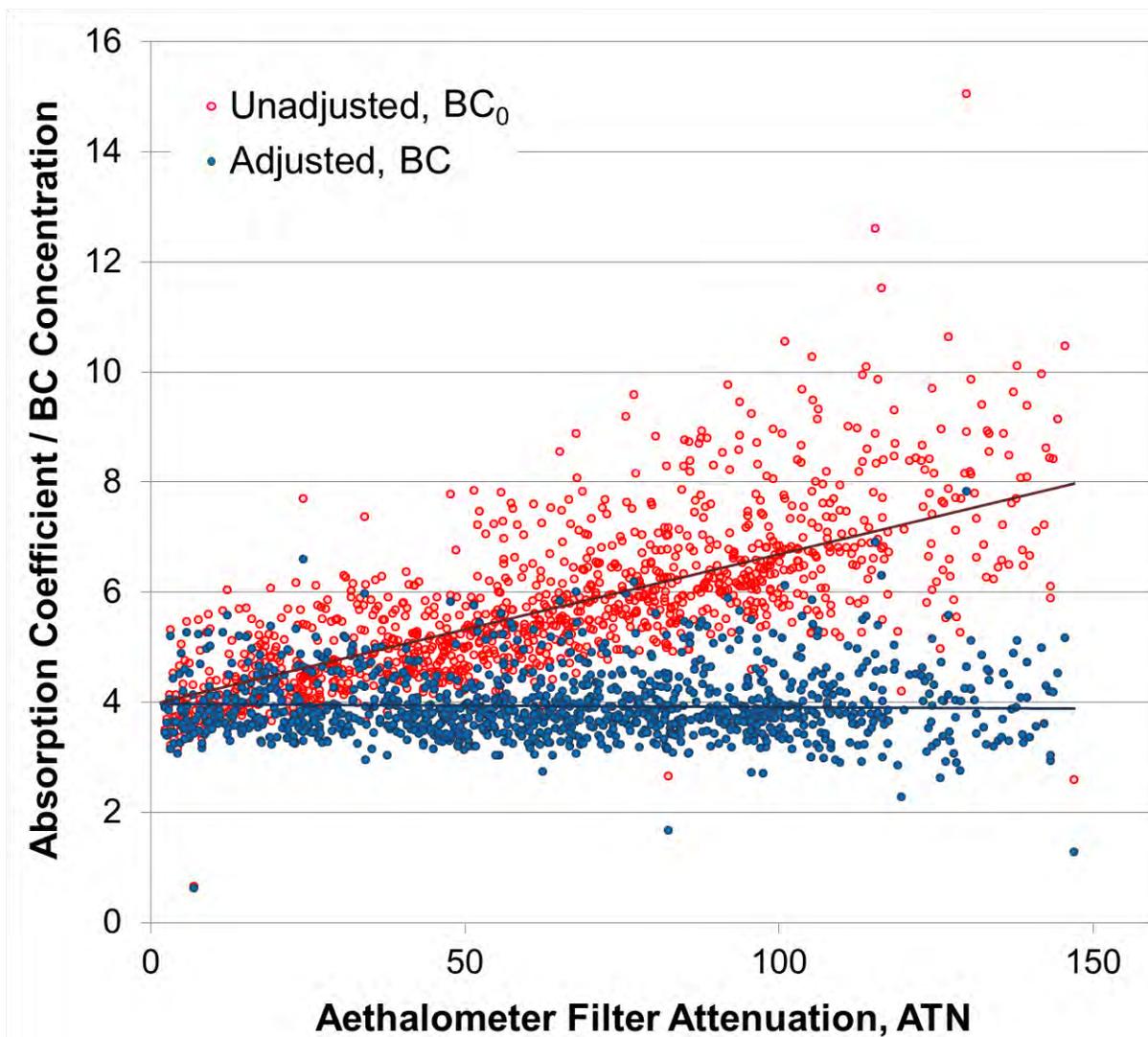
3. RESULTS

3.1 Changes to Port Truck Age Distribution. The age distribution of the truck fleet operating at the Port of Oakland changed dramatically over a short period of time, as shown in Figure 9. Prior to implementation of the Drayage Truck Regulation, median engine age was 11 years, and only 2% of trucks had 2007 or newer model engines with DPFs. After full implementation of the first phase of the regulation, 99% of trucks were equipped with DPFs, 9% had 2010 or newer model engines with DPFs and SCR systems, and median engine age decreased to 6 years.

3.2 Changes in Fleet-Average Emission Factors. Fleet-average emission factors are presented in Table 2, as measured at the Port of Oakland in 2009, 2011, and 2013. Emission factors broken down more finely into four truck categories based on engine model year and installed control technologies are also reported in Table 2 and shown in Figures 10 and 12-15: (1) trucks retrofitted with DPFs, (2) trucks without DPFs, (3) trucks with DPFs as original equipment installed by the manufacturer (OEM), and (4) trucks with both DPF and SCR as original equipment. Emission factors partitioned in even greater detail based on DPF installer and engine manufacturer are shown in Figures 11 and A2 in Appendix. In all these presentations of results, indicated uncertainty ranges about the mean values represent the 95% confidence intervals.

3.2.1 Black Carbon. Between 2009 and 2013, average BC emission factors decreased by 76 ± 22% (see Table 2 and Figure 10). Average BC emission factors of trucks equipped with retrofit DPFs (1994-2006 engines) and trucks with OEM DPFs (2007- 2009 engines) were similar, while trucks with 2010 and newer engines had the lowest BC emission factors. On average, trucks with retrofit or OEM DPFs emit 74 ± 30% less BC than trucks without DPFs, and trucks with OEM DPFs and SCR systems (2010 and newer engines) emit 92 ± 32% less BC.

17
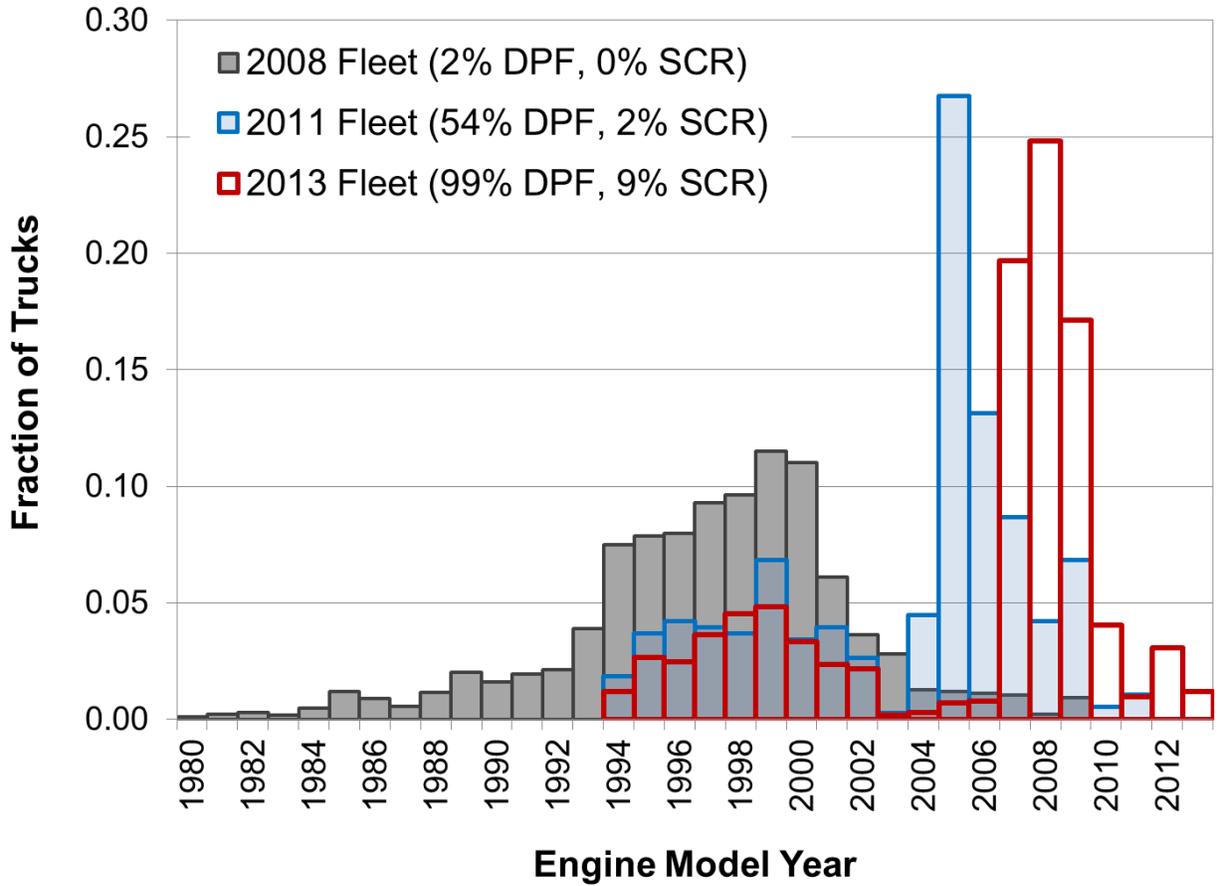
Figure 9. Distribution of truck engine model years at the Port of Oakland (a) prior to the Drayage Truck Regulation in 2008 (BAAQMD, 2009), (b) during implementation of the regulation in 2011, and (c) after full implementation of Phase 1 of the regulation in 2013. Note that (a) is derived from a survey of port trucks conducted prior to our emission measurements, whereas (b) and (c) correspond directly to trucks measured in this study.

Table 2. Summary of results for the fleet over time and by engine control technology, including the median engine model year for each category and the average pollutant emission factors with 95% confidence intervals.

Fleet or Truck Category	Range of Engine Model Years	Median Engine Model Year	BC (g kg^{-1})	NO$_x$ (g kg^{-1})	NO$_2$ (g kg^{-1})	NO$_2$/NO$_x$ Emission Ratio	UFP (10^{15} particles kg^{-1})
2009 Fleet[1] (2% DPF, 0% SCR)	1970 - 2009	1997	1.15 ± 0.19	33 ± 2	1.0 ± 0.7	0.03 ± 0.02	N/A
2011 Fleet (54% DPF, 2% SCR)	1994 - 2011	2004	0.67 ± 0.14	18 ± 1	2.1 ± 0.4	0.12 ± 0.03	3.6 ± 0.8
2013 Fleet (99% DPF, 9% SCR)	1994 - 2013	2007	0.28 ± 0.05	15 ± 1	2.8 ± 0.2	0.18 ± 0.03	2.5 ± 0.5
Retrofit DPF	1994 - 2006	1998	0.32 ± 0.06	26 ± 1	3.9 ± 0.4	0.15 ± 0.02	2.6 ± 0.8
No DPF	2004 - 2006	2005	1.11 ± 0.26	17 ± 2	0.6 ± 0.3	0.03 ± 0.02	4.7 ± 1.0
OEM DPF	2007 - 2009	2008	0.26 ± 0.06	12 ± 1	2.7 ± 0.3	0.23 ± 0.04	2.5 ± 0.6
OEM DPF + SCR	2010 - 2013	2011	0.09 ± 0.04	5 ± 1	1.1 ± 0.3	0.22 ± 0.12	1.6 ± 1.2

[1]Adapted from Dallmann et al. (2011): 2009 emission factors have been adjusted to account for CO_2 monitor response issue and to use a consistent adjustment for time-dependent response of the aethalometer. NO$_2$ emission factor in 2009 estimated based on the NO$_2$/NO$_x$ emission ratio measured in 2011 for trucks without DPFs, as indicated in the text.
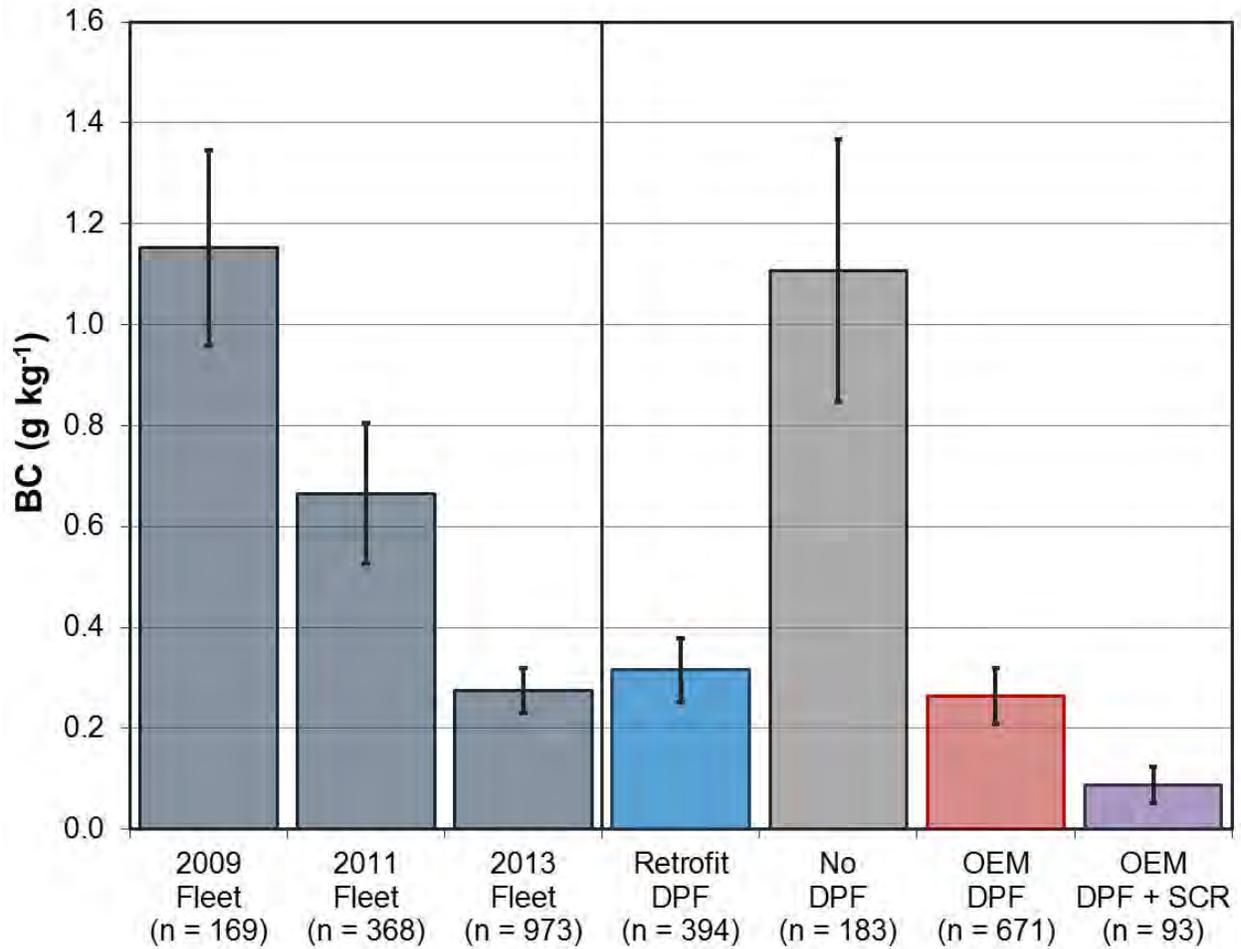
Figure 10. Fleet-average BC emission factors by year of field sampling (bars on left) and average emission factors by truck category (bars on right; includes 2011 and 2013 data only). The number of analyzed trucks is indicated in parentheses on the category axis and the error bars reflect 95% confidence intervals about the mean.

Figure 11. (a) BC and (b) NO_2 emission factors by suppliers of DPF retrofits and OEM engines, with 95% confidence intervals indicated.

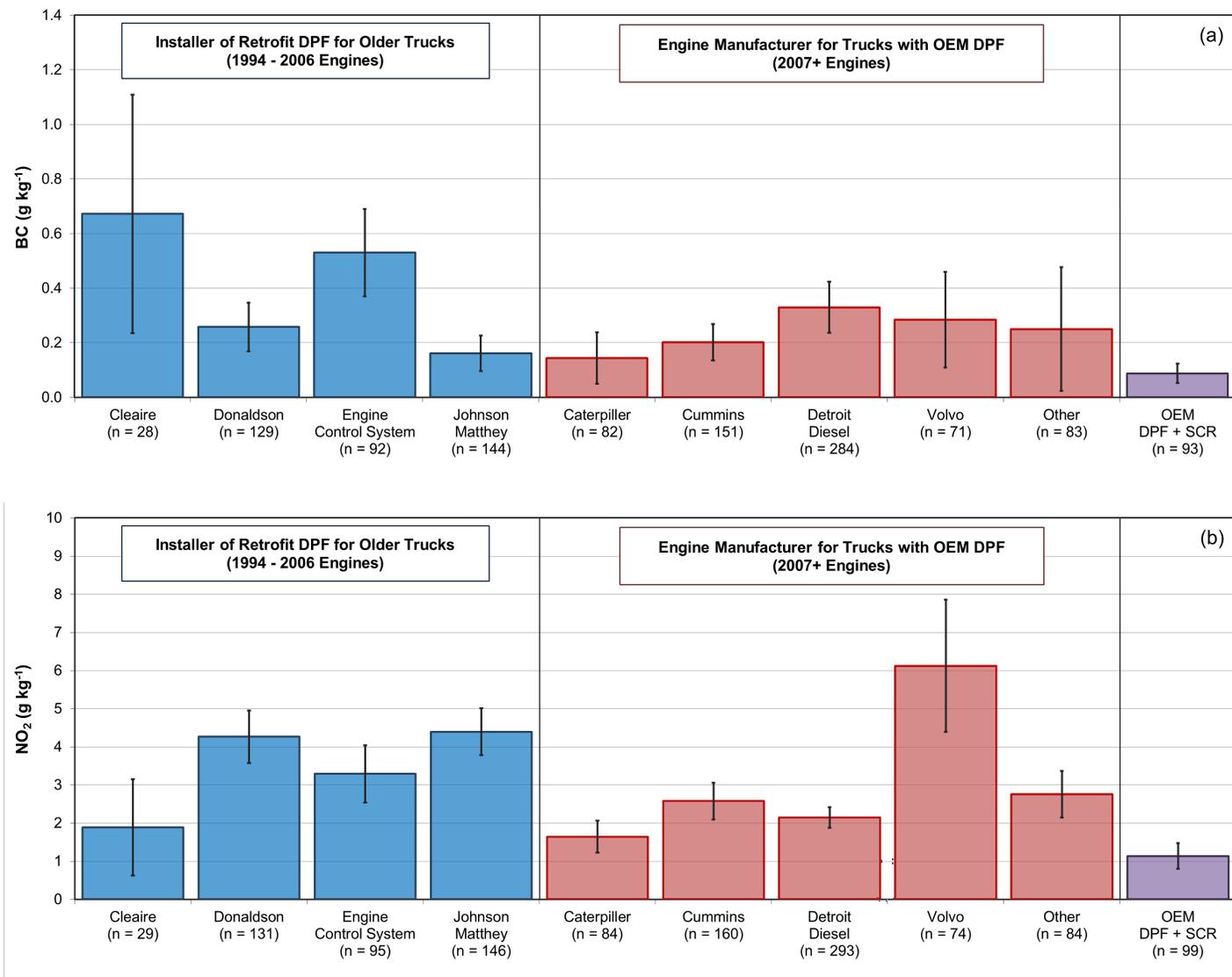Figure 11a shows separate BC emission factors for trucks with retrofit DPFs provided by four different suppliers/installers. Also shown are the BC emissions from trucks equipped with OEM DPFs (2007-2009 engines), supplied as original equipment by four major engine manufacturers, plus an "other" category that groups together small numbers of trucks with engines supplied by various other manufacturers. Finally, for comparison, the lowest BC emission factors in this figure are for trucks with 2010 and newer engines with both OEM DPFs and SCR. No disaggregation by manufacturer is shown for the newest engines because the sample size is relatively small (n = 99 trucks). BC emission factors for trucks with Engine Control System filters are higher than for those with DPFs installed by Donaldson and Johnson Matthey. BC emission factors for trucks with Cleaire retrofit filters are higher than those from trucks with Johnson Matthey filter retrofits but not higher than those installed by Donaldson at the 95% confidence level. In contrast to the retrofit cases, none of the differences in BC emission factors among trucks with original equipment DPFs are statistically significant.

3.2.2 Nitrogen Oxides. The fleet-average NO_x emission factor decreased by 53 ± 8% between 2009 and 2013, as shown in Table 2 and Figure 12. The NO_2/NO_x emission ratio in 2009 was not measured, but is assumed here to be the same as found for trucks without DPFs measured in this study in 2011, using results for 187 trucks ($NO_2/NO_x = 0.03 ± 0.02$, see Table 2). The emission factor for NO_2 in 2009 was estimated by combining this ratio with the NO_x emission factor measured in 2009. The NO_2/NO_x emission ratio increased dramatically as the use of DPFs to control exhaust PM emissions became universal for Port drayage trucks: NO_2 rose from 3 to 18% of total NO_x between 2009 and 2013 (see Table 2).

Comparing the four truck categories (Figure 13), trucks retrofitted with DPFs had the highest NO_2 emission rate, which was 7 times higher than that from trucks without DPFs. The

22

NO_2/NO_x emission ratios were greatest for newer trucks with OEM DPFs (Figure 14), with and without SCR systems, but engines with SCR systems had the lowest NO_2 emission rate of the three categories of DPF-equipped trucks. Despite having a similarly elevated NO_2/NO_x emission ratio as observed for other DPF-equipped trucks, the NO_2 emission rate for trucks with SCR systems (2010 and newer engines) was the same as that determined for the pre-DPF era Port truck fleet measured in 2009. This observation indicates the effectiveness of SCR in mitigating the undesired increase in primary NO_2 emissions associated with use of diesel particle filters to control exhaust PM emissions.

While there were no significant differences in total NO_x emission factors among trucks retrofitted with DPFs from four different suppliers (Figure A2), the NO_2 emission factors for trucks retrofitted with DPFs by Donaldson and Johnson Matthey were significantly higher than for trucks with Cleaire retrofit DPFs, as shown in Figure 11b. Examination of emissions from newer trucks with OEM DPFs (2007-2009 model year engines) distinguished by truck engine manufacturer revealed that trucks with Volvo engines emitted more NO_x and NO_2 (Figures A2 and 11b) compared to engines supplied by other manufacturers. Further, the Volvo engine NO_2 emission rate exceeded that of older trucks retrofitted with DPFs (Figure 11b).

Figure 12. Fleet-average NO$_x$ emission factors by sampling year (bars on left) and average emission factors by truck category (bars on right; includes 2011 and 2013 data only). The number of analyzed trucks is indicated in parentheses on the category axis, and error bars reflect 95% confidence intervals about the mean.
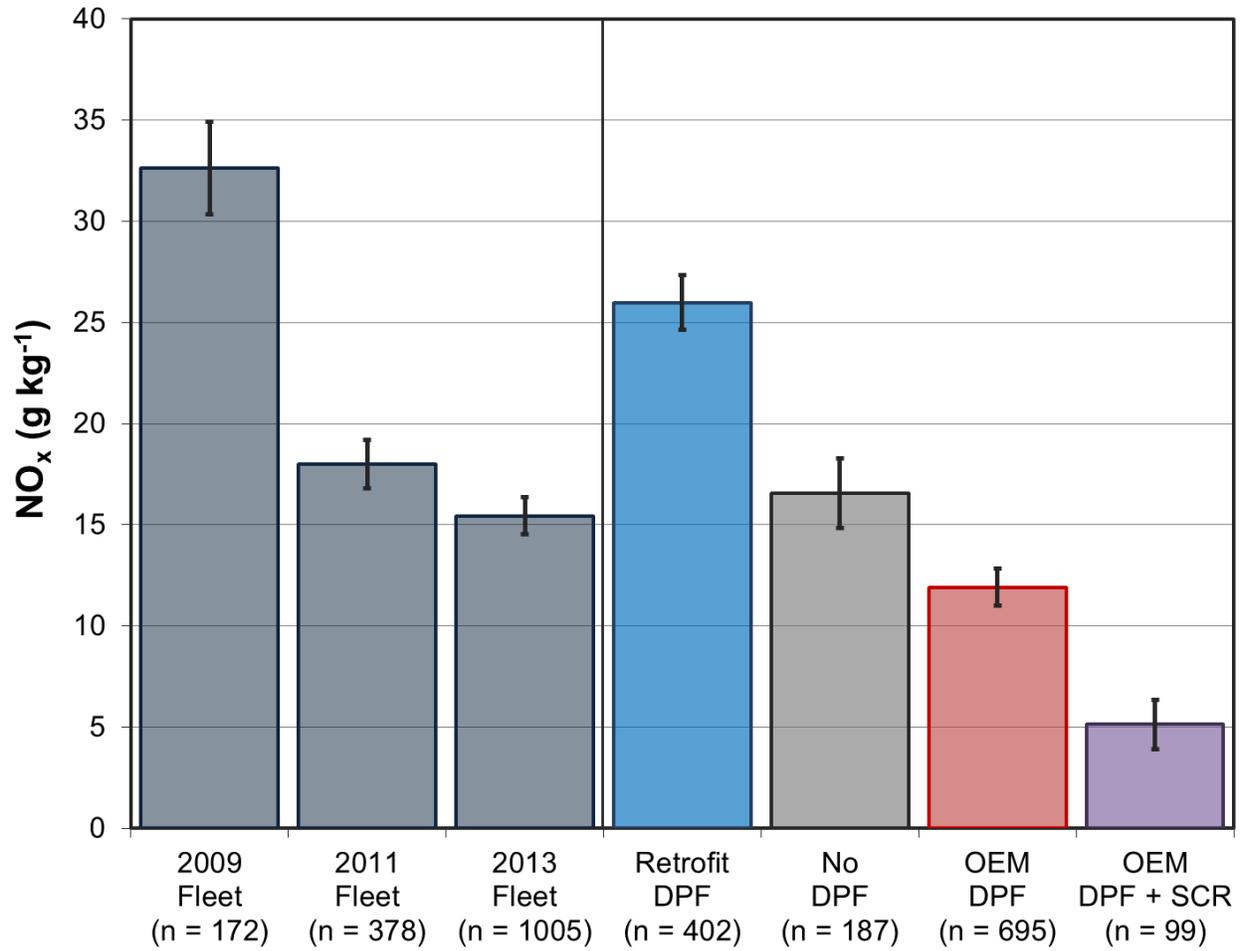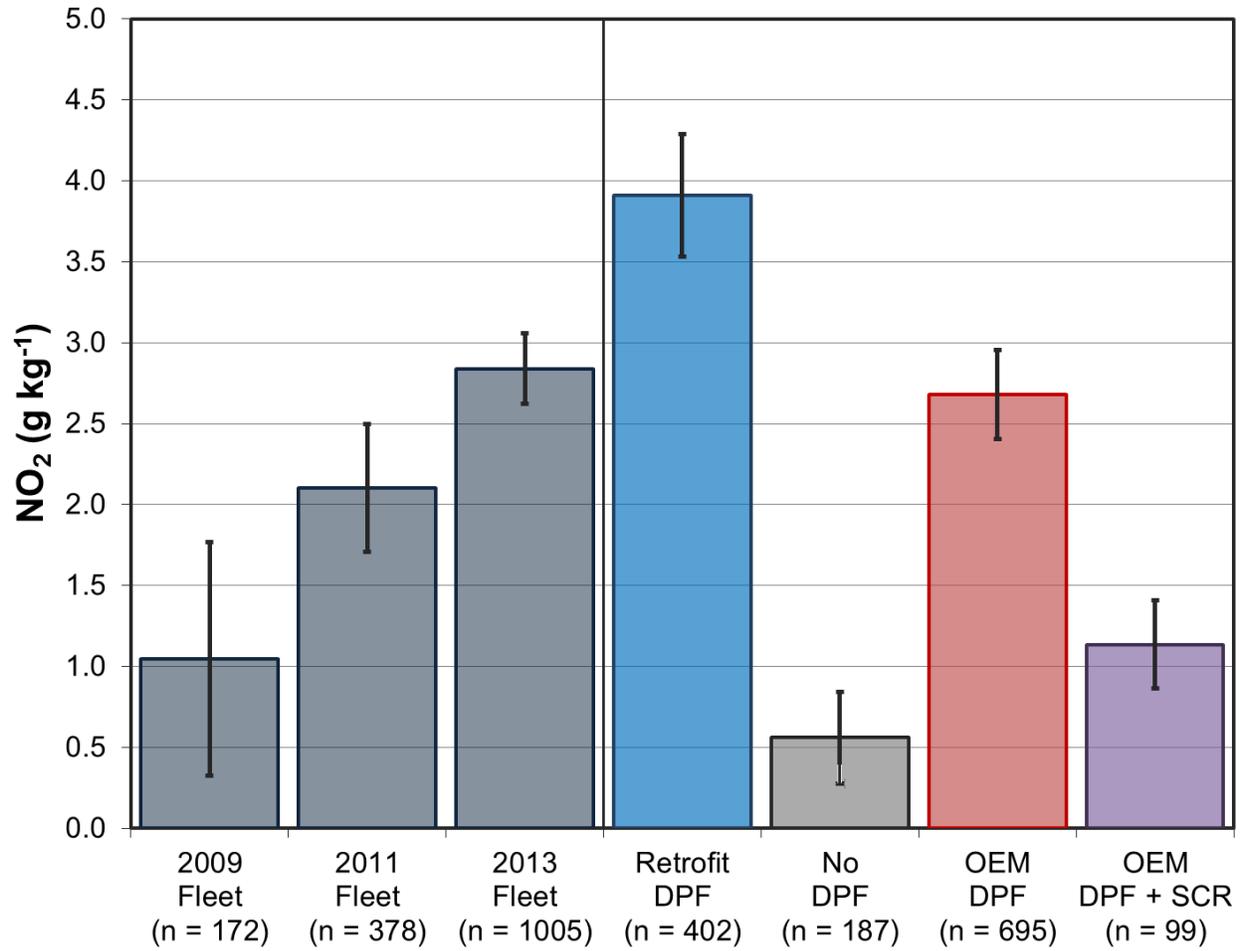
Figure 13. Fleet-average NO$_2$ emission factors by sampling year (bars on left) and average emission factors by truck category (bars on right; includes 2011 and 2013 data only). The number of analyzed trucks is indicated in parentheses on the category axis and the error bars reflect 95% confidence intervals about the mean.

Figure 14. Fleet-average NO_2/NO_x emission ratios by sampling year (bars on left) and average emission ratios by truck category (bars on right; includes 2011 and 2013 data only). The number of analyzed trucks is indicated in parentheses on the category axis and the error bars reflect 95% confidence intervals about the mean.
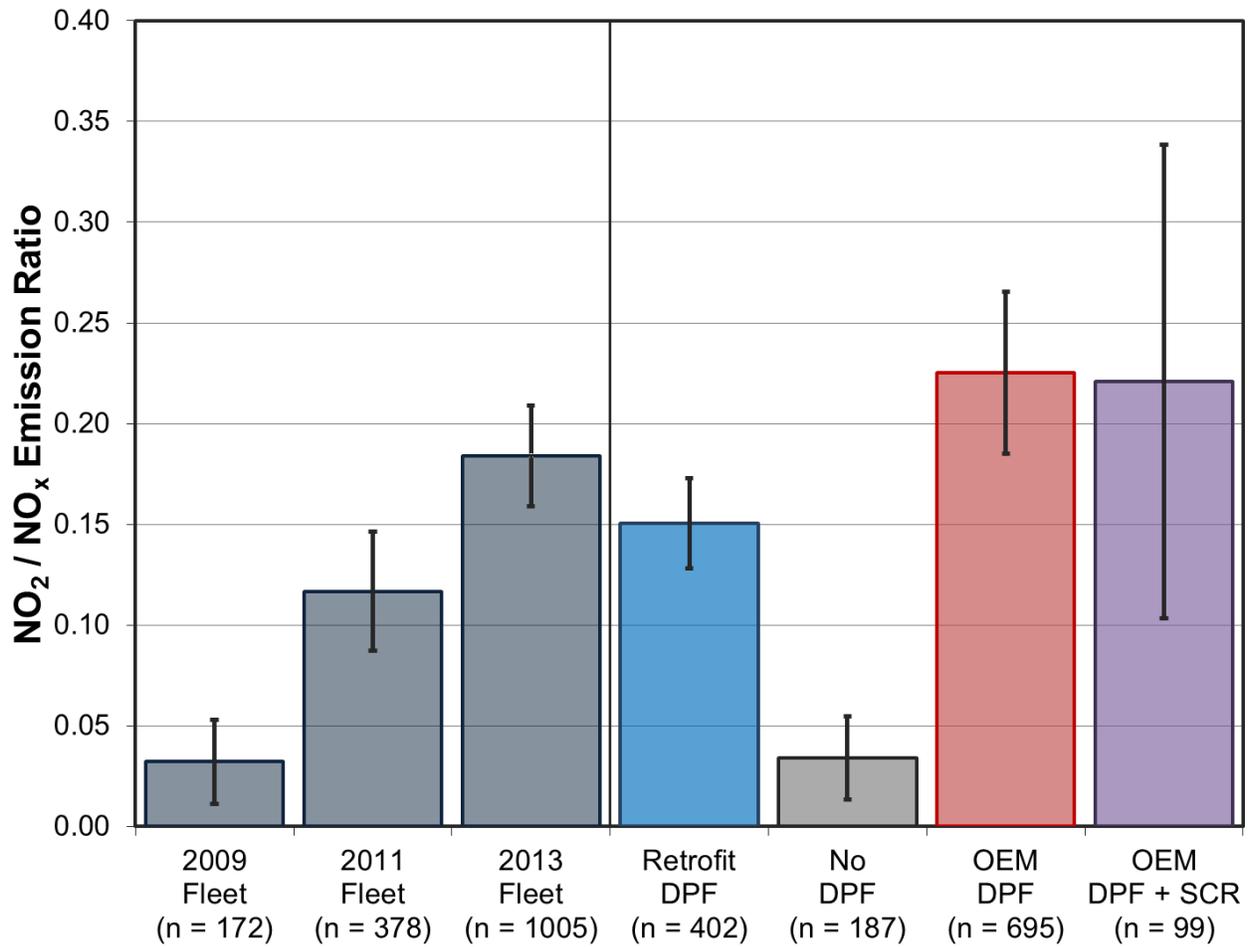
3.2.3 Ultrafine Particles. The Port fleet-average UFP number emission rate decreased $31 \pm 27\%$ between 2011 and 2013 (see Table 2 and Figure 15). It is possible that the reduction in UFP emissions was larger than the above calculation would suggest. While the UFP emission rate was not measured in 2009, the emission rate measured for trucks without DPFs, $(4.7 \pm 1.0) \times 10^{15}$ kg^{-1}, provides an estimate of the UFP emission rate from Port trucks in 2009. (Note that the BC emission factor for trucks without DPFs measured in 2011 was about the same as that for the 2009 fleet-average.) Using the higher baseline estimate for UFP, the estimated reduction in the particle number emission factor between 2009 to 2013 increases to $47 \pm 25\%$. Trucks with 2010 and newer engines had the lowest UFP emission rates, emitting one-third the number of UFP per kg of fuel burned compared to trucks without DPFs.

UFP emission factors presented above are based on particle numbers measured with an ultrafine water-based CPC (WCPC). Results were highly correlated ($R^2 = 0.9$) but approximately 1.5 times higher, on average, than UFP emission factors for the same trucks calculated from measurements made using a butanol-based CPC (Table A2 and Figure A3). The correlation was not as good when comparing values of UFP derived using ultrafine WCPC and FMPS measurement methods ($R^2 = 0.5$). On average, emission factors based on the WCPC were 1.8 times higher than those based on the FMPS. The lower values derived from FMPS measurements may be partly due to the fact that the ultrafine WCPC measures particles as small as 2.5 nm, whereas the lower size limit of the FMPS is 5.6 nm. This result agrees with the observation of Jeong and Evans (2009) that WCPC measurements of particle number concentration exceed FMPS measurements, but is counter to the finding reported by Zimmerman et al. (2014) that the FMPS overstates particle number concentrations during highly time-resolved measurements of high emitting vehicles.
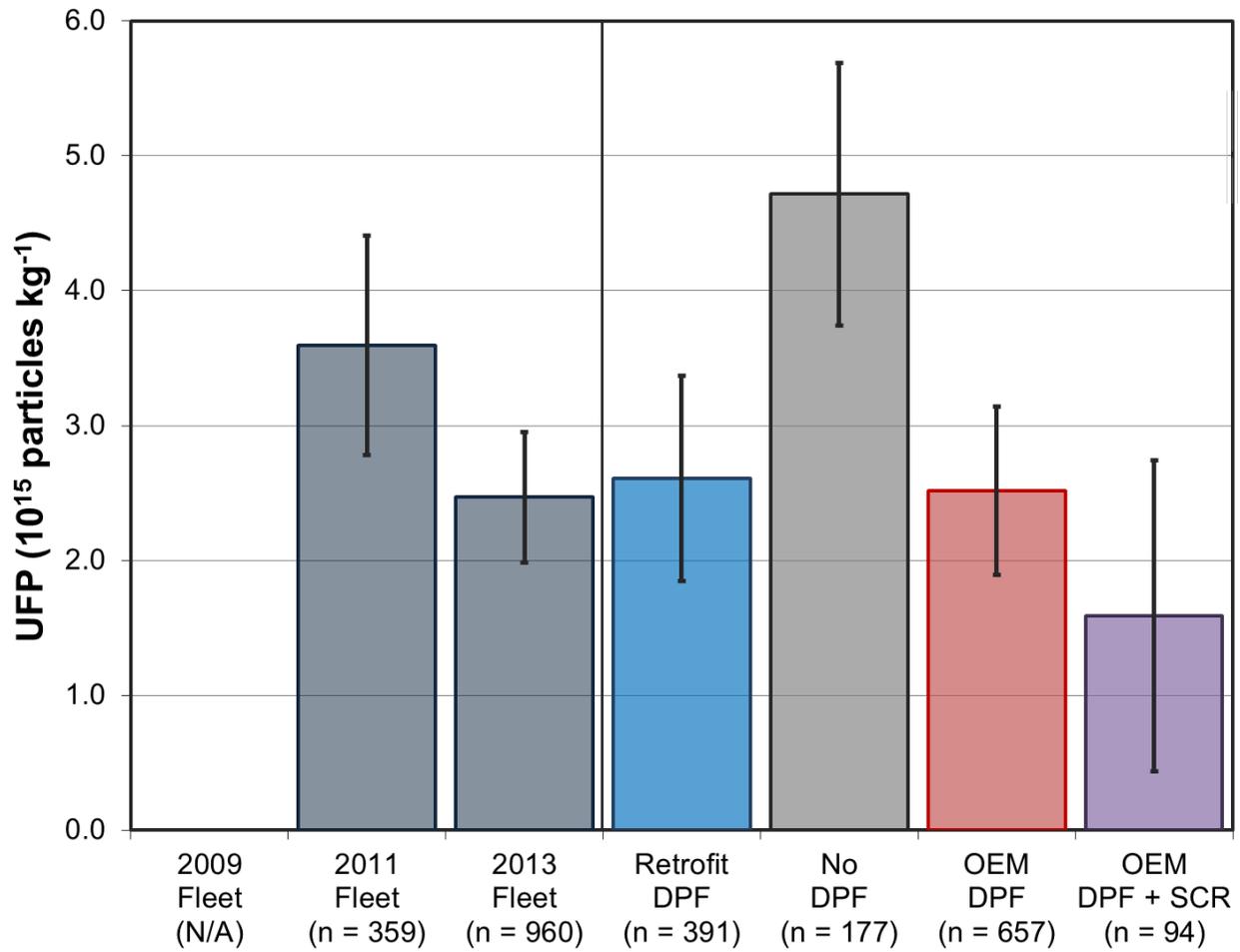
Figure 15. Fleet-average UFP number emission factors by sampling year (bars on left) and average number emission factors by truck category (bars on right; includes 2011 and 2013 data only). The number of analyzed trucks is indicated in parentheses on the category axis. Error bars reflect 95% confidence intervals about the mean.

Emission factor-weighted, normalized particle size distributions determined with Equation 2 for each truck were averaged across each particle diameter bin to determine a characteristic UFP emission rate distribution for each truck category, with results as shown in Figure 16. Distributions were weighted by both FMPS- and WCPC-derived UFP emission factors. The FMPS weighting provides a self-consistent normalization whereas the WCPC weighting may more accurately reflect the UFP emission rate. As shown in Figure 16, trucks emitted particles ranging in size between 6 nm (the lower bound of the FMPS) and 300 nm. DPFs appear to be most effective in reducing the emission rate of particles larger than approximately 15 nm: on average, the emission rates of such particles from trucks without filters exceeded those from DPF-equipped trucks by a factor of about two. These data further indicate that OEM DPF-equipped trucks (2007-2009 engines) had higher emissions of nucleation mode particles (~10 nm in diameter) than either older trucks retrofitted with DPFs or trucks with 2010 and newer engines.

Particle size distributions for all four sub-groups of trucks were characterized by an apparent peak in the smallest size bin of the FMPS (Figure 16). We present these data as measured by the FMPS, though it is unclear if they are truly indicative of a peak in the number concentration of particles smaller than the lower sizing limit of the FMPS.

Assuming spherical particles of uniform density (1 g cm^{-3}) across the entire particle size range, the particle number emission rate distributions were converted into mass emission rate distributions, as shown in Figure 17. The PM emission factor for each category of truck could ideally be estimated from the integrated area under each respective mass emission rate distribution. However, the sensitivity of the FMPS decreases by an order of magnitude in going from the smallest to largest size bins, resulting in inherent noise in the upper size bins that is

magnified when number concentrations are converted to mass concentrations (Figure 17).

Therefore, total PM mass emission rates corresponding to each truck category were determined

assuming lognormal distributions and doubling the area to the left of the assumed peak value of

each mass distribution (typically around 200 nm). The resulting estimated PM emission factor

for trucks without particle filters was approximately 3 times greater, on average, than that for

DPF-equipped trucks. Trucks with OEM DPF and SCR systems had the lowest PM mass

emission factor, followed by trucks with OEM DPFs and retrofit DPFs.

Figure 16a. The characteristic normalized particle number emission rate distribution for each engine control technology, as determined with the FMPS PN emission factor in Equation 2.

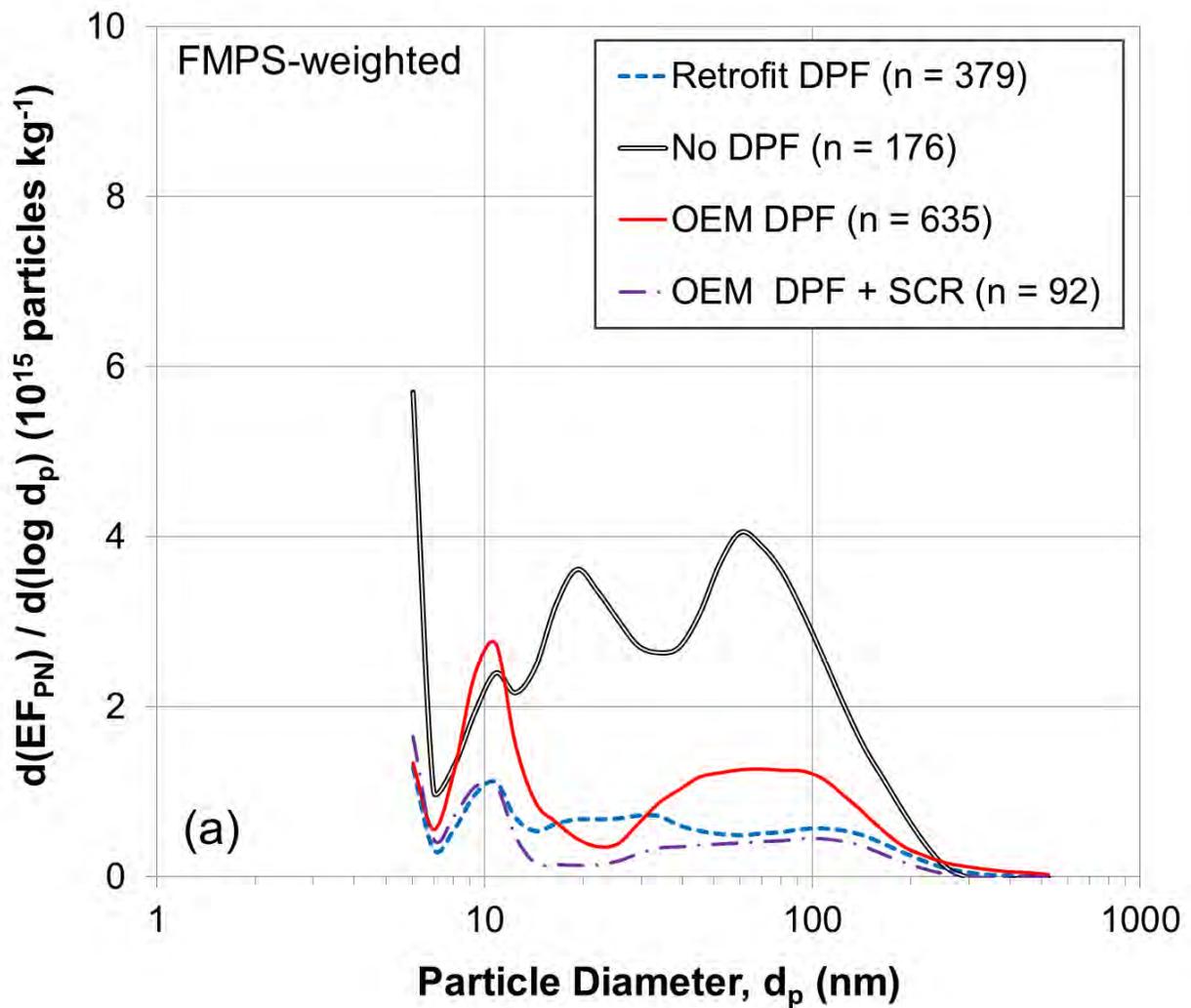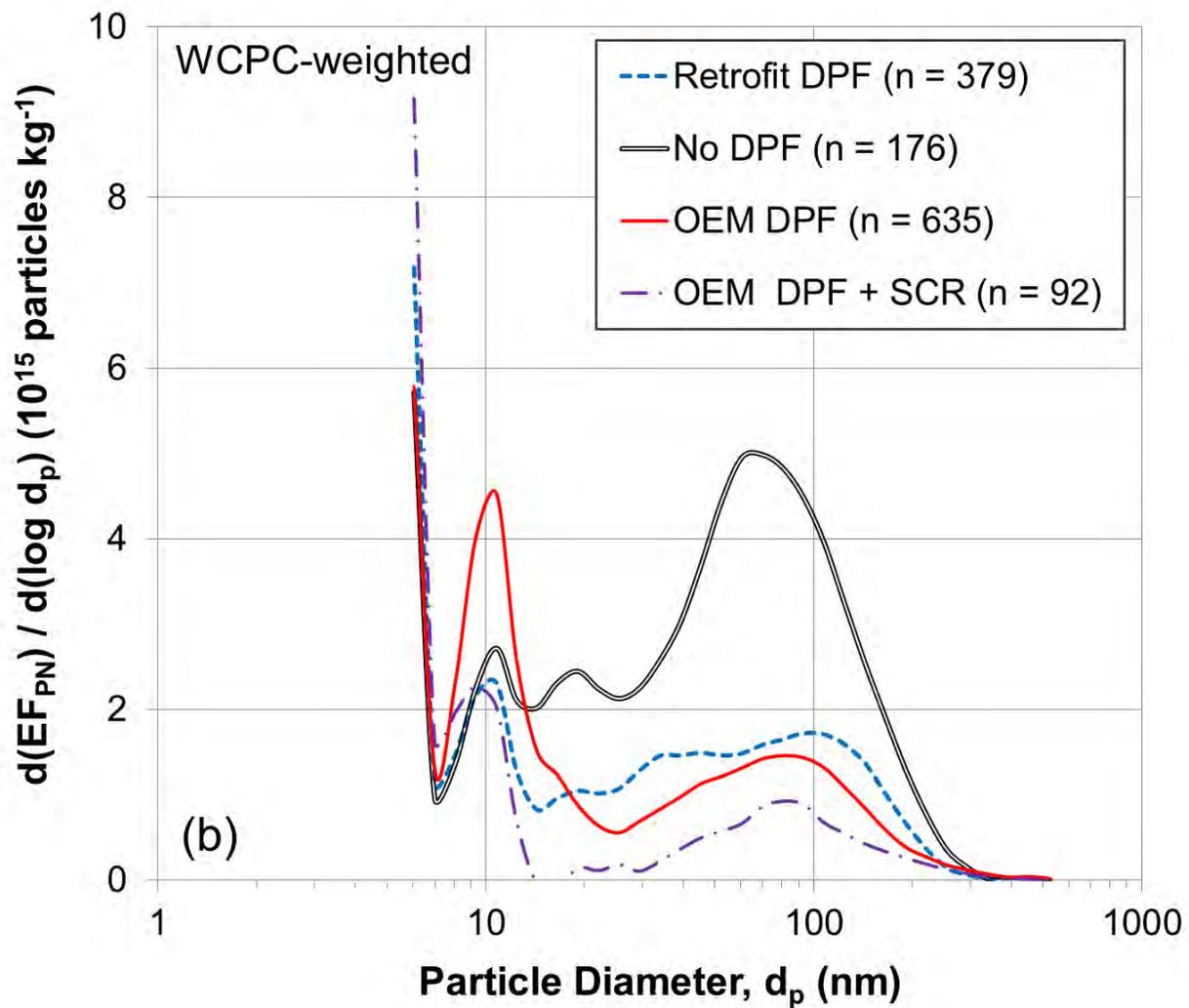**Figure 16b.** The characteristic normalized particle number emission rate distribution for each engine control technology, as determined with the WCPC PN emission factor in Equation 2.

Figure 17a. The characteristic normalized particle mass distributions for each engine control technology, assuming spherical particles of uniform density (1 g cm^{-3}) across the size range. The average PM emission estimated from each mass emission rate distribution is noted.

Comparing estimated PM emission factors with average BC emission factor for each truck category gives BC/PM emission ratios. Using the FMPS-derived PM emission factor, the BC/PM ratio for trucks without DPFs and those with retrofit DPFs is ~0.65, which agrees with previous studies (Ban-Weiss et al., 2008), while the ratio for OEM DPF-equipped trucks is smaller by a factor of about 2 (BC/PM = 0.34). The WCPC-derived ratios are lower: the BC/PM

ratio is 0.24 for retrofit DPF trucks, 0.40 for trucks without DPFs, 0.30 for OEM DPF-equipped

trucks, and 0.15 for trucks with OEM DPF and SCR systems. Even though the better sensitivity

of the WCPC to smaller particles may more accurately capture total UFP emissions, the self-

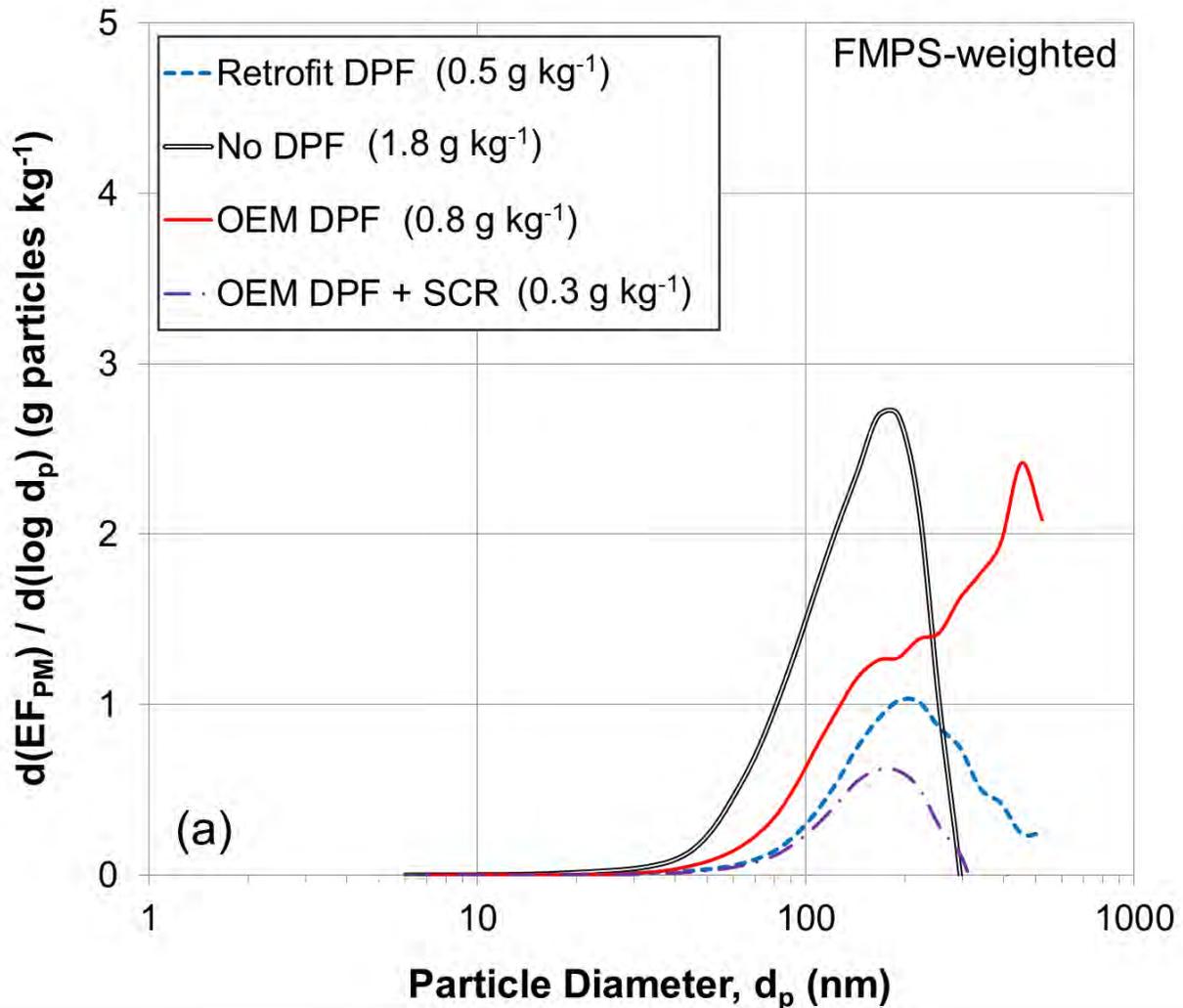consistency of the FMPS-weighting provides a better estimate of PM mass emissions.
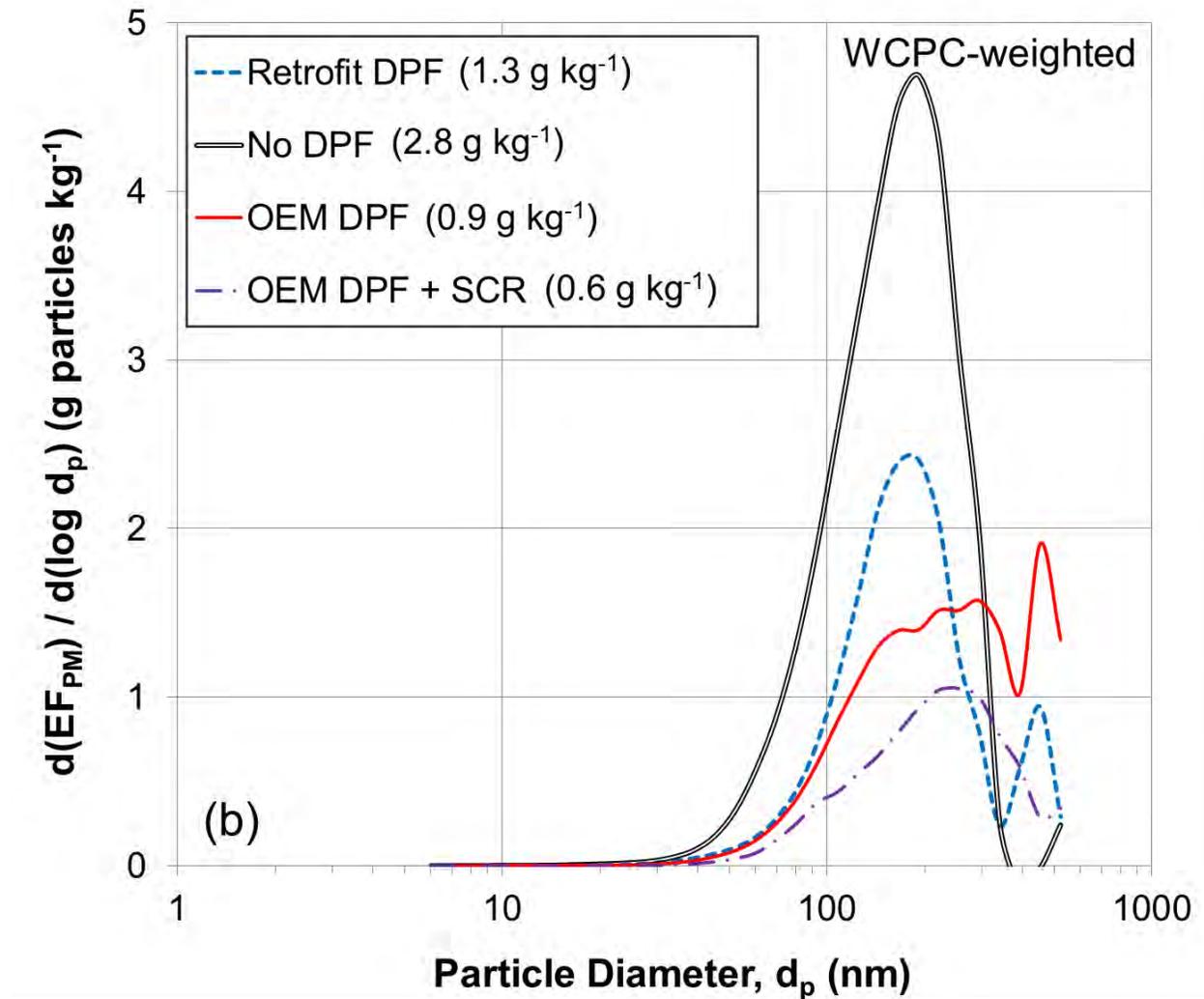
Figure 17b. The characteristic normalized particle mass distributions for each engine control technology, assuming spherical particles of uniform density (1 g cm^{-3}) across the size range. The average PM emission estimated from each mass emission rate distribution is noted.
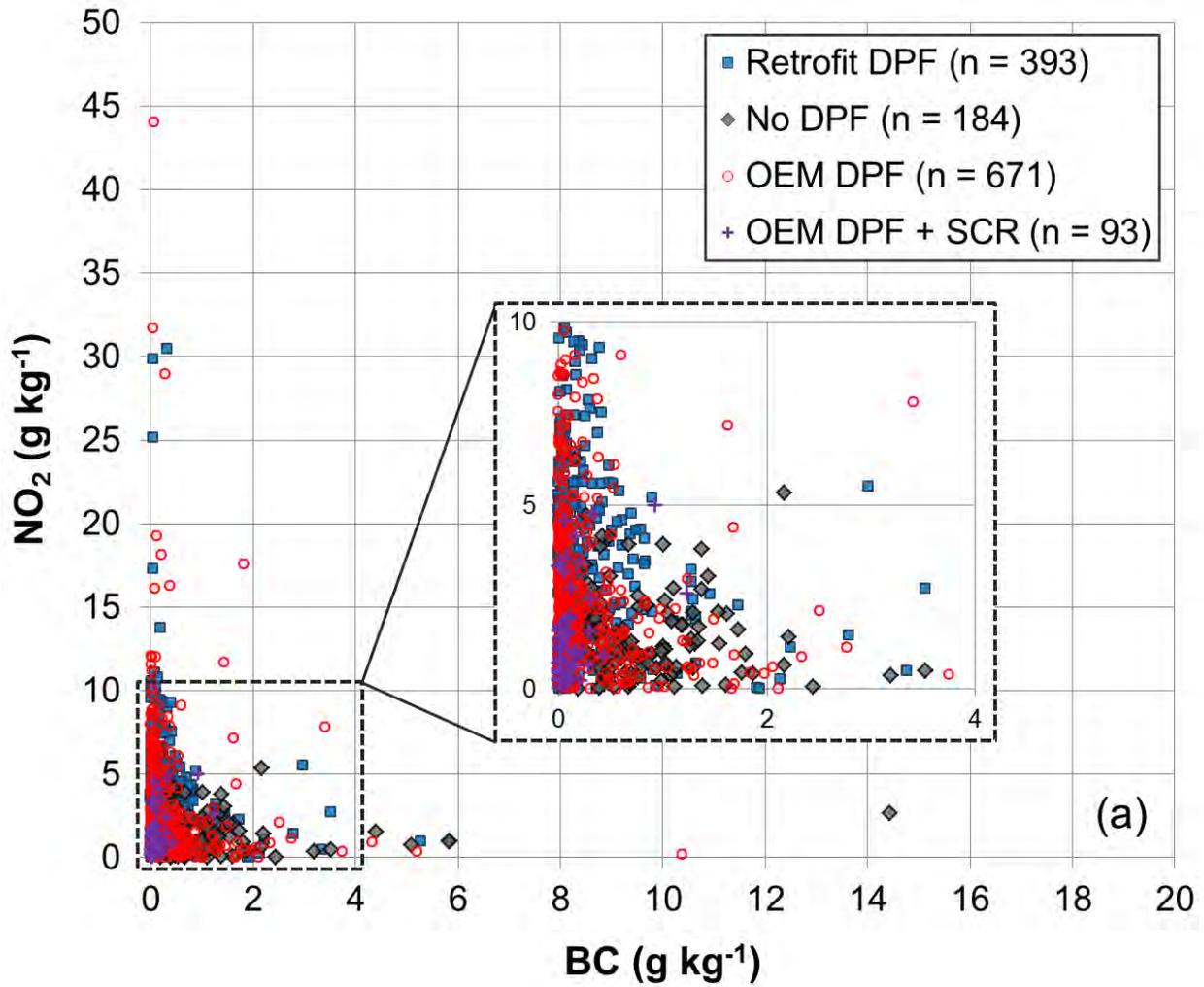
Figure 18a. Relationship between emissions of BC and NO₂ by truck category. The highest emitters of BC tend to have low emissions of NO₂ and UFP, and vice versa, across engine model years and installed control technologies.

Figure 18b. Relationship between emissions of BC and UFP by truck category. The highest emitters of BC tend to have low emissions of NO_2 and UFP, and vice versa, across engine model years and installed control technologies.

3.2.4 High Emitters. In general, the highest emitters of BC tended to have low emissions of NO_2 and UFP—and vice versa—regardless of the type or vi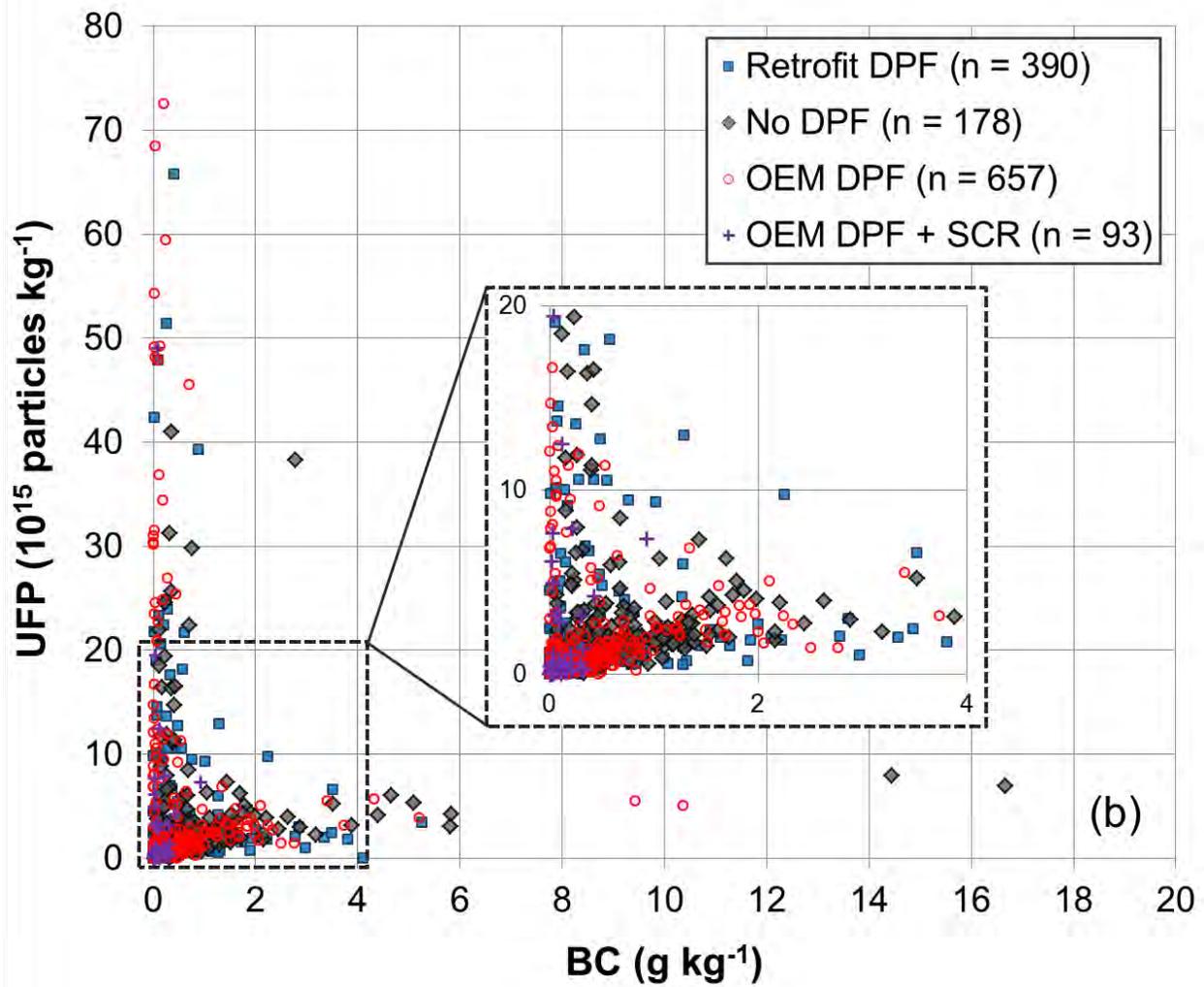ntage of installed emission control equipment (see Figure 18). An inverse relationship between BC and UFP has been observed previously (Ban-Weiss et al., 2009) and is consistent with the hypothesis that UFP formation is suppressed when large amounts of BC are present: the availability of existing (BC) particle surface area means that condensation onto existing particle surfaces is favored over nucleation to form new UFP (Kittelson, 1998). The relationship between BC and NO_2 can be explained by the reduction of BC mass with highly catalyzed DPFs, which would tend to increase NO_2. This inverse relationship is consistent with the BC and NO_2 emissions trends of trucks with retrofit DPFs shown in Figure 11.

3.3 Changes in Emission Factor Distributions. As fleet-average emission factors have decreased for Port drayage trucks, emission factor distributions have become increasingly skewed, such that a small fraction of the fleet is responsible for an increasing fraction of total emissions, as shown in Figures 19 and 20. Particle-related emission factor distributions are more skewed than for nitrogen oxides. In 2013, the highest emitting 10% of trucks were responsible for 65% of total BC and 80% of total UFP, compared to 32 and 37% of total NO_x and NO_2 emissions, respectively. NO_x emission factor distributions are also starting to show skewness, especially as the number of engines equipped with SCR has begun to increase.

An alternative version of these distributions is presented in Figure 21, which sorts the emission factors within each truck category from lowest to highest for both BC and NO_x. Figure 21a shows that the top 5% of BC emitters within the retrofit and OEM DPF truck categories exceeded the 2009 fleet average of 1.15 g BC kg^{-1}, while no truck with 2010 and newer engines had emissions above this baseline value. As in Figures 19b and 20, the distributions of NO_x

emission factors shown in Figure 21b are less skewed, and the trend between truck categories

follows the corresponding engine model years. Only one high emitting truck equipped with an

OEM DPF and SCR system exceeded the 2009 fleet-average value of 33 g NO_x kg^{-1}.

Emission factor distributions are shown in Figure 22, separately for each engine model

year, as measured in 2011 and 2013. Trucks equipped with OEM DPFs and SCR systems not

only had the lowest BC and NO_x emission factors, but also showed the least amount of

variability in measured emission rates. Variability in emission factors for many individual engine

model years increased in the 2013 field sampling campaign relative to 2011, suggesting

degradation or failure of some installed emission control systems over time. Our analysis

identified two OEM DPF trucks as the two highest BC emitters in 2013 (Figure 22a). These

trucks had emission factors of ~10 g BC kg^{-1} when sampled in 2013, which is significantly

higher than the fleet-average of 0.28 ± 0.05 g kg^{-1} and is approximately twice the level observed

for the next highest emitting trucks. In this case, the two high emitting OEM DPF trucks

represented 0.2% of the fleet based on number, but are responsible for 7% of total BC emissions

(Figure 20). This analysis also shows that trucks with 2007-2009 engines frequently emit NO_x at

levels that are similar to what is observed from older trucks, even though the average emission

rate for older trucks is approximately twice as great (Figure 12). As such, even though average

emissions of BC and NO_x have decreased, some newer trucks were observed to emit BC and

NO_x at high levels.

3.4 Emissions Variability. In the results presented above, the 95% confidence intervals provide

a measure of emission factor variability observed in this study. Many trucks passed by our

sampling location two or more times, and this replicate sampling provides a further opportunity

to evaluate emissions variability for individual trucks. Repeat measurements of 207 individual

trucks are plotted in Figure 23 against the average emission factor for the corresponding truck. Scatter about the indicated 1:1 line is due to emission factor variability for individual trucks sampled repeatedly. R^2 values for NO_x and BC are both relatively high, whereas corresponding values for NO_2 and UFP are lower. This analysis indicates that NO_x and BC emission factors are less variable than NO_2 and UFP. Measurement uncertainties may contribute to variability in NO_2 emission factors due to the need to calculate NO_2 by difference between NO_x and NO. Variability in UFP emission factors is much greater than observed during laboratory evaluation of the plume capture sampling method and is most likely due to real variability in UFP emission rates from individual trucks. Fleet-average results for UFP are still meaningful, but the variability in UFP emissions may confound snapshot-type measurements to characterize emissions of this pollutant from individual trucks.

Figure 19a. Cumulative emissions for the fleet with trucks ranked from highest to lowest for BC emissions over the implementation of the Drayage Truck Regulation.

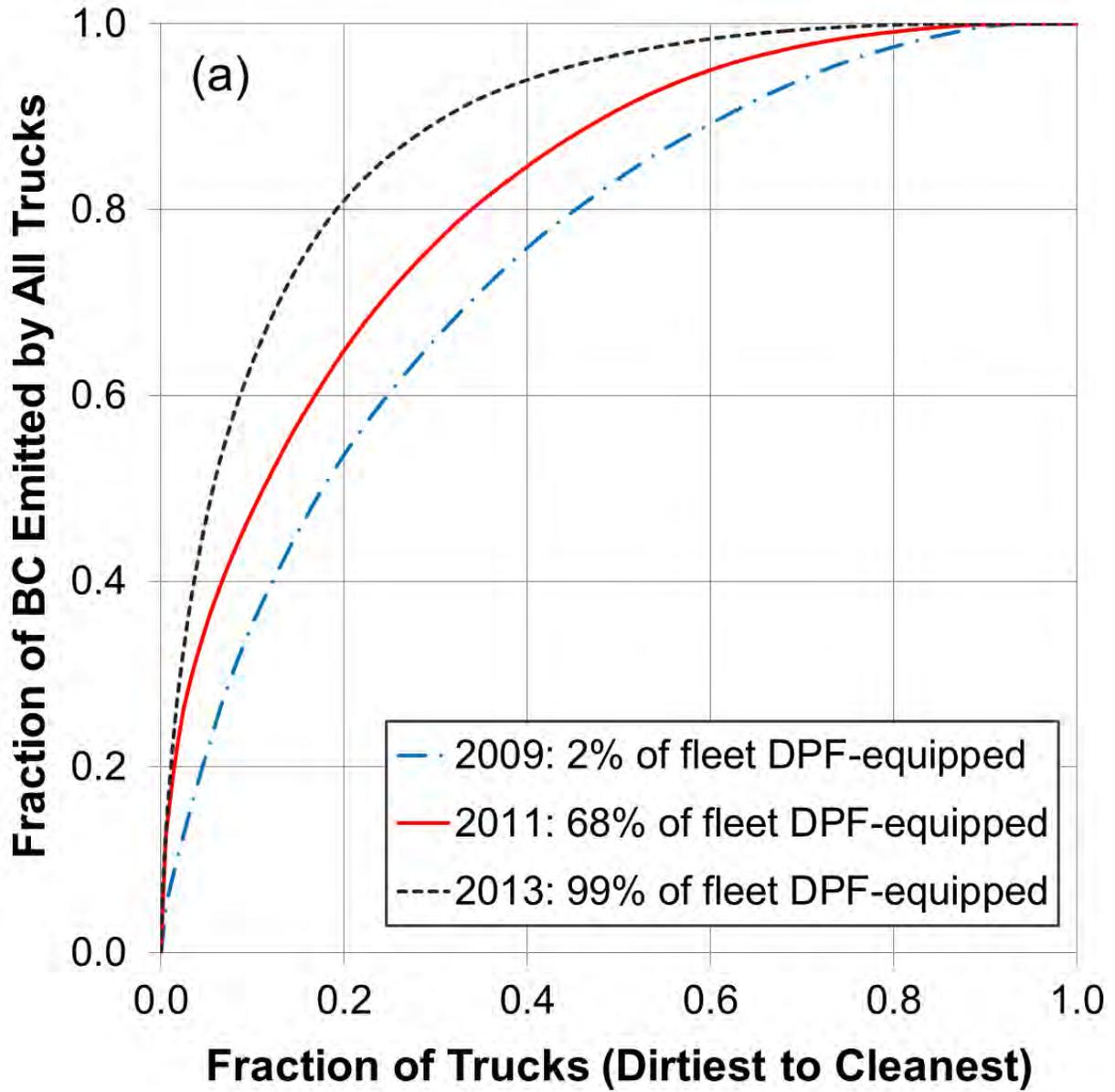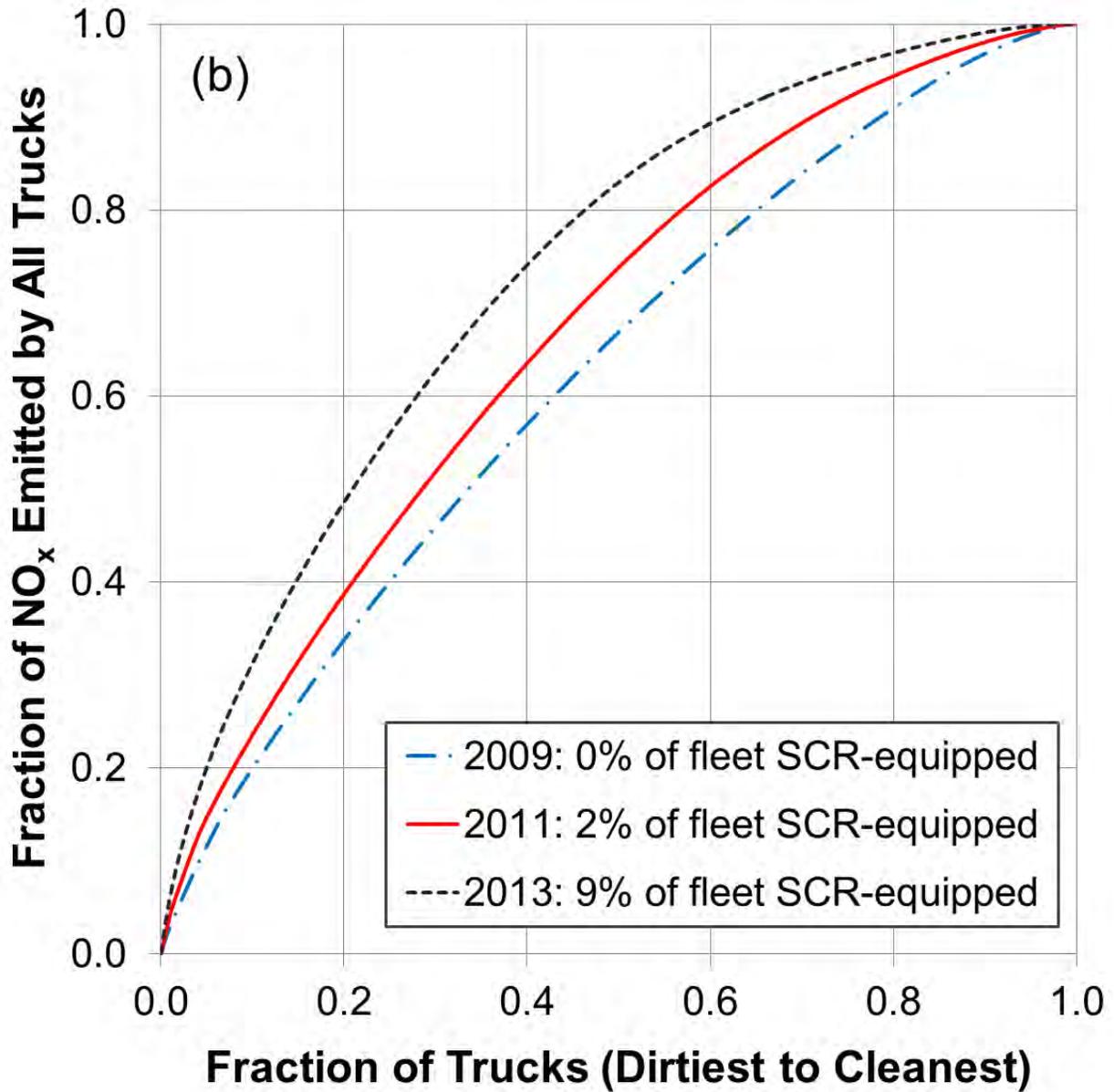**Figure 19b.** Cumulative emissions for the fleet with trucks ranked from highest to lowest for NO$_x$ emissions over the implementation of the Drayage Truck Regulation.
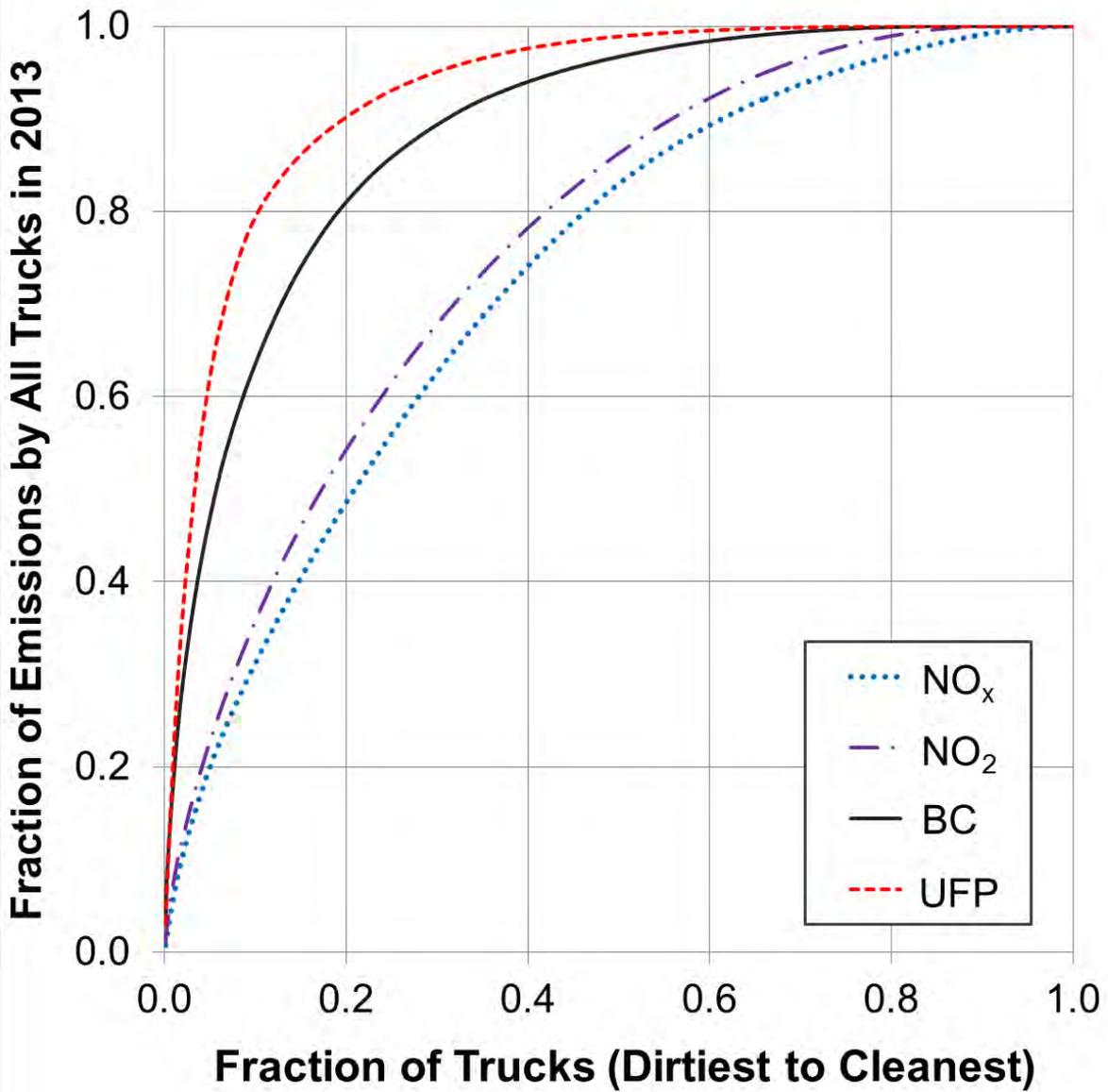
Figure 20. Cumulative pollutant emissions for the 2013 truck fleet after the full implementation of Phase 1 of the Drayage Truck Regulation, with trucks ranked from highest to lowest in terms of emission factors for each of the pollutants.
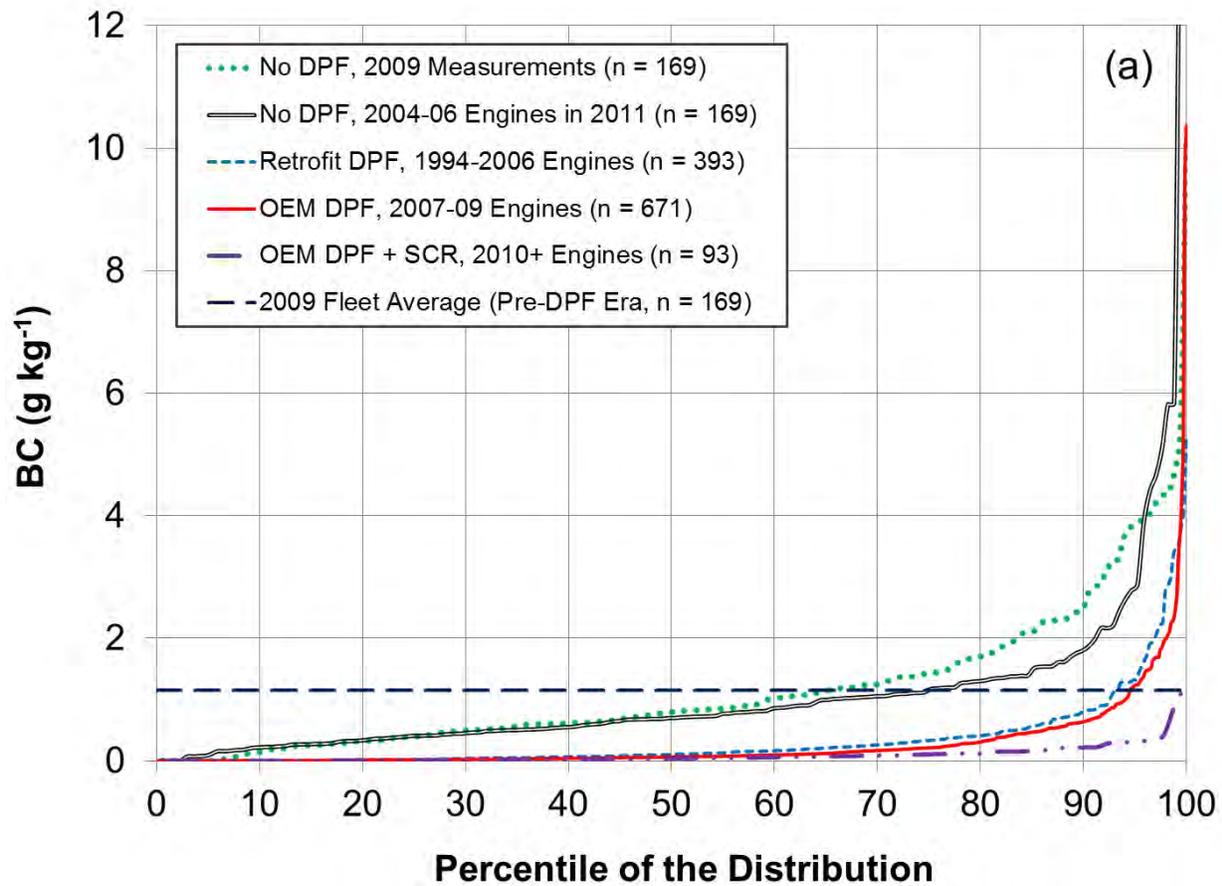
Figure 21a. Distribution of BC emissions as a function of sampling period and truck category, with trucks ranked from cleanest to highest-emitting (moving from left to right). The fleet-average emission factor measured in 2009 is indicated by the horizontal dashed line.
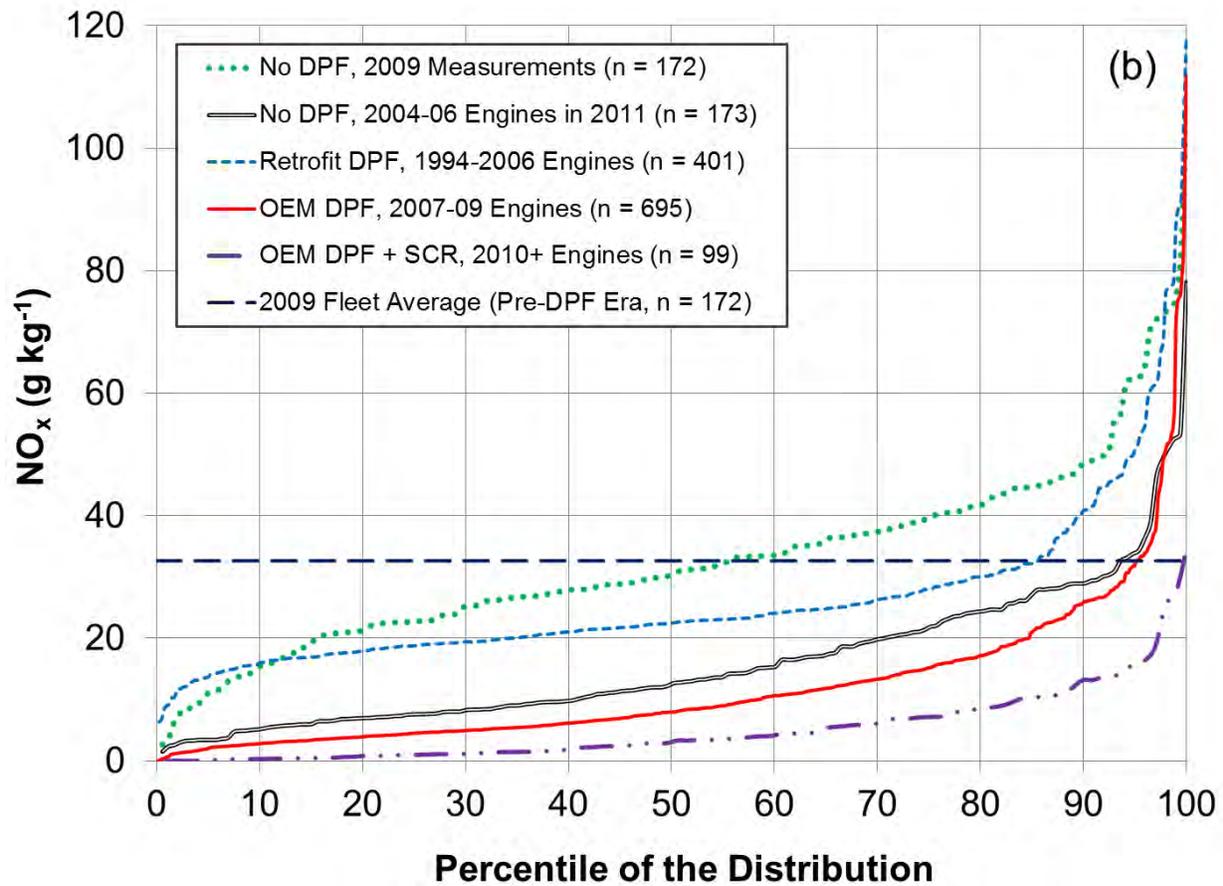
Figure 21b. Distribution of NO_x emissions as a function of sampling period and truck category, with trucks ranked from cleanest to highest-emitting. The fleet-average emission factor measured in 2009 is indicated by the horizontal dashed line.
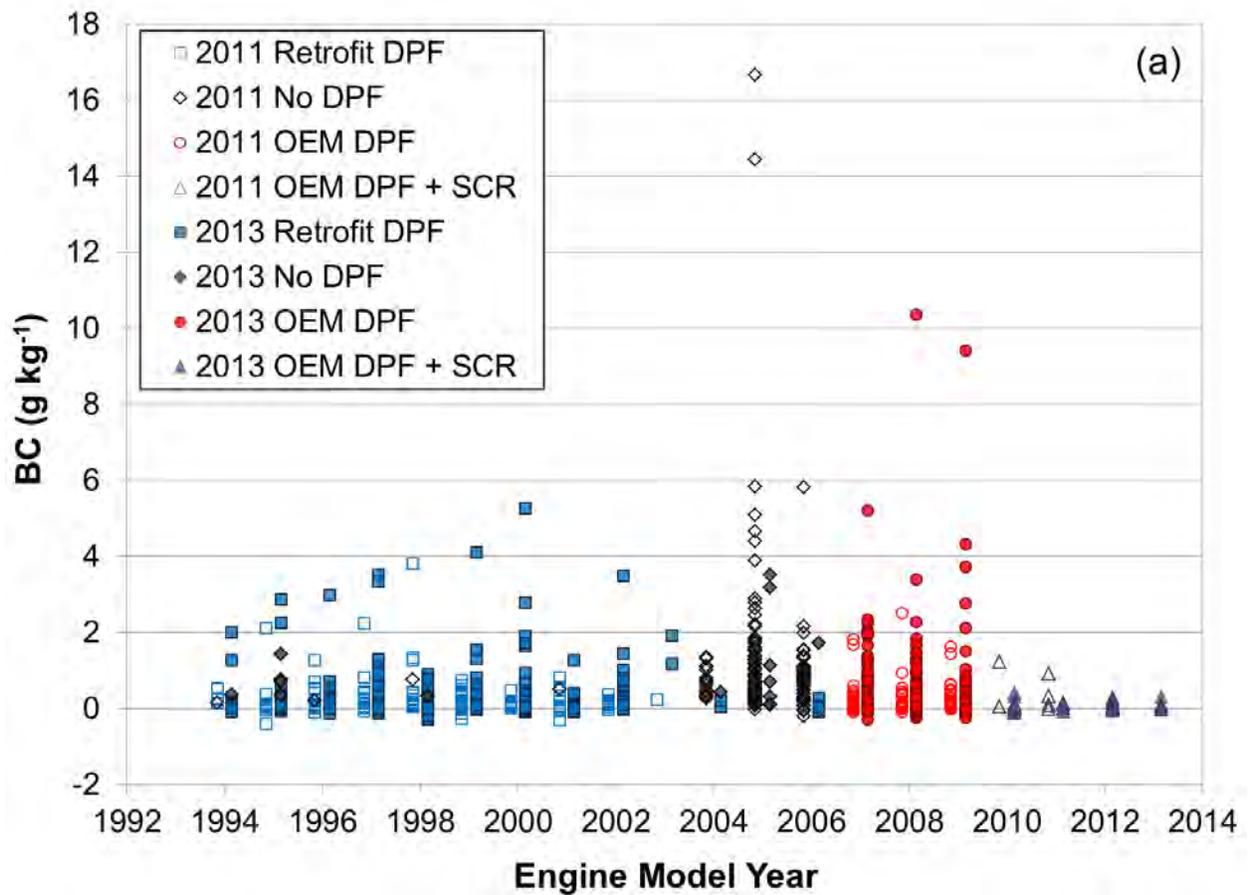
Figure 22a. Distribution of BC emission factors across engine model years for each individually measured truck. The truck categories as measured in 2011 and 2013 are shown separately and indicated in the legend.
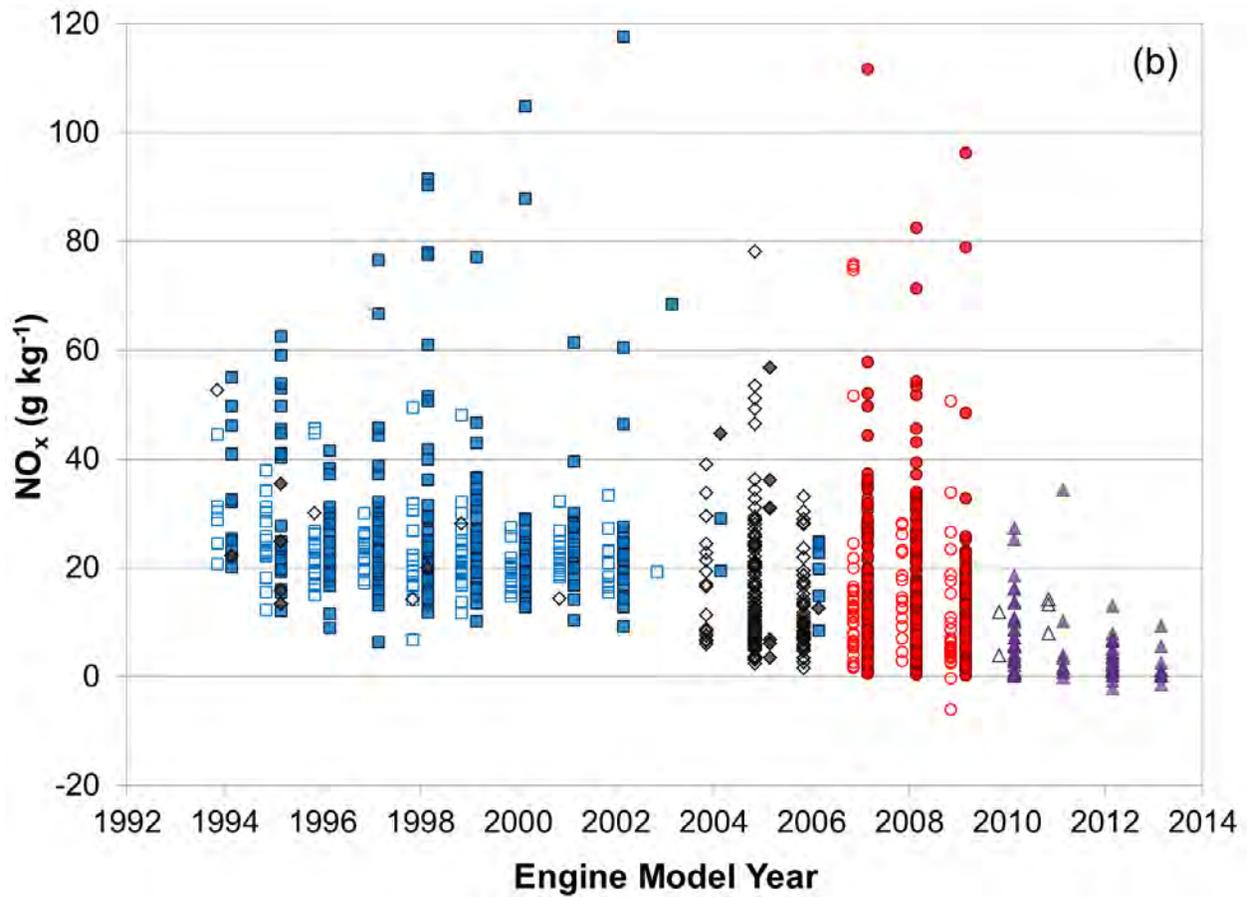
Figure 22b. Distribution of NO$_x$ emission factors across engine model years for each individually measured truck. The truck categories as measured in 2011 and 2013 are shown separately and indicated in the legend above.
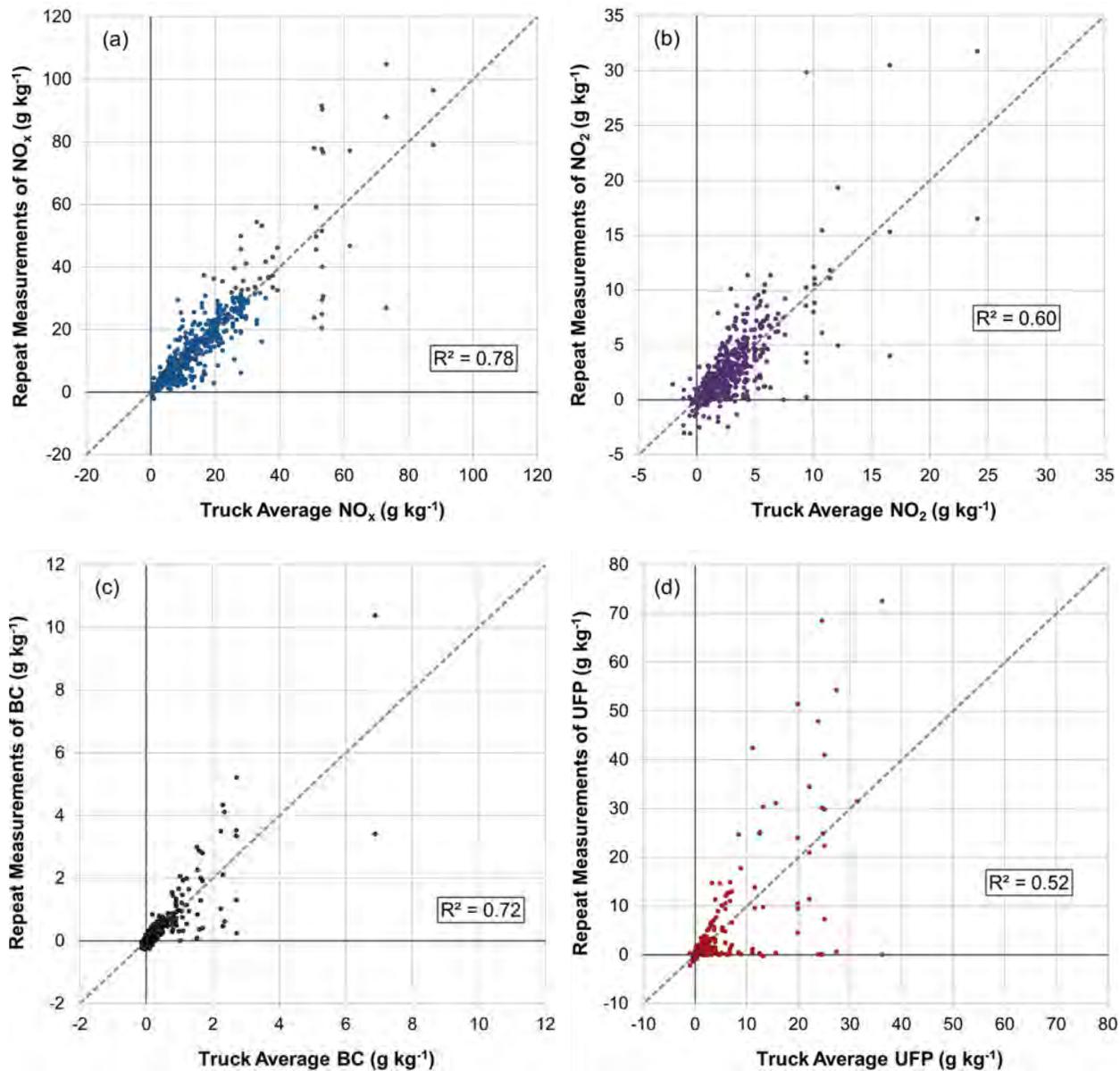
Figure 23. Results of repeat measurements for 207 individual trucks; each individual measurement is plotted on the vertical axis against the corresponding average emission factor for each truck on the horizontal axis. The dashed gray line indicates a 1:1 line and the repeatability of the measurement is indicated by the scatter about this line.

4. DISCUSSION

Measured reductions in BC and NO_x emission factors of 76 ± 22 and $53 \pm 8\%$, respectively, were found in this study following the complete implementation of the first phase of the Drayage Truck Regulation. These reductions are larger than the corresponding initial reductions of 54 ± 11 and $41 \pm 5\%$ reported by Dallmann et al. (2011) based on measurements made during 2009 and 2010. The larger reductions reported here correspond to more widespread (near universal) use of DPFs and further modernization of the truck fleet at the Port of Oakland. Emission reductions observed at the Port over ~3.5 years, from November 2009 to March 2013, are nearly double the 39 and 30% reductions in BC and NO_x emission factors measured for trucks at the Caldecott Tunnel over a period of 9 years, from 1997 to 2006 (Ban-Weiss et al., 2008). The Caldecott Tunnel truck fleet was not subject to accelerated retrofit/replacement requirements during the time span studied by Ban-Weiss et al. Thus, it is clear that changes in emission factors from the Port truck fleet have taken place on an accelerated schedule, and emission reductions have progressed well beyond what would be expected due to normal fleet turnover alone.

The introduction of newer engines and adoption of after-treatment exhaust controls such as DPFs and SCR systems have reduced fleet emissions of NO_x, BC, and UFP, but have led to increased emissions of NO_2 and an increase in the NO_2/NO_x emission ratio from 0.03 to 0.18. This increase in NO_2 is due to the passive regeneration scheme that is part of many diesel particle filter systems, in which NO present in the exhaust is deliberately oxidized to NO_2.

There are good prospects that increases in NO_2 emissions reported here will be mitigated by the eventual replacement of pre-2007 engines. Trucks equipped with 2010 and newer engines

emitted $71 \pm 15\%$ less NO_2 and $72 \pm 27\%$ less BC than trucks with retrofit DPFs, and $58 \pm 17\%$ less NO_2 and $67 \pm 29\%$ less BC than trucks with 2007-2009 engines. Thus, as the Port truck fleet continues to shift towards trucks equipped with both OEM DPF and SCR systems, the reduction in BC emission factors could rise to the 90% range on a fleet-average basis, with little if any increase in NO_2 emissions.

The emission factor-weighted particle number size distribution for trucks without DPFs were trimodal and included broad peaks around 10, 20, and 80 nm, with the peak at 80 nm dominating the distribution. DPFs most effectively removed particles larger than 15 nm. Thus, in addition to reducing the UFP number emission factor, DPFs alter the size distribution of emitted particles: the 10 nm peak becomes dominant for trucks with DPFs. DPFs on newer (2007+) trucks include active filter regeneration, and emissions of 5-10 nm particles are known to increase during active regeneration events (Dwyer et al., 2010). The extent to which active regeneration events may have affected UFP emission rates in our study is unknown. Despite previously expressed concerns that use of DPFs could lead to increases in total UFP emissions, all categories of DPF systems examined in this study showed reductions rather than increases in total particle number as well as PM mass emission rates, relative to a comparison group of trucks measured in this study in 2011 with 2004-2006 engines and no DPF system installed.

Differences in operating conditions can influence pollutant emission rates. This variability can be studied in the laboratory using engine or chassis dynamometer tests over different driving cycles, but these tests are costly and are generally limited to a small sample size. In-use emission evaluations as per the current study, on the other hand, can assess the emission performance of thousands of trucks under real-world conditions. These measurements provide emissions snapshots, though, and do not fully capture the emissions variability for those

trucks. In the current study, hundreds of trucks drove by the sampling location multiple times, and we used these repeat measurements to characterize emissions variability. Overall, UFP emissions were more variable than emissions of BC or NO_x, as indicated by the wider confidence intervals about the fleet, truck category, and installer/manufacturer averages (Figures 10-15 and A2, Table 3) and by the greater scatter about the 1:1 trendline in the analysis of repeat measurements for individual trucks (Figure 23).

As the drayage truck fleet has become cleaner on a fleet-average basis, emission factor distributions have become increasingly skewed. A smaller portion of the fleet is now responsible for a larger fraction of total pollutant emissions. Identification and follow-up for this small fraction of high-emitting trucks within the fleet could become an effective method of further reducing fleet emissions in the future. For instance, if the top 10% of BC emitters from the 2013 fleet when ranked from dirtiest to cleanest (see Figure 20) were removed, then fleet-average BC emissions would decrease from 0.28 ± 0.05 to 0.11 ± 0.01 g kg^{-1}. The trucks that comprise this high-emitting sub-fleet have an average BC emission factor of 1.80 ± 0.10 g kg^{-1}, which was approximately six times the fleet average value in 2013. A number of the trucks included in this top 10% were measured multiple times at the Port, and 64% of the repeat measurements were also included in the top 10% of BC emissions. This suggests that the high-emitter problem for BC is more often chronic rather than intermittent in nature. If all repeat measurements of BC by trucks classified as in the dirtiest 10% of the 2013 fleet are included, the average BC emission factor remains much higher than the fleet average at 1.36 ± 0.10 g kg^{-1}. Elimination of these high-emitting trucks (e.g., through repair or replacement) would further increase the BC reduction measured in this study between 2009 to 2011from $76 \pm 21\%$ (shown in Figure 10) to $90 \pm 22\%$.

In conclusion, widespread application of DPF systems has substantially reduced particle emissions from heavy-duty diesel trucks near the Port of Oakland. Related increases in emissions of primary NO_2 will be offset using SCR systems. Total UFP emissions have decreased as a result of using DPF systems to control exhaust PM, with beneficial effects most clearly apparent at particle sizes of 15 nm and larger.

5. RECOMMENDATIONS

Since the completion of this study in 2013, Phase 2 of the Drayage Truck Regulation has required the replacement of nearly one-third of the Port of Oakland truck fleet (see Table 1 and Figure 9). Specifically, Phase 2 required replacement of 2006 and older model year engines, all of which had been previously retrofitted with DPFs, with 2007 or newer engines. Emission factors of NO_x and NO_2 are lower for 2007 and newer engines than retrofitted pre-2007 engines, and these emissions as well as BC are lowest for 2010 and newer engines (see Figures 10, 12, and 13). Most notably, NO_2 emissions for 2010 and newer engines are comparable to the pre-regulation fleet measured in 2009. Thus, the additional pollutant reductions that have occurred since 2013, and especially the mitigation of increased NO_2 emissions, depend on the extent to which pre-2007 engines were replaced with either 2007-2009 or 2010 and newer engines.

Additional measurements near the Port of Oakland are recommended to verify the BC, NO_x, and NO_2 emissions reductions associated with this significant further modernization of the Port truck fleet. In addition to measuring the effects of the complete regulation (Phases 1 and 2) on fleet composition and emissions, the supplemental measurements would enable further evaluation of emissions from SCR-equipped trucks as more 2010 and newer engines enter the fleet. Additional measurements would also provide an opportunity to assess the potential impacts and prevalence of DPF system failures or degradation as a function of increasing truck/engine age. Additional species that may be affected by use of SCR, including nitrous oxide (N_2O) and ammonia (NH_3), which were not included in the current study, could also be measured. This added data near the Port would facilitate a more robust evaluation of emissions from DPF- and SCR-equipped trucks near the Port of Oakland.

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APPENDIX

Table A1. Instrumentation used to measure truck exhaust emissions in this study.

Parameter	Instrument	Time Resolution
CO_2 concentration	Nondispersive infrared gas analyzer (LI-COR LI-7000)	2 Hz
NO, NO_x concentrations (the difference of which gives NO_2 concentration)	Two chemiluminescent analyzers (ECO Physics CLD-64)	2 Hz
BC concentration	Aethalometer (Magee Scientific AE16)	1 Hz
BC concentration	Photoacoustic absorption spectrometer (PAS) with reciprocal nephelometer (custom)	1 Hz
Particle number concentration*	Ultrafine, water-based condensation particle counter (TSI 3788); lower size limit of 2.5 nm and noted as ultrafine WCPC in above text and as PUW in Table A2 and Figure A3	2 Hz
Particle number concentration	Ultrafine, butanol-based condensation particle counter (TSI 3776); lower size limit of 2.5 nm and noted as ultrafine BCPC in above text and as PUB in Table A2 and Figure A3	10 Hz
Particle number concentration, dilution factor	Two general purpose, high concentration water-based condensation particle counters (TSI 3783); lower size limit of 7 nm and noted as high concentration WCPC in above text and as PSW or PHW in Table A2 and Figure A3	2 Hz
Particle number concentration and size distribution	Fast mobility particle sizer (TSI 3091); lower size limit of 5.6 nm and noted as FMPS in above text and in Table A2 and Figure A3	1 Hz

*The reported UFP emission factors in Table 2 and Figures 15, 16b, 17b, 18b, 20, and 23d are derived from this CPC.

Table A2. Summary of ultrafine particle (UFP) emission factors measured using four different CPCs and the FMPS, for drayage trucks operating at the Port of Oakland in 2011 and 2013. Emission factor units are number of particles emitted per kg of diesel fuel burned. Uncertainty bounds provide 95% confidence intervals about the mean. See Table A1 for more detailed descriptions of the individual CPCs.

Fleet or Truck Category	PUW (10^{15} particles kg^{-1})	PUB (10^{15} particles kg^{-1})	PSW (10^{15} particles kg^{-1})	PHW (10^{15} particles kg^{-1})	FMPS (10^{15} particles kg^{-1})
2011 Fleet (54% DPF, 2% SCR)	3.6 ± 0.8	2.5 ± 0.5	3.8 ± 2.5	2.5 ± 0.9	2.9 ± 0.9
2013 Fleet (99% DPF, 9% SCR)	2.5 ± 0.5	2.0 ± 0.4	1.8 ± 0.3	1.9 ± 0.5	1.3 ± 0.3
Retrofit DPF (1994-2006)	2.6 ± 0.8	1.8 ± 0.5	2.2 ± 0.9	1.6 ± 0.6	1.0 ± 0.4
No DPF (2004-2006)	4.7 ± 1.0	3.9 ± 0.8	3.9 ± 1.9	2.9 ± 0.8	4.3 ± 1.2
OEM DPF (2007-2009)	2.5 ± 0.6	2.0 ± 0.5	1.9 ± 0.6	2.4 ± 0.8	1.7 ± 0.5
OEM DPF + SCR (2010-2013)	1.6 ± 1.2	1.0 ± 1.0	1.1 ± 0.6	0.7 ± 0.4	0.7 ± 0.5

Figure A1. Map of the Port of Oakland, Union Pacific (UP) and BNSF rail yards, nearby community of West Oakland, and sampling location (noted by a star).
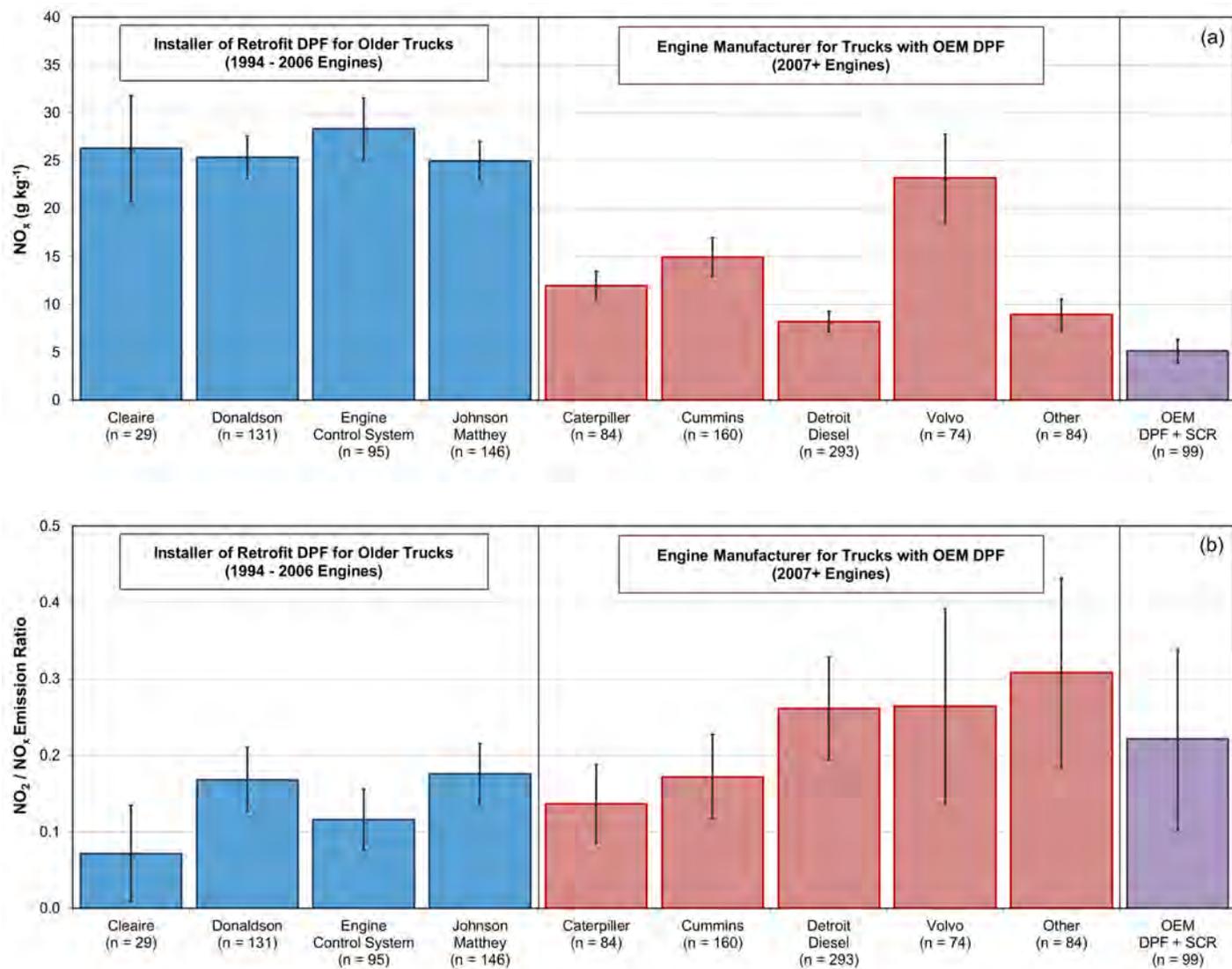
Figure A2. Average (a) NO_x emission factors and (b) NO_2/NO_x emission ratios, separated by major suppliers of DPF retrofits and OEM engines, with 95% confidence intervals indicated.
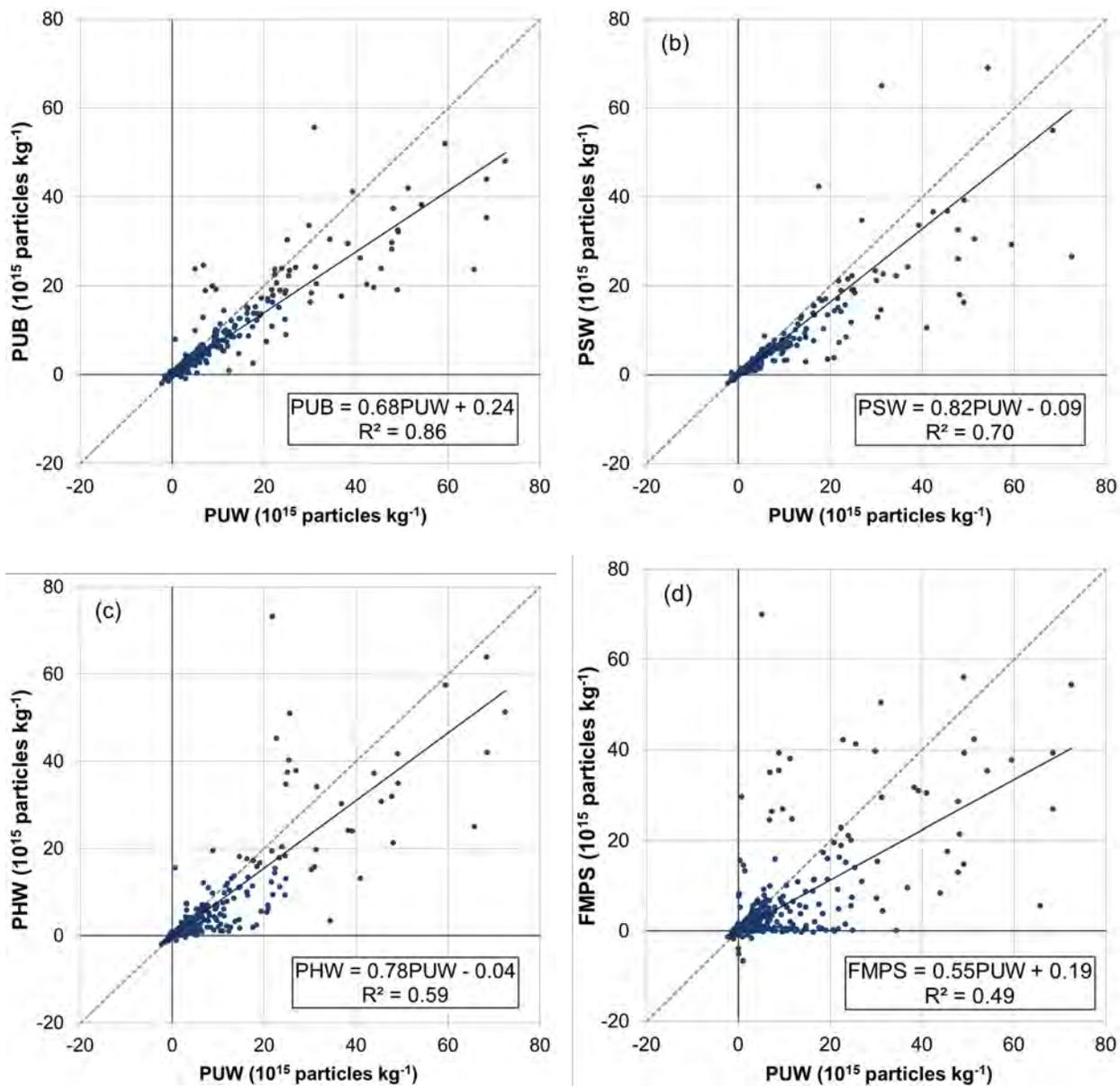
Figure A3. Comparison of UFP emission factors determined from each of four CPCs and the FMPS, using a water CPC-derived UFP emission rate as the independent variable. The dashed line is the 1:1 diagonal; linear best fit lines and regression statistics are also shown for each plot.

59