EFFECTS OF EXHAUST AFTER-TREATMENT AND FLEET MODERNIZATION ON PORT OF OAKLAND TRUCK EMISSIONS FOLLOWING COMPLETE IMPLEMENTATION OF CALIFORNIA'S DRAYAGE TRUCK REGULATION

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ABSTRACT

UC Berkeley previously reported on emission impacts of Phase 1 of the Statewide Drayage Truck Regulation, which increased the prevalence of diesel particle filters (DPFs) on drayage trucks serving the Port of Oakland, from 2 to 99% of trucks between 2009 and 2013. The present study reports on further changes corresponding to Phase 2 of this regulation, which took effect in January 2014. Between 2013 and 2015, the fraction of Oakland drayage trucks equipped with selective catalytic reduction (SCR) systems increased from 9 to 25%, and all pre-2007 model year engines were replaced with 2007 and newer engines. Coincident with these changes, the fleet-average nitrogen oxide (NO_x) emission factor decreased by 36 ± 7%. The increased use of SCR has partially mitigated a previously observed increase in nitrogen dioxide (NO_2) emitted by DPF-equipped trucks, with a 23 ± 10% reduction in the fleet-average NO_2 emission factor between 2013 and 2015. However, the NO_2 emission rate from trucks equipped with both DPF and SCR is still approximately twice that of trucks without these emission controls. Surprisingly, the fleet-average emission rate of black carbon (BC) increased by 12 ± 35% between 2013 and 2015, despite the fact that virtually all trucks were equipped with DPF across this period. The increase was driven by a 50% increase of the BC emission rate for 2007–2009 model year engines between 2013 and 2015. It appears that some particle filters are failing as they age, and thus a small fraction of DPF-equipped trucks are responsible for a large fraction of BC emissions. The combined impact of both phases of the Drayage Truck Regulation continues to be positive, with reductions of 70 ± 9% for NO_x, 73 ± 22% for BC, and 74 ± 27% for particle number emission factors, between 2009 and 2015. In order to maintain the air quality benefits of these emission control system investments, improved filter design/durability and maintenance/repair programs may be needed.

1. INTRODUCTION

The effects of truck fleet modernization and diesel particle filter (DPF) retrofits due to Phase 1 of the Statewide Drayage Truck Regulation were examined through direct measurements of truck emissions at the Port of Oakland (Preble et al., 2015). The phased compliance schedule for the Regulation is summarized below in Table 1.

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

Preble et al. measured pollutant concentrations at high time resolution in the exhaust plumes of a total of ~1,400 drayage trucks as they drove toward the Port on a major access road in November 2011 and March 2013. Emission factors were matched to data from a statewide drayage truck registry, including engine make, engine model year, and installed emission control equipment, using recorded license plates for each truck. Between 2009 and 2013, Phase 1 of the Drayage Truck Regulation led to an increase in Port trucks equipped with particle filters 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 ± 22% and 53 ± 8% for black carbon (BC) and nitrogen oxides (NOx), 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. Engines retrofitted with particle filters (pre-2007 model years) and engines equipped with filters as original equipment (model years 2007–2009) had NO_2 emission factors that were 7 and 5 times higher, respectively, than trucks without particle filters. The newest trucks with 2010 model year and newer engines, equipped with both DPFs and selective catalytic reduction (SCR) for NO_x control, emitted less than 10% of the BC, approximately the same NO_2, and less than 35% of the particle number (PN) as the truck fleet measured in 2009 prior to implementation of the drayage truck regulation.

Starting in 2014, Phase 2 of the Drayage Truck Regulation has required the replacement of nearly 30% of the Port of Oakland truck fleet. Specifically, Phase 2 required replacement of 1994 to 2006 model year engines, which had been previously retrofitted with DPFs, with 2007 or newer engines. Changes in BC and NO_x emissions, and in particular effects on primary NO_2 emissions, depend on the extent to which pre-2007 engines were replaced with either 2007–2009 (DPF only) or 2010 and newer (DPF + SCR) engines.

Drayage truck emissions at the Port of Oakland were measured again in 2015 to quantify changes in BC, NO_x, and NO_2 emissions associated with further modernization of the drayage truck fleet. The specific objectives of this study are to:

1. Compare fleet-average emission factors for NO_x, NO_2, BC, and PN measured in 2013 and 2015 to quantify effects of Phase 2 of the Drayage Truck Regulation.
2. Compare field data collected in 2009, 2011, 2013, and 2015 to describe evolution in truck engine age distribution and emissions.
3. Evaluate emissions from SCR-equipped trucks now that more 2010 and newer engines have entered the Port truck fleet. These trucks were scarcely present at the Port in our

2011 study, and comprised less than 10% of the fleet during our 2013 study. Given that SCR systems may be inactive when truck speeds and exhaust temperatures are relatively low, the 2015 measurements provide data on SCR performance under low-speed driving conditions.

4. Measure nitrous oxide (N_2O) emission factors. N_2O was not included in our previous studies at the Port, but is now of interest due to wider use of SCR systems. N_2O is a potent greenhouse gas that can be formed as a by-product of NO_x reduction with SCR.
5. Describe frequency of DPF system failures and deterioration of emission control system performance as a function of engine age by examining emission factor distributions (i.e., mean, skewness, and frequency and magnitude of high emitters). The prevalence of high-emitters in the first generation of DPF-equipped engines, 2007–2009 engine model years, had already started to increase in 2013 compared to 2011. Observations of the rate of DPF system failures in 2015, after two further years of in-service aging, are useful in tracking the durability of diesel exhaust emission control systems as a function of time.

2. METHODS

This study was conducted in September 2015 and followed a sampling methodology similar to that previously employed at the Port of Oakland and reported in Dallmann et al. (2011) and Preble et al. (2015). Excerpts from Preble et al. (2015) are included below, modified to describe any changes in the equipment used in the 2015 field study. Pollutant concentrations in the exhaust plumes of individual trucks were measured from an instrumented van that was positioned on an overpass. Westbound trucks heading toward the Port drove underneath on a major access road (Figure 1). Exhaust/ambient air mixtures sampled above the roadway were delivered to the van via a flexible aluminum duct. Concentrations of CO_2, NO_x, nitric oxide (NO), NO_2, BC, fine particle mass (PM), PN, and N_2O were measured at 1 Hz or faster, using instruments listed in Table A1 of the Appendix. A license plate recognition camera (GeoVision Inc., Taipei, Taiwan; GV-Hybrid LPR Camera 10R) at roadway level recorded truck license plates, which were later verified and then matched to data from California's Drayage Truck Registry, including engine model year and installed emission control devices.

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 (Ban-Weiss et al., 2009):

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

The emission factor for pollutant P (E_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 concentration peak to account for the fact that instruments operated with 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]* both are expressed in mass concentration units (e.g., μg m^{-3}), the ratio compares the relative abundances of pollutant P and CO_2 in the exhaust. The weight fraction of carbon in diesel fuel (w_c = 0.87) is used to convert the pollutant/carbon mass ratio to an emission factor expressed per unit mass of fuel burned (Ban-Weiss et al., 2009), and the factor *44/12* converts CO_2 to carbon mass. This carbon balance method for calculating emission factors assumes that all fuel carbon is converted to CO_2 during combustion, with negligible amounts of CO and VOC emitted relative to CO_2 (Dallmann et al., 2012).

NO_2 emission factors for each truck were computed (i) as the difference of NO_x and NO emission factors, which were measured simultaneously using two separate chemiluminescent analyzers, and (ii) via direct measurements of NO_2 using a sensitive optical absorption technique. NO_x emission factors were calculated using the molecular weight of NO_2. Unless noted otherwise, reported NO_2 emission factors were calculated by the difference method.

Emission factors were computed for trucks when the peak CO_2 concentration rose more than 7% above baseline roadway concentrations, following Dallmann et al. (2011). The baseline was taken to be the concentration measured just prior to the passage of a truck, with the timing determined from the roadway level video. Emission factors were computed only when the CO_2 peak could be definitively attributed to a single truck. Thus, no plume analyses were attempted when multiple trucks drove by at the same time or in close succession. In cases where CO_2

plume capture was successful but without clearly detectable peaks for other pollutants, near-zero values of emission factors were still computed.

Figure 1. Instrumented van positioned on an overpass sampling the exhaust from a truck en route to the Port of Oakland.

The particle sampling configuration shown in Figure A1 of the Appendix was designed to minimize wall losses by inertial separation and diffusion. Laminar flow was maintained in all sampling lines to minimize particle losses due to turbulence. An in-line dilution system was used to avoid exceeding the concentration limit of the ultrafine, butanol-based condensation particle counter (CPC) that measured PN concentrations. This system consisted of filtered recirculating flow in a closed loop with controlled flow for a dilution ratio of 6.5–9.8, depending on the flow set point.

Previously reported changes in PN emission rates in Preble et al. (2015) were derived from an ultrafine, water-based CPC. Table S3 of the Supporting Information for Preble et al. (2015) also reported average PN emission factors that were calculated using a butanol-based CPC in 2011 and 2013. In order to make direct comparisons with the most recent 2015 measurements using a butanol-based CPC, we refer to the butanol-based CPC values of Preble et al. in the Results and Discussion below.

An aerosol photoacoustic absorption spectrometer (PAS) was used in conjunction with an aethalometer to measure BC concentrations. The former instrument aided in post-processing the aethalometer data to control for the aethalometer's filter loading artifact, in which time-dependent behavior 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:

$$BC = \left[\frac{BC_o}{a \exp\left(\frac{-ATN}{100}\right) + (1 - a)} \right] \quad (2)$$

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. The site-specific value for *a* previously reported in Preble et al. (2015) was validated for these 2015 measurements by plotting the ratio of the light absorption coefficient measured with the PAS and the BC concentration measured with the aethalometer, as shown in Figure A2.

The suite of analyzers used in these 2015 measurements included duplicate measures of nitrogen dioxide and fine particle mass (Table A1). Figures A3–A5 show comparisons between these measurement methods. NO_2 concentrations were measured *by difference* via two chemiluminescent analyzers with one monitor measuring total NO_x and the other measuring NO, as well as *directly* with a monitor that used a cavity-attenuation phase shift technique. Overall, these two measurements of NO_2 were highly correlated ($R^2 = 0.93$) with a slope near unity and near-zero intercept (Figure A3).

Fine particle mass (PM) concentrations were measured with a DustTrak and a Dekati Mass Monitor (DMM). The DustTrak uses light scattering to infer particle mass concentrations. This method requires calibration to control for differences in aerosol optical properties. The factory setting assumes that the aerosol being measured is Arizona road dust, which is coarser and gives a higher fraction of scattered light compared to the darker and strongly-absorbing particles emitted in heavy-duty diesel engine exhaust. Calibrating the DustTrak specifically to the truck exhaust sampled at the Port of Oakland was not possible, as the particle mass collected on filters during plume sampling was insufficient to create a complete calibration curve for the analyzer. These measurements are still qualitatively useful, but they may not accurately represent fine PM emission rates. The DMM uses electrical low-pressure impaction to determine

concentrations of fine PM, and thus should not require aerosol-specific calibration. The agreement between these two measures was strong when derived emission factors were less than 5 g kg^{-1} (R^2 = 0.83), but the agreement became weaker when larger emission factors were also included in the analysis (R^2 = 0.65), as shown in Figure A4. Fine PM emission factors determined with the DMM tended to be smaller than those calculated from the DustTrak measurements. This disparity may be due to the difference in measurement method between the two analyzers. There is also a difference in the particle size-selective inlets for these two analyzers. The DustTrak measured PM_2, as larger particles were excluded using a cyclone located upstream at the beginning of the aerosol sampling line (Figure A1). The DMM included an internal cyclone and provides mass data for $PM_{1.2}$. The difference in particle mass between these two cut points is likely to be small in this study, though, as diesel exhaust PM emissions consist almost entirely of sub-micron particles (Preble et al. 2015). Overall, the DMM-derived fine PM emission factors correlated better with the corresponding BC emission factors than the DustTrak-derived values (Figure A5). However, fine PM emission factors derived from both analyzers can be smaller than the corresponding BC emission factors (Figure A5 and Tables A2–A3). Since BC is a component of PM, the PM reading should be at least as large as BC, and caution is therefore needed in comparing absolute values of fine PM and BC emission rates reported here.

Prior to the field study, all instruments and data loggers 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 conditions was used to verify the plume capture sampling and carbon balance

analysis methods used in this study. The NO_2 conversion efficiency of both chemiluminescent NO_x analyzers was evaluated using ozone titration tests to ensure accuracy of total NO_x (NO + NO_2) measurements. Multipoint calibrations were conducted for all gas analyzers prior to deployment in the field. During field deployment, these calibrations were verified with zero and span checks at the beginning and end of each day of sampling.

3. RESULTS AND DISCUSSION

License plate data obtained from the Drayage Truck Registry allowed for the categorization of each sampled truck into one of four control technologies: (1) 1996–2006 model year engines without DPFs, (2) 1996–2006 model year engines with retrofit DPFs, (3) 2007–2009 model year engines with DPFs, and (4) 2010–2016 model year engines with DPFs and SCR. Unless explicitly described as a retrofit, DPF and SCR systems were installed as original equipment on trucks when they were first sold as new.

In the results and discussion that follow below, model years refer to the engine rather than the truck chassis. All reported uncertainty ranges are based on the sampled fleet and represent 95% confidence intervals about the mean measured values. Also, in figures and tables, the number of trucks (n) evaluated for mean and confidence interval values is reported.

3.1 Changes to Port Truck Age Distribution. As the Drayage Truck Regulation was implemented, the truck fleet serving the Port of Oakland rapidly modernized (Figure 2). In 2008, prior to the regulation, the median engine age was 11 years and only 2% of Port trucks were equipped with DPFs (BAAQMD, 2009). By early 2013, after Phase 1 was completely in effect, 99% of trucks were equipped with filters, a small fraction of the fleet was also equipped with SCR systems for NO_x control, and the median engine age had decreased to 6 years (Preble et al., 2015). Phase 2 of the regulation required the replacement of pre-2007 engines, which comprised 29% of the fleet in 2013, with 2007 or newer engines. In 2015, 74% of trucks had 2007–2009 engines with DPFs, 25% had 2010 or newer engines with both DPF and SCR systems, and the median engine age was 7 years (Table 2, Figure 2). No trucks equipped with a retrofit DPF were observed in 2015, consistent with regulatory requirements.

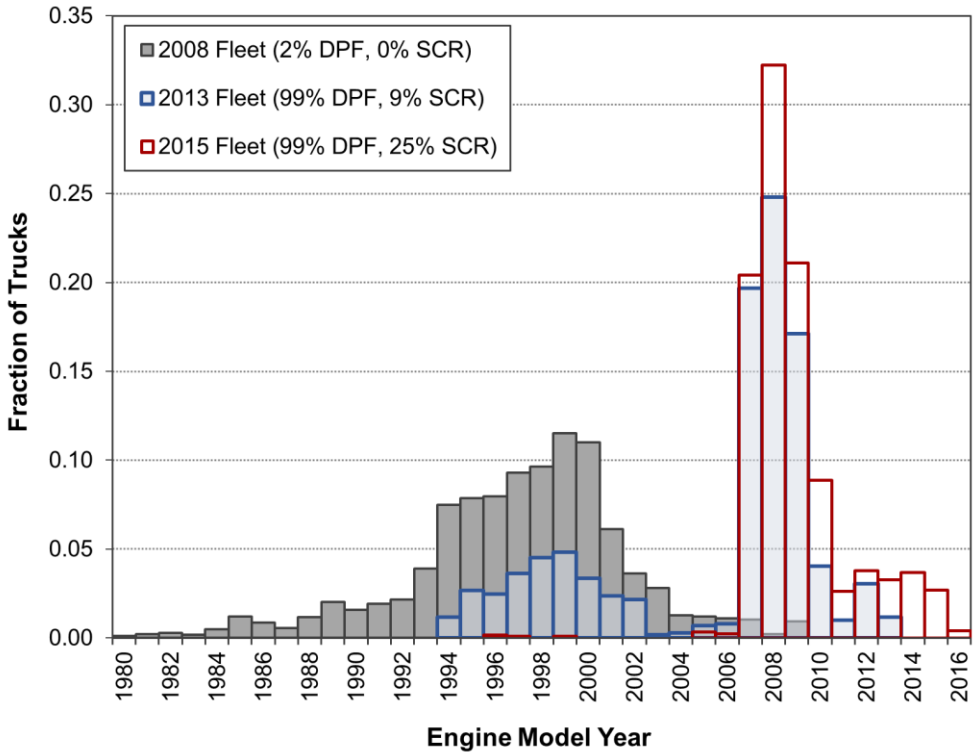

Figure 2. Distribution of truck engine model years at the Port of Oakland prior to the Statewide Drayage Truck Regulation in 2008 (BAAWMD, 2009), after full implementation of Phase 1 of the regulation in 2013 (Preble et al., 2015), and after full implementation of Phase 2 of the regulation in 2015. Note that the model year distribution from 2008 is based on truck chassis; some engines may be one year older than the chassis.

Table 2. Prevalence of exhaust emission control systems for the Port of Oakland drayage truck fleet after Phase 1 of the Drayage Truck Regulation was in effect in 2013 and after Phase 2 was implemented in 2015.

Control Category	Engine Model Years	2013 Fleet (N = 1016)	2015 Fleet (N = 1219)
No DPF	1994–2006	1% (n = 15)	1% (n = 11)
Retrofit DPF	1994–2006	28% (n = 281)	0% (n = 0)
DPF	2007–2009	62% (n = 626)	74% (n = 904)
DPF + SCR	2010–2016	9% (n = 94)	25% (n = 304)

3.2 Changes in Emission Rates over Time. Fleet-average emission factors measured as the Drayage Truck Regulation was implemented are reported in Table A2 of the Appendix and shown below in Figures 3–7. The pre-regulation fleet is characterized using 2009 field measurements from Dallmann et al. (2011). The values for 2009 presented here have been adjusted to account for a CO_2 sampling artifact and the site-specific aethalometer correction, as described in Preble et al. (2015). The 2011 and 2013 measurements took place during the middle and at the end of the Phase 1 regulation, and have been reported previously in Preble et al. (2015). The 2015 measurements took place after full implementation of Phase 2 requirements.

Average pollutant emission rates by emission control category are also presented in Table A2 and Figures 3–7. The category-average emission factors for all pollutants as measured in each of the three sampling years can also be found in Table A2 of the Appendix.

3.2.1 Nitrogen Oxides. As shown in Figure 3a, the fleet-average emission rate of total NO_x decreased by 70 ± 9% between 2009 and 2015, compared to the initial 53 ± 8% reduction

achieved by the first phase of the Drayage Truck Regulation. Between Phases 1 and 2 of the regulation, NO_x emissions decreased by 36 ± 7%. These NO_x emission reductions are the result of rapid fleet modernization. Given that newer engines must adhere to more stringent emission limits, the shift in median engine model year from 1997 to 2008, and associated changes in emission controls, led to the large reductions in fleet-average NO_x emissions. A comparison of NO_x emissions by control technology highlights the cause of this dramatic reduction: the newest trucks equipped with both DPF and SCR systems emitted 76 ± 7% less NO_x compared to older trucks with 1994–2006 model year engines, the truck category with the highest observed NO_x emission rates in 2011 and 2013.

As a result of the near-universal adoption of DPFs to control the fleet's PM emissions, fleet-average emissions of NO_2 doubled between 2009 and 2015 (Figure 3b). This overall change is less than the previously observed increase over Phase 1 of the regulation (Preble et al. 2015), as the wider use of SCR over Phase 2 led to a decrease of 23 ± 10% in emitted NO_2 between 2013 and 2015. As such, SCR systems have partially mitigated this undesirable DPF-related increase in emitted tailpipe NO_2. Even so, the emission rate of NO_2 for the newest trucks with both DPF and SCR systems is nearly double that found for trucks without DPFs.

As a result of this increased NO_2 emission rate with corresponding decrease in total NO_x emitted, the fleet-average NO_2/NO_x emission ratio increased from 0.03 ± 0.02 in 2009 to 0.22 ± 0.02 in 2015 (Figure 4). The NO_2/NO_x ratio is highest for 2007–2009 DPF-equipped trucks. Newer trucks also equipped with SCR systems have significantly lower average NO_x and NO_2 emission factors, even though the subsequent category-average NO_2/NO_x ratio does not show a significant decrease relative to the 2007–2009 trucks (Figures 3 and 4).

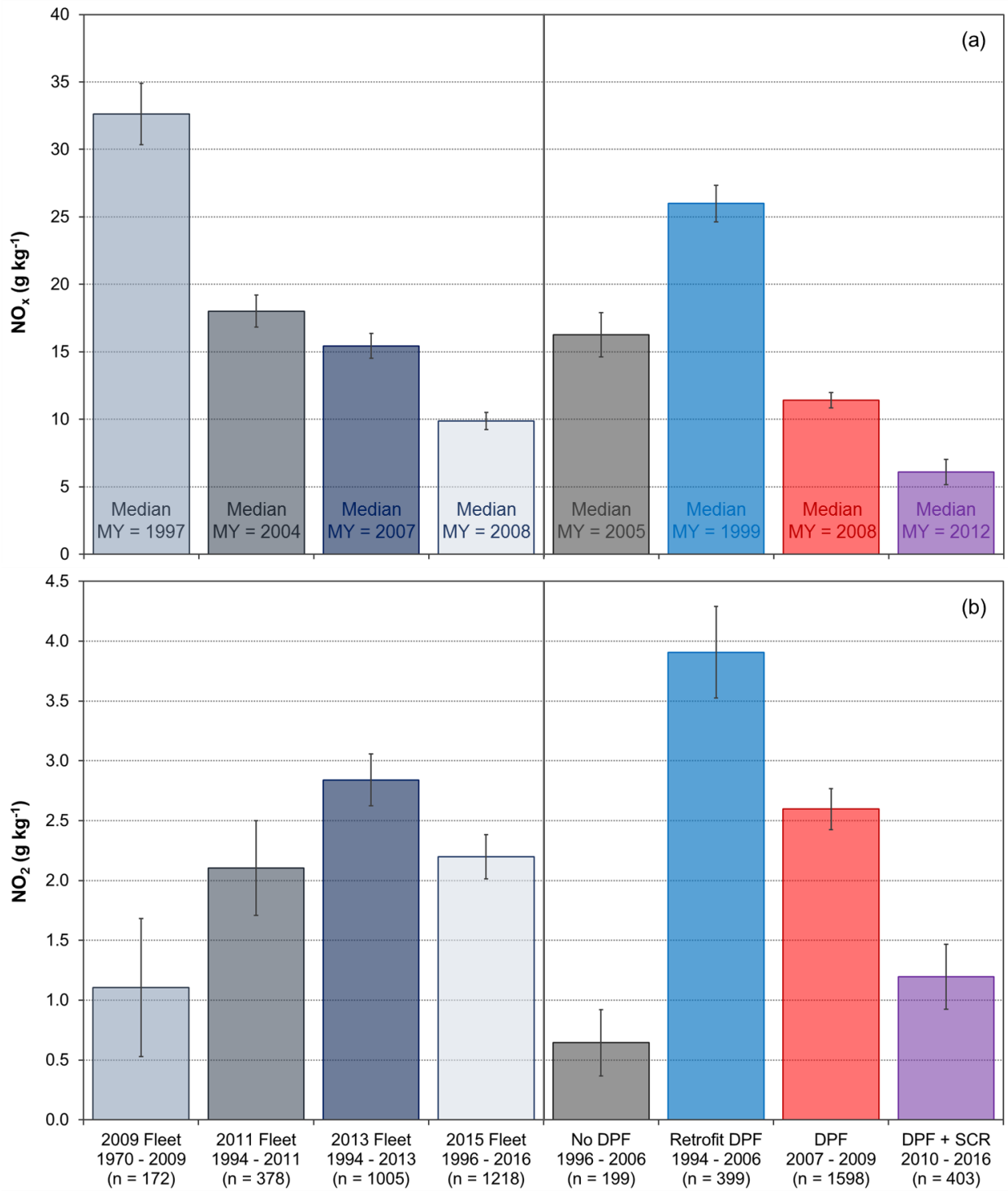

Figure 3. Fleet-average emission factors for (a) NO_x and (b) NO_2 as measured in 2009, 2011, 2013, and 2015 at the Port of Oakland (left panel), as well as category-average values for combined 2011 + 2013 + 2015 measurements (right panel).

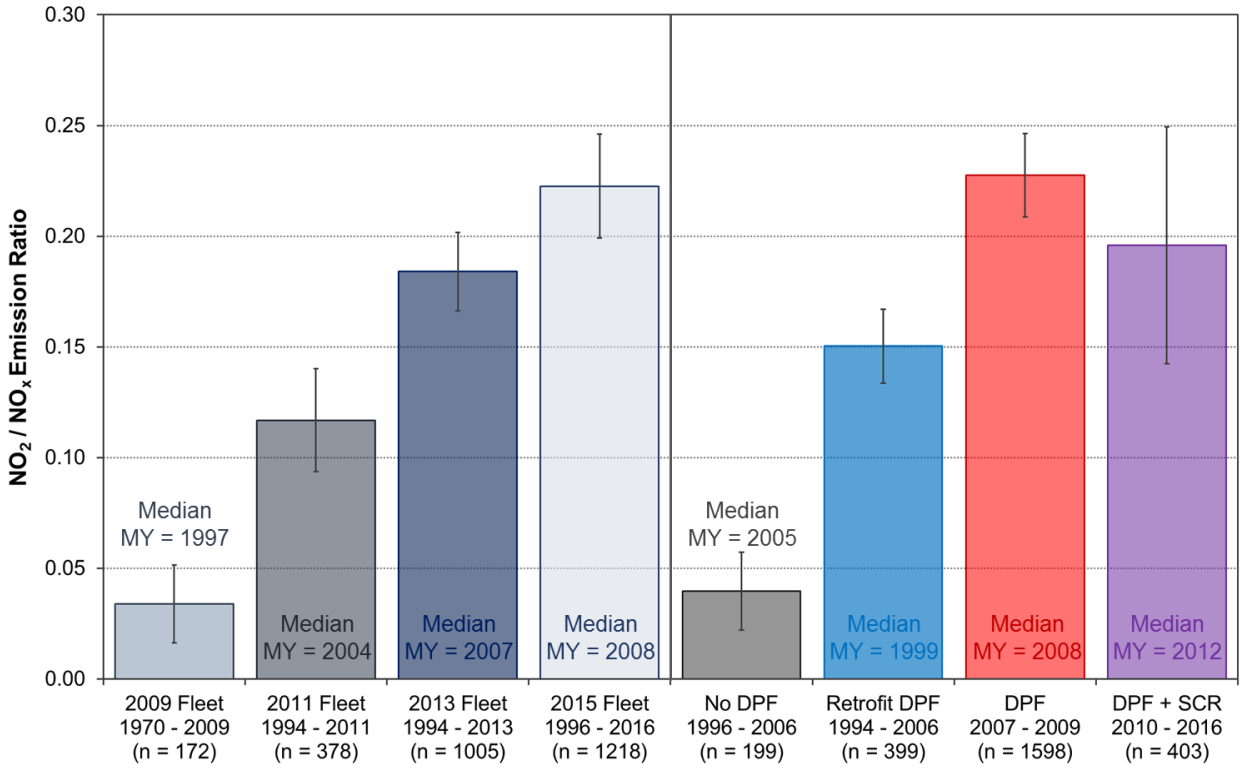

Figure 4. Fleet-average emission ratios of NO_2/NOx as measured in 2009, 2011, 2013, and 2015 at the Port of Oakland (left panel), as well as category-average values for combined 2011 + 2013 + 2015 measurements (right panel).

3.2.2 Black Carbon. Between 2009 and 2015, the fleet-average emission rate of BC decreased by 73 ± 22%, compared to the 76 ± 22% reduction previously found over Phase 1 between 2009 and 2013 (Figure 5). This slightly smaller overall reduction is the result of a 12 ± 35% *increase* in the fleet-average BC emission rate between 2013 and 2015. This result is surprising given that the fraction of the drayage fleet equipped with DPFs remained at 99% across both sampling years. By looking at the changes in the category-average values over time, this BC increase can be attributed to an increase in the average for 2007–2009 model year engines. The median emission rate for this category of trucks remained relatively constant (0.05 g kg^{-1} in 2013 versus 0.04 g kg^{-1} in 2015), but the mean increased by a factor of 1.5 from 0.26 to 0.39 g kg^{-1} over the

same period (Figure 6). This finding indicates that some diesel particle filters are deteriorating as they age, especially for 2007–2009 engines (Figure 6). The distribution of BC emissions across individual trucks and the importance of high emitters are discussed in more detail later in this report.

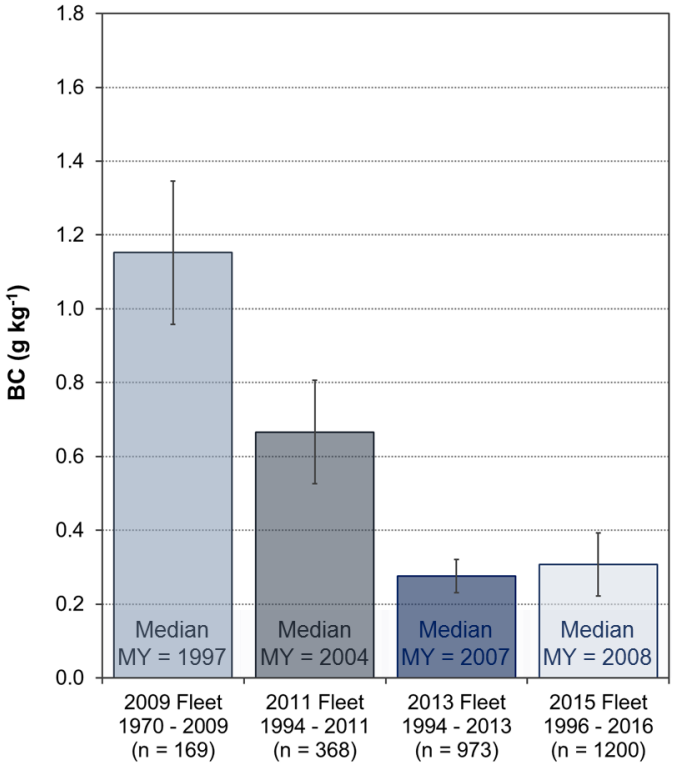

Figure 5. Fleet-average BC emission factors as measured in 2009, 2011, 2013, and 2015 at the Port of Oakland.

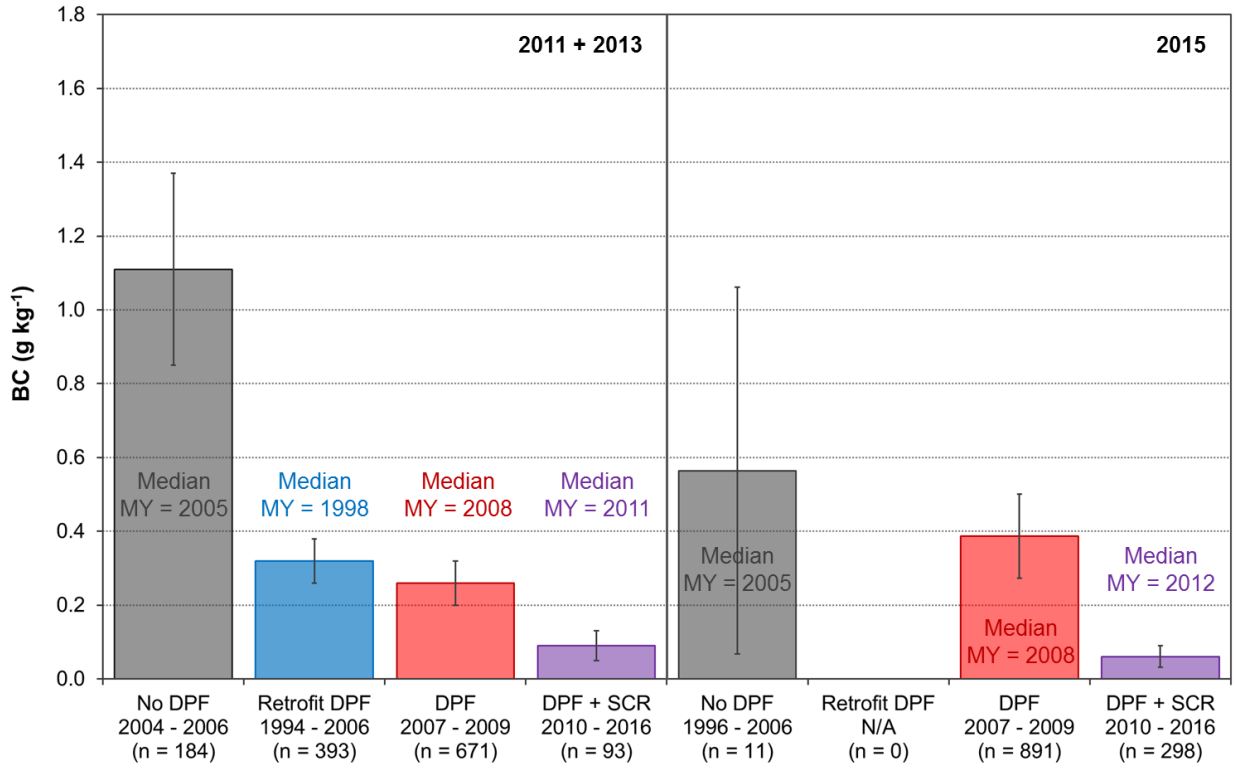

Figure 6. Average BC emission factors as a function of installed emission control systems, for the combined 2011 + 2013 data reported previously by Preble et al. (left panel) compared to average values measured in 2015 as part of the current study (right panel).

3.2.3 Fine Particle Mass. Fine particle mass (PM) was measured with a DustTrak but not reported in our previous field campaigns (Dallmann et al., 2011; Preble et al., 2015). The reason is that this light-scattering based instrument was not calibrated to measure diesel PM emissions (see discussion in Methods section). Therefore, the PM emission factors presented here are used to examine trends rather than define absolute magnitudes of PM emission rates.

As shown in Figure 7, the fleet-average PM emission rate decreased by 72 ± 24% between 2009 and 2015. This reduction is smaller than the reduction of 79 ± 23% between 2009 and 2013. As observed for BC, the average PM emission rate increased between 2013 and 2015.

The increase in PM can be attributed to a doubling of the average PM emission rate for 2007–2009 model year engines with DPFs between 2013 and 2015 (Figure 8).

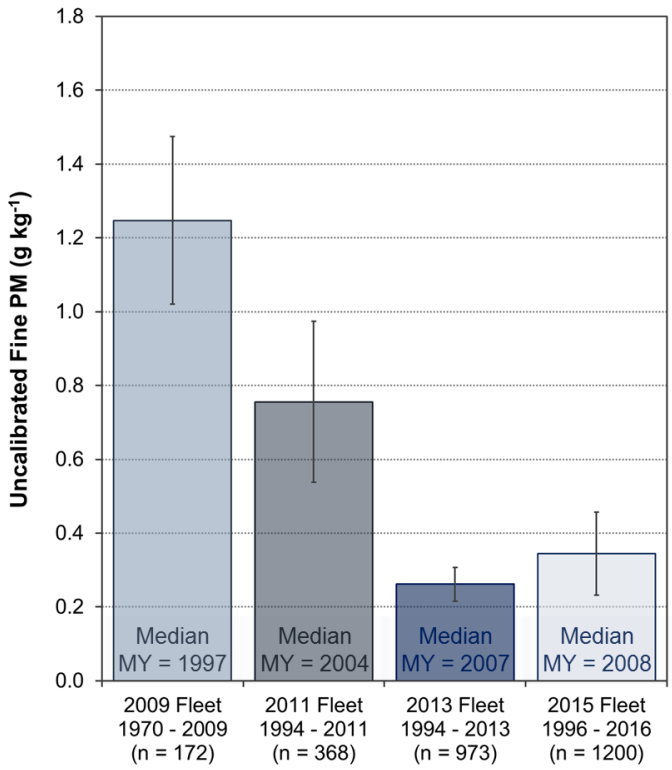

Figure 7. Fleet-average fine PM emission factors as measured in 2009, 2011, 2013, and 2015 at the Port of Oakland. Emission factors are derived from fine PM concentrations measured with an uncalibrated instrument (see text), so the focus is the emission trend over time rather than absolute values in a specific year.

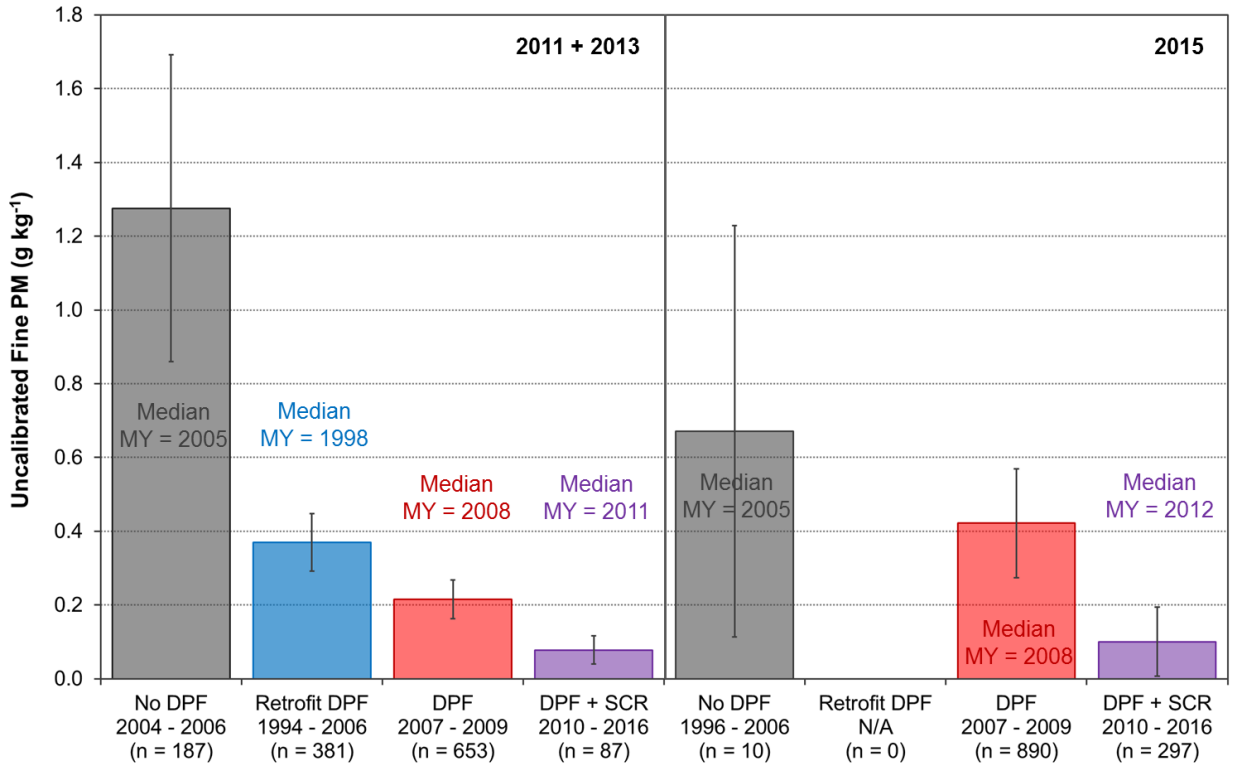

Figure 8. Average fine PM emission factors by control category type for the combined 2011 + 2013 data (left panel) compared to the average values found in the current study, as measured in 2015 (right panel). Note that these emission factors are derived from fine PM concentrations measured with an uncalibrated instrument (see text), so the emission trends by control category rather than the absolute values are considered to be relevant to the fleet's PM emissions.

3.2.4 Particle Number. The fleet-average particle number emission rate decreased by 74 ± 27% between 2009 and 2015, indicating further progress in reducing emissions relative to the initial reduction of 49 ± 17% measured in 2013 relative to the same 2009 baseline (Figure 9). Between Phases 1 and 2, the fleet-average PN emission rate decreased by 49 ± 25%. As previously reported in Preble et al. (2015), under these driving conditions and at this location, trucks equipped with DPFs have significantly lower PN emission rates compared to trucks without filters. The newest engines that are equipped with both DPF and SCR systems have the lowest

PN emission factors, emitting one-fourth the number of particles per kg of fuel burned compared to trucks without filters.

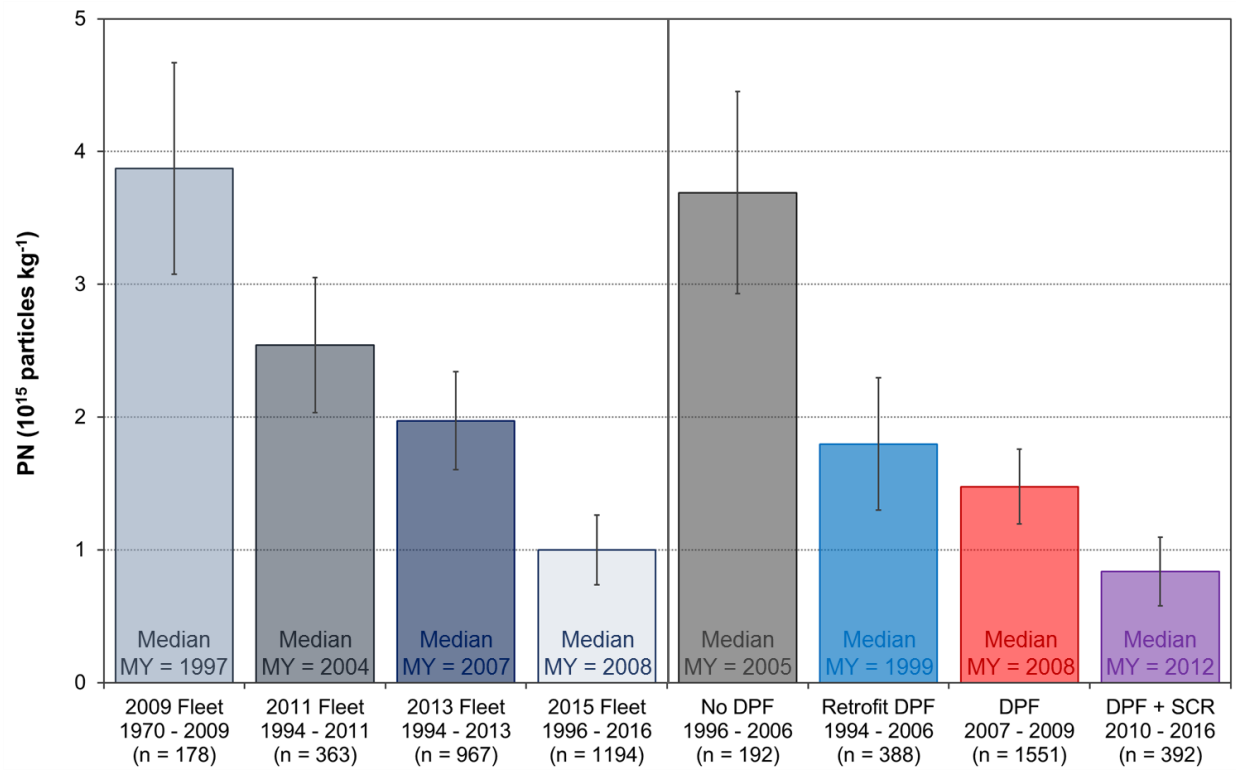

Figure 9. Fleet-average PN emission factors as measured in 2009, 2011, 2013, and 2015 at the Port of Oakland (left panel), as well as category-average values for combined 2011 + 2013 + 2015 measurements (right panel).

3.3 Nitrous Oxide. Selective catalytic reduction systems have the potential to form nitrous oxide (N_2O), a potent greenhouse gas, via oxidation of ammonia by excess NO_2 and oxygen in the system (DieselNet, 2016). During Phase 1 of the regulation, this was of minimal concern given the small fraction of the fleet equipped with SCR in 2013 (<10%). However, with the increase in SCR use driven by the implementation of Phase 2 of the regulation, N_2O has become a pollutant of interest and was measured in the 2015 campaign. The fleet-average emission rate of N_2O in

2015 was 0.16 ± 0.03 g kg^{-1}, with SCR-equipped engines emitting nearly 7 times more N_2O per kg of fuel burned than those trucks without SCR systems (Figure 10). These fleet- and category-average N_2O emission rates are below California's 0.1 g bhp-hr^{-1} limit for 2014 and newer model year engines (CARB, 2014), which equates to 0.6 g kg^{-1} assuming a brake specific fuel consumption of 0.17 kg hp-hr^{-1} (Heywood, 1998). Across the truck fleet measured in 2015, the measured N_2O emission rate for 6% of trucks exceeded this California standard, of which 75% were SCR-equipped trucks. Of the 300 trucks with SCR systems measured in 2015, 18% exceeded the California N_2O emission limit. Though N_2O was not measured in prior campaigns, these results indicate that the fleet-average N_2O emission factor has increased markedly between 2009 and 2015 as the fraction of Port of Oakland drayage trucks equipped with SCR increased from 0 to 25%.

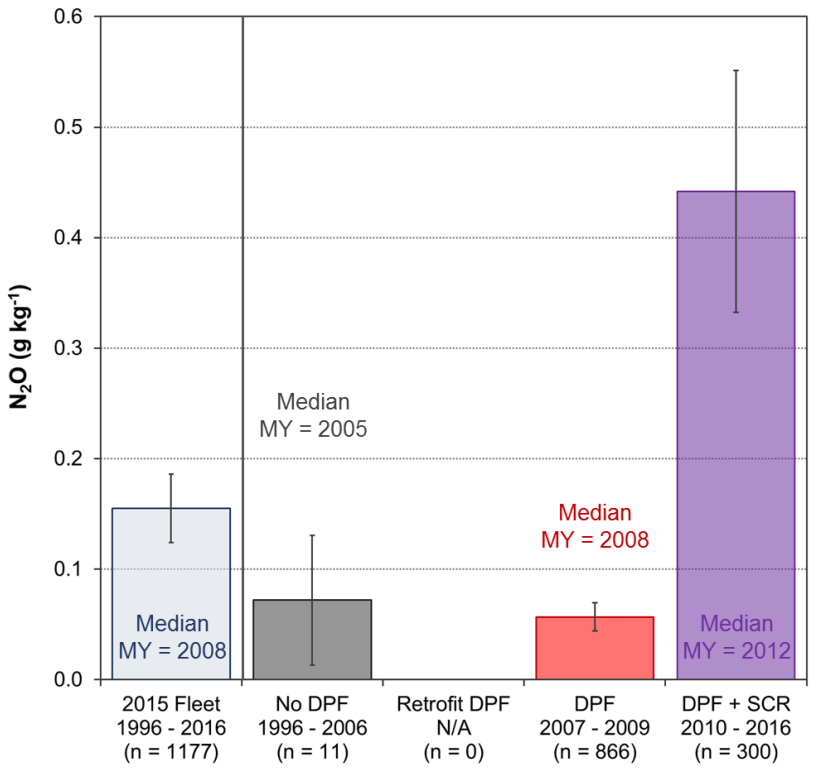

Figure 10. Fleet-average (left panel) and control category-average (right panel) N_2O emission factors as measured in 2015 at the Port of Oakland.

3.4 Emission Factor Distributions. In order to gain more insight into the fleet- and category-average trends presented above, the distributions of measured emission factors across different sampling years are presented in Figures 11–13. These distributions are presented in two ways: (i) cumulative emission factor distributions of fleet emissions, and (ii) box-and-whisker distributions of emission factors by engine model year.

As adoption of DPF and SCR systems increased and fleet-average BC and NO_x emission factors decreased, the distribution of emissions within the Port of Oakland truck fleet have become increasingly skewed (Figure 11). Progressively smaller minorities of trucks have become responsible for increasing fractions of total BC and NO_x emissions. Particle emissions are more skewed than nitrogen oxide emissions (Figure 11). In 2013, the highest-emitting 10% of the fleet was responsible for 32% of NO_x and 65% of BC emissions, though different trucks comprise the high-emitting sub-group for each pollutant (Preble et al., 2015). In 2015, the skewness of these distributions increased modestly for total NO_x, with the top 10% of trucks responsible for 37% of total emissions. The increase in high-emitter contribution was more substantial for BC, with the dirtiest 10% of the fleet responsible for 77% of total emissions (Figure 11). Between 2013 and 2015, there was no change in the skewness of the PN emission factor distribution, with the top 10% of trucks emitting ~80% of total PN (Figure 12). Emissions of N_2O by the 2015 drayage truck fleet were also quite skewed, with 10% of trucks responsible for 71% of emitted N_2O (Figure 12). The skewness in N_2O emissions is expected, given that SCR-equipped trucks are the major source but only 25% of Port trucks had SCR systems installed as of 2015. As previously stated, 6% of the 2015 fleet had N_2O emission rates in excess of the California standard of 0.6 g kg^{-1} (see section 3.3 of this report); these high N_2O emitter trucks were responsible for 59% of the fleet's total N_2O emissions.

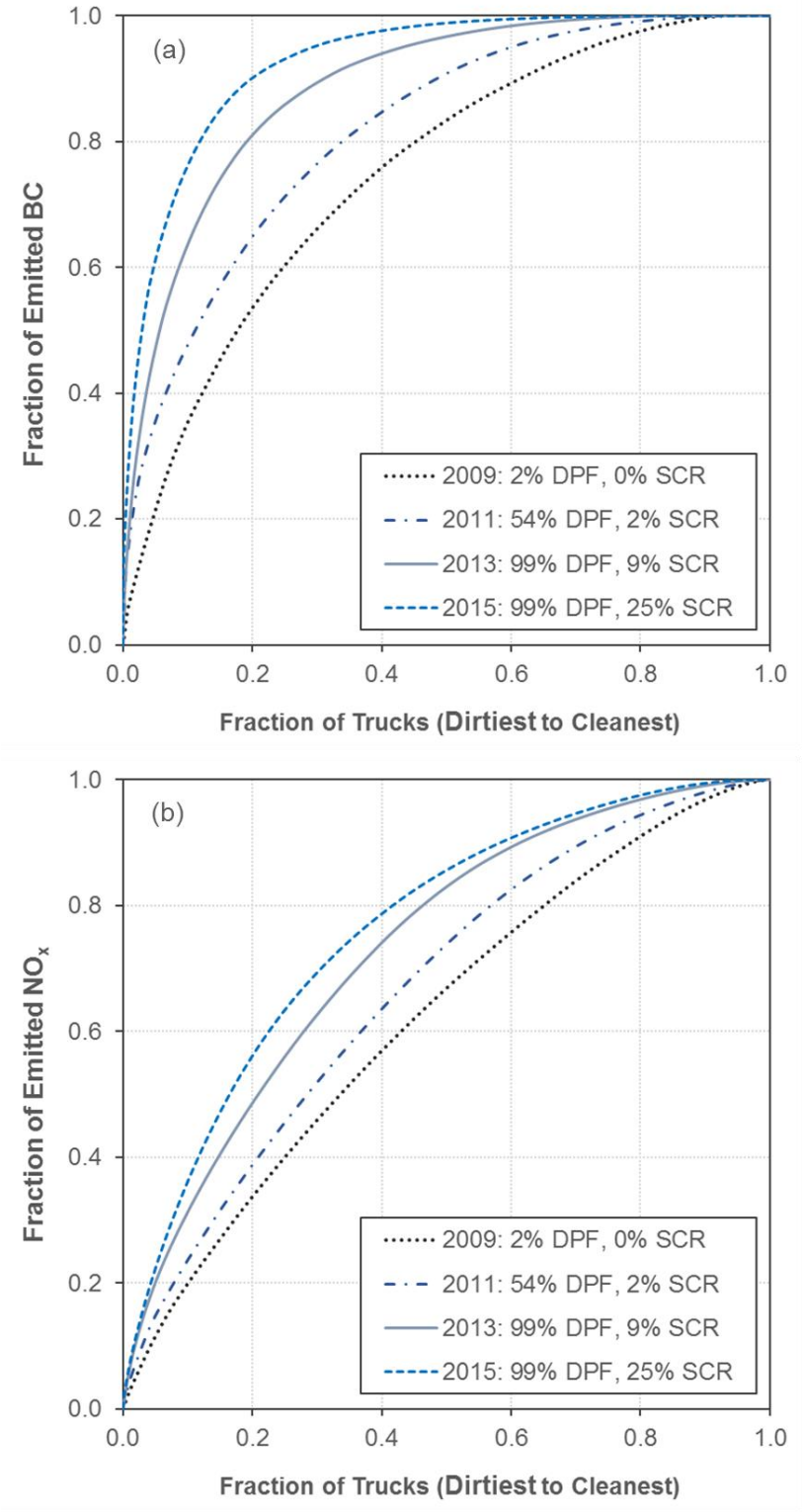

Figure 11. Cumulative (a) BC and (b) NO_x emission factor distributions as measured in 2009, 2011, 2013, and 2015 at the Port of Oakland, in which emission factors for individual trucks were ranked from highest to lowest.

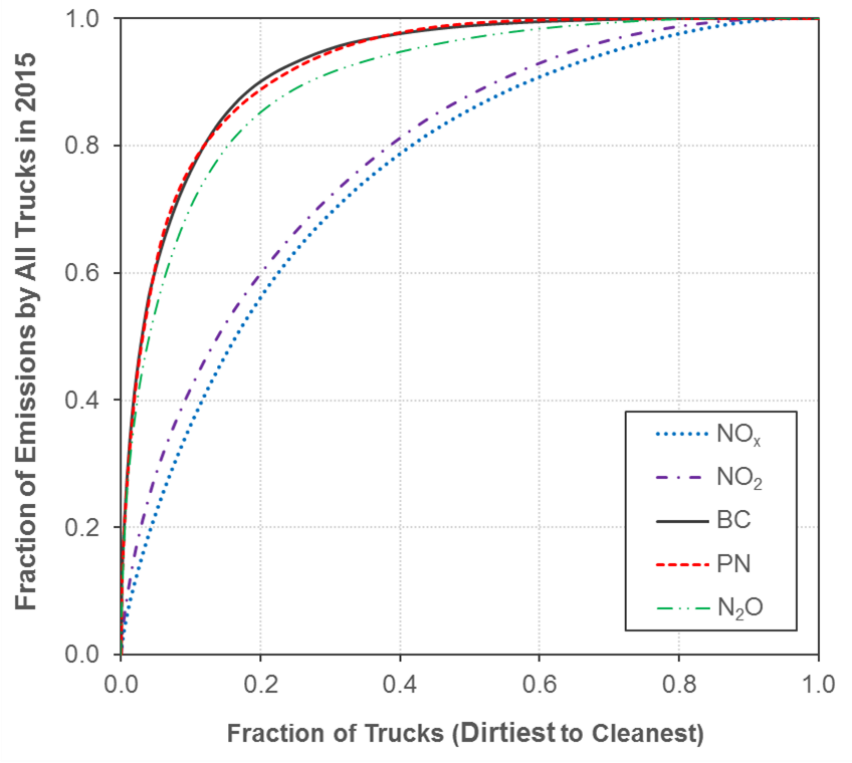

Figure 12. Cumulative emission factor distributions for NO_x, NO_2, BC, PN, and N_2O as measured in 2015 at the Port of Oakland, in which emission factors for individual trucks were ranked from highest to lowest.

BC and NO_x emission factor distributions are shown by model year for all trucks originally equipped with DPFs (i.e., 2007 and newer engines), as measured in 2013 and 2015, in Figure 13. In both field studies, BC emission factor distributions for 2007–2009 engines are more skewed than for 2010 and newer engines, as indicated by mean values that often greatly exceed the 90th percentile for these model years. Emission factor distributions for 2007 and 2008 engines also show greater spread in 2015 than in 2013, as seen with the wider boxes-and-whiskers (Figure 13a). Also for 2007 and 2008 model year engines, the mean BC emission rate increased in 2015 relative to each corresponding 2013 value, while the median value remained approximately the same. These BC distributions follow the same trends as the category-average

results previously presented in Figure 6 and also highlight the increasing impact of high emitters relative to the general fleet. These observations for 2007 and 2008 engines indicate that deteriorated DPF system performance is an issue for at least some trucks.

The mean values for NO_x emission factors as a function of engine model year shown in Figure 13b generally fall between the median and the 75th percentile levels. This is consistent with the earlier finding of a less skewed distribution for NO_x compared to BC (Figures 11 and 12). NO_x emissions for some SCR-equipped trucks were higher than the 75th percentile value for 2007–2009 engines without SCR, indicating that some SCR systems may not have been operational when exhaust was sampled. There is no evidence of a shift in NO_x emission factor distributions between 2013 and 2015 for trucks without SCR (i.e., 2007–2009 engines). Data for trucks with 2010 and newer engines are limited. The measurements suggest a shift to higher emission levels for 2011 but not 2010 or 2012 engines. There were few 2013 model year engines observed in the 2013 field study, so apparent changes in NO_x emissions for that model year should be interpreted with caution.

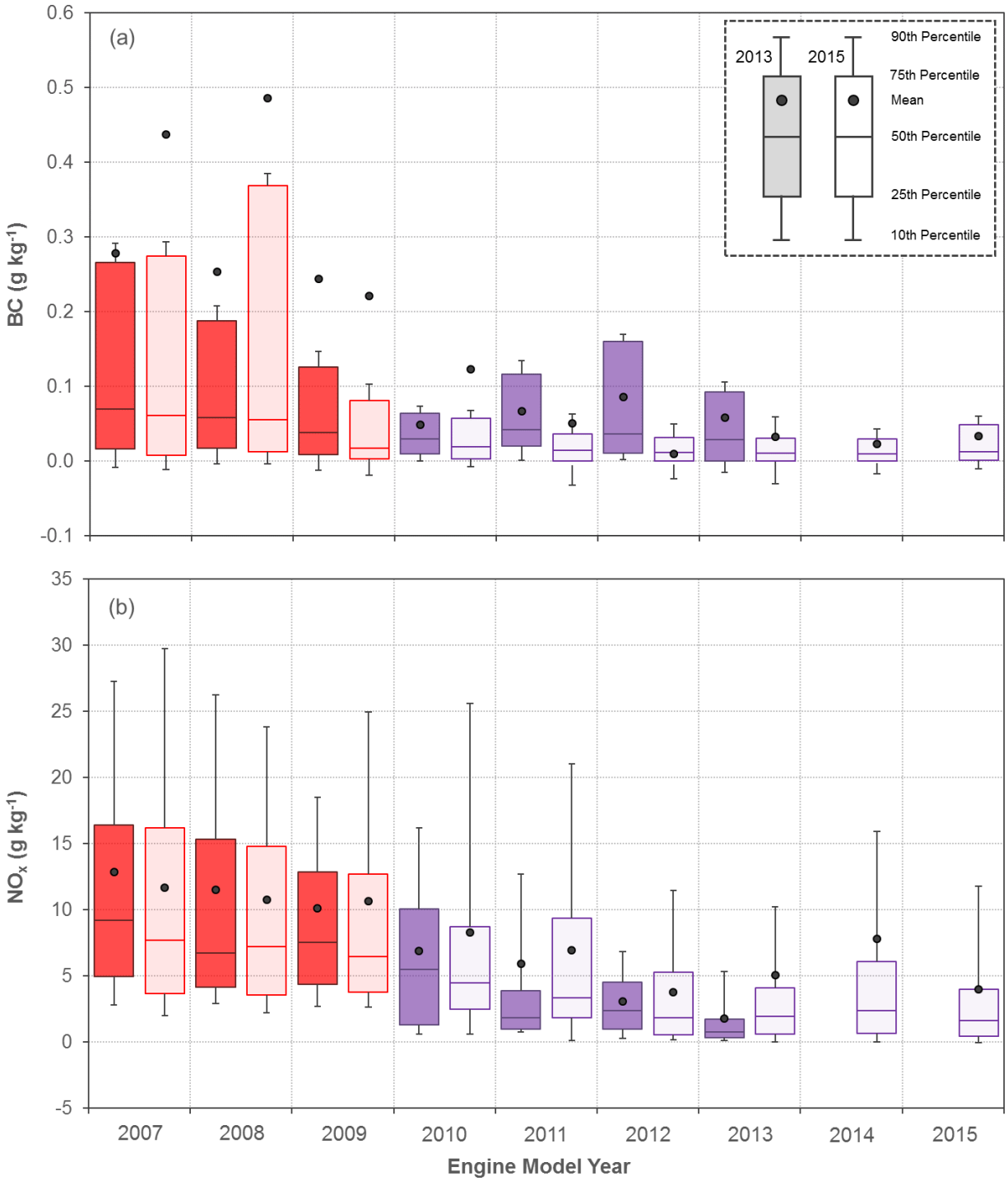

Figure 13. Distribution of (a) BC and (b) NO_x emission factors by engine model year for DPF-equipped trucks as measured in 2013 (left bar in each pair) and 2015 (right bar in each pair). Trucks with 2007–2009 engines are shown in red; those with 2010 and newer engines, equipped with both DPF and SCR, are shown in purple.

3.5 DPF Deterioration. Figures 6 and 13a indicate that performance of some DPFs installed on 2007–2009 engines has deteriorated over time, whereas DPFs on 2010 and newer engines have remained effective at reducing emitted BC. Figures 14–16 delve further into the issue of DPF system failures as the age of engines and filter systems increase.

The distributions of BC emission factors for 2007–2009 engines over time are shown in Figure 14. High emitters are defined as those trucks with BC emission factors greater than 1.1 g kg^{-1}, the category-average for diesel trucks without filters (Preble et al., 2015). This value is 18 times the applicable PM emission standard of 0.01 g $hp\text{-}hr^{-1}$ for 2007 and newer engines, assuming brake specific fuel consumption of 225 g kWh^{-1} or 0.17 kg $hp\text{-}hr^{-1}$ (Heywood, 1998). This definition of high emitter is not equivalent to the regulatory opacity-based standard that is employed by CARB's Inspection and Maintenance program. As highlighted in Figure 14, the fraction of 2007–2009 trucks classified as high-emitters in all three measurement years was similar, between 6–8% of the truck category.

Figure 15 shows the contribution of high emitters to the total truck fleet cumulative emissions shown earlier in Figure 11b. In 2009, when only 2% of the fleet was DPF-equipped, 33% of trucks in the fleet were high emitters, and these trucks were responsible for 69% of total BC emissions. When DPFs were used to control emissions throughout the truck fleet in 2013, 52% of BC emissions could be attributed to high emitters that constituted 6% of the fleet. In 2015, the fraction of the fleet identified as high-emitting remained at 6%, but those trucks were responsible for 65% of total BC emissions. Of the 73 trucks that comprised the high-emitter fraction in 2015, 93% were 2007–2009 engines with DPFs, 4% were 2010 and newer engines with DPF and SCR systems, and only 3% were trucks without filters.

To summarize, a small minority (less than 10%) of the Port drayage trucks observed in 2015 were responsible for the majority of BC emissions, and the highest emitters were mostly DPF-equipped 2007–2009 model year engines. Whereas the fraction of high-emitting trucks in the 2007–2009 model year category has remained approximately constant, the emission rates for these high-emitting trucks appear to be increasing over time.

BC emissions were broken down by engine manufacturer for the 2007–2009 trucks sampled in the 2015 study to see if there were differences in performance and durability of these particle filters (Figure 16). The most common engines installed in this category of trucks were supplied by Detroit Diesel, Cummins, and Volvo, together representing 81% of the 2007–2009 model year engines in the Port fleet. Trucks with Volvo engines had significantly lower BC emissions than trucks with Detroit Diesel, Cummins, and Mercedes Benz engines (Figure 16a). Trucks with Detroit Diesel, Cummins, and Mercedes Benz engines also had the highest fractions of trucks identified as high-emitters. Between these three manufacturers, though, there was no significant difference in average BC emitted by those high emitters (Figure 16b). The average BC emission rate by these high emitters was nearly an order of magnitude greater than the 2015 fleet-average, however, and a factor of 7 times greater than the 2007–2009 engine model year category average observed in 2015.

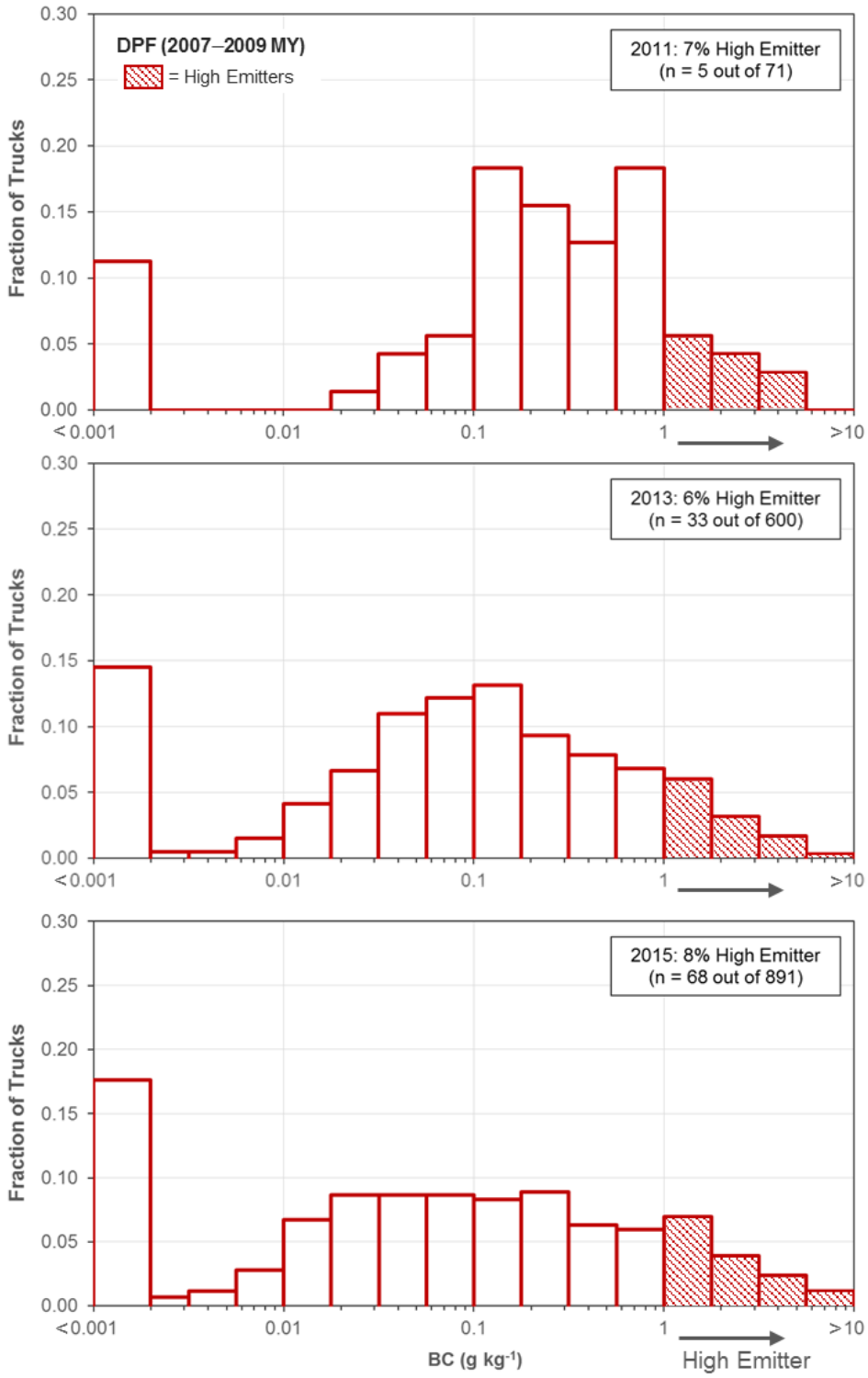

Figure 14. Distribution of BC emission factors for 2007–2009 model year engines equipped with DPFs, as measured in 2015. High-emitting trucks are defined as having a BC emission factor greater than the category-average of 1.1 g kg^{-1} for trucks without DPF (see text).

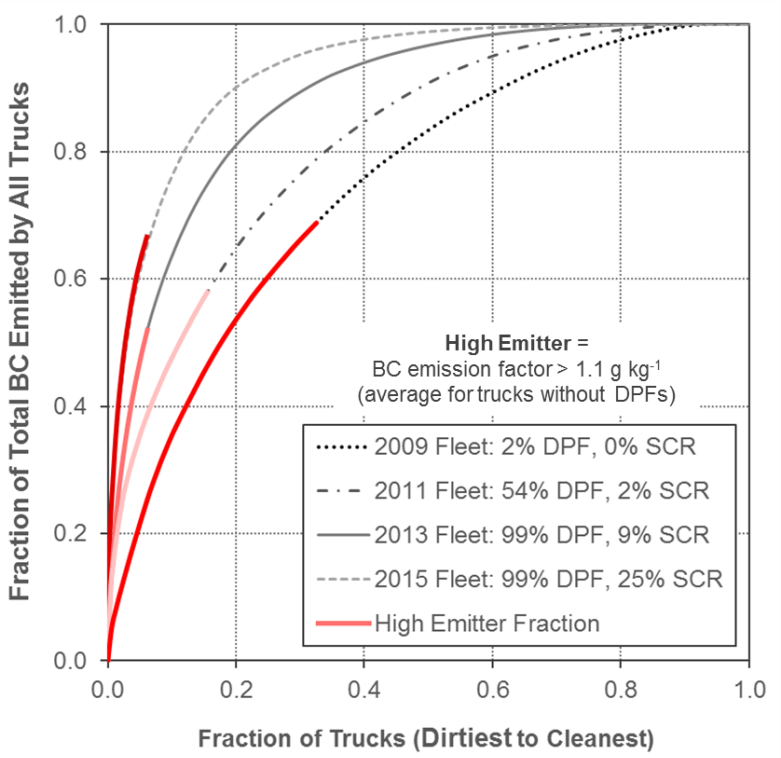

Figure 15. Cumulative emission factor distribution for BC over time as the Drayage Truck Regulation was implemented at the Port of Oakland. Emission factors for individual trucks were ranked from highest to lowest, with high-emitters defined as those trucks emitting BC at levels greater than the category-average of 1.1 g kg^{-1} for trucks without DPF (see text).

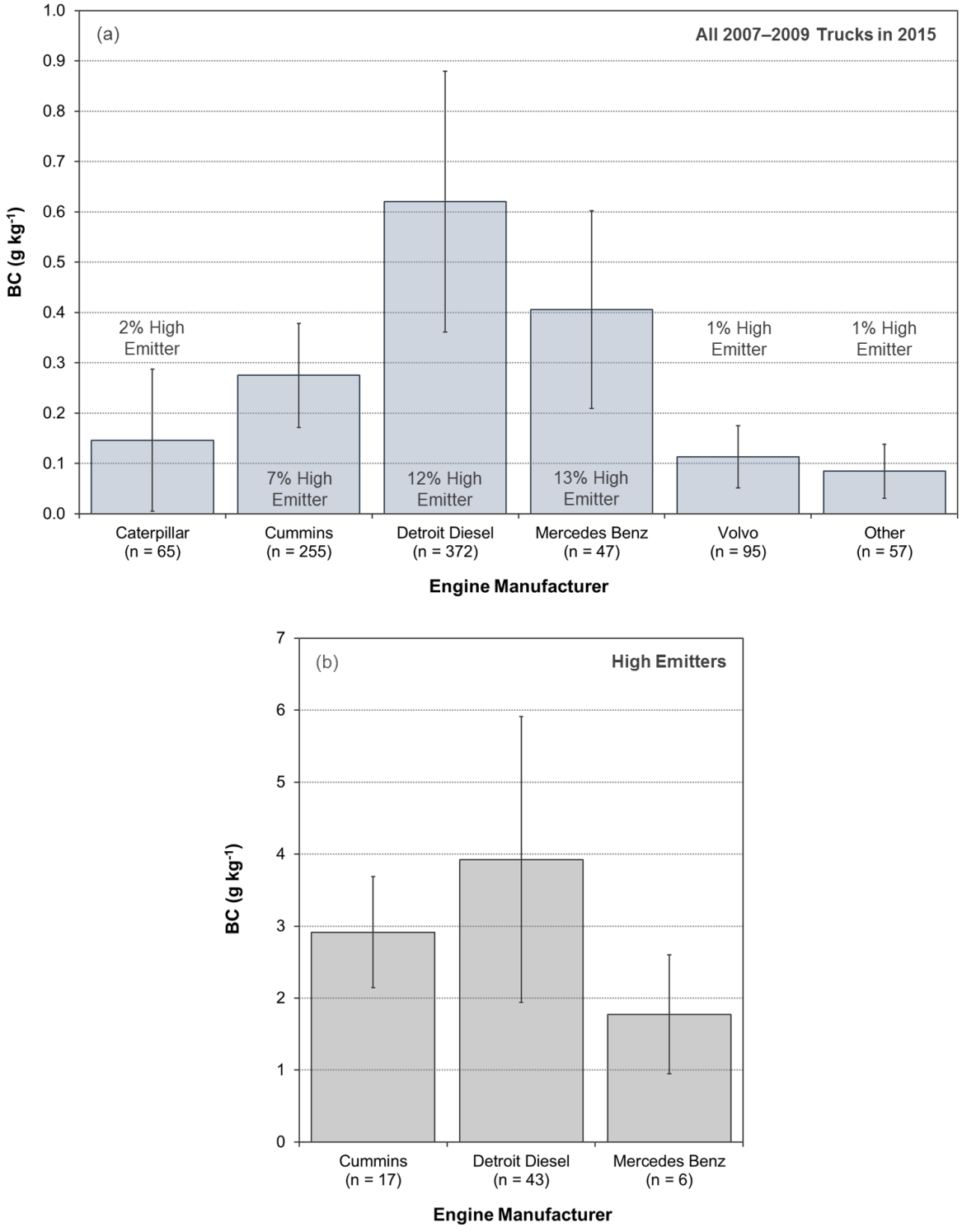

Figure 16. Average BC emission factors by engine manufacturer for (a) all 2007–2009 trucks measured in 2015, and (b) those 2007–2009 trucks classified as high emitters. The definition of a high-emitter used here is trucks emitting BC at levels above 1.1 g kg^{-1}, which is characteristic of trucks without DPF in prior studies at this site (see text).

3.6 Emissions Variability. The exhaust from 224 individual trucks was sampled 2 to 9 times during the 2015 campaign. Figures 17 and 18 plot individual emission factors against the average emission factor for each truck across six of the pollutants measured. The variability in repeat emission factor measurements for individual trucks is indicated by scatter about the one-to-one line. BC, PM, and NO_x have the best correlations and show the least scatter (Figures 17a and 18a–b), whereas PN shows the most scatter (Figure 18c).

Further analysis of these repeat measurements also indicates that high emitters of BC tend to be more chronic rather than intermittent in nature. Of the 73 trucks classified as high emitters in 2015, emissions were sampled more than once from 23 of these trucks, and 61% of the repeat measurements also qualified as high-emission events. The average BC emission factor for all 73 high emitter trucks sampled in 2015 was 3.38 ± 1.19 g kg^{-1}. For those 23 high emitter trucks with replicate measurements, the average BC emission rate when all repeat measurements are included (i.e., including the 40% that didn't qualify has high-emission events when measured again) is slightly lower at 2.65 ± 0.82 g kg^{-1}, but still 8.6 times greater than the 2015 fleet-average of 0.31 ± 0.09 g kg^{-1}.

If all high-emitters were removed from the 2015 fleet, the fleet-average emission rate would decrease to 0.11 ± 0.01 g kg^{-1}. Such a decrease would have led to a *reduction* of $61 \pm 20\%$ between Phases 1 and 2 of the Drayage Truck Regulation, rather than the observed $12 \pm 35\%$ *increase*. The combined effect of both phases of the regulation would then have been an $91 \pm 23\%$ decrease in emitted BC between 2009 and 2015. Greater efforts to prevent trucks from becoming high-emitters in the first place (e.g., via improved durability of emission control

systems), and/or identifying and repairing high-emitting trucks, should be considered as possible next steps for reducing BC emissions.

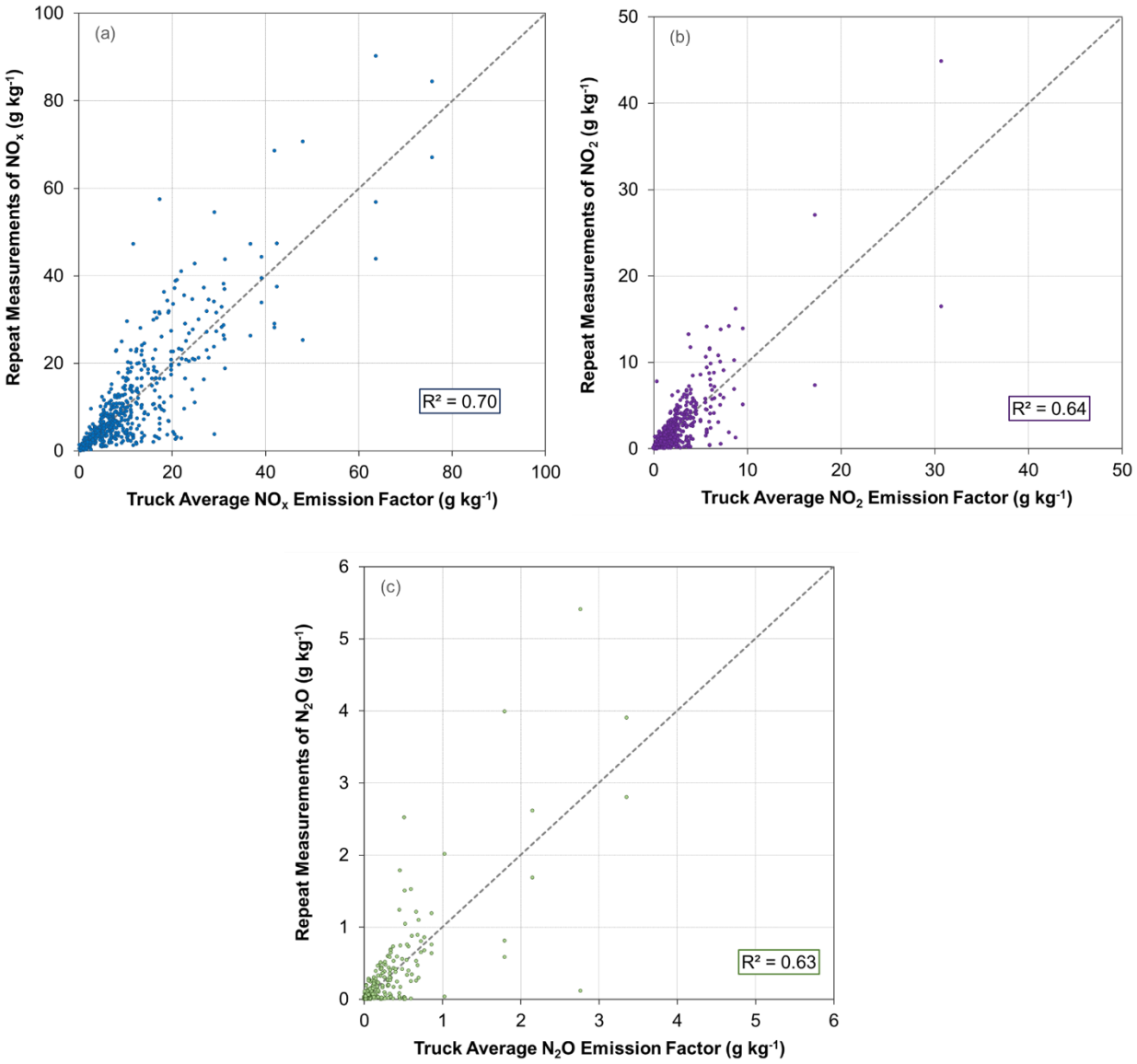

Figure 17. Repeat (a) NO_x, (b) NO_2 (determined by difference), and (c) N_2O measurements for 224 trucks, with each individual measurement plotted on the vertical axis against the corresponding average emission factor for the same truck on the horizontal axis. Repeatability of the measurement is indicated by the degree of scatter about the dashed 1:1 line.

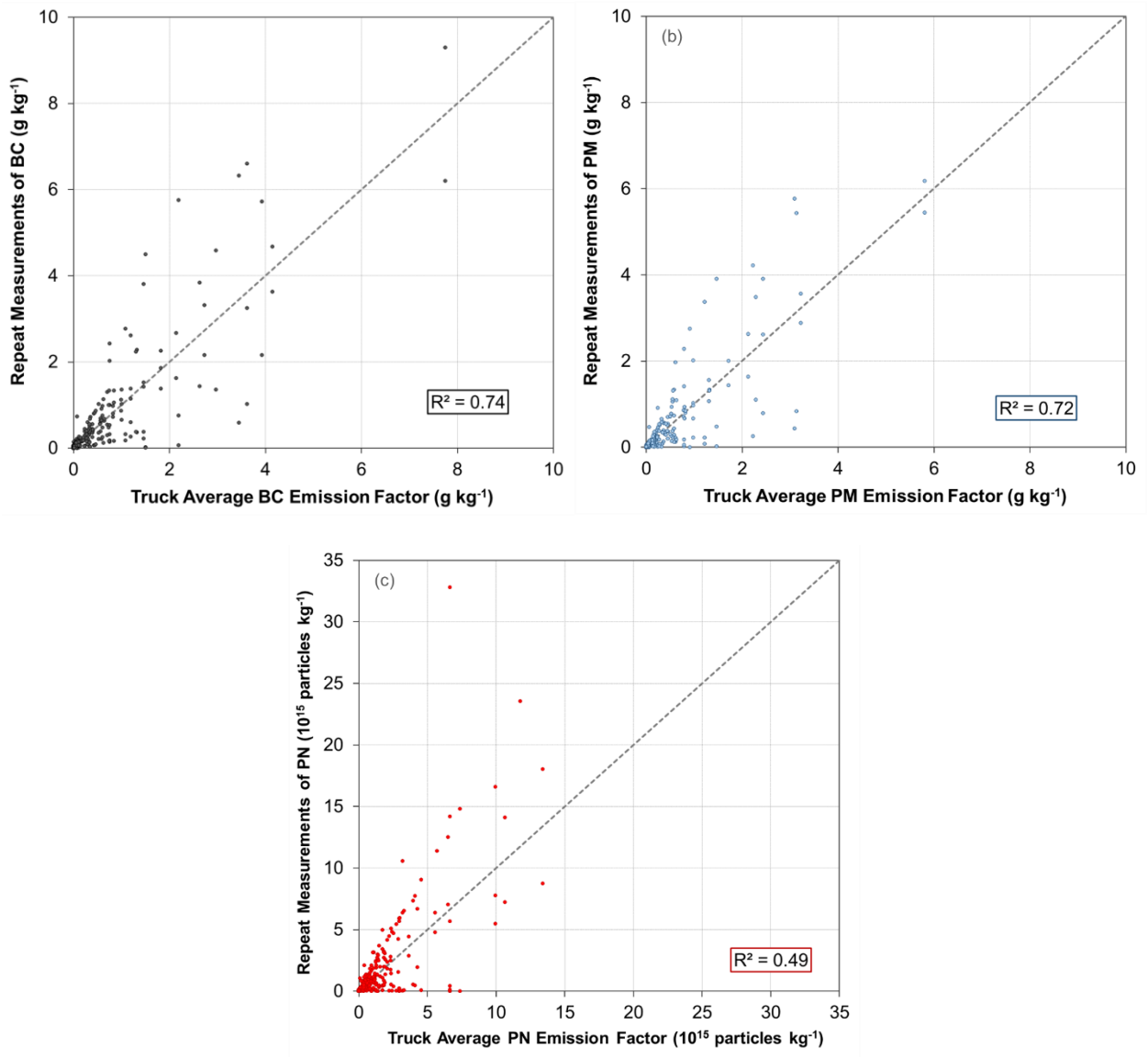

Figure 18. Repeat (a) BC, (b) fine PM (derived from the DMM measurements), and (c) PN measurements for 224 trucks, with each individual measurement plotted on the vertical axis against the corresponding average emission factor for the same truck on the horizontal axis. Repeatability of the measurement is indicated by the degree of scatter about the dashed 1:1 line. A single high-emitting outlier was removed from these three plots.

4. CONCLUSIONS

The Statewide Drayage Truck Regulation has markedly changed the drayage truck fleet at the Port of Oakland. Between 2009 and 2015, the truck fleet turned over rapidly: the median engine age decreased from 11 to 7 years, the fraction of the fleet equipped with DPFs increased from 2 to 99%, and adoption of SCR increased from 0 to 25%. As a result of this modernization of the truck fleet, average emission rates of NO_x, BC, and PN decreased by 70 ± 9%, 73 ± 22%, and 74 ± 27%, respectively. The more widespread use of SCR after Phase 2 of the regulation was implemented has helped achieve further reductions of nitrogen oxides emissions. Between 2013 and 2015, the fraction of the fleet equipped with SCR increased from 9 to 25%, which caused the fleet-average emission rates of NO_x and NO_2 to decrease by 36 ± 7% and 23 ± 10%, respectively. An undesirable increase in fleet-average NO_2 emissions that resulted from widespread adoption of DPFs has been partially mitigated by increased use of SCR.

Between 2013 and 2015, use of DPFs to control particulate matter emissions was stable and near-universal for trucks at the Port, but the fleet-average BC emission factor nevertheless increased by 12 ± 35%. This unexpected increase was caused by a factor of 1.5 increase of the BC emission rate from 2007–2009 model year engines over the same period. Our findings indicate that performance of the installed emissions control systems on some of these engines is deteriorating over time. As a result, trucks with DPF system failures and/or other emissions-related malfunctions are now responsible for a majority of the BC emitted by the Port truck fleet. Based on these observations, future efforts to reduce BC emissions should aim to improve durability/reduce the failure rate of installed emission control systems.

5. REFERENCES

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6. APPENDIX

Table A1. Instrumentation used to measure truck exhaust emissions in this 2015 study.

Parameter	Instrument	Time Resolution
CO_2	Nondispersive infrared gas analyzer (LI-COR LI-7000)	2–5 Hz
NO, NO_x (the difference of which gives NO_2)	Two chemiluminescent analyzers (ECO Physics CLD-64)	2 Hz
NO_2	Cavity attenuated phase shift (CAPS) technique for direct absorption (Aerodyne CAPS NO_2)	1 Hz
BC	Aethalometer (Magee Scientific AE16) and photoacoustic absorption spectrometer (PAS) with reciprocal nephelometer (custom)	1 Hz
PM_2	Light scattering laser photometer (DustTrak II 8530) with PM cut-point established via upstream cyclone (BGI, Inc.)	1 Hz
$PM_{1.2}$	Electrical low pressure impaction (Dekati Mass Monitor 230)	1 Hz
PN	Ultrafine, butanol-based condensation particle counter (TSI 3776); lower size limit of 2.5 nm	2 Hz
N_2O	Off-axis integrated cavity output spectroscopy technique for cavity enhanced absorption (LGR, Enhanced Performance model)	1 Hz

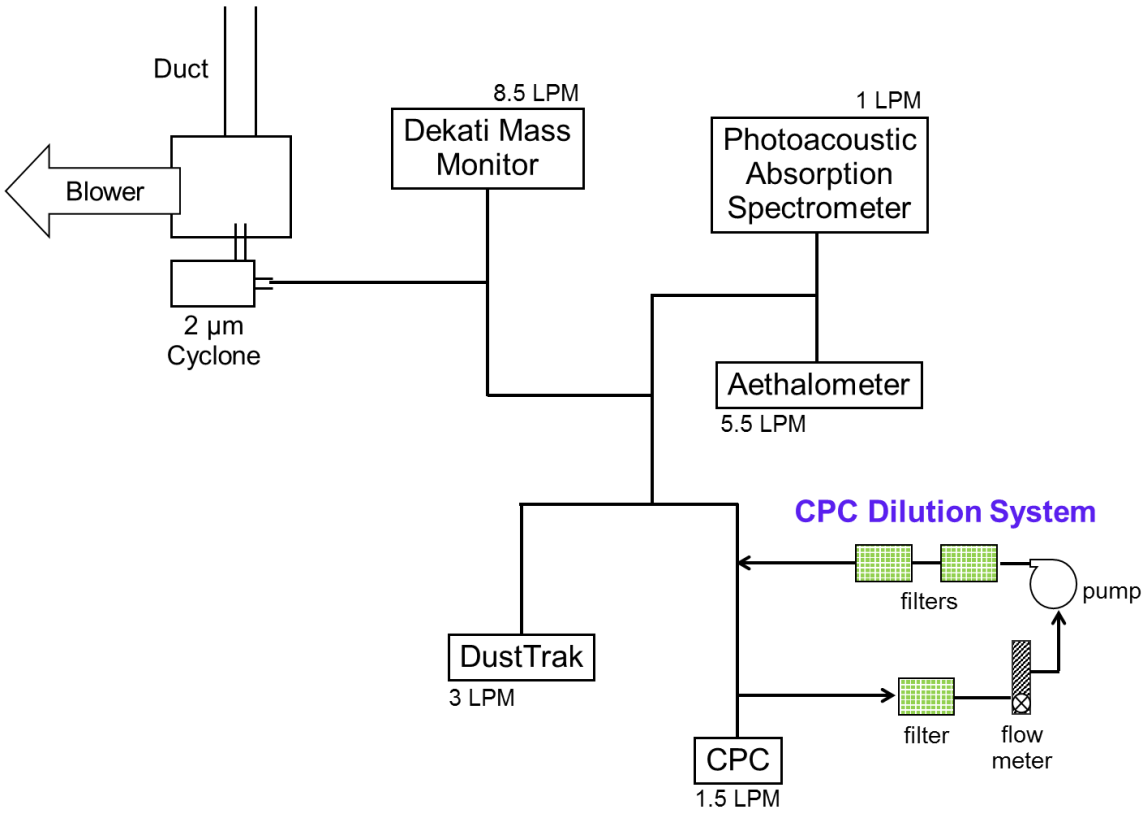

Figure A1. Schematic of particle sampling instruments and their flow rates, as well as the in-line dilution system for the condensation particle counter.

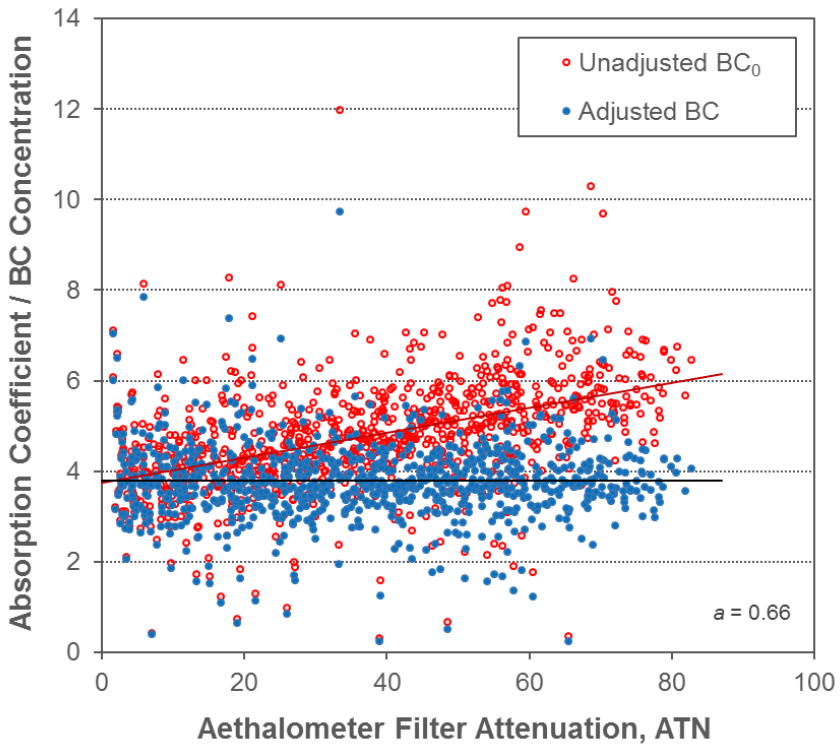

Figure A2. Verification of site-specific aethalometer correction parameter, a = 0.66, as reported in Preble et al. (2015). Here, BC concentration measured using an aethalometer and absorption coefficient measured using a photoacoustic absorption spectrometer (PAS) are compared. Unadjusted BC concentrations (BC_0) depend on aethalometer filter loading (i.e., ATN) whereas the adjusted BC concentrations (BC) are independent of ATN.

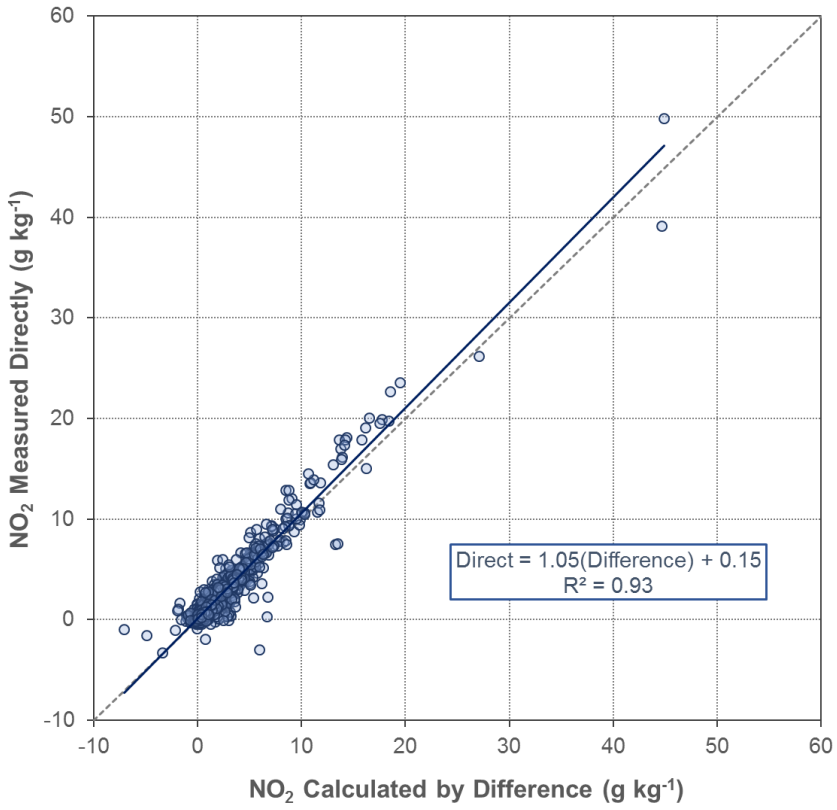

Figure A3. Nitrogen dioxide (NO_2) emission factors determined by difference with two chemiluminescent analyzers versus directly by absorption for individual heavy-duty diesel trucks at the Port of Oakland in 2015.

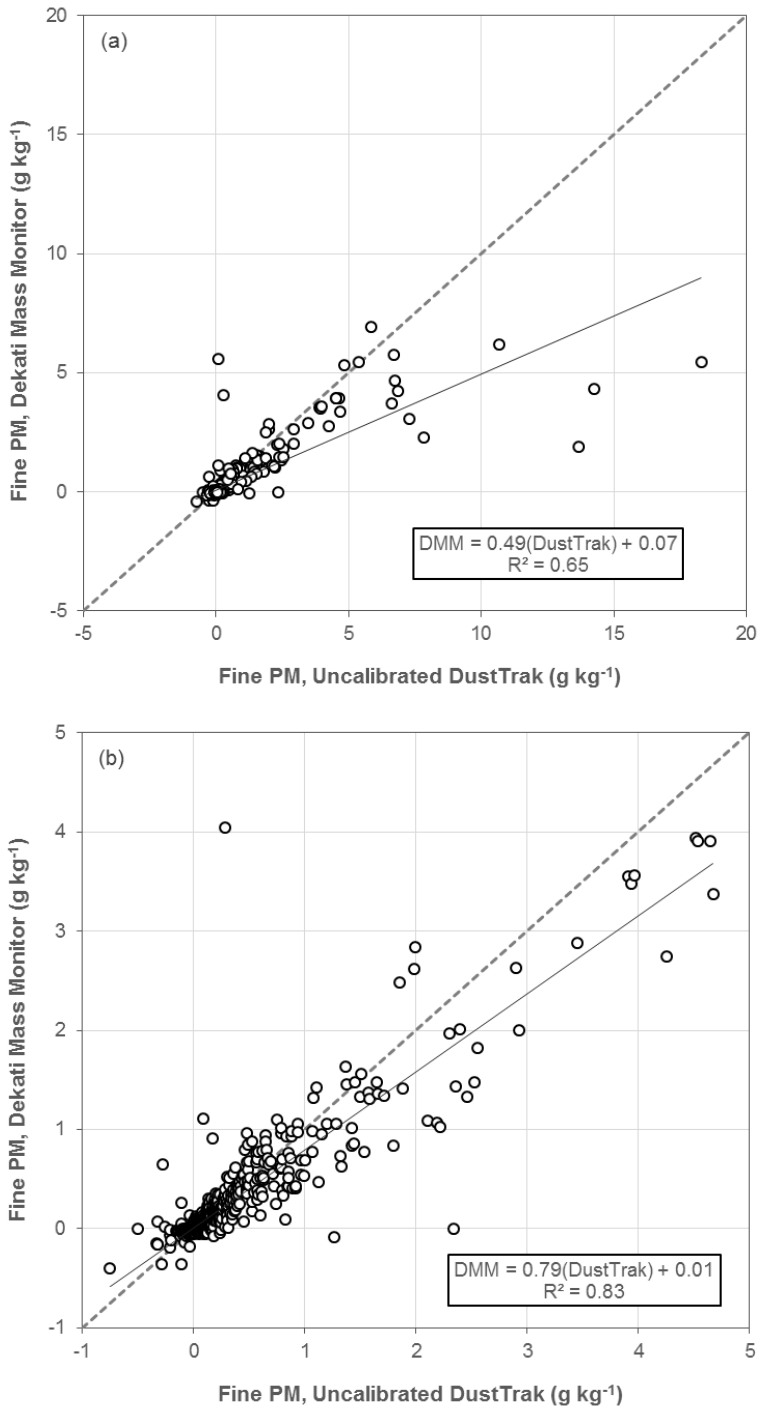

Figure A4. Fine particle mass (PM) emission factors determined with a DustTrak versus a Dekati Mass Monitor (DMM) at the Port of Oakland in 2015. The same data is shown in both (a) and (b), but the regression of (a) includes all data whereas the linear regression for (b) isolates data where BC emission factors are <5 g kg^{-1}. Note that the DustTrak measurement was not calibrated to heavy-duty diesel truck exhaust. Also, the DustTrak measured PM_2 while the DMM measured $PM_{1.2}$.

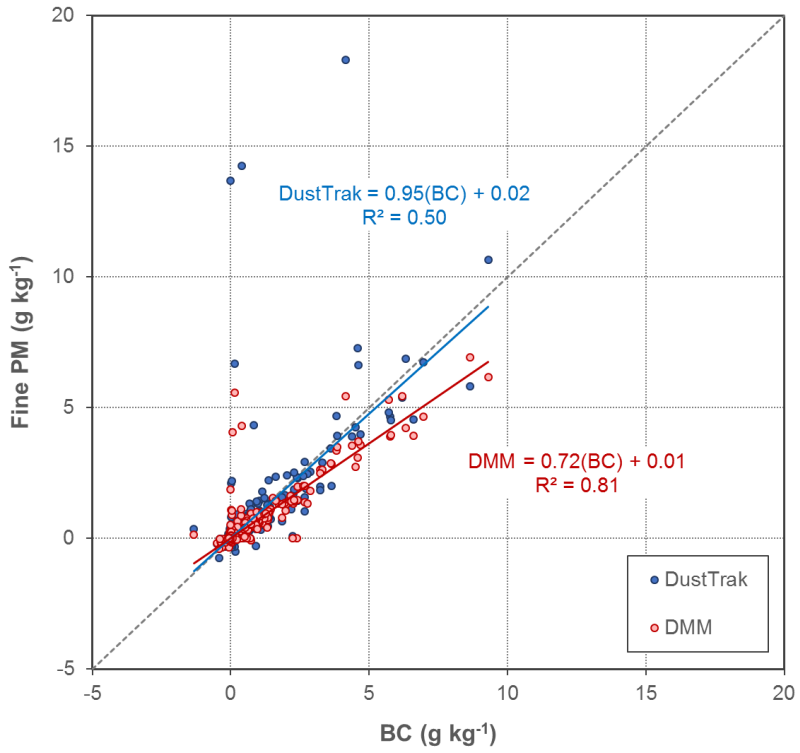

Figure A5. Fine particle mass (PM) emission factors determined with a DustTrak and a Dekati Mass Monitor (DMM) versus corresponding black carbon (BC) emission factors found for individual heavy-duty diesel trucks sampled at the Port of Oakland in 2015.

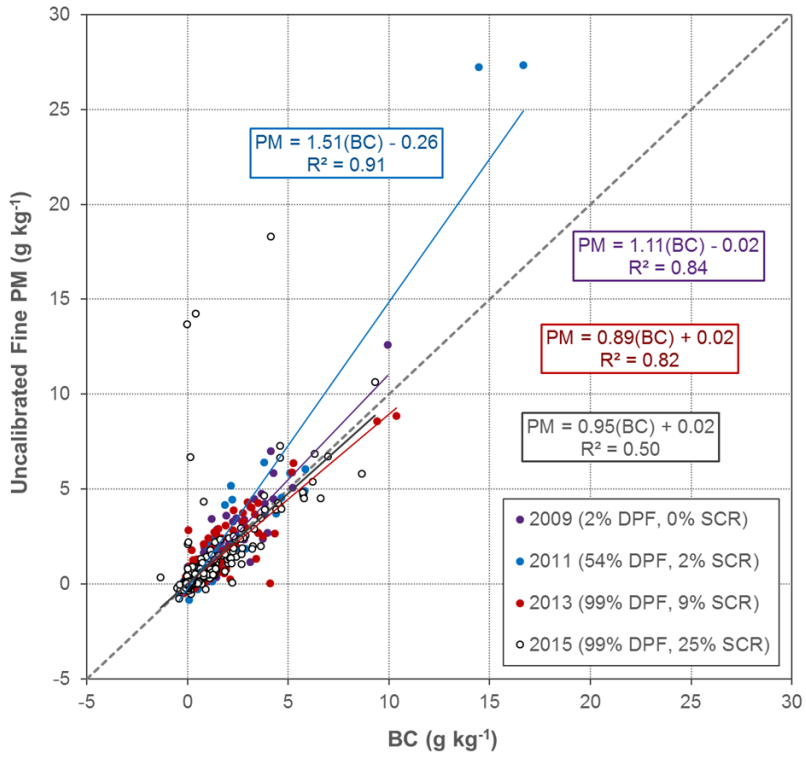

Figure A6. Scatter plot of fine PM and BC emission factors as measured in 2009, 2011, 2013, and 2015 at the Port of Oakland. Note that the fine PM emission factors are derived from concentrations measured with an uncalibrated instrument (see text), so the emission trend over time rather than the absolute values is considered to be relevant to the fleet's PM emissions.

Table A2. Average emission factors (± 95% confidence intervals) for the drayage truck fleet characterized by engine control technology and engine model year, as measured in 2009, 2011, 2013, and 2015 at the Port of Oakland.

Sampling Year	Fleet or Truck Category	Range of Engine Model Years	Median Engine Model Year	Number of Trucks	NO_x (g kg^{-1})	NO_2 (g kg^{-1})	NO_2/NO_x Emission Ratio	BC (g kg^{-1})	Fine PM (g kg^{-1})	PN (10^{15} particles kg^{-1})
2009	Fleet (2% DPF, 0% SCR)	1970–2009	1997	169–172	32.6 ± 2.3	1.11 ± 0.58	0.03 ± 0.02	1.15 ± 0.19	1.25 ± 0.23	3.9 ± 0.8
2011	Fleet (54% DPF, 2% SCR)	1994–2011	2004	363–368	18.0 ± 1.2	2.09 ± 0.40	0.12 ± 0.02	0.67 ± 0.14	0.76 ± 0.22	2.6 ± 0.5
	No DPF	2004–2006	2005	166–174	15.9 ± 1.7	0.51 ± 0.30	0.03 ± 0.02	1.12 ± 0.28	1.28 ± 0.45	3.3 ± 0.7
	Retrofit DPF	1994–2003	1999	119–124	23.0 ± 1.2	3.39 ± 0.50	0.15 ± 0.02	0.26 ± 0.09	0.34 ± 0.14	1.4 ± 0.8
	DPF	2007–2009	2008	70–75	15.0 ± 3.6	3.48 ± 1.50	0.23 ± 0.11	0.31 ± 0.11	0.25 ± 0.11	3.0 ± 1.6
	DPF + SCR	2010–2011	2001	5–6	10.2 ± 3.8	3.77 ± 1.40	0.37 ± 0.19	0.44 ± 0.41	0.20 ± 0.30	1.1 ± 1.6

2013	Fleet (99% DPF, 9% SCR)	1994–2013	2007	934–1005	15.4 ± 0.9	2.84 ± 0.22	0.18 ± 0.02	0.28 ± 0.05	0.26 ± 0.05	2.0 ± 0.4
	No DPF	2004–2006	2005	14–15	24.0 ± 7.7	1.12 ± 0.81	0.05 ± 0.04	1.01 ± 0.54	1.18 ± 0.61	10.6 ± 5.4
	Retrofit DPF	1994–2006	1998	258–281	27.3 ± 1.9	4.14 ± 0.50	0.15 ± 0.02	0.34 ± 0.08	0.39 ± 0.09	2.0 ± 0.6
	DPF	2007–2009	2008	581–626	11.5 ± 0.9	2.58 ± 0.25	0.22 ± 0.03	0.26 ± 0.06	0.21 ± 0.06	1.9 ± 0.5
	DPF + SCR	2010–2013	2011	81–94	4.9 ± 1.2	1.00 ± 0.32	0.20 ± 0.08	0.06 ± 0.02	0.07 ± 0.04	1.0 ± 0.6
2015	Fleet (99% DPF, 25% SCR)	1996–2016	2008	1194–1218	9.9 ± 0.6	2.20 ± 0.19	0.22 ± 0.02	0.31 ± 0.09	0.34 ± 0.11	1.0 ± 0.3
	No DPF	1996–2006	2005	10–11	11.5 ± 4.8	2.08 ± 0.90	0.18 ± 0.11	0.56 ± 0.50	0.67 ± 0.56	0.7 ± 0.3
	Retrofit DPF	N/A								
	DPF	2007–2009	2008	871–903	11.0 ± 0.7	2.53 ± 0.22	0.23 ± 0.02	0.39 ± 0.11	0.42 ± 0.15	1.1 ± 0.3
	DPF + SCR	2010–2016	2012	286–304	6.4 ± 1.2	1.22 ± 0.34	0.19 ± 0.06	0.06 ± 0.03	0.10 ± 0.09	0.8 ± 0.3

Table A3. Average emission factors (± 95% confidence intervals) for the drayage truck fleet characterized by engine control technology and engine model year, as measured in 2015 at the Port of Oakland.

Sampling Year	Fleet or Truck Category	Range of Engine Model Years	Median Engine Model Year	Number of Trucks	NO_2 by Difference (g kg^{-1})	NO_2 Direct (g kg^{-1})	Fine PM, DustTrak (g kg^{-1})	Fine PM, DMM (g kg^{-1})	N_2O (g kg^{-1})
2015	Fleet (99% DPF, 25% SCR)	1996–2016	2008	1095–1218	2.20 ± 0.19	2.45 ± 0.20	0.34 ± 0.11	0.22 ± 0.04	0.16 ± 0.03
	No DPF	1996–2006	2005	10–11	2.08 ± 0.90	2.46 ± 1.06	0.67 ± 0.56	0.33 ± 0.36	0.07 ± 0.06
	Retrofit DPF	N/A							
	DPF	2007–2009	2008	811–903	2.53 ± 0.22	2.87 ± 0.24	0.42 ± 0.15	0.26 ± 0.05	0.06 ± 0.01
	DPF + SCR	2010–2016	2012	274–304	1.22 ± 0.34	1.23 ± 0.32	0.10 ± 0.09	0.07 ± 0.05	0.44 ± 0.11