Does California’s EMFAC2017 vehicle emissions model underpredict California light-duty gasoline vehicle NO\textsubscript{x} emissions?

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ABSTRACT

On-road remote sensing measurements of light and medium-duty gasoline vehicles collected within California’s South Coast Air Basin since 1999 generally fall within the range of observed summer ambient molar NO\textsubscript{x}/CO measurements collected during morning rush hours. Compared with ambient and on-road emissions, the California Air Resources Board EMFAC model underpredicts 2018 gasoline vehicle NO\textsubscript{x} emission factors by more than a factor of 2.6. Contributing to these differences is that vehicles older than model year 2006 have NO\textsubscript{x} emission deterioration rates that are up to four times higher on-road than predicted by the EMFAC model. A fuel-based inventory using the 2018 on-road gasoline emission factors for CO and NO\textsubscript{x} results in total CO emissions similar to the basin inventory but NO\textsubscript{x} emissions that are 74% higher than the inventory. The higher NO\textsubscript{x} emission estimates from on-road gasoline vehicle measurements make their contribution to the inventory slightly larger than heavy-duty diesel vehicles. We have found LEV I (1994–2003) gasoline vehicles are a major source of these on-road emissions and that significant NO\textsubscript{x} reductions in the South Coast Air Basin are being overlooked by not targeting the high emitters for removal.

Implications: A comparison of ambient and on-road vehicle molar NO\textsubscript{x}/CO ratios collected in California’s South Coast Air Basin with those predicted by California’s EMFAC2017 vehicle emissions model shows that the model significantly underpredicts NO\textsubscript{x} emission factors for gasoline vehicles. This results in a 74% underestimate of the contribution of gasoline vehicles to the basin’s NO\textsubscript{x} inventory, with the contribution from gasoline vehicles comparable to that from heavy-duty diesel trucks. This likely means that current projections for future NO\textsubscript{x} emission reductions from mobile sources in the basin will not be realized unless additional NO\textsubscript{x} reductions are obtained from older gasoline vehicles.

Introduction

With the introduction of the National Ambient Air Quality Standards (NAAQS) in the early 1970s as part of the Clean Air Act, the U.S. began establishing networks of ambient air monitors in urban areas across the country (U. S. Environmental Protection Agency). The NAAQS were intended to limit common pollutants found in outdoor air that were considered to be harmful to public health. The species to be monitored were carbon monoxide (CO), lead, nitrogen dioxide (NO\textsubscript{2}), particulate matter, ozone and sulfur dioxide. California’s South Coast Air Basin (SoCAB), which includes Los Angeles, has over the years experienced elevated levels of most of these species but through tremendous reductions in mobile and stationary source emissions over the last 60 yr now only exceeds levels for particulate matter and ozone (Pollack et al. 2013; United States Environmental Protection Agency 2020a, 2020b; Warneke et al. 2012).

Since Haagen-Smit first documented the presence of ozone in Los Angeles’ air, reducing it to healthful levels has proven to be a difficult task (Haagen-Smit, Bradley, and Fox 1953; Haagen-Smit and Fox 1954). Unlike most of the criteria pollutants, ozone is not directly emitted but is a secondary product formed in a nonlinear reaction involving volatile organic compounds (VOC), carbon monoxide (CO) and oxides of nitrogen (NO\textsubscript{x} = NO + NO\textsubscript{2}) in sunlight (Finlayson-Pitts and Pitts 2000; Stedman 2004). Despite significant reductions in ozone levels in the SoCAB, with the 2015 revisions of the NAAQS 8-hr ozone rule that lowered the standard to 70ppb (fourth-highest daily maximum, averaged across 3 consecutive years), compliance is not expected for many decades (Fujita et al. 2013; Parrish et al. 2017; U. S. Environmental Protection Agency 2015).

Because of the interplay between VOC and NO\textsubscript{x} emissions, ozone production can be limited by either species depending on the atmospheric chemistry at
a particular location (Stedman 2004). HC-limited ozone formation, as documented by increases in ozone formation on weekends when NOₓ emissions from diesel vehicles decrease significantly, has previously been the predominate mechanism in the SoCAB; however, some recent observations and models are predicting this to be changing in some areas of the basin to a NOₓ-limited regime (Baidar et al. 2015; Chinkin et al. 2003; Fujita et al. 2016; Laughner and Cohen 2019; Pollack et al. 2012; South Coast Air Quality Management District 2017). This has shifted the focus of California regulatory agencies to significantly lowering NOₓ emissions (45% reduction beyond current control measures by 2023) as called for in the SoCAB 2016 State Implementation Plan (South Coast Air Quality Management District 2017).

Achieving the desired NOₓ reductions relies on an accurate local emissions inventory. On-road vehicles in 2018 were estimated to emit approximately 50% of the SoCAB NOₓ emissions, with diesel trucks estimated to contribute the larger share (60%) of this total (see supporting material). In California, on-road vehicle emissions contributions to the inventory are estimated using the California Air Resources Board developed EMission FACtors (EMFAC) vehicle emissions model. The current version, EMFAC2017, combines vehicle emission factors (grams per mile and per start) for selected pollutants with vehicle activity (miles driven and starts) to estimate total emissions (California Air Resources Board 2020b).

Using SoCAB ambient air monitor measurements and on-road vehicle emission measurements, we examine the SoCAB on-road mobile source NOₓ emission inventory, and in particular the on-road gasoline and diesel apportionment. The apportionment of NOₓ emissions is particularly important since any policies aimed toward reducing NOₓ emissions need to be targeted appropriately in order to be effective. Absent this the expected reductions will not materialize and the needed improvements in ozone levels will be pushed even further into the future.

**Experimental methods**

**Ambient measurements**

Long-term (1960–2010) ambient molar NOₓ/CO trends in California’s SoCAB were described by Pollack et al. for a number of field measurement campaigns and two basin surface network monitoring sites (Azusa and Upland) (Pollack et al. 2013). Hassler et al. extended this NOₓ/CO trend through 2015 and added eight additional surface network monitoring sites (La Habra, Long Beach, Magnolia, Mira Loma, North Main, Pomona, Reseda and Rubidoux) (Hassler et al. 2016). The NOₓ/CO ratios were determined by bivariate least squares linear regression using only summer (May–September), non-holiday, weekday morning (05:00–09:00 local time) hourly ambient measurements. Ratios were only reported for sites where two-thirds of the possible number of hourly data existed and where the resulting NOₓ/CO correlation coefficient was greater than or equal to 0.5 ($r^2 \geq 0.5$) helping to restrict the measurements to fresh local motor vehicle emissions. We have extended this data record using this approach through 2018, using data from nine of the 10 sites as the Magnolia site ceased operation at the end of 2014.

**On-road measurements**

On-road vehicle tailpipe exhaust measurements have been collected with a remote sensor developed at the University of Denver named Fuel Efficiency Automobile Test (FEAT) (Bishop and Stedman 1996). FEAT is composed of an infrared (IR) and ultraviolet (UV) light source placed across a single lane roadway from four nondispersive IR and one (NO only for the pre-2008 data sets) or two dispersive UV detectors (includes NO₂ starting in 2008) that allow the measurement of vehicle exhaust gases as a molar ratio to exhaust CO₂ (i.e., CO/CO₂, NO/CO₂, NO₂/CO₂, etc.) (Burgard et al. 2006a). Each measured species ratio is scaled using certified (±2% accuracy) gas cylinder ratios measured daily as needed at each site by FEAT. This corrects for variations in instrument sensitivity and most importantly ambient CO₂ levels caused by changes in atmospheric pressure, temperature and background pollutants. The molar ratios can also be converted into fuel-based emission factors of grams of pollutant per kg of fuel by the carbon balance method. This uses a carbon mass fraction for the fuel of 0.86 and a doubling of the HC/CO₂ ratio to normalize the reading with a flame ionization detector and compensate for the weak IR absorbance of many aromatic compounds (Singer et al. 1998). Each measurement includes a video image of the license plate of the vehicle that is manually transcribed and used to retrieve non-personal vehicle information (i.e. age and type) from the California registration records that is combined with the emission measurements into a final database for analysis.

On-road emission measurements have been collected using FEAT from light and medium-duty vehicles in California’s SoCAB since 1989. However, NO measurements were not collected until the late 1990s when the instrumentation for collecting those measurements was developed (Popp, Bishop, and Stedman 1999). Beginning in 1999, emission measurements have been collected at
the on-ramp from southbound La Brea Ave. to eastbound I-10, about midway between downtown LA and Santa Monica. To date there have been eight data sets collected at this West Los Angeles location (1999, 2001, 2003, 2005, 2008, 2013, 2015 and 2018) that includes more than 165,000 vehicle emission measurements (Bishop et al. 2010; Bishop and Stedman 2008, 2015). The 1999–2005 measurements were made in the fall of each year. Beginning with the 2008 measurements we began collecting emission measurements of NO₂, allowing for the reporting of vehicle NOₓ emissions, and the measurement dates switched to the spring (Burgard et al. 2006b). Though only a single site within the basin, other researchers have shown that these measurements are representative of basin-wide emissions (Hassler et al. 2016; Kim et al. 2016; Nowak et al. 2012; Pollack et al. 2013).

Seven additional data sets, also collected within the basin since 1999, in Riverside (1999, 2000 and 2001), at the intersection of I-710 and SR91 (1999), Van Nuys (2010) and at two sites in Lynwood (2018) are included in this analysis (Bishop 2019; Bishop et al. 2012; Bishop and Stedman 2008). Heavy-duty diesel trucks with either elevated or ground-level exhaust emissions were also measured in the spring of 2017 at the Peralta weigh station on SR 91 in the Anaheim Hills of the SoCAB (Haugen et al. 2018). All of the databases used in this study, as well as many others compiled by the University of Denver, are available at www.feat.biochem.du.edu.

**EMFAC2017 modeling**

California’s EMFAC2017 vehicle emissions factor model was run using the online EMFAC2017 web database v1.0.2 (https://www.arb.ca.gov/emfac/2017/). For comparison with the ambient measurement ratios, summer emissions (online model allows summer or annual estimates) for the SoCAB were modeled for years 2001, 2003, 2005, 2008, 2010, 2013, 2015 and 2018 to match the West Los Angeles remote sensing measurement years. This version of the EMFAC model only predicts emissions back to calendar year 2000, so we did not compare modeled emissions to the 1999 measurements.

Running exhaust molar ratios were calculated for each year by summing the model predicted short tons per day for the 13 gasoline vehicle types output by EMFAC. The tons were converted into grams and the grams into moles of each pollutant and then ratioed. Model years were aggregated by the model for each vehicle type and predictions were generated for two speeds, aggregated over all of the drive cycles included in the model and at a fixed 20 mph. The latter most closely matches the average speed observed at the West Los Angeles FEAT on-road measurement site. The supporting material includes a sample output of this process (see Table S6).

Annual CO and NOₓ running exhaust emission factors (grams/kilogram of fuel) for 2018 were calculated by model year for gasoline-powered light-duty passenger vehicles (model type LDA) and trucks (fuel consumption weighted composite of model vehicle types LDT1, LDT2 and MDV) from the tons/day predicted by the EMFAC model. The chosen truck types cover the weight classes observed at the West Los Angeles site and account for more than 87% of the predicted truck gasoline fuel consumption by the model. Aggregated speeds were used for these calculations as the model does not provide fuel consumption at a fixed speed setting. The model predicted tons/day for each model year were converted into grams/day, the predicted gallons of gasoline consumed were converted into kilograms assuming a density of 0.75 g/mL for California reformulated gasoline and the two ratioed for each pollutant. The supporting material includes examples of the 2018 calculations (see Tables S8 and S9).

Annual fuel-specific NOₓ emission factors for the 2017 heavy-duty diesel truck measurements were calculated using EMFAC2017 in a similar manner. A fuel weighted composite of diesel trucks included all medium–heavy-duty diesel trucks greater than 26,000 lb (EMFAC model type T6) and all heavy–heavy-duty diesel trucks (EMFAC model type T7) except those using the Truck and Bus rule agricultural provision (T7 Ag) with aggregated speeds (California Air Resources Board 2018). The density of ultra-low sulfur diesel fuel was assumed to be 0.86 g/mL.

**Results**

**NOₓ/CO trends**

Maintaining the graphical approach used in Hassler et al., Figure 1 plots the log of the molar NOₓ/CO ratios for the California SoCAB surface network sites against measurement year for the 1983–2018 time period (Hassler et al. 2016). The solid lines in Figure 1 represent a quadratic fit applied to the 1983–2009 ambient measurement ratios and a best fit straight line for the remaining 2010–2018 ratios. The scatter in these measurements reflects the distribution differences between the gasoline and diesel fleets, vehicle operating characteristics and any observed contributions from stationary or non-road sources. It is expected that gasoline vehicles will figure more prominently in the CO emission contributions while diesel vehicles will contribute a larger fraction of the NOₓ emissions.
Figure 1. Molar NOx/CO emission ratios from California’s SoCAB ambient monitors (○), on-road measured average ratios for gasoline vehicles from six basin locations and EMFAC2017 running exhaust modeled ratios for gasoline-only vehicles with aggregated speed or fixed at 20mph versus measurement year. A quadratic fit is shown for the 1983–2009 ambient data and a best fit straight line for the 2010–2018 measurements. Uncertainties for the FEAT measurements are standard error of the mean calculated using the daily means.

The ambient ratios increase steadily until about 2010 after which they level out and then decrease. The rise is consistent with the observation by Pollack et al. that vehicle CO emissions decreased at about twice the rate of NOx emissions over the earlier time period (Pollack et al. 2013). Since 2010, on-road NOx emissions have seen significant reductions from both the light-duty gasoline and heavy-duty diesel fleets (Bishop and Haugen 2018; Haugen et al. 2018). These reductions have been driven by the introduction of LEV II light-duty vehicles in 2009 (phased in between 2004 and 2009 in California) and by the phase-in of selective catalytic reduction systems for NOx control in heavy-duty diesel trucks beginning with 2011 trucks.

FEAT molar NOx/CO ratios for the gasoline portion of each site’s fleet are plotted against measurement year for the 15 data sets collected in the SoCAB since 1999. For the data sets collected prior to 2008, the molar NOx/CO ratios only include measurements for the moles of NO. However, since we have restricted these comparisons to only gasoline-powered vehicles this should only slightly (<1%) underestimate the true NOx/CO ratios as gasoline engines emit little NOx. The number of diesel vehicles with ground level exhaust at these on-road sites is small (1.5–3%) and their inclusion does, as expected, increase the ratios (~8% on average) but does not significantly change the results (Table S7 in the supporting material). Uncertainties displayed are standard error of the mean calculated from the daily means.

This is not a direct comparison with the ambient measurements, as the ratios derived from the on-road measurements exclude diesel-powered vehicles. However, in general, the on-road measurements fall within the lower range of the observed ambient NOx/CO ratios, rising along with them until the 2010 peak. The figure indicates that after 2010 the on-road NOx/CO ratios continue to increase slightly and do not show the decreases observed in the ambient ratios.

As noted, summer EMFAC2017 molar NOx/CO ratios were calculated from the model for the SoCAB gasoline fleet using two speed selections: (1) a static 20 mph and (2) aggregated speeds; these are shown in Figure 1. The aggregated speed setting produces slightly higher NOx/CO ratios than the static 20 mph speed for each of the years modeled. The EMFAC2017 predicted ratios begin in 2001 with values that are in general agreement with the ambient and the on-road measurements but then steadily decrease in the following years. Noticeably absent from the modeled ratio predictions is the rising ratio values found in the ambient and on-road measurements between 2001 and 2010. The 2018 molar NOx/CO ratios estimated by EMFAC are 1.8 (aggregated speed) and 2.4 (20 mph fixed speed) times lower than the average ambient measurements.

Because we are comparing ratios it is not immediately clear whether the disagreement between the model and the ambient measurements is the result of an under-prediction of NOx or an over prediction of CO. Since the on-road measured ratios generally fall within the range of the ambient measurements, we will use the on-road data from the West Los Angeles site to investigate potential differences with the model predictions.
Gasoline vehicle emissions comparison

Figure 2 compares the fuel-specific CO emissions for the 2018 West Los Angeles FEAT measurements and the EMFAC2017 predicted emission factors by model year for gasoline light-duty passenger vehicles (top graph) and light-duty trucks (bottom graph). EMFAC2017 light-duty trucks, as previously mentioned, are a fuel use weighted composite emission factor for the model types LDT1, LDT2 and MDV. The model emission factors were calculated on an annual basis for the SoCAB using aggregated speeds since that speed setting predicted the higher ratio values (see supporting material). Uncertainties for the FEAT data are standard error of the mean determined from the daily measurements. Because of the decline in the number of vehicle measurements with age and their increasing uncertainty the on-road measurement plotted at model year 1990 is the average for all 1987–1993 vehicles.

The emissions by model year comparisons between the on-road measurements and the model predictions, within the uncertainties, are in generally good agreement. We can calculate an age-normalized fleet mean emissions for the EMFAC2017 model output using the fleet CO measurement fractions by model year and vehicle type observed in the 2018 West Los Angeles measurements to further compare the overall agreement (see Table S8). Mean fuel-specific EMFAC2017 CO emissions are slightly over-predicted (+11%) for passenger vehicles (10.5 vs. 9.3 ± 0.5 gCO/kg of fuel) and are slightly underpredicted (−20%) for the light-duty trucks (11.1 vs. 13.9 ± 0.6 gCO/kg of fuel). When combined, the fleet means are not statistically different at the 95% CI (10.7 vs. 11.2 ± 0.6 gCO/kg of fuel).

Figure 3 is the companion graph comparing the fuel-specific NOx emissions by model year for gasoline light-duty passenger vehicles (top) and light-duty trucks

![Figure 2](image-url)  
**Figure 2.** Fuel-specific CO emissions for the 2018 West Los Angeles gasoline fleet (Δ) and the EMFAC2017 predicted 2018 SoCAB gasoline fleet (model type LDA) emissions (●) versus model year for light-duty passenger vehicles (top panel) and trucks (bottom panel). EMFAC2017 truck emission factors are a fuel use weighted composite of model types LDT1, LDT2 and MDV. Uncertainties for the FEAT measurements are standard error of the mean calculated using the daily means.

![Figure 3](image-url)  
**Figure 3.** Fuel-specific NOx emissions for the 2018 West Los Angeles gasoline fleet (Δ) and the EMFAC2017 predicted 2018 SoCAB gasoline fleet (model type LDA) emissions (●) versus model year for light-duty passenger vehicles (top panel) and trucks (bottom panel). EMFAC2017 truck emission factors are a fuel use weighted composite of model types LDT1, LDT2 and MDV. Uncertainties for the FEAT measurements are standard error of the mean calculated using the daily means.
(bottom) for the 2018 FEAT West Los Angeles measurements and the EMFAC2017 model predictions. The NO\textsubscript{x} comparison is quite good for the 2009 and newer model year vehicles that all have near-zero emissions. Unlike the CO emissions comparison, however, the observed on-road gNO\textsubscript{x}/kg of fuel emissions for both vehicle types are substantially higher than predicted by the EMFAC2017 model for 2008 and older models. For passenger vehicles (0.7 vs. 1.8 ± 0.1 gNO\textsubscript{x}/kg of fuel) and trucks (1.1 vs. 3.0 ± 0.1 gNO\textsubscript{x}/kg of fuel) the on-road age normalized mean emissions are factors of 2.6 and 2.7 times higher than predicted by the model. It is these differences in light-duty gasoline NO\textsubscript{x} emissions that are the likely explanation for the differences observed in the EMFAC NO\textsubscript{x}/CO ratio comparison with the ambient and on-road measurements (see Figure 1).

Fujita et al. reported on an underprediction of NO\textsubscript{x} emissions with an earlier version of the model, EMFAC2007, for measurements collected in a Van Nuys, CA tunnel in the summer of 2010 (Fujita et al. 2012). EMFAC2007 underreported NO\textsubscript{x} emissions by factors of 1.2 and 1.4 for the median weekday and weekend measurements, respectively. However, the underprediction was larger (factor of 1.8) for a Sunday morning measurement when the fleet was almost exclusively gasoline-powered vehicles (8 diesel vehicles in 1290 total vehicles) while CO emissions were accurately predicted.

A study to model PM\textsubscript{2.5} levels in California’s San Joaquin Valley came to a similar conclusion using the 2014 version of EMFAC (Kleeman, Kumar, and Dhiman 2019). Observed ambient NO\textsubscript{x} emissions in Fresno in the winter of 2013 were significantly higher during the morning rush hour period than predicted by the model even after including soil NO\textsubscript{x} emissions. Concentrations of total reactive nitrogen were consistently underpredicted due to insufficient levels of NO\textsubscript{x} emissions. The authors concluded that there is an “unknown source of NO\textsubscript{x} emissions that is not currently represented in the emissions inventory” (Kleeman, Kumar, and Dhiman 2019).

On-road measurements collected in Fresno in 2008, prior to the recession, found an older light-duty fleet (~8.7 yr) than data collected during the same campaign in San Jose (~7.9 yr) and West Los Angeles (7.3 yr) (Bishop et al. 2010). The 2008–2009 recession increased fleet age by 2 yr at the West Los Angeles site as observed in 2013 and we would expect fleets in the San Joaquin Valley to have aged at least this much (Bishop and Stedman 2014). This would amplify the number of LEV I vehicles (1994–2003 model years) in the San Joaquin Valley during the 2013 measurements that we have found to have NO\textsubscript{x} emissions most underpredicted by the EMFAC model which could be the source for the missing NO\textsubscript{x} emissions.

One possible explanation for the differences observed in the fuel-specific NO\textsubscript{x} emissions for the 2005 and older model year vehicles is that emission deterioration rates are actually larger on-road than assumed in EMFAC. Using the multiple years of emission measurements collected at the West Los Angeles site and plotting fuel-specific NO\textsubscript{x} emissions for the gasoline fleet versus vehicle age at the time of measurement by model year we can estimate the on-road NO\textsubscript{x} emissions deterioration rates and compare those with model predictions using the same multi-year approach (Bishop and Stedman 2008). The supporting material details the calculation process, and Figure S1 shows the graphs for the multi-year on-road and model predicted NO\textsubscript{x} emission factors. We have used the year 2000 in the EMFAC2017 model for the eighth data point as a substitute for the 1999 on-road measurements since the year 1999 cannot be modeled.

Figure 4 plots the emissions deterioration rates in gNO\textsubscript{x}/kg of fuel/year versus model year for the gasoline fleet obtained from the fitting results using the data shown in Figure S1. The uncertainties plotted are the standard error of the least squares fit for each model year. The deterioration rate comparison has a similar trend within the uncertainties for the 2007 and newer model years but the on-road deterioration rates, like the fuel-specific emissions (see Figure 3), are much higher than the EMFAC estimates for the 2006 and older model year vehicles. The EMFAC2017 estimated emissions deterioration rates do not show any increases until the 2003 model year vehicles. This may be the result of the model assuming California LEV II
vehicle emission deterioration rates through the 2004 model year vehicles. This is when their introduction into the California fleet was to begin but we would not expect them to be a significant fraction of the fleet until later model years (DieselNet 2018). For the oldest model year vehicles (pre-1999) the final deterioration rates are approximately a factor of 4 higher (~0.5 vs. ~0.125 g NOx/kg of fuel/year) for the on-road measurements. For perspective 1996 model year vehicles with a 0.49 gNOx/kg of fuel/year emissions deterioration rate represents a 5.8%/yr emissions increase or an emissions doubling time of 12 yr. The lower rates predicted by the model are likely the result of the NOx emission factors appearing to be capped around 10 gNOx/kg of Fuel for 1996–1989 model year vehicles (see Figure S1) which we do not observe in the on-road measurements.

Fuel-based inventory

The SoCAB inventory is a critical piece of information that is used to shape regulatory policy for future emission reductions toward the goal of achieving compliance with the NAAQS. To estimate the extent of the NOx underprediction we have constructed a NOx fuel-based inventory for the SoCAB using our on-road emission measurements for light-duty gasoline and heavy-duty diesel vehicles. On-road heavy-duty diesel truck emissions were measured at the Peralta weigh station on SR 91 in the spring of 2017 (Haugen et al. 2018). Figure S2 shows the fuel-specific NOx emissions by model year comparison for the on-road heavy-duty diesel measurements and the EMFAC2017 predictions. Annual EMFAC2017 emission factors were calculated using aggregated speeds and a fuel weighted composite emission factor for year 2017 using the diesel truck types previously described. The on-road emissions are generally higher than the model predictions but comparison of estimated mean emissions using the Peralta model year distribution for both sets of emission factors results in only a 20% difference (10 vs. 12.5 ± 0.6 gNOx/kg) which overall is good agreement.

Following the methodology of Hassler et al., we have calculated the 2018 daily gasoline and diesel fuel consumption for the SoCAB in kilograms/day using annual fuel sales data for the State of California (Table S10 in the supporting material) (Hassler et al. 2016). These values are multiplied by the mean on-road emission factors measured at the West Los Angeles site (for the gasoline fleet) in 2018 and the Peralta weigh station in 2017 (heavy-duty diesel fleet) to estimate short tons/day emissions in the Basin. Table 1 compares the on-road fuel-based inventory with the 2018 Annual California Air Resources Board inventory predicted by the online California Emission Projections and Analysis Model (CEPAM) emission tool for the SoCAB (see supporting material) (California Air Resources Board 2020a). The fuel-based inventory indicates 74% more NOx emissions from the light-duty gasoline fleet than accounted for in the inventory despite the fact that the fuel-based inventory does not include idle and starting emissions. The light-duty CO and heavy-duty diesel NOx emissions inventory comparisons are in better agreement with the fuel-based results.

The fuel-based inventory estimates increase the light-duty gasoline NOx emissions to being on par, or slightly higher than, the heavy-duty NOx emissions. This is not a new observation as Kim et al. previously reported this for the 2010 SoCAB fleet (Kim et al. 2016). It does, however, raise the question as to whether the extra NOx emissions increase the total NOx inventory or are they offset by overestimates in other categories? The 2018 total inventory estimates of 1741 tons CO/day and 356 tons NOx/day result in a molar NOx/CO ratio of 0.12 which is within the spread of the morning ambient measurements and supports the two totals (see Table S5 in the supporting material). In addition, Morris et al. showed good agreement between OMI satellite NO2 observations and the 2018 basin NOx inventory again supporting the NOx total and suggesting that overestimates exist in other inventory categories (Morris, Parker, and Stocekenius 2019).

Figure 5 is a plot of the percent of the total gNOx/kg of Fuel emissions contributed by vehicle type and fuel, compared with the vehicle age distribution, by model year for the 2018 West Los Angeles on-road measurements. The newest model LEV II vehicles account for the largest percentage of vehicles in the fleet (2009 and newer ~62% of fleet) but only a small minority of the emissions (~11% of fuel-based NOx). The simple addition of lower emitting new vehicles, even if they are zero emitting

| Table 1. 2018 inventory comparison for California’s South Coast Air Basin. |
|---------------------------|---------------------------|-----------------------------|
| South Coast Air Basin | West Los Angeles (Gasoline) | Peralta Weigh Station (Diesel) |
| kg Fuel/day\(^{a}\) | (4.8 ± 0.3) × 10\(^7\) | (8.1 ± 0.7) × 10\(^6\) |
| On-road GNO\(_x\)/kg of fuel\(^{b}\) | 2.3 ± 0.1 | 12.5 ± 0.6 |
| Fuel-based NO\(_x\) tons/day\(^{c}\) | 122 ± 13 | 111 ± 15 |
| CEPAM\(^{d}\) NO\(_x\) tons/day\(^{c}\) | 70 | 111 |
| On-road gCO/kg of fuel\(^{b}\) | 11.2 ± 0.2 | 5.9 ± 0.9 |
| Fuel-based CO tons/day\(^{c}\) | 592 ± 47 | 53 ± 13 |
| CEPAM\(^{d}\) CO tons/day\(^{c}\) | 620 | 24 |
| Notes. | | |
| \(^{a}\)Derived from State annual fuel sales from Hassler et al. (2016) see supporting material. |
| \(^{b}\)Uncertainties are standard error of the mean derived from the daily measurements. |
| \(^{c}\)Short tons; 1 ton = 0.907 metric tons. |
| \(^{d}\)Uncertainties are the combined uncertainty from the fuel and emission factors. |
| \(^{e}\)California Emission Projections and Analysis Model. |
vehicles, will not appreciably change the light-duty NO\textsubscript{x} emissions distribution and will not provide any significant changes to the NO\textsubscript{x} inventory. The majority of the light-duty NO\textsubscript{x} emissions are found in the older LEV I vehicles. LEV I vehicles (1994–2003) are approximately 16% of the fleet observed at the West Los Angeles site but account for half of the fuel-specific NO\textsubscript{x} emissions. The highest emitting 10% of the LEV I vehicles are responsible for more than half of these emissions. Even small reductions in this segment of the fleet will yield large reductions in NO\textsubscript{x} emissions. Currently 2001 model year vehicles, as an example, have a year over year percentage removal from the fleet of 8.3% (−8.5 year half-life). The desire for large reductions in the NO\textsubscript{x} inventory in the SoCAB will require an extraordinary effort to achieve and LEV I gasoline vehicles appear to be a significant source. It is unlikely that we can expect the elimination of these vehicles from the fleet through natural attrition, especially within the current economic downturn, will occur very fast. This suggests that other methods need to be explored to hasten the removal of high NO\textsubscript{x} emitters from the fleet.

NO\textsubscript{x} reductions from heavy-duty diesel vehicles but admittedly believe that reductions from other sources will be necessary to achieve these targets.

Ambient molar NO\textsubscript{x}/CO ratios collected on weekdays during the morning rush hour in the SoCAB were compared with those estimated for gasoline-powered vehicles by California’s EMFAC2017 emissions model and on-road emission measurements collected from gasoline-powered vehicles at sites within the SoCAB. The ambient molar NO\textsubscript{x}/CO ratios steadily increase until around 2010 when they level off and then begin to decline through 2018. Both the EMFAC2017 predictions and the on-road emission measurements from 1999 to 2001 have ratios that are along the lower edge of the ambient measurements. However, after 2001 the EMFAC2017 and the on-road NO\textsubscript{x}/CO ratios diverge with the model predicted ratios decreasing significantly through 2018 and ending up factors of 1.8 to 2.4 below the 2018 average. The on-road measured ratios increase along with the ambient measurements until their peak and then increase slightly through 2018.

The difference between the two sets of ratios was found to most likely be that the EMFAC2017 model underestimates the NO\textsubscript{x} emission factors for the gasoline fleet as the CO emission factor comparison was good. Comparisons with 2018 on-road measurements collected in West Los Angeles found that the fuel-specific NO\textsubscript{x} emission factor comparison was good for the 2009 and newer model year vehicles that all have near-zero emissions. However, for the 2008 and older models, the observed on-road gNO\textsubscript{x}/kg of fuel emissions for both passenger vehicles and trucks increase at a higher rate than predicted by the EMFAC2017 model. For gasoline passenger vehicles (0.7 vs. 1.8 ± 0.1 gNO\textsubscript{x}/kg of fuel) and trucks (1.1 vs. 3.0 ± 0.1 gNO\textsubscript{x}/kg of fuel), the on-road age normalized mean emissions are factors of 2.6 and 2.7 times higher than predicted by the model. For the oldest model year vehicles (pre-1999) we found NO\textsubscript{x} emission deterioration rates that were approximately a factor of 4 higher (~0.5 vs. ~0.125 gNO\textsubscript{x}/kg of fuel/year) for the on-road measurements, a likely cause of the underprediction.

A fuel-based inventory for the 2018 SoCAB constructed using the on-road measurements from the West Los Angeles site indicates that there is 74% more NO\textsubscript{x} emissions from the light-duty gasoline fleet than represented in the inventory. These estimates imply that NO\textsubscript{x} emissions from light-duty gasoline vehicles are comparable or slightly higher than those from heavy-duty vehicles. The majority of the light-duty NO\textsubscript{x} emissions are found in the older LEV I (1994–2003) vehicles, which make up approximately 16% of the fleet observed in 2018 at the West Los Angeles site but account for half of the fuel-specific NO\textsubscript{x} emissions. The emissions of

**Summary**

State and regional air quality officials depend on air basin inventories for designing emission reduction plans to help a region comply with the NAAQS. Because the time frames involved in these plans are typically long and the implementation costs are often high any errors in the inventory can result in costly missteps and lost time if the emissions reductions anticipated are not achieved. The SoCAB currently experiences some of the nation’s highest ozone levels; as a result local and state regulations are targeting large
these vehicles are as expected skewed with the highest emitting 10% responsible for more than half of the LEV I contribution or 25% of the total NOx emissions.

The underreporting of NOx emission factors by the EMFAC model has been reported by other researchers; however, the newer iterations continue to carry forward this problem. This issue significantly changes the NOx emissions distribution for mobile sources in the SoCAB and will lead to an overestimation in the percent reductions that can be achieved in lowering diesel vehicle NOx emissions. In addition, it overlooks the possibility of significant NOx emission reductions by targeting the removal of older high emitting gasoline vehicles.

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Disclosure statement

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