

Decadal Trends in Air Pollutant Emissions from Motor Vehicles in Central California

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Abstract

Motor vehicles are an important source of air pollution on spatial scales ranging from urban to global. Over the past decade, vehicle emissions have changed due to improvements in emission control systems, introduction of reformulated fuels, and increased driving. On-road use of diesel fuel in the U.S. grew at a rate that was ~3 times higher than for gasoline, while efforts to control nitrogen oxide (NO_x) emissions from diesel engines lagged. Using fuel sales data together with measurements of on-road emission rates and ambient pollutant concentrations, we estimate vehicle emissions of carbon monoxide (CO), non-methane organic compounds (NMOC), and nitrogen oxide (NO_x) in California in 1990 and 2000. We focus specifically on Central California including the San Francisco Bay area, the San Joaquin Valley, and Sacramento. Reductions in NO_x emissions in this region from light-duty passenger vehicles were offset by increases in emissions from heavy-duty diesel trucks. Growth in vehicle travel has been especially rapid in the San Joaquin Valley, where in the year 2000 diesel trucks accounted for ~60% of on-road vehicle NO_x emissions. Fuel-based estimates of vehicle emissions developed in the present study are compared with predictions of California's vehicle emissions model (EMFAC). The most important uncertainty highlighted by this comparison is the magnitude of the change in NO_x emissions between 1990 and 2000, and underlying changes in diesel engine activity and NO_x emission factors over this time period.

Introduction

Motor vehicles are a major source of emissions of carbon monoxide, volatile organic compounds (VOC), and oxides of nitrogen (NO_x) on urban to global scales. Over the past decade, emissions have been affected by more stringent emission standards for new vehicles, fleet turnover, increases in the number of vehicles and the amount of driving, and reformulation of gasoline and diesel fuel. On-road vehicles now account for 51% of CO, 29% of VOC, and 34% of NO_x emissions in the U.S. (EPA, 2001). These pollutants react in the presence of sunlight to form tropospheric ozone. Vehicles also contribute to particulate matter in the atmosphere both by direct emissions of soot and other primary particles, and through gas-to-particle conversion of gaseous precursor emissions. Developing accurate air pollutant emission estimates and tracking changes in vehicle emissions over time are critical to understanding and controlling air pollution problems.

While the number of light-duty passenger vehicles on the road increased during the 1990s, improved emission control technologies and reformulated gasoline led to reductions in emission rates for pollutants other than CO_2 during the same period (Kirchstetter et al., 1999a). Emission controls such as exhaust gas recirculation, three-way catalytic converters, activated carbon canisters, and on-board diagnostic systems are standard on new light-duty vehicles (Sawyer et al., 2000). Changes made to gasoline include reduction of vapor pressure; elimination of lead; use of detergent additives; reduction of benzene, total aromatics, olefins, and sulfur contents; and addition of oxygenates (see Kirchstetter et al., 1999a). Roadway tunnel measurements and changes in ratios of pollutant concentrations measured in ambient air indicate that more progress was made in reducing CO and VOC than NO_x emissions during the 1990s (Kirchstetter et al., 1999a; Parrish et al., 2002).

As shown in Figure 1, consumption of both gasoline and diesel fuel by on-road vehicles has been growing. While the absolute amount of gasoline consumed exceeds on-road consumption of diesel fuel in the U.S. by a wide margin, note that the relative rate of growth for diesel fuel has been higher than for gasoline by a factor of about 3. With respect to exhaust emissions, three-way catalytic converters that reduce NO_x were introduced in the U.S. light-duty vehicle fleet starting in the early 1980s, whereas emission standards for NO_x from heavy-duty diesel engines did not take effect until the 1990s. To reduce NO_x emissions from engines sold during the 1990s, heavy-duty engine manufacturers relied mainly on changes in the timing of diesel fuel injection, rather than post-combustion treatment of the exhaust gases. Unfortunately NO_x emission rates from diesel engines were not reduced as intended (Yanowitz et al., 2000). Taken together, success in controlling light-duty vehicle emissions, rapid growth in on-road diesel fuel consumption, and unsuccessful efforts to control diesel NO_x emission rates imply that the importance of diesel engines as a source of NO_x emissions must be growing.

In this study, measurements that encompass a large number of on-road vehicles are used to define emission factors that are expressed per unit of fuel burned (Singer and Harley, 1996). Fuel sales data from tax records are combined with emission factors to estimate total emissions. Compared to conventional modeling approaches in which emissions are estimated based on distance traveled by each vehicle, the fuel-based approach has the advantage that emission factors normalized to fuel consumption vary less over the wide range of vehicle weights and driving conditions that are seen on the road (Pierson et al., 1996; Yanowitz et al., 2000; Kean et al., 2003). Heywood (1988) comments specifically on the usefulness of a fuel-based approach to estimating diesel NO_x emissions, noting that much of the fuel burns under nearly stoichiometric conditions in diesel engines, even though the overall air/fuel ratio varies with changes in engine

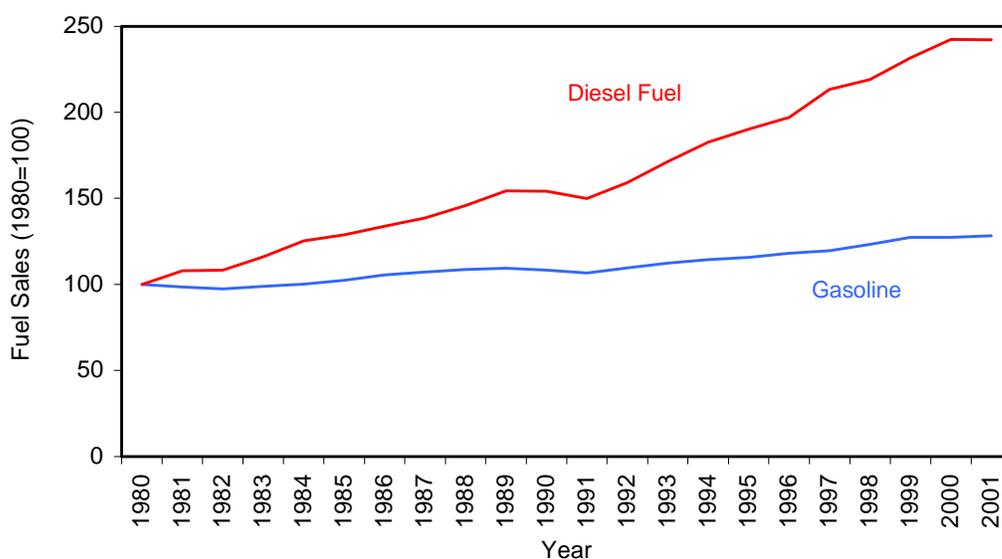


Figure 1. National trends in consumption of gasoline and diesel fuel by on-road vehicles. The base year is 1980, which is set to a reference value of 100 for each fuel. Source: Federal Highway Administration *Highway Statistics*.

load. Note also that all engine emission rates are infinite at idle when expressed per km driven; this singularity in emission factors is avoided by quantifying emissions per unit of fuel burned.

The objective of this research is to use a fuel-based approach combined with ambient pollutant concentration ratios to estimate CO, NMOC, and NO_x emissions from on-road vehicles in central California. We assess how emissions have changed between 1990 and 2000, and explore how motor vehicle emissions vary by location. Fuel-based estimates of motor vehicle emissions in 1990 and 2000 are compared to corresponding predictions of California's motor vehicle emissions model known as EMFAC (CARB, 2002).

Methods

Fuel Sales and Regional Apportionment. Three regions within central California are the focus of this work: the San Francisco Bay area, the San Joaquin Valley, and Sacramento County. Summertime air pollution problems resulting from photochemical processing of vehicle exhaust and other emissions are present throughout this domain, and are especially severe in the San Joaquin Valley.

Emissions estimates were based on state records of taxable fuel sales for on-road vehicles in 1990 and 2000. Fuel consumption was apportioned based on separate estimates of passenger vehicle and heavy-duty diesel truck travel by county in 1990 and 1998, the most recent year for which detailed counts of truck travel were available. Emission factors were derived from on-road infrared remote sensing and tunnel measurements. Ambient ratios of NMOC/CO were used to help estimate NMOC emissions.

On-Road Remote Sensing. CO emission factors from light-duty vehicles were measured by on-road remote sensing. Using infrared and ultraviolet spectroscopy (Popp et al., 1999), roadside remote sensing instruments measure CO, hydrocarbon, and NO_x concentration ratios to CO₂ in individual tailpipe emission plumes. A carbon balance together with known gasoline properties were used to calculate the CO emission factor for each vehicle in grams of CO emitted per liter of gasoline burned (for details see Singer and Harley, 1996).

Remote sensing field studies of vehicle emissions in central California were conducted in the summers of 1991, 1999, and 2000. The 1991 study by Stedman et al. (1994) included measurements of vehicle emissions at five Bay area sites. The 1991 results were adjusted upward by 10% to estimate CO emission factors for 1990. This adjustment is based on the rate of decrease in CO emission factors observed on-road during the 1990s by Kean et al. (2000). More

recent roadside emission surveys were performed in 1999 by Stedman and coworkers, and by California Air Resources Board personnel as part of the summer 2000 Central California Ozone Study. The 1999-2000 remote sensing surveys include measurements of exhaust gas concentrations for nearly 200 000 vehicles at 58 sites. Locations of the remote sensing sites and other details are listed in the appendix. Measurements from 1999 were linked to model year information for each vehicle, determined by matching license plate numbers to vehicle registration records. In those cases, a fuel economy-weighted emission factor was calculated, which places greater weight on emissions from vehicles that use more fuel to travel the same distance (Singer and Harley, 1996). For heavy-duty diesel trucks which are a minor source of CO emissions, an emission factor of $16 \pm 2.5 \text{ g L}^{-1}$ was used, which is the average of results from tunnel and remote sensing studies conducted during the 1990s (Bishop et al., 2001).

NMOC/CO Ambient Ratios. Emissions of NMOC were estimated using ambient concentration ratios (NMOC/CO), together with fuel-based estimates of CO emissions described above. This was done because some categories of evaporative emissions are not observed during on-road driving, and because instrument-specific adjustment of raw remote sensing hydrocarbon measurements is needed to reflect accurately all of the organic carbon mass emitted (Singer et al., 1999). Ambient NMOC concentrations were measured in air samples that were collected in steel canisters. Morning rush hour ambient air measurements were selected for this analysis because this time period captures peak commuter traffic when emissions are fresh and photochemical reactions are of minimal concern.

Co-located ambient NMOC and CO concentrations were analyzed from special studies in 1990, 1999, and 2000. Marr et al. (2002) present NMOC/CO ratios from the summer of 1990 from 16 sites in central California. More recently, NMOC concentrations were measured at six

sites in the San Francisco Bay Area in the summer of 1999, and two sites in Sacramento and three sites in the San Joaquin Valley in the summer of 2000. Air samples were collected during the period 6-9 AM in stainless steel canisters, and were analyzed by the Monitoring and Laboratory Division of the California Air Resources Board using gas chromatography to quantify NMOC concentrations. Matching CO concentrations were obtained from a data archive maintained by the California Air Resources Board.

NO_x Emission Factors. NO_x emissions were estimated as the product of fuel consumption and separate emission factors for both light- and heavy-duty vehicles. Unlike CO and NMOC, a significant fraction of on-road vehicle NO_x emissions comes from diesel engines. Separate light- and heavy-duty NO_x emission factors were derived from on-road measurements that were made between 1990 and 2001. Chassis dynamometer testing and on-road measurements indicate that heavy-duty diesel NO_x emission factors did not change significantly during the 1990s (Yanowitz et al., 2000), so a NO_x emission factor of $32 \pm 3 \text{ g L}^{-1}$ was used for both 1990 and 2000 (this is an average of values reported by Pierson et al., 1996; Kirchstetter et al., 1999b, Jiménez et al., 2000; and Bishop et al., 2001). For light-duty vehicles in 1990, a NO_x emission factor of $8.4 \pm 1.8 \text{ g L}^{-1}$ was used throughout central California, as described previously by Marr et al. (2002). For 2000, the light-duty vehicle NO_x emission factor was estimated from measurements made in the Caldecott Tunnel in the San Francisco Bay area in the summers of 1999 and 2001 (Kean et al., 2000; Kean et al., 2002), and from remote sensing studies conducted in central California during 1999 and 2000, as described above.

Emission Inventory. Total gasoline and diesel fuel sales in California were apportioned spatially based on the percent of statewide vehicle travel in each county and air basin (Caltrans, 1999; Caltrans, 2002). Total gasoline sales were reduced by 3% to exclude consumption by off-

road vehicles, and then increased by 4% to account for higher than average gasoline sales and driving during summer months (Singer and Harley, 2000). The basis for the small upward summer driving season adjustment in gasoline sales is monthly reports of in-state taxable fuel shipments. Tax-exempt diesel fuel sold for use in off-road engines was excluded from the present inventory calculations.

CO and NO_x emissions were estimated by multiplying the volumes of fuel sold by the corresponding emission factors described above. NMOC emissions were estimated from total CO emissions (including running exhaust plus EMFAC estimates of excess CO emissions associated with vehicle starting), multiplied by NMOC/CO concentration ratios. Molar NMOC/CO ratios were converted to mass ratios using 14 g (mol C)⁻¹ for NMOC and 28 g mol⁻¹ for CO. This assumes that two hydrogen atoms are present on average per carbon atom in NMOC. Ambient NMOC concentrations measured from 6-9 AM reflect running exhaust, idle, and cold start emissions, as well as hot soak and some running loss evaporative emissions. Diurnal and resting evaporative losses are not well-represented in the ambient ratios because NMOC measurements are for morning hours only.

Results and Discussion

Fuel Consumption. Mirroring national trends shown in Figure 1, fuel consumption by on-road vehicles in California grew between 1990 and 2000, with larger relative increases for diesel fuel than for gasoline. In central California (see Table 1), growth in both gasoline and diesel fuel consumption outpaced the statewide average, led mainly by rapid growth in the San Joaquin Valley. Fuel consumption estimates shown in Table 1 agree well with corresponding predictions of the EMFAC model for gasoline. In both 1990 and 2000, gasoline consumption estimates for the Bay area and Sacramento agree to within $\pm 3\%$. For the San Joaquin Valley, the two 1990 estimates of gasoline consumption match exactly, though EMFAC predicts a 33% increase in gasoline consumption between 1990 and 2000, whereas the present study indicates a smaller increase of 16%.

Table 1. Fuel consumption by on-road motor vehicles, 1990 and 2000.

Region	Gasoline (10^9 L)			Diesel (10^9 L)		
	1990	2000	Change	1990	2000	Change
Sacramento County	1.74	1.90	+9%	0.18	0.29	+61%
San Francisco Bay area	9.71	10.88	+12%	0.86	1.11	+29%
San Joaquin Valley	4.29	4.98	+16%	1.34	2.15	+60%
Central California ^a	15.8	17.8	+13%	2.4	3.6	+49%
California state total	50.8	56.0	+10%	6.9	10.0	+45%

^aSum of the above three regions.

Estimates of on-road diesel fuel consumption in central California do not agree as well as gasoline estimates discussed above. Diesel fuel estimates are compared in Figure 2. The best agreement is seen for the San Joaquin Valley in 1990, and region-wide total diesel fuel consumption in all of central California in 2000. Baseline estimates of diesel fuel consumption in 1990 differ by a factor of about 2 for both the Bay area and Sacramento. For the Bay area, despite large differences in the 1990 baseline, similar increases in diesel fuel consumption of 27 and 29% are estimated to have occurred between 1990 and 2000. For the San Joaquin Valley, EMFAC indicates an increase of 39% in diesel fuel consumption over 10 years, whereas the present study suggest a larger increase of 60%. The differences in growth estimates are further apart for Sacramento county: 26% predicted by EMFAC versus 61% in the present study. These differences in diesel engine activity estimates have important implications for the sub-regional distribution of NO_x emissions within central California, and the relative importance of diesel engines as a contributor to total NO_x from on-road vehicles. Overall for central California, EMFAC predicts a 31% increase in diesel fuel consumption between 1990 and 2000, whereas the present study indicates a larger increase of 49%.

Though the statewide total for diesel fuel consumption includes travel on both the state highway system and on local roads, the method used here for spatial apportionment of diesel fuel consumption relies on counts of diesel truck traffic on the state highway system only, and so may under-represent diesel fuel consumption in the more heavily urbanized areas such as San Francisco and Sacramento. On the other hand, the traffic counts and travel demand models used to provide input data to the EMFAC model typically focus on total vehicle counts, which are dominated by light-duty passenger vehicles. More attention is needed to differences between

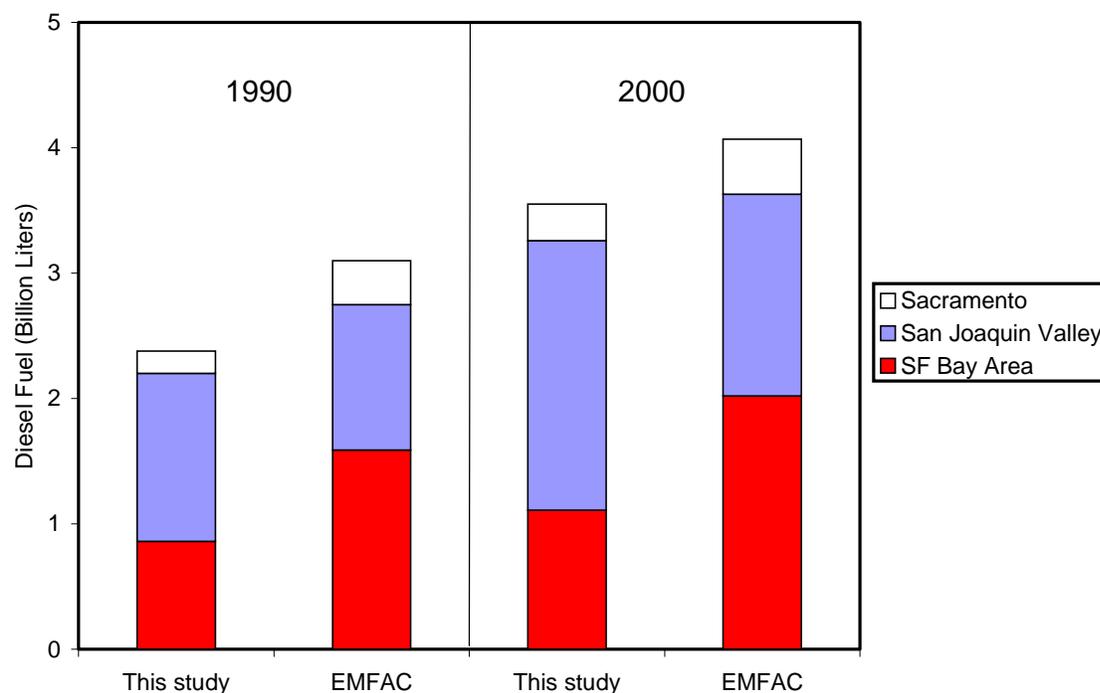


Figure 2. On-road diesel fuel consumption in Central California in 1990 and 2000.

heavy-duty diesel truck and light-duty passenger vehicle activity patterns on time scales ranging from diurnal and weekly to decadal.

Emission Factors. CO emission factors used in this study are summarized in Table 2.

Data for the year 2000 indicate that typical emission factors for vehicles in Central California lie within the range 30 to 50 grams of CO emitted per liter of fuel burned. Emission factors for CO in the San Francisco Bay area are based on results from comparatively small numbers of roadside sampling sites, so relative uncertainties are larger compared to Sacramento and the San Joaquin Valley where many more sites were surveyed. Emission measurements in the San Joaquin Valley were performed in the rapidly growing cities of Fresno and Stockton. Individual sampling locations and numbers of vehicles observed at each site are listed in the appendix.

Table 2. Exhaust emission factors for gasoline-powered motor vehicles, 1990 and 2000.

Region	Year	Sample size ^a (# of sites)	Emission Factor ^b (g L ⁻¹)	
			CO	NO _x
San Francisco Bay area	1990	4	63 ± 25	8.4 ± 1.8 ^c
San Francisco Bay area	2000	4	31 ± 14	5.5 ± 2.3
Sacramento County	2000	27	42 ± 4	5.4 ± 0.5
San Joaquin Valley	2000	23	50 ± 6	7.5 ± 0.9

^aNumber of on-road sampling sites. Only sites where at least 800 vehicles were measured are included here.

^bValues of the emission factors shown here are averaged over on-road measurement sites within the specified region. Stated uncertainties are 95% confidence intervals for the mean.

^cThe remote sensor used in 1990 did not measure NO_x emissions, so tunnel data were used for this pollutant (see text).

Site-average CO emission factors, where all vehicles measured are weighted equally regardless of fuel economy, do not differ greatly from fuel economy-weighted emission factors. Both methods of calculating average CO emission factors were compared at seven sites where vehicle model year information was available. When vehicle age and associated average fuel economy are ignored, the average CO emission factor at a site is lower than the fuel economy-weighted value by about 4%. Determining fuel economy-weighted emission factors at all sites would require considerable additional effort to capture and analyze license plate data to determine the model year of each vehicle. For the purposes of the present study, the fuel economy weighting of emission factors makes little difference to the results. However for other applications of remote sensing data such as assessment of inspection and maintenance programs, information about individual vehicles may be essential.

Information on vehicle age distributions derived from license plate data also may be useful to understanding differences in average CO emissions among sub-regions within Central California. However, Figure 3 shows that vehicle age alone is not sufficient to explain all site-to-site differences in emissions. Figure 3 presents average CO emission factors by model year for three different areas that were surveyed by Stedman and coworkers in 1999: Southern California (average of 3 Los Angeles area remote sensing sites), Sacramento (average of three sites), and San Jose (one site) in the San Francisco Bay area. For a given model year, average CO emission factors by model year are highest in southern California and lowest in Sacramento with few exceptions. The largest differences are for pre-1990 model years. Possible explanations include higher accumulated mileage for vehicles of the same model year, a less well-maintained vehicle fleet in southern California compared to the other areas, differences in driving conditions and ambient temperature, and bias in the remote sensing measurements.

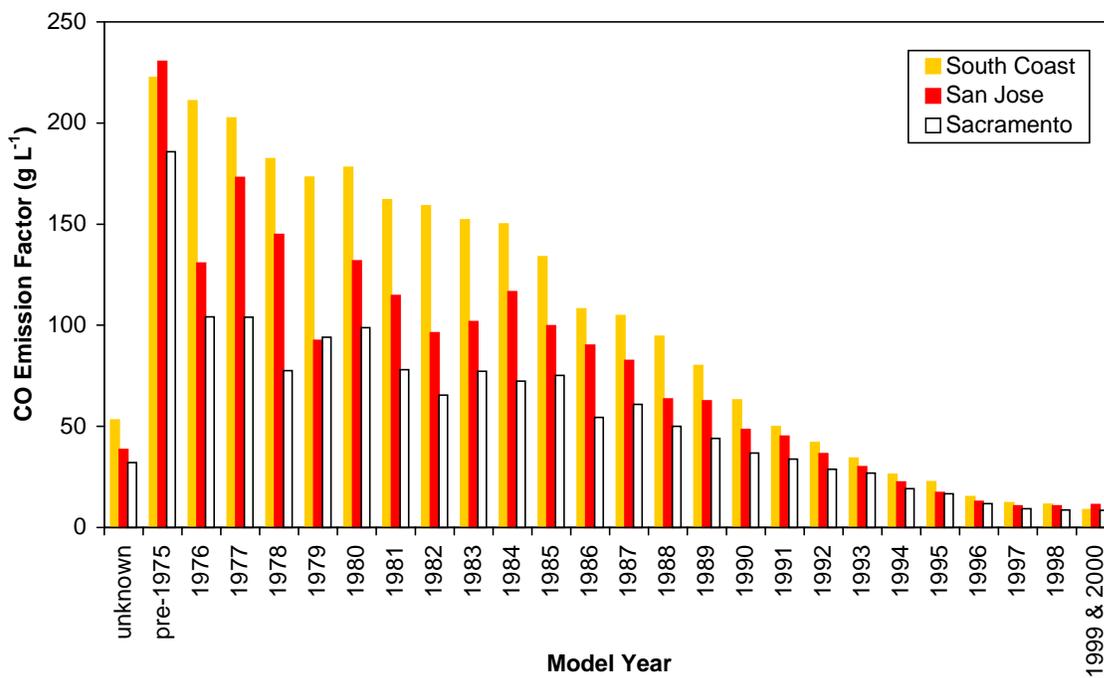


Figure 3. CO emission factor versus model year in three regions of California.

Ambient Pollutant Ratios. Figure 4 shows concentrations of ambient NMOC versus CO measured in central California in 1990 and 2000. Also shown are results of linear regression analyses conducted separately for 1990 and 2000. The best-fit slopes give NMOC/CO emission ratios in moles of organic carbon per mole of CO. The NMOC/CO ratio decreased by 19% between 1990 and 2000. This indicates that between 1990 and 2000, the decrease in NMOC emissions has been larger than the decrease in CO emissions. Though it is a minor source of uncertainty, ambient NMOC/CO ratios shown in Figure 4 could be linked more directly to vehicle-related emissions by excluding ethane and propane from NMOC concentrations in the regression analysis. The main benefit of doing this would be improved R^2 and reduced uncertainty in the NMOC/CO ratio.

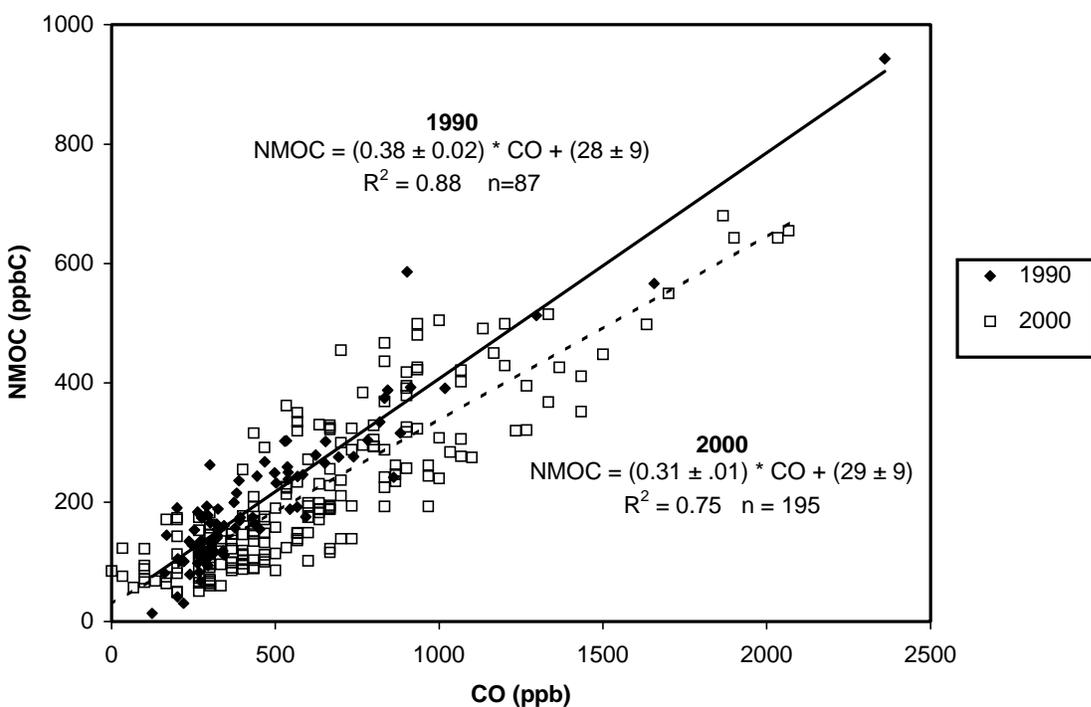


Figure 4. Linear regression of NMOC v. CO concentrations in central California, 1990 and 2000.

Emission Inventory. Table 3 presents emission inventory estimates developed using the fuel-based approach, with comparison to EMFAC model estimates for corresponding years (1990 and 2000), pollutants, and geographic locations. On-road measurements of vehicle emissions used here to determine emission factors do not include much of the excess emissions associated with vehicle starting. Therefore in Table 3, starting emissions are listed separately from running exhaust emissions for CO and NO_x, so that direct comparisons can be made between fuel-based and EMFAC estimates of running emissions. While NO_x emissions are subdivided into gasoline and diesel engine exhaust contributions in Table 3, this is not done for the other pollutants because gasoline engines are the overwhelmingly dominant mobile source of CO and NMOC. A minor amount of diesel NO_x (~3% of the total) associated with idling engines is also lumped with vehicle start emissions in Table 3.

For carbon monoxide, fuel-based estimates of running emissions in 1990 are about half those predicted by the EMFAC model. In 2000, running exhaust emissions are in good agreement. Small differences between the emission estimates for 2000 could be explained by excess CO emissions associated with high-speed/high-load driving which are under-represented in the remote sensing data. Capturing excess CO emissions associated with high-load driving also would reduce differences in 1990 estimates of running CO emissions. Uncertainties in fuel-based CO emission estimates are high throughout Central California in 1990, due to the small number (4) of roadside sampling sites for which emissions data were available. Another possible source of bias for CO in 1990 is that all of the emissions data comes from Bay Area roadside sites, and the effect of vehicle fleet differences in other areas is neglected. The uncertainties in fuel-based estimates of CO emissions for 2000 are relatively low in Sacramento and the San Joaquin Valley where many roadside sampling sites were visited, and remain high in the Bay

Area where the number of different sampling sites remained small. Large reductions in CO emissions during the period between 1990 and 2000 have been documented elsewhere (see for example Kirchstetter et al., 1999 and Kean et al., 2002). Unfortunately in the present study the likely range of CO emission estimates for 1990, accounting for uncertainty bounds, is large enough that it overlaps with corresponding emission estimates for 2000. EMFAC indicates that vehicle starting accounted for ~20% of exhaust CO emissions in both 1990 and 2000. The importance of start emissions is expected to be higher during winter months when engines and catalytic converters take longer to warm up, and when automotive air conditioning systems are not much used.

The most important source of uncertainty in the fuel-based estimates of NMOC presented in Table 3 is the underlying CO emission estimates (recall NMOC emissions were estimated by ratio to CO emissions). Estimates of NMOC emissions for 1990 agree, while for 2000, the fuel-based estimates are greater than those predicted by EMFAC by 40-70% in the San Joaquin Valley and Sacramento. Note that if EMFAC rather than fuel-based estimates of CO emissions are used as the basis for estimating NMOC, then higher estimates of NMOC emissions would be obtained especially for 1990.

Comparisons between emission estimates for NMOC are complicated by the presence of non-tailpipe evaporative emissions: some of these emissions are not well represented in the NMOC/CO ratios determined from ambient air samples collected during morning hours. The present study already lists diurnal and resting loss evaporative emissions separately under the “evap” column in Table 3, as those emissions occur mostly outside the 6-9 AM period. Running losses are by far the largest category of evaporative emissions according to EMFAC model estimates; these evaporative emissions occur while vehicle engines are operating, in contrast to

Table 3. On-road motor vehicle emission inventory estimates.

	1990			2000		
	running exhaust		start/idle	running exhaust		start/idle
	this work ^a	EMFAC ^b	EMFAC	this work	EMFAC	EMFAC
CO (10³ kg day⁻¹)						
Sacramento county	310±120	637	180	232±22	257	79
San Francisco Bay	1710±680	3175	802	970±440	1343	411
San Joaquin Valley	800±320	1906	467	780±90	910	209
Central California	2820±1100	5718	1449	1980±450	2510	698
NO_x (10³ kg day⁻¹)						
Sacramento county	56±9	76	7.3	53±4	54	6
gasoline engines	40±9	49	6.5	28±3	27	5
diesel engines	16±2	27	0.8	25±2	27	1
San Francisco Bay	300±50	385	33	261±70	277	31
gasoline engines	220±50	264	30	164±69	151	27
diesel engines	75±7	121	3	97±9	126	4
San Joaquin Valley	216±24	219	17	290±22	187	17
gasoline engines	99±21	127	14	102±12	86	13
diesel engines	117±11	92	3	188±18	101	4
Central California	570±80	680	57	600±70	518	54
gasoline engines	360±80	440	50	290±70	264	45
diesel engines	210±20	240	7	310±20	254	9

Table 3 (continued). On-road motor vehicle emission inventory estimates.

	1990			2000		
	emissions ^c		evap ^d	emissions ^c		evap ^d
	this work ^a	EMFAC ^b	EMFAC	this work	EMFAC	EMFAC
NMOC (10³ kg day⁻¹)						
Sacramento County	90±40	83	6	48±5	31	3
San Francisco Bay	480±190	409	22	210±100	165	13
San Joaquin Valley	240±100	224	14	152±18	96	9
Central California	810±330	716	42	410±100	292	25

^aFuel-based emission estimates developed in the present study.

^bPredictions obtained using the EMFAC model (EMFAC 2002, version 2.2).

^cFor NMOC, emissions estimates presented in the table include running and starting exhaust emissions, hot soak, and running loss evaporative emissions.

^dIncludes resting loss and diurnal evaporative emissions that are not well-represented by the morning ambient NMOC/CO ratio method used to estimate emissions in the present study.

diurnal and resting loss emissions which occur mainly while vehicles are parked. Examples of running losses include fuel leaks in the engine compartment, and unburned fuel that “blows by” the piston into the engine crankcase. Crankcase vapors may be vented to the atmosphere if positive crankcase ventilation (PCV) control systems malfunction. Gasoline vapors also may be emitted to the atmosphere if heated fuel is returned from the engine compartment back to the fuel tank. V. Hughes (ARB, personal communication, 2003) commented that significant increases in running loss evaporative emissions can occur on hot summer afternoons. In that case, some of

the running loss emissions should be transferred from the “emissions” to the “evap” column in Table 3. It would follow that the fuel-based estimates would exceed the remaining EMFAC estimates by a larger amount (e.g, by up to an additional 50×10^3 kg/day for the Bay Area in summer 2000).

There was little overall change in motor vehicle emissions of NO_x between 1990 and 2000. While NO_x emitted from gasoline-powered vehicles probably decreased, NO_x emissions from diesel trucks grew by ~50%. Therefore, the contribution of diesel engines to total vehicular NO_x emissions in Central California increased significantly: from 35% of the on-road total in 1990 to 50% as of 2000. The increase in NO_x emissions from diesel trucks is due to a large increase in diesel fuel consumption during this period, and a lack of progress in reducing the NO_x emission factor for heavy-duty vehicles. Increased diesel fuel consumption and unsuccessful heavy-duty engine NO_x control efforts have offset progress in controlling emissions from the light-duty sector.

There are important regional differences in NO_x emissions from on-road vehicles, with diesel trucks accounting for a larger fraction of emissions in the San Joaquin Valley than in other parts of central California. Between 1990 and 2000 diesel exhaust contributions to total vehicular NO_x emissions increased: from 24 to 35% in the San Francisco Bay area, from 26 to 44% in Sacramento, and from 52 to 60% in the San Joaquin Valley.

Uncertainties in NO_x emissions estimates for diesel engines in the present study arise in defining and apportioning the amount of diesel engine activity. As heavy diesel trucks can travel long distances (~1000 miles) between refueling stops, there is concern that long-haul truckers may purchase lower-cost diesel fuel before entering the state. Estimates of diesel fuel exports (33 million gallons) and imports (202 million gallons) have been made for California based on

international fuel tax agreement (IFTA) returns filed by inter-state truckers (K. Beile, State Board of Equalization, personal communication, 1997). Net diesel fuel imports of 169 million gallons amounted to ~8% of taxable diesel fuel sales in California as of 1996. Note however, that fuel imports and exports are already accounted for in tabulations of taxable diesel fuel sales (existing law, section 60115 of the Revenue and Taxation Code, requires inter-state truckers who have not already paid the excise tax at the pump by purchasing diesel fuel in California, to report and pay their liability through quarterly IFTA returns). Therefore no adjustments were made to reported taxable diesel fuel sales in the present study.

Another uncertainty in measuring diesel fuel use is the separation between taxable on-road and tax-free off-road uses. Tax laws are enforced at highway weigh stations to ensure that “red dye” (tax-free) diesel fuel is not being used in on-road driving. California has developed a separate model for estimating off-road engine activity and emissions. That approach involves estimating unregistered off-road engine populations using new engine sales data each year, estimating rated horsepower, load factors, and hours of use per engine, and multiplying these quantities together to get total hp-hrs of engine activity. Unfortunately, there is no mass balance check on the total amount of diesel fuel consumed by on + off-road engines, which should in principle be reconciled with known total production of distillate fuels at refineries. Kean et al. (2000) analyzed surveys of diesel fuel wholesalers conducted by the U.S. Energy Information Administration, and found that off-road engine activity appears to have been significantly overstated in past emission inventories. This issue may be a source of compensating error in California emission inventories, if overstated emissions from off-road engines make up for understated NO_x emissions from on-road diesel engines.

In Figure 5, emission estimates of the present study for calendar year 2000 are compared to EMFAC model predictions. Agreement between emission estimates in the present study has improved considerably compared to past assessments using older versions of the EMFAC model (e.g., Pierson et al., 1990; Fujita et al., 1992; Singer and Harley, 1996).

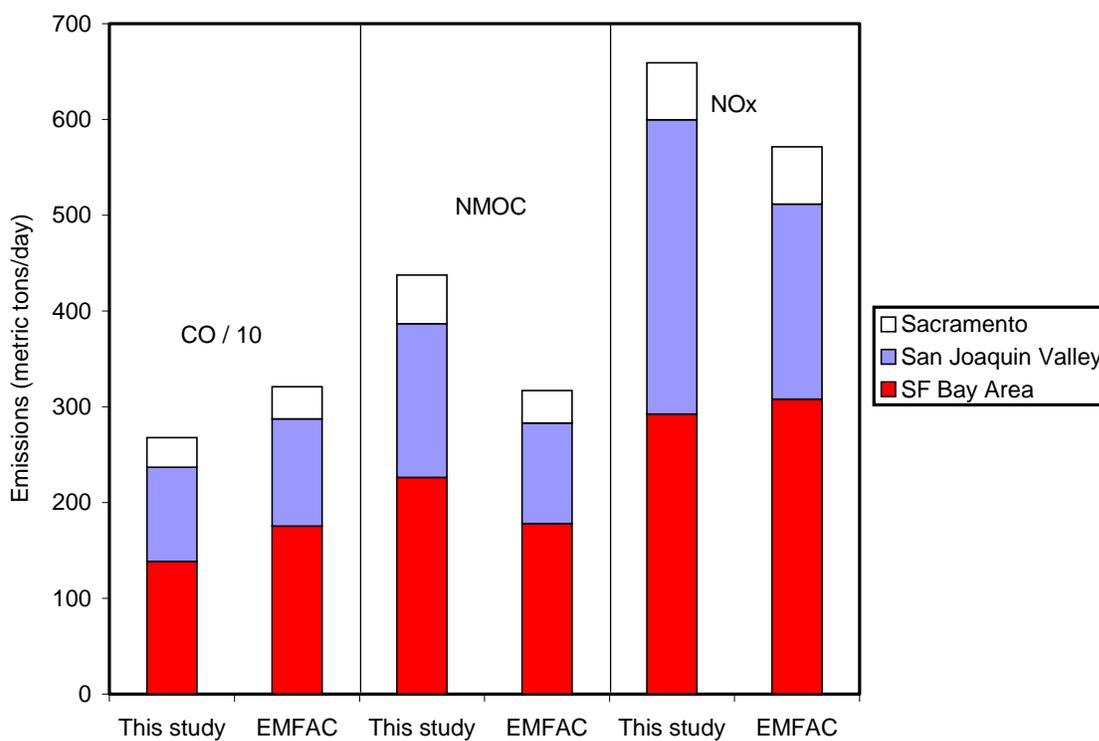


Figure 5. Motor vehicle emission inventory for central California in 2000, estimated using fuel-based methods (this study) and the EMFAC model.

CO emissions estimates shown in Figure 5 are in good agreement. The lower CO estimate of the present study may be due to the small number of Bay area roadside sampling sites visited during the summer 2000 remote sensing study. NMOC emissions estimated in this study are about 40% higher than the corresponding EMFAC model prediction for all of central

California. Furthermore, atmospheric NMOC/CO ratios are higher than the ratio of corresponding EMFAC emission estimates by 56%. Therefore, if the EMFAC estimate of CO emissions is accepted in lieu of the fuel-based estimate, NMOC emissions estimated using ambient NMOC/CO ratios would be even higher than estimates of the present study indicated in Figure 5.

Inspection of Figure 5 indicates that NO_x emissions estimates agree well, except in the San Joaquin Valley where fuel-based estimates of the present study are higher by 52%. Both EMFAC and the present study agree on the amount of NO_x emitted from gasoline engines in all sub-regions within central California. Therefore remaining differences are the result of uncertainties relating to diesel engine activity and emissions. Even in the Bay area and Sacramento, some fortuitous compensating errors may be present such that differences in both diesel engine activity and NO_x emission factors offset one another. Compared to the present study, EMFAC specifies a higher level of diesel engine activity in the Bay area and Sacramento, but predicts a lower NO_x emission factor.

Conclusions

Overall on-road vehicle emissions of NO_x did not change much between 1990 and 2000, due to growth in fuel use and emissions from heavy-duty diesel trucks. As light-duty vehicles grow cleaner and their contribution to the total emission inventory declines, NO_x emissions from heavy-duty diesel trucks and off-road engines will continue to increase in importance. Though post-combustion emission controls will be installed on new diesel engines beginning in the 2007 model year to help address the NO_x problem, it will take additional time to retire or retrofit older engines that are already on the road.

Recommendations

- 1. Improved emission inventory tools are needed for tracking the growth and different spatial and temporal patterns of activity for diesel engines.** Diesel engines will continue to increase in relative importance as a source of NO_x emissions in California. Heavy diesel truck activity differs from gasoline engines on diurnal, weekly, and decadal time scales, and in its spatial patterns as well. Current travel demand models used to estimate vehicle activity are based on population and employment and were designed for estimating commuter peak period traffic. Diesel truck traffic is still commonly modeled as a specified fraction of total traffic at all times and locations. This assumption is contradicted by numerous studies including Dreher and Harley (1998), Marr et al. (2002), and ARB's review of truck activity data for studies of the weekend ozone effect.
- 2. Off-road diesel engine activity and emissions in California should be reassessed.** California should consider using a fuel-based approach for estimating off-road diesel engine emissions, as EPA is now proposing in the new MOVES model, which includes both on-road and off-road engines in a single integrated modeling system.
- 3. Semi-continuous data from automated gas chromatograph (GC) systems should be used to examine how VOC species vary by time of day and with temperature.** Uncertainties remain in the relative and absolute importance of exhaust vs. evaporative emissions as sources of VOC emissions. Further research is needed to help improve the understanding of emissions variations as a function of time of day and ambient conditions.

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Appendix

The following pages list the roadside field sampling sites where vehicle emissions were measured. Separate pages describe data for the San Francisco Bay area (1990 and 2000), Sacramento County (2000 only), and the San Joaquin Valley (2000 only). Information provided for each site includes the number of vehicles observed per day of sampling, average CO and NOx emission factors measured at the site, and the source of the data. Most of the data are from a summer 2000 field survey conducted by California Air Resources Board staff from El Monte. Field sites with fewer than 800 vehicles observed are not included in this listing.

Summary of Central California On-Road Vehicle Emission Sampling Sites

Area/City	Sampling Site	Date	Sample Size	Average Emissions (g/L)		Data source	Instr. ID
				CO	NOx		
SAN FRANCISCO BAY AREA - CIRCA 1990							
Sunnyvale	EB Evelyn, 200 m west of Mathilda	15-Jul-1991	1076	58.2	N/A	Stedman	FEAT
Hayward	EB Winton, 400 m west of Hesperian	16-Jul-1991	3525	67.1	N/A	Stedman	FEAT
Berkeley	EB Ashby just west of MLK	17-Jul-1991	3416	66.1	N/A	Stedman	FEAT
Lafayette	SB Pleasant Hill Rd south of Hwy 24	18-Jul-1991	1718	36.9	N/A	Stedman	FEAT
				1991 CO	1990 CO		
		Average	2434	57.1	63.4		
		Standard error		7.0	7.8	N=4	
		95% CI		22.3	24.8	3.182	t statistic
SAN FRANCISCO BAY AREA - CIRCA 2000							
Sunnyvale	NB Hwy 280 ramp to NB Hwy 880	1999	44100	40.6	5.5	Stedman	FEAT
Berkeley	EB Hwy 24 Caldecott Tunnel	1999 & 2001	8400	35.2	4.5	Kean et al	TEI
Livermore	NB Vasco ramp to WB Hwy 580	23-Aug-2000	1753	26.7	7.5	CARB	ESP
San Jose	Hwy 101 ramp to EB Capitol Expwy	24-Aug-2000	3951	21.3	4.4	CARB	ESP
		Average	14551	31.0	5.5		
		Standard error		4.3	0.7	N=4	
		95% CI		13.7	2.3	3.182	t statistic

Summary of Central California On-Road Vehicle Emission Sampling Sites

Area/City	Sampling Site	Date	Sample Size	Average Emissions (g/L)		Data source	Instr. ID
				CO	NOx		
SACRAMENTO COUNTY							
Sacramento	EB Hwy 50 to Sunrise Blvd	1999	24700	30.8	4.2	Stedman	FEAT
Sacramento	SB Hwy 99 to EB Florin Rd	1999	5200	32.5	6.7	Wenzel	ESP
Sacramento	WB Arden Way to WB Business 80	1999	19000	29.9	6.5	Wenzel	ESP
Sacramento	SB Bradshaw to WB Hwy 50	1-Aug-2000	1120	40.8	7.6	CARB	FEAT 3005
Sacramento	SB Zinfandel to WB Hwy 50	2-Aug-2000	1542	48.6	8.0	CARB	FEAT 3005
Sacramento	NB Bradshaw to EB Hwy 50	3-Aug-2000	2967	42.9	6.3	CARB	FEAT 3005
Sacramento	WB Hwy 50 to SB Watt	4-Aug-2000	916	30.9	6.7	CARB	FEAT 3005
Sacramento	EB Hwy 50 to NB Watt	7-Aug-2000	5877	36.5	4.3	CARB	FEAT 3005
Sacramento	SB Sunrise to WB Hwy 50	8-Aug-2000	5618	37.7	6.1	CARB	FEAT 3005
Sacramento	SB Hwy 5 to EB Hwy 80	9-Aug-2000	2317	48.3	5.7	CARB	FEAT 3005
Sacramento	EB Stockton to NB Hwy 99	10-Aug-2000	2646	34.4	4.8	CARB	FEAT 3005
Sacramento	SB Bradshaw to WB Hwy 50	1-Aug-2000	1042	44.7	4.0	CARB	FEAT 3006
Sacramento	NB Bradshaw to EB Hwy 50	3-Aug-2000	1434	46.1	3.5	CARB	FEAT 3006
Sacramento	WB Hwy 50 to SB Watt	4-Aug-2000	931	30.6	6.8	CARB	FEAT 3006
Sacramento	EB Hwy 50 to SB Watt	7-Aug-2000	2549	33.7	3.2	CARB	FEAT 3006
Sacramento	WB Pocket to NB Hwy 5	8-Aug-2000	1259	41.9	6.4	CARB	FEAT 3006
Sacramento	EB Florin to NB Hwy 99	9-Aug-2000	1720	51.5	5.5	CARB	FEAT 3006
Sacramento	SB Hwy 5 to EB Florin	10-Aug-2000	2262	32.2	3.5	CARB	FEAT 3006
Sacramento	WB Garden to SB Hwy 5	22-Aug-2000	2980	46.3	5.3	CARB	MDL
Sacramento	WB El Camino to SB Hwy 5	23-Aug-2000	1567	50.9	5.2	CARB	MDL
Sacramento	NB Watt to WB Hwy 80	24-Aug-2000	2313	63.2	4.8	CARB	MDL
Sacramento	WB Richards to SB Hwy 5	25-Aug-2000	1792	50.9	5.6	CARB	MDL
Sacramento	SB Bradshaw to EB Hwy 50	28-Aug-2000	903	36.5	4.7	CARB	MDL
Sacramento	SB Norwood to WB Hwy 80	30-Aug-2000	1781	54.2	6.3	CARB	MDL
Sacramento	WB Hwy 50 to SB Watt	7-Sep-2000	1069	38.1	4.6	CARB	MDL
Sacramento	Airport to SB Hwy 5	8-Sep-2000	3913	32.6	5.5	CARB	MDL
Sacramento	EB Hwy 80 to NB Hwy 5	11-Sep-2000	1491	56.8	3.3	CARB	MDL
		Average	3737	41.6	5.4		
		Standard error		1.8	0.3	N=27	
		95% CI		3.6	0.5	2.056	t statistic

Summary of Central California On-Road Vehicle Emission Sampling Sites

Area/City	Sampling Site	Date	Sample Size	Average Emissions (g/L)		Data source	Instr. ID
				CO	NOx		
SAN JOAQUIN VALLEY							
Fresno	NB Hwy 41 south of Avenue 14	7-Sep-2000	2342	28.8	6.5	CARB	FEAT 3005
Fresno	SB Hwy 41 south of Avenue 14	8-Sep-2000	1013	33.5	5.8	CARB	FEAT 3005
Fresno	NB Golden State fo WB Hwy 180	11-Sep-2000	2102	55.2	5.6	CARB	FEAT 3005
Fresno	Central to NB Hwy 99	14-Sep-2000	2398	60.7	6.4	CARB	FEAT 3005
Fresno	Ventura to NB Hwy 99	15-Sep-2000	827	76.0	7.3	CARB	FEAT 3005
Stockton	Fremont to SB Hwy 5	29-Aug-2000	4982	59.7	10.3	CARB	ESP
Stockton	Charter to NB Hwy 5	30-Aug-2000	4870	41.0	9.8	CARB	ESP
Stockton	EB Hammer to SB Hwy 99	31-Aug-2000	5642	47.7	8.7	CARB	ESP
Fresno	Shaw to SB Hwy 99	6-Sep-2000	2486	45.8	9.3	CARB	ESP
Fresno	WB Herndon to SB Hwy 41	7-Sep-2000	4658	34.9	8.6	CARB	ESP
Fresno	NB Hwy 99 to EB Hwy 41/180	8-Sep-2000	2398	41.6	8.1	CARB	ESP
Fresno	McKinley to SB Hwy 41	12-Sep-2000	1710	36.8	8.1	CARB	ESP
Fresno	WB McKinley to NB Hwy 41	13-Sep-2000	1998	50.4	8.4	CARB	ESP
Fresno	NB Cedar to WB Hwy 180	14-Sep-2000	1260	58.2	9.6	CARB	ESP
Fresno	WB Shaw to NB Hwy 41	22-Sep-2000	1289	37.2	7.8	CARB	ESP
Fresno	WB Bullard to NB Hwy 41	25-Sep-2000	1366	34.1	7.7	CARB	ESP
Fresno	WB Jensen to NB Hwy 99	28-Sep-2000	2202	33.7	8.6	CARB	ESP
Stockton	EB Hammer to NB Hwy 99	13-Sep-2000	1197	56.5	5.4	CARB	MDL
Stockton	EB Cherokee to SB Hwy 99	14-Sep-2000	1899	65.9	6.9	CARB	MDL
Fresno	WB Shaw to NB Hwy 41	19-Sep-2000	1719	63.8	8.4	CARB	MDL
Fresno	WB White Bridge to EB Hwy 180	21-Sep-2000	892	60.1	1.3	CARB	MDL
Fresno	WB McKinley to SB Hwy 99	22-Sep-2000	1262	68.0	8.9	CARB	MDL
Fresno	WB Shields to NB Hwy 41	25-Sep-2000	1562	58.5	4.3	CARB	MDL
Average			2264	49.9	7.5		
Standard error				2.8	0.4	N=23	
95% CI				5.8	0.9	2.074	t statistic

Note: remote sensing sites with N<800 were deleted from the above tables before calculating average emission factors.