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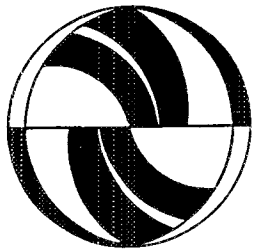
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A Fuel-Based Motor Vehicle Emission Inventory

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A Fuel-Based Motor Vehicle Emission Inventory

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ABSTRACT

A fuel-based methodology for calculating motor vehicle emission inventories is presented. In the fuel-based method, emission factors are normalized to fuel consumption and expressed as grams of pollutant emitted per gallon of gasoline burned. Fleet-average emission factors are calculated from the measured on-road emissions of a large, random sample of vehicles. Gasoline use is known at the state level from sales tax data, and may be disaggregated to individual air basins. A fuel-based motor vehicle CO inventory was calculated for the South Coast Air Basin in California for summer 1991. Emission factors were calculated from remote sensing measurements of more than 70,000 in-use vehicles. Stabilized exhaust emissions of CO were estimated to be 4400 tons/day for cars and 1500 tons/day for light-duty and medium-duty trucks, with an estimated uncertainty of $\pm 20\%$ for cars and $\pm 30\%$ for trucks. Total motor vehicle CO emissions, including incremental start emissions and emissions from heavy-duty vehicles were estimated to be 7900 tons/day. Fuel-based inventory estimates were greater than those of California's MVEI 7F model by factors of 2.2 for cars and 2.6 for trucks. A draft version of California's MVEI 7G model, which includes increased contributions from high-emitting vehicles and off-cycle emissions, predicted CO emissions which closely matched the fuel-based inventory. An analysis of CO mass emissions as a function of vehicle age

revealed that cars and trucks which were ten or more years old were responsible for 58% of stabilized exhaust CO emissions from all cars and trucks.

INTRODUCTION

Accurate emission inventories are needed to understand and control air pollution problems. Recent national air pollutant emission estimates published by the U.S. Environmental Protection Agency (EPA) indicate that in 1993 on-road vehicles contributed 62% of all carbon monoxide (CO) emissions, 32% of all nitrogen oxides (NO_x) emissions, and 26% of all volatile organic compound (VOC) emissions in the United States.¹ The contribution of motor vehicle emissions is even greater within urban areas.² An accurate motor vehicle emission inventory (MVEI) is therefore essential for a correct understanding of air pollution problems.

At present, the MVEI is calculated using travel-based models which combine gram-per-mile emission factors with activity data, expressed as vehicle miles travelled (VMT), for an array of vehicle subgroups. Activity and emission factors are resolved by vehicle class (e.g., light-duty passenger car, heavy-duty truck, etc.), engine/emissions control technology, and age. Emission factors are derived from dynamometer testing of vehicles recruited from the in-use fleet through mailings to registered vehicle owners. Travel demand models are used to estimate total VMT, as well as the breakdown of VMT by vehicle type and speed. Since gram-per-mile emissions vary significantly with engine load and vehicle speed, emission factors and activity data must be calculated for an array of vehicle speeds.

Emission inventories are often inconsistent with emissions measured from on-road vehicles. In 1987, emission factors for CO and HC measured from vehicles at the Sherman Way tunnel in Van Nuys, CA were found to be higher than those predicted by California's EMFAC 7C emission factor model by factors of approximately 3 and 4, respectively. NO_x emission factors predicted by EMFAC 7C agreed well with the on-road measurements.³⁻⁵ Fujita et al.⁶ arrived at similar conclusions by analyzing measured ambient pollutant concentrations from the 1987 Southern California Air Quality Study.⁶ Emission factor ratios predicted by the EMFAC 7F model compared more favorably to

IMPLICATIONS

The fuel-based methodology presented in this study may be used to calculate a motor vehicle emission inventory for any geographic area for which emissions data are available. The fuel-based approach requires fuel use data, which is obtained from state tax records, and on-road emissions data, which are becoming more available as remote sensing devices are used in Inspection & Maintenance programs around the country. Currently, the fuel-based method may be used to verify official emission inventories of CO. With further development of the methodology and advances in remote sensing capabilities for NO_x and HC, the fuel-based approach could become the standard emission inventory methodology.

on-road measurements at the Caldecott Tunnel in Oakland, CA in the summer of 1994.⁷ EMFAC 7F correctly predicted the VOC/NO_x ratio but still underpredicted the CO/NO_x ratio by a factor of 1.5-2.2. Comparisons between predictions from EPA's MOBILE model and measured on-road vehicle emissions have been reported by Pierson et al. for the Fort McHenry tunnel in Baltimore, MD, and the Tuscarora tunnel on the Pennsylvania turnpike.⁸

Since inventories of vehicle emissions are so uncertain, development of an independent method of calculating the MVEI has been identified as a high priority for air quality research.⁹ The alternate approach should be based on emission factors measured from large, random samples of on-road vehicles and on activity data which can be verified independently. In the present study, a fuel-based methodology is proposed in place of the traditional travel-based approach. In a fuel-based inventory, emission factors are normalized to fuel consumption rather than miles travelled, and activity is measured as the amount of fuel consumed. Fuel-based emission factors are calculated from on-road emissions measurements such as those from remote sensors and tunnel studies. Precise fuel use data are readily available from records of taxes paid when fuel is sold.

Much has been learned about real-world vehicle emissions by use of infrared remote sensors which measure the CO/CO₂ and HC/CO₂ ratios in the exhaust of individual vehicles as they drive by the sensors.¹⁰⁻¹⁶ Newer remote sensors also include an ultraviolet channel for measuring the NO/CO₂ ratio in vehicle exhaust.¹⁷ Remote sensing studies show that total fleet emissions are dominated by the extremely high emissions of a small fraction of on-road vehicles. For both CO and HC, it is typical to find that less than 10% of the fleet is responsible for half of the total exhaust emissions and that less than 20% of the fleet is responsible for 80% of the emissions. Although the CO high-emitters are not necessarily the same vehicles as the HC high-emitters, there is significant overlap between the groups.¹⁸ A similarly skewed distribution in vehicular NO emissions has been reported.¹⁷ Calculation of accurate emission factors for use in inventory calculations must therefore include high-emitting vehicles, weighted according to their presence in the on-road fleet.

Discrepancies between model predictions and real-world observations may result from limitations of the current MVEI methodology. Because of the expense and time involved in dynamometer testing, the sample of vehicles tested as part of the current MVEI methodology may be too small to represent adequately each combination of vehicle class, technology type, and age. More importantly, the test fleet may not be representative of the on-road fleet since the highest-emitting vehicles—those in a severe state of disrepair and those with emission control systems that have been

tampered with—are less likely to be submitted voluntarily for emissions testing. Insufficient representation of high-emitting vehicles in the test fleet leads to predicted fleet-average emission factors that are too low.

Current methods of estimating VMT lead to uncertainty in the MVEI. The California Department of Transportation (Caltrans) estimates total statewide VMT from fuel use data and an estimate of the average fuel economy for all vehicles in the state.¹⁹ VMT estimates for each air basin are developed using travel demand models. Basin-wide VMT estimates are checked against measurements of travel across access points (e.g., bridges) and compared to total statewide VMT. Additional uncertainty in the MVEI is introduced when VMT is resolved to vehicle subgroups and by vehicle speed.

Since gram-per-mile emissions vary significantly with vehicle speed and engine load, correction factors are required to predict emissions under conditions that differ from the urban driving schedule defined in the Federal Test Procedure. Speed correction factors are derived by testing vehicles on an array of driving schedules with different average speeds. Roadway grade also affects engine load, but current models do not account for this effect.

A major advantage of the methodology presented here is that fuel-based emission factors vary much less than travel-based emission factors as driving mode changes. In a remote sensing study of 23 vehicles, Stedman et al.^{14,20} measured the CO and HC emissions as the vehicles were driven through a series of 10 modes (idle, cruises of 5, 15, 30, and 45 mph, light, medium, and hard accelerations, and two identical decelerations). CO emissions were lowest and least variable for the cruises and light and medium accelerations, somewhat higher and more variable for the idle and deceleration runs, and significantly higher during hard acceleration. HC emissions were comparable for all the modes except for the decelerations, where significantly higher and more variable emissions were observed. In a study of driving mode effects on a large, in-use vehicle fleet, Pierson et al.⁸ measured emission factors for uphill (loaded) and downhill driving on a 3.76% grade in the Fort McHenry tunnel. On a gram-per-mile basis, uphill emissions were higher than downhill emissions by factors of 1.52, 1.86, and 2.19 for non-methane hydrocarbons (NMHC), CO, and NO_x, respectively. When normalized to fuel consumption, uphill to downhill emission factor ratios were 0.95 for NMHC, 1.19 for CO, and 1.38 for NO_x.

The objectives of this study were to describe a fuel-based methodology for calculating motor vehicle emissions; to apply this methodology to calculate a 1991 summertime inventory of running exhaust CO emissions for California's South Coast Air Basin; and to compare the fuel-based inventory with official California estimates of motor vehicle emissions.

METHOD

A fuel-based emission inventory uses emission factors normalized to fuel consumption (i.e., grams of pollutant emitted per gallon of fuel burned). Average emission factors for subgroups of vehicles are weighted by the fraction of total fuel used by each vehicle subgroup to obtain an overall fleet-average emission factor. The fleet-average emission factor is multiplied by regional fuel sales to compute pollutant emissions.

Emission Factors

By carbon balance, it is possible to relate the amount of pollutant emitted to the amount of fuel burned if the molar exhaust concentrations of CO₂, CO, and HC are measured. An emission factor E_p for pollutant P can be computed as follows:

$$E_p = \left(\frac{[P]}{[CO_2] + [CO] + [HC]} \right) \cdot \frac{w_c \rho_f M_p}{12} \quad (1)$$

where E_p is in units of grams of pollutant P emitted per unit volume of fuel consumed, $[P]$ is the exhaust concentration of pollutant P, w_c is the carbon weight fraction of the fuel, ρ_f is the fuel density, and M_p is the molecular weight of P. The denominator of Equation 1 represents a sum of carbon atoms in the exhaust; the factor of 12 is the atomic mass of carbon. For example, suppose the pollutant P of interest is carbon monoxide, and that infrared remote sensing measurements of $Q_1 = [CO]/[CO_2]$ and $Q_2 = [HC]/[CO_2]$ are available. Then

$$E_{CO} = \left(\frac{Q_1}{1 + Q_1 + 3Q_2} \right) \cdot w_c \rho_f \left(\frac{28}{12} \right) \quad (2)$$

The equation is written for an exhaust hydrocarbon concentration expressed on a propane-equivalent basis; the factor of 3 in the denominator is needed to convert from propane molecules to carbon atoms. Although remote sensors measure Q_1 and Q_2 directly, remote sensing data are generally reported as exhaust gas concentrations such as %CO and %HC. Q_1 and Q_2 may be back-calculated as the ratios of these values to the %CO₂ value which is also available, but not always presented or discussed in the literature. Each remote sensor measurement is coupled to a video image of the vehicle license plate. License plate numbers are matched to registration records from the state Department of Motor Vehicles (DMV) to obtain the information needed to classify each vehicle. In California, vehicle registration records include the vehicle make, model, model year, fuel type, body code (which identifies the functional capabilities of the vehicle, e.g., pick-up truck, school bus, passenger vehicle, etc.), body type (e.g. four-door, hatchback, station wagon, etc.), vehicle identification number, and whether the vehicle is registered for commercial or private use. With this information it is possible to disaggregate the vehicle fleet and compute average emission factors for subgroups of vehicles. In this study, vehicles are grouped by

model year i and vehicle class j . Thus, E_{pij} represents the average emission factor for all vehicles of model year i and vehicle class j .

Vehicle emissions also can be measured in roadway tunnels, where elevated levels of motor vehicle exhaust are present. Tunnel measurements provide composite emission factors for the entire fleet of vehicles travelling through the tunnel. If the fleet composition within the tunnel is variable over time, separate emission factors can be derived for different vehicle types.²¹ Using this approach, emission factors can be determined for vehicle types which are difficult to measure with remote sensors, and for pollutants other than CO and HC. For example, Pierson et al.⁸ used this technique to compute emission factors for heavy-duty trucks at the Tuscarora tunnel, where trucks varied from 6% to 80% of the vehicles travelling through the tunnel at various times of day.

Vehicle Activity

In a fuel-based inventory, vehicle activity is measured by fuel use. Precise fuel sales data are available at the statewide level from tax records. Calculation of the emission inventory for individual air basins requires that fuel use be resolved to the same spatial scale. Fuel use can be apportioned by tracking fuel shipments from major suppliers, through surveys of filling stations, or by considering the breakdowns of population

Table 1. Fuel economy data for cars and trucks.^a

Model Year	Fuel economy (mpg)	
	Cars	Trucks
Pre-75 ^b	14.2	13.7
1975	15.8	13.7
1976	17.5	14.4
1977	18.3	15.6
1978	19.9	15.2
1979	20.3	14.7
1980	23.5	18.6
1981	25.1	20.1
1982	26.0	20.5
1983	25.9	20.9
1984	26.3	20.5
1985	27.0	20.6
1986	27.9	21.4
1987	28.1	21.6
1988	28.6	21.2
1989	28.1	20.9
1990	27.7	20.7
1991	28.0	21.3

^a Murrell et al.²² Combined city/highway fuel economy estimates. Values listed here have not been adjusted to account for lower fuel economy observed in real-world driving.

^b All vehicles of model year 1974 and earlier. The fuel economy for cars shown here is for the 1974 model year from AAMA Facts and Figures.⁴⁰ The fuel economy for pre-1975 model year trucks is approximated using the same value reported for the 1975 model year.

and of the number of registered vehicles among all air basins in the state.

The fuel used in each air basin must be apportioned among vehicle subgroups. Fuel-based emission factors cannot be weighted by VMT fractions because different vehicles use different amounts of fuel to travel the same distance. For example, light-duty trucks use more fuel on average than light-duty passenger cars per mile travelled.²² In addition, vehicle fuel economy has improved dramatically over the last two decades, as shown in Table 1. Each year, EPA publishes average fuel economy values, weighted by new vehicle sales, for all vehicles of each class (cars and trucks), for all vehicles sold by each manufacturer, and for foreign and domestic vehicles.²² The actual fuel economy realized by vehicles on the road varies with vehicle model, but is lower on average than unadjusted EPA estimates by about 20%.²³ In 1985, the average shortfall associated with light-duty trucks was 20.1%, compared to a shortfall of 18.7% for cars. No significant trends in fuel economy shortfall have been correlated to automobile size or to vehicle age.²³ Since the calculation of fuel use by each vehicle subgroup requires only that the relative fuel economies be known (note that the fuel economy M_{ij} appears in the summations in both the numerator and the denominator of Equation 3, below), EPA fuel economy values were used without adjustment for the present analysis.

The travel fractions v_{ij} of each vehicle subgroup are measured directly as the frequencies at which vehicles of each subgroup pass a remote sensor. For example, if a total sample of N vehicles drive by a remote sensor, then $v_{ij} = n_{ij}/N$ where n_{ij} is the count of vehicles in subgroup (i,j) . Using average fuel economy M_{ij} and vehicle travel fraction v_{ij} , the fraction f_{ij} of total fuel used by each vehicle subgroup (i,j) is:

$$f_{ij} = \frac{(v_{ij} / M_{ij})}{\sum_{j=1}^C \sum_{i=Y_1}^{Y_n} (v_{ij} / M_{ij})} \quad (3)$$

where Y_1 is the model year of the oldest vehicles, Y_n is the most recent model year, and C is the number of vehicle classes being considered.

Combining Activity and Emission Factors

The overall fleet-average emission factor for pollutant P ,

$$\bar{E}_P = \sum_{j=1}^C \sum_{i=Y_1}^{Y_n} f_{ij} E_{Pij} \quad (4)$$

is multiplied by total fuel use to compute vehicular emissions of pollutant P . Emissions can be calculated by vehicle class by applying Equations 3 and 4 separately for each class j .

APPLICATION

The fuel-based methodology described above was applied to California's South Coast Air Basin. Stabilized exhaust emissions of carbon monoxide were calculated for

gasoline-powered light-duty cars and light/medium-duty trucks for the summer of 1991.

Emission factors were calculated from remote sensing measurements made by Stedman, Bishop, and co-workers from the University of Denver as part of a study of on-road CO and HC emissions in California.¹⁴ The complete data set, as received from the University of Denver, contained 91,679 valid CO and HC measurements matched to vehicle registration records.²⁴ Vehicles were sampled at 13 sites in the Los Angeles and San Francisco Bay areas during May through July of 1991. Because the present application is for the SoCAB, only data obtained from the Los Angeles area sites were used. One of the sites, a parking lot, was excluded because vehicles were measured while operating in cold start mode. Summary descriptions of the remaining seven sites, as reported by Stedman et al.,¹⁴ are given below.

Rosemead: Rosemead Blvd. north of the cloverleaf intersection with the Pomona Freeway (I-60) in south El Monte. Rosemead Blvd. is a flat, six-lane divided highway with traffic signals and a posted speed limit of 50 mph; however, vehicles were measured while travelling at speeds ranging from nearly 0 to 50 mph. During the monitoring, all south-bound traffic was funneled into a single lane to increase the measurement rate. Because this site was used for a random pullover study which lasted about two weeks, more data are available for the Rosemead site than for all of the other sites combined. General Motors Research used their own remote sensor to obtain side-by-side measurements at this site; these results have been reported elsewhere.²⁵

Peck: Interchange of Peck Road to I-10. Driving modes consisted of moderate accelerations on the on-ramp and decelerations on the curved off-ramp.

Beach: Beach Boulevard to Southbound I-405. Two remote sensors were placed beyond the metering lights on the curved, 2% uphill grade. Vehicles were monitored as they accelerated past both units to merge with the freeway. Heavy congestion on the freeway restricted traffic flow during the morning hours.

Lynwood: Long Beach Blvd. one block north of Norton. Long Beach Blvd. is a level, four-lane surface street with light traffic and average speeds of 10-25 mph. In the afternoon, one of the lanes was closed so that more vehicles could be sampled.

El Segundo: El Segundo Blvd. to Southbound I-405. Two instruments were located past the metering lights on an uphill on-ramp, approximately 20 and 100 feet from the ramp exit. Vehicles were observed to be gently accelerating as they passed the first sensor, and in cruise mode as they passed the second sensor.

Broadway: Northbound Broadway to northbound I-101. Two units were used to monitor vehicles along a downhill on-ramp. Driving modes included light accelerations and cruises at 20-40 mph.

Vermont: Southbound Vermont Ave. to westbound I-10. Vehicles were monitored as they accelerated past the first unit, located on a steep (~5% slope) uphill on-ramp, and as they cruised or decelerated slightly past a second sensor, located 140 feet away on a more gentle slope.

Using fuel-code data from the vehicle registration records, all vehicles burning fuels other than gasoline were excluded. Of 80,775 measurements at the seven sampling sites, 1,424 measurements of non-gasoline vehicles were excluded. 1,189 measurements were matched to registration records with no fuel code designated; these vehicles were assumed to be predominantly gasoline-powered and thus included in the calculation. One hundred and ten measurements were matched to vehicles registered as 1992 model year vehicles. These measurements were grouped together with the 4,690 measurements of 1991 vehicles.

Vehicles were classified as either cars or trucks according to body code and/or body type information in the registration records. The truck category includes both light-duty and medium-duty trucks. It was not possible to track these truck classes separately because the division is based on gross vehicle weight, a parameter that was not included with the registration data. However, since remote sensors are designed to sample emissions from vehicle tailpipes that are about 10-12 inches from the ground, light-duty trucks were sampled more efficiently than medium-duty trucks. Emissions could not be measured from heavy-duty vehicles with elevated exhaust pipes.

On many of the sampling days, a single remote sensor was used. For one day of sampling at the Rosemead site, and for all sampling at the Beach, Broadway, El Segundo, and

Vermont sites, two instruments were positioned in series to measure the same stream of vehicles. This arrangement allowed for a greater fraction of the passing vehicles to be sampled at least once, and for a large fraction of the vehicles to be sampled twice. Duplicate measurements for a single vehicle pass were averaged together and subsequently considered as a single measurement.

A gram-per-gallon emission factor was calculated for each remote sensor measurement using Equation 2 along with fuel properties of industry average gasoline, i.e., a carbon weight fraction $w_c = 0.87$ and a fuel density $\rho_f = 750$ g/l.²⁶ Average emission factors were computed for vehicles of each subgroup (*i,j*) at each site. All vehicles of 1974 vintage and older were grouped together, and as mentioned previously, 1992 vehicles were grouped with 1991 vehicles. Table 2 shows the total sample size and the mean model year for cars and trucks at each site.

Differences in average emissions between sites can be explained in large part by differences in average vehicle age.¹⁴ Age differences can be controlled for by calculating emission factors for each model year at each site and comparing only vehicles of the same age. Average emission factors for each model year of cars at each site are plotted in Figure 1. Figure 1 shows that vehicles of the same age have comparable emissions even when measured at different sites. The coefficient of variation of emission factors measured at the seven sites is similar for all model years of cars. The analogous plot for trucks exhibits the same trend of increasing emissions with vehicle age, but shows greater year-to-year fluctuations at each site because the truck sample was significantly smaller than the car sample, as indicated in Table 2.

Figure 1 also shows that emission factors measured at the Rosemead site are lower overall than those measured at the other sites. Rosemead emission factors are lowest or second lowest for every model year of cars back through 1979, with the exception of 1982, when Rosemead cars had the third lowest average emission factor. Recent model year vehicles at the Vermont Avenue site have emissions which are consistently higher than vehicles of the same age at the other sites. Emission factors for Vermont vehicles are 32% to 69% higher than the mean of the emission factors measured at the other six sites for vehicle model years 1983-1991. The positive offset in emission factors for recent model year vehicles at the Vermont site may have resulted from a portion of the fleet experiencing enrichment effects. Enrichment occurs when the computers of modern technology vehicles command a rich fuel-air ratio for increased power during high-load driving. Emissions of CO and HC from properly-functioning vehicles can increase by 2 to 3 orders of magnitude when the vehicles operate in commanded enrichment mode.^{27,28}

California statewide gasoline sales in 1991 totalled 13.2×10^9 gallons.²⁹ It has been estimated that 2.7% of this total was purchased for use in off-road engines such as farm

Table 2. Vehicle counts and mean model year for Los Angeles sites.^a

Location	Cars measured	Mean car model year ^b	Trucks measured	Mean truck model year ^b
Beach	6103	1986.0	1538	1985.8
Broadway	3372	1985.8	586	1985.8
El Segundo	2925	1985.7	732	1986.5
Lynwood	1374	1981.9	419	1982.1
Peck	4167	1984.4	946	1984.3
Rosemead	32297	1985.0	10425	1985.0
Vermont	5053	1984.4	804	1985.2
Totals	55291	1985.0	15450	1985.0

^a Data from 1991 remote sensing study of Stedman et al.¹⁴

^b Pre-1975 vehicles were considered as having a model year of 1974.

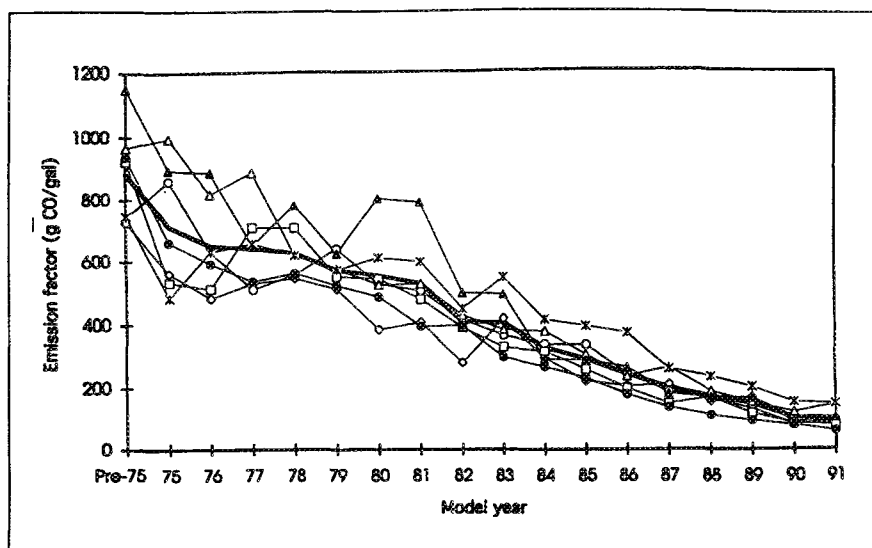


Figure 1. Average CO emission factors for cars of each model year at each site. The sites are: Beach (open diamond), Broadway (open circle), El Segundo (open triangle), Lynwood (filled triangle), Peck (open square), Rosemead (filled circle), Vermont (asterisk). The 7-site average emission factor data are plotted as a solid line. Data from the Rosemead, Vermont, and Lynwood sites are discussed further in the text.

equipment, construction equipment, and boat engines;³⁰ this amount was deducted from total gasoline sales. The fraction of fuel used in the SoCAB was calculated by considering the county-by-county breakdowns of population and registered vehicles. On a population basis, the SoCAB includes all of Orange county, 98% of Los Angeles county, 81% of San Bernardino county, and 72% of Riverside county.³¹ In 1991, 44% of the people residing in California lived in the SoCAB.^{32,33} Similarly in 1991, 40% of California vehicles were registered in the SoCAB.³² Using the average of population and vehicle registration data, 42% of statewide fuel use was apportioned to the SoCAB. Heavy-duty trucks and motorcycles, which are not included in this inventory calculation, were estimated to consume 11% of the gasoline in the SoCAB.³⁴ This amount was subtracted from the total on-road vehicle gasoline use. Gasoline used by cars, light-duty trucks, and medium-duty trucks in the SoCAB was therefore calculated to be 13.2×10^6 gallons per day. Using data from Tables 3 and 4, fuel use by cars was calculated to be 10.1×10^6 gallons per day and fuel use by trucks was calculated to be 3.1×10^6 gallons per day.

RESULTS

Two emission inventory calculations were performed: Lower-bound values were calculated using emission factors measured at the Rosemead site, and a best-estimate of the true emission inventory was calculated using an equal weighting of the emission factors measured at each of the seven remote sensing sites. Measured emission factors are presented in Table 3 for each model year of cars. Similar emission factor data for trucks are presented in Table 4. Estimates of the

uncertainty associated with the emission factors are included in Tables 3 and 4. For the Rosemead emission factors, the error estimates represent the standard error of the mean, a measure of the degree to which the sample mean represents the mean for the entire population (fleet) of vehicles. The uncertainty associated with the 7-site average emission factors is stated as ± 1 standard deviation, a measure of the site-to-site variability in average emission factors for each model year. Note that measurements from the Rosemead site make up about two-thirds of all the data and that the remaining data are not equally distributed among the other six sites, as indicated in Table 2.

Travel fractions for each model year of cars and trucks are presented in Tables 3 and 4, respectively. These values were calculated by averaging observed travel fractions from all seven sites for vehicles

of each model year. Site-by-site age distribution data for cars are plotted in Figure 2. According to the observed vehicle distributions, cars accounted for 81% and trucks 19% of the travel for these two vehicle classes. Travel fractions shown in Tables 3 and 4 were converted to fuel use fractions using Equation 3 and the fuel economy data presented in Table 1. As expected because of their lower fuel economy, trucks accounted for a greater fraction of fuel used (24%) than of distance traveled (19%).

Overall average emission factors for cars and trucks at the Rosemead site and for all seven sites combined are presented in Table 5. A correction factor was used to account for the bias of using only those remote sensing measurements which were matched to DMV registration records. Registration data were not available for out-of-state vehicles and for vehicles with unreadable or missing license plates. The correction factor was derived from Rosemead data, where the average CO concentration for 60,487 matched and unmatched vehicle measurements was 0.86%, whereas the average CO concentration for vehicles matched to DMV registration records (42,546 measurements) was 0.79%.¹⁴ The class-average emission factors were therefore increased by a factor of 1.09.

Final emission inventory results for summer 1991 are presented in Table 5; corresponding predictions of the California Air Resources Board (CARB) MVEI 7F model are shown for comparison. The fuel-based estimate of stabilized exhaust CO emitted by cars and light/medium-duty trucks is 2.3 times higher than the official emission inventory estimates of the MVEI 7F model. Fuel-based estimates are 2.2 and 2.6 times higher than MVEI 7F estimates for cars and trucks, respectively. Lower-bound inventory estimates—based on

Table 3. Measured travel fraction and emission factor data for cars in the SoCAB during 1991.

Model Year	Travel fraction ^a (%)	Fuel fraction ^b (%)	Rosemead		7-Site Average	
			Vehicles measured	Emissions ^c (g CO/gallon)	Vehicles measured	Emissions ^d (g CO/gallon)
Pre-75	3.80 ± 1.73	6.31	1533	936 ± 20	2509	885 ± 156
1975	0.67 ± 0.39	1.00	243	660 ± 47	416	712 ± 202
1976	1.20 ± 0.98	1.61	441	592 ± 32	736	649 ± 150
1977	1.87 ± 1.34	2.41	660	535 ± 25	1117	638 ± 132
1978	2.80 ± 2.15	3.32	1010	560 ± 20	1696	626 ± 87
1979	3.09 ± 2.17	3.58	1133	520 ± 20	1911	569 ± 49
1980	3.40 ± 2.12	3.41	1107	485 ± 19	1992	553 ± 129
1981	3.33 ± 1.22	3.12	1283	394 ± 17	2176	528 ± 135
1982	3.47 ± 0.82	3.14	1301	396 ± 16	2267	408 ± 67
1983	3.60 ± 0.51	3.27	1414	298 ± 14	2430	406 ± 89
1984	5.09 ± 0.75	4.55	2177	267 ± 10	3648	329 ± 53
1985	5.79 ± 0.90	5.05	2475	230 ± 9	4155	290 ± 62
1986	6.37 ± 1.05	5.38	2482	179 ± 7	4355	243 ± 65
1987	7.49 ± 1.89	6.27	2942	138 ± 5	5197	200 ± 50
1988	7.53 ± 1.91	6.20	3289	111 ± 4	5485	170 ± 37
1989	8.70 ± 2.80	7.29	3688	95 ± 3	6311	148 ± 36
1990	7.64 ± 2.49	6.50	3271	79 ± 3	5554	102 ± 27
1991 ^e	4.88 ± 1.81	4.10	1848	60 ± 4	3336	101 ± 33
All years	80.7	76.5	32297	314 ^f	55291	364 ^f

^a Observed $v_i = n_i/N$ averaged over all 7 sites ± 1 standard deviation.^b Percent of total gasoline use, computed using equation 3.^c Average emission factor measured at Rosemead site ± 1 standard error.^d Average of emission factors measured at 7 Los Angeles sites ± 1 standard deviation.^e Includes some 1992 vehicles; see text for discussion.^f Average emission factor for all model years calculated using equation 4.**Table 4.** Measured travel fraction and emission factor data for trucks in the SoCAB during 1991.

Model Year	Travel fraction ^a (%)	Fuel fraction ^b (%)	Rosemead		7-Site Average	
			Vehicles measured	Emissions ^c (g CO/gallon)	Vehicles measured	Emissions ^d (g CO/gallon)
Pre-75	1.30 ± 3.95	2.23	693	872 ± 29	973	899 ± 248
1975	0.21 ± 0.43	0.36	113	725 ± 70	166	959 ± 367
1976	0.33 ± 0.88	0.53	153	860 ± 70	229	733 ± 169
1977	0.41 ± 1.13	0.61	215	849 ± 59	309	817 ± 142
1978	0.56 ± 2.04	0.87	304	545 ± 37	425	583 ± 254
1979	0.70 ± 1.53	1.12	362	625 ± 38	521	636 ± 181
1980	0.48 ± 1.16	0.61	233	750 ± 54	345	798 ± 186
1981	0.51 ± 0.91	0.59	249	485 ± 43	369	660 ± 116
1982	0.50 ± 0.66	0.57	260	464 ± 36	391	648 ± 297
1983	0.69 ± 1.47	0.78	296	573 ± 42	461	535 ± 160
1984	1.03 ± 0.95	1.19	633	414 ± 24	901	421 ± 101
1985	1.60 ± 1.38	1.83	869	347 ± 19	1304	354 ± 100
1986	1.94 ± 0.92	2.13	1101	248 ± 14	1602	338 ± 46
1987	1.91 ± 1.50	2.08	1030	170 ± 11	1550	288 ± 108
1988	2.01 ± 3.05	2.23	1085	111 ± 8	1651	187 ± 67
1989	2.29 ± 2.59	2.58	1263	96 ± 7	1895	151 ± 53
1990	1.90 ± 2.91	2.16	1014	64 ± 6	1544	124 ± 70
1991 ^e	0.94 ± 0.48	1.04	552	54 ± 5	814	89 ± 63
All years	19.3	23.5	10425	362 ^f	15450	415 ^f

NOTES:

^a Observed $v_i = n_i/N$ averaged over all 7 sites ± 1 standard deviation.^b Percent of total gasoline use, computed using equation 3.^c Average emission factor measured at Rosemead site ± 1 standard error.^d Average of emission factors measured at 7 Los Angeles sites ± 1 standard deviation.^e Includes some 1992 vehicles; see text for discussion.^f Average emission factor for all model years calculated using equation 4.

emission factors measured at the Rosemead site—are higher than MVEI 7F predictions by factors of 1.9 for cars and 2.3 for trucks. In Table 6, the fuel-based inventory of stabilized CO exhaust emissions is combined with MVEI 7F estimates of incremental start emissions and emissions from other vehicle types, such as heavy-duty vehicles. Total on-road vehicle CO emissions in the SoCAB were computed to be 7900 tons/day.

Figure 3 shows the mass of CO emitted by cars of each model year. Cars which were ten or more years old (model year ≤ 1981) contributed 59% of the stabilized exhaust CO from all cars. Similarly, trucks that were ten or more years old were responsible for 55% of the stabilized exhaust CO from all trucks. Despite the large contribution to total emissions from older vehicles, it should be noted that malfunctioning new vehicles can emit much more than well-tuned older vehicles.¹⁸ Figure 3 also shows the cumulative percentage of total CO as a function of model year, calculated using both fuel-weighted and travel-weighted activity distributions. Since travel-weighted calculations ignore the upward trend in fuel economy between 1975 and 1990 vehicles, past analyses have understated the fraction of emissions contributed by older, less fuel-efficient vehicles.

DISCUSSION

Accuracy of Fuel-Based Inventory

The accuracy of a fuel-based emission inventory depends primarily on how well the vehicles and driving modes from which emission factors were measured represent the entire area under study. In the present case, seven sites were used

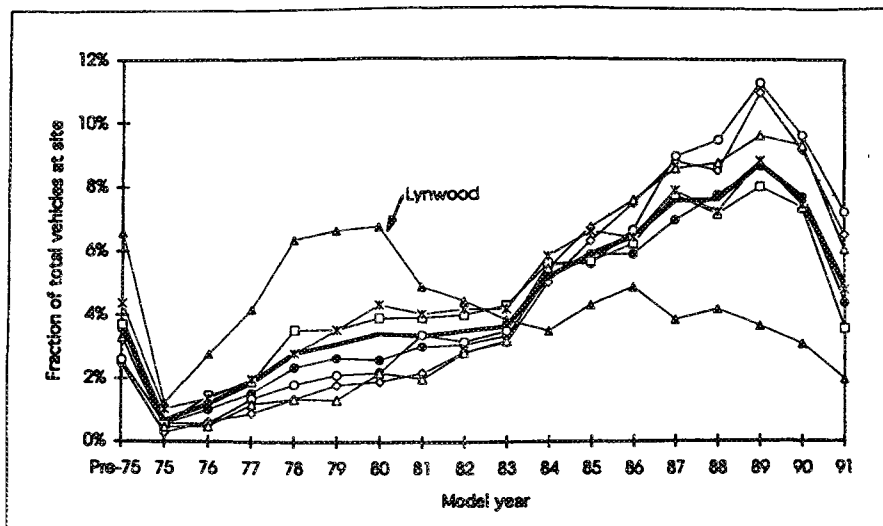


Figure 2. Observed age distributions for cars at each sampling site. Lynwood (filled triangle) is identified by arrow. Symbols for other sites are: Beach (open diamond), Broadway (open circle), El Segundo (open triangle), Peck (open square), Rosemead (filled circle), and Vermont (asterisk). The 7-site average age distribution is plotted as a solid line.

to represent the vehicle population of the South Coast Air Basin. The sites were well distributed geographically, and six of the seven sites were on high-volume traffic segments. The seventh site, in Lynwood, was located in a low-income neighborhood where emissions increased with age more sharply than at other sites.^{13,14} Site-to-site differences in the percentage of foreign versus domestic vehicles and in the vehicle age distribution suggest that the sampling sites represented a range of socioeconomic levels.

Driving conditions varied from site to site and with time of day at the same site. However, most vehicles were sampled while cruising, lightly accelerating, or lightly decelerating—modes for which gram-per-gallon CO emissions are similar, as discussed in the introduction. At the Rosemead site, most vehicles were measured while cruising at moderate speeds on a level roadway, the driving mode which leads to lowest CO emissions. In contrast, one of the two remote sensors at the Vermont site measured

Table 5. Stabilized CO exhaust emissions, SoCAB, summer 1991.

	Emission factors ^a		Gasoline used (10 ⁶ gal/day)	Fuel-based Inventory		MVEI 7F (tons/day)	Ratio of Best-estimate to MVEI 7F
	Rosemead (g CO/gal)	7-site		Lower-bound ^b (tons/day)	Best-estimate ^c (tons/day)		
Cars	342	397 ± 80 ^d	10.1	3800	4400 ± 900	1963	2.2 ± 0.5
Trucks	394	452 ± 134 ^d	3.1	1300	1500 ± 450	595	2.6 ± 0.8 ^e
Totals	354	410 ± 93 ^f	13.2	5100	5900 ± 1350	2558	2.3 ± 0.5

^a Emission factors from Tables 3 and 4 have been increased by a factor of 1.09 (see text).

^b Calculated using Rosemead emission factors.

^c Calculated using 7-site average emission factors.

^d Uncertainty bounds calculated as ±20% of class-average emission factor for cars and ±30% of class-average emission factor for trucks. (See text for discussion of uncertainty)

^e Ratio calculated using unrounded emissions estimate of 1537 tons/day

^f Total fleet-average emission factor and uncertainty bounds were calculated by weighting the emission factor and uncertainty for each vehicle class by the fractions of fuel used by each class.

Table 6. On-road vehicle CO Emission Inventory, SoCAB, summer 1991.

	MVEI 7F Inventory			Starts/ Total	Fuel-Based Inventory		
	Start ^a Emissions (tons/day)	Stabilized Exhaust (tons/day)	Total CO Emissions (tons/day)		Stabilized Exhaust (tons/day)	Total CO ^b Emissions (tons/day)	Starts/ Total
Cars	976	1963	2939	33%	4400 ± 900	5400	18%
Trucks	289	595	884	33%	1500 ± 450	1800	16%
HD+MC ^c	5	515	520	—	500	500	—
Diesel ^d	3	144	147	—	150	150	—
Total	1273	3217	4490	29%	6600 ± 1350	7900	16%

^a Incremental cold + hot start emissions from MVEI 7F.³⁴

^b Incorporates MVEI 7F estimates for incremental start emissions, heavy-duty gasoline-powered trucks, motorcycles, and all diesel vehicles so that total motor vehicle CO inventory can be calculated. No uncertainty estimates are available for the MVEI 7F emissions estimates. Thus, uncertainty in the total reflects only the uncertainty in estimates of stabilized emissions for cars and trucks.

^c Heavy-duty gasoline-powered trucks plus motorcycles. MVEI 7F provides no separate estimates for start emissions from HD gasoline-powered trucks.

^d All diesel vehicles, including cars, light-duty trucks, heavy-duty trucks, and urban buses.

vehicles as they were accelerating on a 5% uphill grade. Depending on the rate of acceleration, such moderate to high-load driving conditions can lead to increased gram-per-gallon CO emissions in most vehicles and likely caused some hard accelerating vehicles at this site to enter enrichment mode.

The lower-bound emission inventory presented in Table 5 was calculated using emission factors measured (at the Rosemead site) from a very large sample of vehicles operating in the mode which corresponds to lowest CO emissions. Best-estimate inventory estimates were calculated using an equal weighting of data from all seven remote sensing sites. Therefore, a fleet with a higher than average percentage of poorly maintained older vehicles (Lynwood) and a site where one of two remote sensors measured vehicles during moderate to high-load driving conditions (Vermont) were each included with a one-seventh weighting factor.

The accuracy of fleet-average emission factors also depends on the accuracy of the age distribution used to weight emissions data from each vehicle model year. Age distributions for cars and trucks were calculated by weighting equally the age distributions observed at the seven remote sensing sites. These age distributions, shown in Tables 3 and 4, agree well with those developed by CARB for use with the MVEI 7F model, as discussed below. Uncertainty in the average travel fractions for each model year results from site-to-site variations in the car-to-truck ratio, and from the atypical age distribution of the Lynwood site, where a large percentage of older vehicles was observed. As shown in Figure 2, age distributions for the remaining six sites were similar to one another. Because of smaller sample sizes, uncertainties in the travel fractions by model year for trucks are larger than the corresponding uncertainties for cars.

From this discussion, it is clear that the fuel-based emission inventory estimates are sensitive to the weighting applied to Lynwood data. The Lynwood site is not unique in the Los Angeles area; however, further study is needed to determine whether vehicles similar to the Lynwood fleet account for one-seventh of the SoCAB vehicle fleet. The importance of the weighting factor for Lynwood data was examined by recalculating fleet-average emission factors for cars and trucks, assuming that emission factors and age distributions measured at Lynwood represented only 4% of the entire SoCAB vehicle population. With this adjustment, fleet-average emission factors and corresponding inventory estimates would be only 7% lower than the best-estimate values

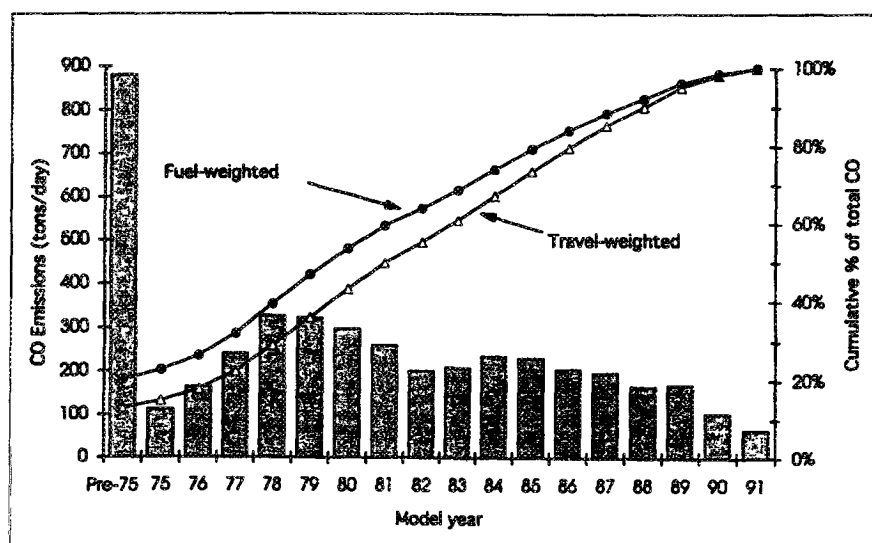


Figure 3. CO mass emissions from cars of each model year. Also shown is the cumulative contribution to total CO emissions as a function of vehicle age, using fuel-weighted and travel-weighted activity distributions.

reported in Table 5. Thus, even if the fleet of vehicles at Lynwood represents a fraction of the SoCAB vehicle population smaller than the initially assumed one-seventh (15%), the CO inventory will not differ substantially from the best-estimate values of Table 5.

Reduced fuel economy for CO high-emitters was not factored into the present inventory calculation. The presence of significant amounts of CO in vehicle exhaust causes a reduction in the thermal efficiency of combustion.³⁵ As a result, high-emitting vehicles obtain lower fuel economy than similar vehicles which have lower CO emissions. Inclusion of fuel economy penalties for CO high-emitters would shift more of the fuel use to the high-emitters and result in higher calculated CO emissions, relative to the values presented in Table 5.

Uncertainty of Inventory Estimates

Uncertainty in the fuel-based inventory results from uncertainty in the emission factors measured for each vehicle model year and from the weighting factors used to combine the emission factor data. In general, large numbers of measurements are required to ensure that average emission factors are measured precisely for each vehicle model year at each sampling site. For example, the large number of measurements at the Rosemead site ensured the precise calculation of emission factors for all vehicle model years at that site. High precision in the measurements at this site is indicated by the small standard errors associated with the yearly emission factors presented in Tables 3 and 4. Smaller vehicle samples resulted in larger standard errors in emission factors at each of the other sites.

Additional uncertainty results from weighting the data from individual sampling sites to calculate a basin-wide fleet-average emission factor. From the preceding discussion of vehicle populations and driving modes included in this study, it is likely that the true average emission factor for all SoCAB cars and trucks of any given model year lies within the range of yearly emission factors measured at the seven sampling locations (see Figure 1 for cars). Thus, an absolute lower-bound on the inventory can be calculated using the lowest measured emission factors for cars and trucks of each model year. Following this approach, it was found that the true inventories for cars and trucks could not be more than 23% and 36% below the best-estimate values presented in Table 5. Repeating the calculation with the highest measured emission factors for each model year produced absolute upper-bound estimates for cars and trucks which were 34 and 49% higher than best-estimate values.

The preceding calculations give insight into the extreme bounds on the inventory. As indicated in Figure 1, individual sites tend to be consistently above or below the 7-site average for all model years. More realistic uncertainty bounds were estimated using the standard deviations of emission

factors for each vehicle model year. Therefore, the upper uncertainty bound was calculated using the mean plus one standard deviation as the emission factor for each car and truck model year. The lower uncertainty bound was calculated using the mean minus one standard deviation as the emission factor for each model year. By this method, uncertainties associated with the best-estimate values were calculated to be $\pm 20\%$ for cars and $\pm 30\%$ for trucks. These uncertainties were applied to the 7-site fleet-average emission factors and CO inventory estimates presented in Table 5.

Comparison Between Fuel-Based and MVEI 7F Methods

Consistent with the fuel-based calculation, MVEI 7F assigns 43% of statewide VMT for cars and trucks to the SoCAB. The age distributions for cars and trucks used in MVEI 7F are similar to the 7-site average age distributions used in the present calculations, as shown in Figure 4. MVEI 7F activity data indicate that 80% of total car and light/medium-duty truck miles are driven by cars, consistent with the 81% travel fraction for cars observed on-road in the SoCAB and used in the fuel-based inventory. MVEI 7F reports fuel usage of 9.5×10^6 gallons per day for cars and 3.6×10^6 gallons per day for trucks. These values are consistent with the fuel use calculations of the present study, although MVEI 7F assigns a somewhat greater fraction of fuel use to trucks. When MVEI 7F fuel use and age distribution data were combined with fuel-based emission factors of the present study, calculated CO emissions were still 2.1 times higher than MVEI 7F predictions for cars and trucks combined. Therefore, differences between MVEI 7F model predictions and the fuel-based inventory presented here result mainly from differences in emission factors.

A draft version of CARB's revised motor vehicle emission inventory model, MVEI 7G, predicts CO emissions which closely match the fuel-based inventory. MVEI 7G estimates that stabilized CO emissions for the SoCAB in summer 1991 were 4704 tons/day for cars and 1587 tons/day for trucks. The more than two-fold increase in predicted CO emissions between versions 7F and 7G of the MVEI model results from the inclusion in MVEI 7G of increased contributions from enrichment mode driving events and from correction factors which account for the under-representation of high-emitting vehicles in the dynamometer emissions testing program.³⁶

Additional CO Emissions

Stabilized exhaust emissions are shown in the context of total vehicular CO emissions in Table 6. In MVEI 7F, cold- and hot-start emissions are calculated from the difference between emission factors when vehicles are started (i.e., before the catalyst reaches operating temperature) and after the engine and emissions control equipment have reached

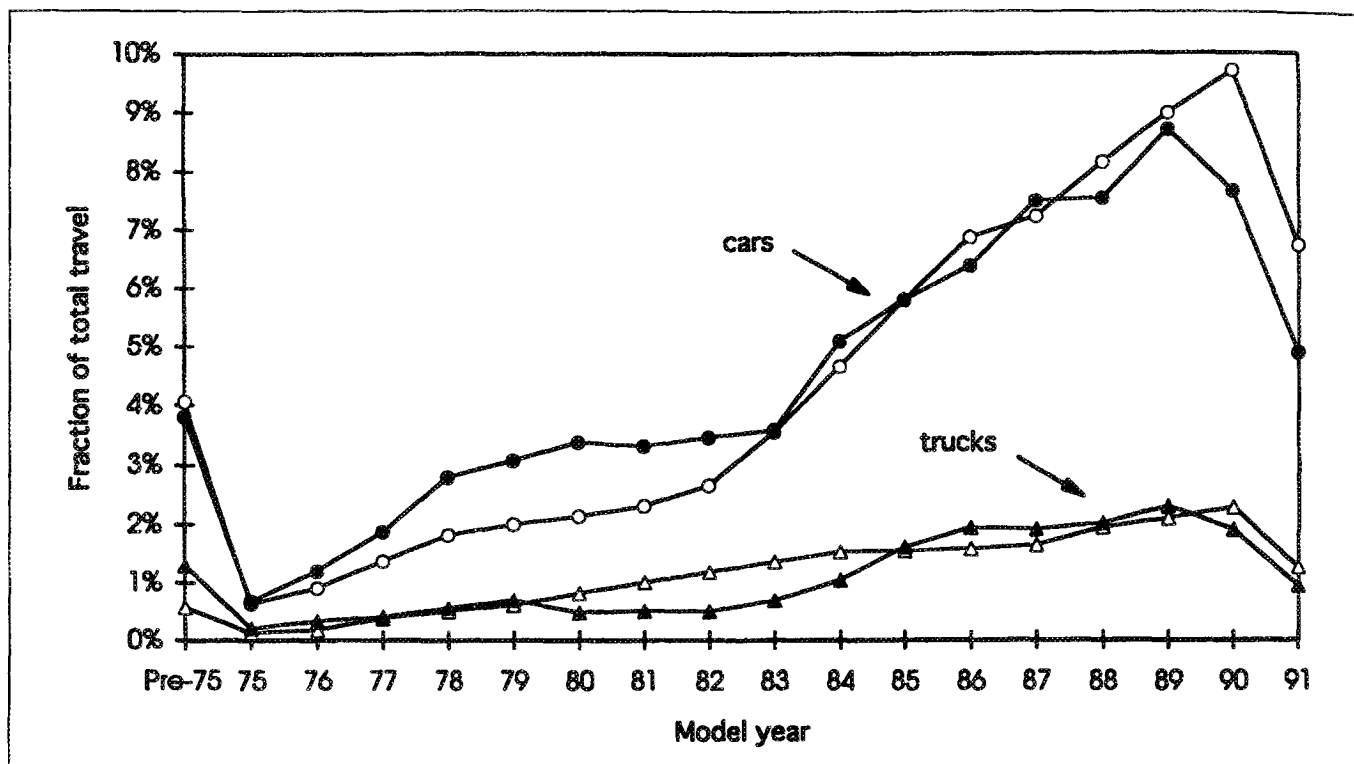


Figure 4. Comparison of car and truck age distributions used in fuel-based (solid symbols) and MVEI 7F (open symbols) calculations.

stable operating conditions. Stabilized emissions are estimated assuming that all travel occurs after vehicles have reached stabilized conditions. Additional emissions which result from vehicle starting are termed incremental start emissions. Analogously, the fuel-based stabilized CO inventory considers that all fuel is used during stabilized vehicle operation and does not include incremental emissions associated with vehicle starting.

According to MVEI 7F, incremental start emissions comprised 33% of all summertime CO emissions from cars and light/medium duty trucks in summer 1991.³⁴ The fuel-based inventory developed in this study indicates that stabilized exhaust emissions are higher than suggested by MVEI 7F. This increase in stabilized emissions reduces the relative importance of incremental start emissions to only 16% to 18% of total summertime CO emissions from cars and trucks, as shown in Table 6. Therefore, MVEI 7F probably overstates the importance of incremental start emissions.

The present fuel-based inventory includes stabilized exhaust emissions from gasoline-powered light-duty cars, light-duty trucks, and medium-duty trucks. According to the inventory totals presented at the right of Table 6, stabilized exhaust emissions from these vehicles accounted for 75% of all motor vehicle CO emissions in the SoCAB. As shown in Table 6, motorcycles and heavy-duty gasoline-powered trucks contributed an estimated 6% and diesel vehicles contributed only 2% of total vehicular CO emissions. Emissions associated with enrichment may be included to some

extent in the present fuel-based inventory because some vehicles measured at the Vermont site were likely operating in enrichment mode. Enrichment is not included in the MVEI 7F estimates; however, results from the Auto/Oil program indicate that inclusion of off-cycle emissions will raise estimates of the exhaust CO inventory by about 9%.³⁷

Comparison to MOBILE

The methods used to quantify emissions in EPA's MOBILE model differ from those used by MVEI 7F. Instead of calculating separately the additional emissions associated with vehicle starts, MOBILE uses a single composite emission factor that reflects a weighted average of cold start, hot start, and stabilized exhaust emissions. Fuel-based inventory results may be compared to MOBILE predictions by specifying 100% stabilized operation within the MOBILE model.³⁸

Future Applications

A fuel-based inventory can be calculated for any pollutant in any region for which representative emission factor and fuel use data are available. Emission factor data are already available for many U.S. locations from tunnel and remote sensing studies. The use of remote sensors in Inspection and Maintenance programs is producing data for additional areas. Exhaust emissions of CO are measured accurately by remote sensors. The use of remote sensing HC measurements for inventory purposes is more complicated. Remote sensors can accurately measure the

total mass of alkanes in vehicle exhaust, but measure only a portion of the olefins and aromatics. Stephens³⁹ has shown that if the speciation of vehicle exhaust is known, a correction factor may be applied to remote sensor measurements to estimate the total mass of exhaust HC emissions. Newer remote sensors are capable of measuring the NO/CO₂ ratio in vehicle exhaust.¹⁷ Such instruments may provide on-road measurements of NO emissions analogous to the currently available CO data. Once the CO inventory is known, emissions of HC, NO_x, and additional pollutants such as formaldehyde and benzene, may be estimated from emission factor ratios measured in roadway tunnels or from concentration ratios measured in ambient air.

CONCLUSIONS

A fuel-based methodology was developed and applied to calculate emissions of CO from cars and light/medium-duty trucks in the South Coast Air Basin. In the summer of 1991, stabilized exhaust CO emissions were calculated to be 4400 ± 900 tons per day for cars and 1500 ± 450 tons per day for trucks. Total CO emissions, including emissions estimates for incremental starts and heavy-duty vehicles from existing models, were estimated to be 7900 tons per day. Total car and truck emissions calculated using the fuel-based methodology were higher than predictions from the MVEI 7F model by a factor of 2.3. Lower-bound fuel-based inventory estimates indicate that MVEI 7F predictions were low by factors of at least 1.9 for cars, 2.3 for trucks, and 2.0 for both vehicle classes combined.

Since the fuel-based methodology uses emission factors measured from large numbers of on-road vehicles, high-emitting vehicles are weighted according to their presence in the on-road fleet. The use of gram-per-gallon instead of gram-per-mile emission factors simplifies the calculation procedure because statewide gasoline use is known, and because CO emission factors normalized to fuel consumption vary only slightly over most driving modes. In present form, the fuel-based approach is useful as an independent method for verifying predictions of traditional emission inventory models. With further development, the fuel-based methodology could become the standard for calculating motor vehicle emission inventories.

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ERRATA

In the April *Journal*, in Table 2 on page 338 in "Comparison of Outdoor and Classroom Ozone Exposures for School Children in Mexico" by Diane R. Gold, George Allen, Andrew Damokosh, Paulina Serrano, Carl Hayes, and Margarita Castillejos, the heading on the fourth column from the left should have read "Windows/doors closed, air cleaner on (n=47)."