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An on-road motor vehicle emissions inventory for Denver: an efficient alternative to modeling

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Abstract

Emission inventories from mobile sources have traditionally been obtained through computational modeling. This method, however, has intrinsic shortcomings in that the factors used incorporate only a limited amount of real-world observations. The agreement between model predictions and measurements has often been poor. Recently, a fuel-based method of obtaining on-road emissions inventories has been developed. This technique calculates emission factors in grams of pollutant per unit of fuel used (kg, gallons or l) from remote sensing measurements. Combining these factors with fuel use data, available from tax records, results in a fuel-based emission inventory. This method for obtaining emission inventories is very economical and an ideal alternative for locations lacking the resources to develop an emissions model. We have used this routine to calculate CO, HC and NO on-road running exhaust emissions inventories for the Denver Metropolitan area during several years when the enhanced I/M program has been in place. These calculations indicate a continually decreasing inventory over the 6 yr study period. The calculations are also compared with results from the recent MOBILE6 model. The modeled inventories are 30–70% higher, 40% lower, and 40–80% higher for CO, HC and NO, respectively.

Keywords: Fuel-based emission factor; Mobile sources; Remote sensing; Emission model comparison

1. Introduction

In order to better monitor and control air pollution, it is essential that one correctly designates the sources of pollution. An emission inventory does that by assigning a specific quantity of pollutant to a source or set of sources. As of 1998, on-road vehicles were believed to be the single largest source for the major atmospheric pollutants, contributing 60% of the carbon monoxide (CO), 44% of the hydrocarbons (HC), and 31% of the oxides of nitrogen (NO_x) to the national emission inventory (USEPA, 2000). Thus, an accurate assessment of emissions from motor vehicles is crucial to understand the air quality of a given region.

Until recently, motor vehicle emission inventories have been travel based; that is, they have been calculated from computational models that use vehicle miles traveled (VMT) estimates and mass per distance emissions factors extrapolated from limited dynamometer testing. While the models may help obtain inventories rather quickly after a large initial investment to develop the model, their accuracy remains uncertain. VMT numbers are also the result of a model applied to scattered and expensive vehicle count data. The predictions from these models do not correlate well with on-road or ambient air data (Singer and Harley, 1996). Furthermore, the development of computer models is quite expensive as it involves gathering of copious amounts of data that may be location-specific. The use of on-road remote sensing emissions data to obtain fuel-based inventories is an alternative. Such an assessment of the emission inventory would be based much more on

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data and less on model predictions and would be considerably less expensive than computer modeling.

Remote sensing involves measurement of fuel-based mass emissions from a statistically significant random sample of vehicles on the road. These qualities eliminate many of the biases seen with the travel-based approach. The vehicle sample is more representative with remote sensing because all types of vehicles that are on the roads are measured randomly. For example, with the travel-based approach, vehicle mileages are estimated from the registered fleet, and the model has been criticized because the dynamometer testing procedure leaves out many high-emitting vehicles, which would not volunteer for government emissions testing (NRC, 2000). In remote sensing data, high-emitting vehicles are weighted according to their presence on a specific road. Furthermore, remote sensing measures emissions of vehicles as they drive on the road; a range of speeds and loads is sampled and real-world emission measurements are obtained. Specifically, this study describes remote sensing measurements of vehicles under loads up to 50 kW (metric ton)$^{-1}$ in terms of vehicle specific power (Jimenez, 1999). This range encompasses the loads present in the US Federal Test Procedure, an important certification cycle for most US vehicles (Jimenez, 1999) and the cycle used to generate a significant portion of MOBILE6 model data.

Also with remote sensing, a proportionate picture of the relative activity of sub-sets of vehicles is obtained since the frequency of measurement is the frequency of travel. Finally, remote sensing is fuel based in that emissions are measured in mass of pollutant per amount of fuel. This type of measurement is less dependent on engine speed and load compared to a travel-based approach, which estimates the amount of pollutant per distance traveled and then requires large vehicle speed corrections and VMT estimates (Pierson et al., 1996; Singer and Harley, 1996).

Singer and Harley (1996) have proposed a method for obtaining fuel-based emission inventories using remote sensing data. Emission rates for individual vehicles are obtained directly from remote sensing pollutant ratios. These grams of pollutant per gallon of fuel (or g/kg) values are averaged for subgroups of vehicles to obtain emission factors for the subgroups. The factors for each subgroup are weighted by the fraction of total fuel used by that subgroup to obtain an overall fleet emission factor. This value is then multiplied by amount of fuel sold to obtain an emission inventory.

The remote sensor used in this study (FEAT) was developed at the University of Denver for measuring the pollutants in motor vehicle exhaust, and has previously been described in detail in the literature (Bishop and Stedman, 1996; Popp et al., 1999). The remote sensor directly measures ratios of CO, HC or NO to CO_2, termed Q, Q' and Q'', respectively. These ratios are constant for a given exhaust plume, and on their own are useful parameters for describing a hydrocarbon combustion system. The measured emission ratios can be directly converted into mass emissions per gallon or kilogram of fuel used. The equations are

$$\frac{\text{gCO}}{\text{kgFUEL}} = \frac{28 \times Q}{Q + 1 + (3 \times 2.2 \times Q')} \times 71.4,$$

$$\frac{\text{gHC}}{\text{kgFUEL}} = \frac{44 \times 2.2 \times Q'}{Q + 1 + (3 \times 2.2 \times Q')} \times 71.4,$$

$$\frac{\text{gNO}}{\text{kgFUEL}} = \frac{30 \times Q''}{Q + 1 + (3 \times 2.2 \times Q')} \times 71.4,$$

where the 28, 44 and 30 are g mol^{-1} for CO, HC (as propane) and NO, respectively, and 71.4 is the moles of carbon per kg of fuel assuming gasoline is stoichiometrically CH_2. It turns out that g kg^{-1} of fuel calculations are very insensitive to the small changes observed in the carbon-to-hydrogen ratio because in all cases the majority of the fuel mass is the (measured) carbon component. The factor of 2.2 is used to normalize HC measurements made by the NDIR detector for comparison to measurements by FID instruments (Singer et al., 1998).

Harley et al. have used the fuel-based method to obtain inventories from several sources including a 1991 summertime inventory of running exhaust CO emissions for the South Coast Air Basin in California (Singer and Harley, 1996), a 1997 summertime inventory of emissions in the Los Angeles area (Singer and Harley, 2000), a heavy-duty diesel truck exhaust emission inventory of fine black carbon particles and NO_x (Dreher and Harley, 1998), and an assessment of off-road diesel engine emissions (Kean et al., 2000).

In this study we have used a similar method to assess the on-road motor vehicle running exhaust emissions inventory for the Denver metropolitan area. This area consists of the six counties that participate in the enhanced Inspection and Maintenance program to reduce automobile emissions. These counties are Adams, Arapahoe, Boulder, Denver, Douglas and Jefferson. The Denver area was estimated by the Colorado Department of Public Health and Environment to emit 1308 tons of CO per day in 1995 using the US EPA's Mobile 5a model. To meet the standard, CO emissions would have to be reduced to an average of 875 tons day^{-1} by 2001. Carbon monoxide, hydrocarbons and nitric oxide are the pollutants being measured and quantified. Inventories for several years were calculated in order to assess progress in emission control.

Furthermore, early in 2002 EPA made public its latest emissions factor model called MOBILE6. Details of the model can be found at www.epa.gov/otaq/m6.htm. It is the sixth generation of such a model issued by the EPA for use by state and local agencies to project automobile emissions. The current generation of the model took several years to construct and presumably a considerable amount of financial resources. MOBILE6 was used here to generate travel-based model emission inventories for the Denver area. These modeled inventories are compared to the fuel-based inventories obtained from remote sensing data.

2. Method

In a manner similar to Singer and Harley (1996), the fuel economy and measurement frequency of different model year car and truck subgroups are used to calculate relative fuel use by each of these subgroups. One can then combine the fuel use with emission factors for each of the subgroups to obtain an overall fleet emission factor. Mathematically, the process is as follows:

$$t_{yv} = \frac{n_{yv}}{N},$$

where y is the model year subgroup, v the vehicle type subgroup, t the fraction of travel of subgroup, n the number of measurements of subgroup, N the total number of measurements.

In other words, given subgroups of model year y and vehicle types v, the fraction of travel of each subgroup (t_{yv}) is the number of measurements of that subgroup (n_{yv}) divided by the total number of measurements (N) during a remote sensing event.

The relative fuel use of each subgroup (f_{yv}) is then given by

$$f_{yv} = \frac{(t_{yv}/F_{yv})}{\sum_{v=V_1}^{V_n} \sum_{y=Y_1}^{Y_n} (t_{yv}/F_{yv})},$$

where F_{yv} is the fuel economy of model year subgroup y and vehicle type v, $Y_1, \ldots, Y_n$ the various model years measured, $V_1, \ldots, V_n$ the vehicle types measured.

Finally, the overall emission factor is given by the product of relative fuel use and measured emission factor for each subgroup summed over all of the subgroups

$$E = \sum_{v=V_1}^{V_n} \sum_{y=Y_1}^{Y_n} f_{yv} E_{yv}.$$

This emission factor is then multiplied by total fuel use to obtain the emission inventory. It should be noted that the measured HC values were offset adjusted in the manner described in Pokharel et al. (2002).

For the modeled inventories, MOBILE6 inputs specific to the Denver area were included where available. Model year registration fractions of the light-duty fleet were derived from remote sensing data. Fig. 1 illustrates travel fraction as a function of age for

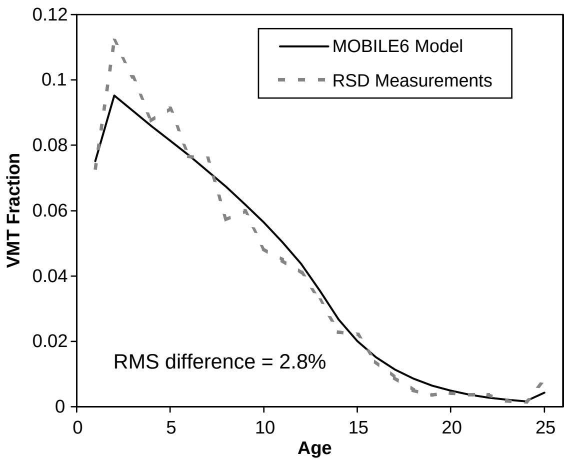

Fig. 1. Relative travel fractions of passenger cars as a function of age. Solid line is the MOBILE6 national average default values. The broken line is the measurement frequency in remote sensing.

passenger cars calculated from frequency of remote sensing measurements. Also shown are the MOBILE6 national defaults. Note that the two profiles are rather similar with an RMS difference of only 2.8%. Travel fraction with age profiles for two classes of light-duty trucks also agreed with national averages, as did the relative travel fractions of the various light-duty vehicle classes. Other model variables such as I/M inputs and fuel properties were obtained from relevant state agencies. The emission factors for running exhaust emissions calculated using MOBILE6, given in grams of pollutant per mile of travel, were combined with vehicle activity data from the Colorado Department of Public Health and Environment—Air Pollution Control Division to obtain travel-based modeled inventories of on-road running exhaust motor vehicle emissions.

3. Results

For the fuel-based inventories we have used the fuel economies given in Singer and Harley (2000) for 1974–1997 cars and light-duty trucks. Pre-1974 vehicles were all assigned the fuel economy of 5.0 km/l and model year 1998 and newer cars and trucks were assigned the economy of 1997 vehicles. The vehicles were divided into these two categories as designated by the Colorado DMV records (PAS and LTK as cars and trucks, respectively). Though there is uncertainty in these fuel economy estimates, a calculation assuming a constant fuel economy for all vehicles resulted in emissions inventory differences of only 3–7% depending on pollutant. Thus, uncertainties in the fuel economy estimates are not a major source of error.

The first set of data analyzed was the measurements made in Denver during the winter of 1999–2000 under Coordinating Research Council (CRC) contract E-23. The structure, contents and preliminary analysis of the data, along with other FEAT remote sensing databases and reports, can be found at www.feat.biochem.du.edu.

When divided by model year and car/truck designation, each subgroup contained anywhere between 10 (1974 model year trucks) and 2000 (1999 model year cars) measurements. From this data set, CO, HC and NO emission factors and the travel frequency were calculated for each model year and vehicle type subgroup. The calculated travel frequencies and fuel economy data were combined using the equation given above to obtain the fraction of fuel used by each subgroup. As indicated, each fraction was multiplied by an emission factor for the subgroup and summed across the subgroups to give fleet gasoline emission factors of 65 g CO, 5.79 g HC and 6.52 g NO/kg fuel.

To determine fuel use in the Denver Metro area, the state fuel sales tax data were used. Such data can be obtained from the Colorado Department of Revenue—Office of Tax Analysis on a monthly basis. For the 1999–2000 fiscal year approximately 5.6 million gallons of gasoline were sold each day in Colorado. To assess the percentage of the state fuel sales used in the project area, population and vehicle registration data were consulted. These statistics were obtained from the Colorado Department of Local Affairs and the Colorado Department of Motor Vehicles, respectively. The two methods allocated an average of 53% of the state gasoline use in the 6 county Denver Metro area; 56% of the population lived in the Denver Metro area in July of 1999, while 50% of the vehicles were registered in the area in 1999. Multiplying this fuel use by the calculated emission factors gave the emission inventory from gasoline vehicles in the Denver metropolitan area: 528 tons day^{-1} CO, 48 tons day^{-1} HC, 58 tons day^{-1} NO.

3.1. Diesel emissions

A non-negligible fraction of total on-road fuel use, and of emissions, comes from diesel vehicles. Heavy-duty diesel trucks account for almost all of the diesel use. Bishop et al. (2001) have measured emission factors for heavy-duty diesel trucks in Colorado. The truck emission factors were 32 g CO, 14 g HC and 24 g NO per kg of fuel. Again in the case of HC, the remote sensor measurements were adjusted in order to obtain total VOC concentration. For diesel fuel the adjustment factor is 2 (Singer et al., 1998).

In the case of heavy-duty trucks, population and vehicle registration data were supplemented with the Colorado Department of Transportation's daily vehicle miles of travel to apportion the total state diesel sales to the six county Denver metro area. Such an analysis allocates 36% of the heavy-duty truck travel, and thus diesel fuel use, to the Denver area. Combining this fuel use with the emission factors above yields the following emission inventory from diesel trucks: 48 tons day^{-1} CO, 21 tons day^{-1} HC and 36 tons day^{-1} of NO.

3.2. Measurement of uncertainty

The availability of an uncertainty estimate is an important advantage of the fuel-based approach to emission inventories. The first source of error arises in the emission factors. At least two mechanisms can contribute to the uncertainty in these factors: noise in the actual emission measurements and emission variability.

We have shown elsewhere (Stedman et al., 1997) that for large fleets of vehicles average RSD emissions can be obtained with a precision of $\pm 2\%$. The absolute accuracy of the measurements depends upon the calibration cylinder, claimed by the manufacturer to be $\pm 2\%$. Possible sources of bias include unreadable plates and unmatchable plates. In the study discussed here, the number of vehicle hits (including heavy-duty diesel trucks that produce three hits each) was 30,139 with 29,000 valid measurements. The number of valid hits with readable license plates was 24,378, and the number with matched plates was 22,986. Though the misses may be a source of bias, the bias is not thought to be large due to the significant plate match rate and the good agreement shown in Fig. 1 between measured and expected fleet characteristics.

Another possible source of bias results from the relationship between emissions and vehicle load. Fuel based values are less sensitive to load than mileage based ones. CO is almost independent of load, HC is more dependent, and NO is most dependent. This study is biased towards higher load sites because larger exhaust plumes are easier to measure and because most fuel is consumed at higher load and, thus, is most relevant to basin-wide emissions. This effect, if load bias were introduced in this study, would not significantly affect CO, would slightly lower HC and raise NO in our estimates.

The rate of emissions of a vehicle depends on the state of the catalyst and other parts of the emission control technology, on the quality of the fuel, the quality of the air and the intake system, and a host of other variables. Measurement at one location limits the driving modes being sampled. In order to incorporate the whole profile of load on the vehicle several sites with varying speed, acceleration, grade, etc. must be sampled. Although we have used the measurements from one site to obtain the average inventory value, we have incorporated seven other sites in the Denver area to obtain an estimate of the uncertainty in the emission factors.

These seven other sites were part of a Denver area emission study during the summer of 2000. This study used the Smart Sign to furnish drivers with information on the state of their vehicle's emissions control system as they drove by (Bishop et al., 2000). The sites were all within the Denver metropolitan area being studied here.

Besides encompassing a wide range of driving modes, these locations incorporate ranges in other variables that affect vehicle emissions. The number of measurements also aid in surveying vehicles from a wide range of socioeconomic backgrounds. Table 1 gives a listing of the measurement sites, their average emissions and their characteristics.

From these weeklong campaigns at the seven sites it is evident that emission rates vary. A statistical analysis of this data, along with the central set of Denver measurements divided into daily averages to asses measurement variability, indicated the following 95% confidence intervals on the mean: 21% for CO and 29% for HC. The seven other campaigns did not include NO measurements. The confidence interval for NO, then, was generated by sampling average values from various remote sensing campaigns conducted under sponsorship of the CRC. This analysis yielded a 95% confidence interval for NO of 16% of the mean.

Another source of uncertainty is in the percentage of state fuel sales used in the study area. Using population and vehicle registration data in the six county area, an uncertainty of $\pm 3\%$ was calculated for gasoline fuel use. Additional uncertainty is present in the amount of diesel fuel consumed in the study area and in the relative fuel economies of the vehicles. Propagation of the errors led to overall uncertainties in tons of pollutant per day of 15% for CO, 18% for HC and 11% for NO. Note that it is assumed that fuel transferred into the study area is equal to that transferred out. Furthermore, fuel used for non-road applications such as farm and construction equipment are taxed separately in Colorado and so are not included in the fuel sale estimates.

3.3. Trend over several years

Measurements have been conducted at the 6th and I-25 site in Denver during several previous years and in 2001. These data were used to obtain gasoline emission factors for those years of measurement. Heavy-duty diesel emissions were only measured in 1999 in the area so those diesel emission factors are used throughout and scaled with yearly fuel sales. The resulting emission inventories are illustrated in Fig. 2. The fuel use and emission factor values for the five separate years are summarized in Table 2.

Table 1
Measurement sites for the Smart Sign study and their characteristics

Site	Start date	Count (All Valid)	Average CO/CO_2 (10^{-2})	Average HC/CO_2 (10^{-4})	Characteristics
6th Ave. and Kipling St., JEFFERSON	31-May-00	31,566	3.7	8.6	Uphill curved on-ramp from major arterial to freeway, moderate socioeconomics.
HW36 and Federal Blvd., ADAMS	27-Jun-00	14,010	4.3	12.9	Off highway loop, incline, moderate socioeconomics.
6th Ave. and I-225, ARAPAHOE	10-Jul-00	10,736	4.4	7.3	Long highway on-ramp, slight incline, accelerating, moderate–low socioeconomics.
Lincoln Ave. and I-25, DOUGLAS	17-Jul-00	18,508	1.8	4.3	Loop on-to highway, flat, upscale socio-economics, no residences within a mile.
Northglenn Town Hall, ADAMS	31-Jul-00	15,403	2.9	7.3	Residential–official two-way road, flat, some slight acceleration.
112th Ave. and Colorado Blvd., ADAMS	3-Aug-00	14,140	4.1	9.6	Residential, two-way road, slight incline, moderate socioeconomics.
Table Mesa Dr. to Foothills Pkwy., BOULDER	14-Aug-00	55,611	3.3	5.7	One way loop between two transit roads, incline, somewhat upscale socioeconomics.

3.4. MOBILE6 model

The release of the MOBILE6 model allowed us to compare our fuel-based inventory results with the latest model predictions. CO, HC and NO_x emission factors were modeled for several years in the Denver Metropolitan area. The factors are for the running exhaust and do not include starting and evaporative emissions. Combining these factors with daily vehicle mileage in the area from the Colorado Department of Public Health and Environment—Air Pollution Control Division, travel-based model inventories were obtained. These inventories are included in Fig. 2.

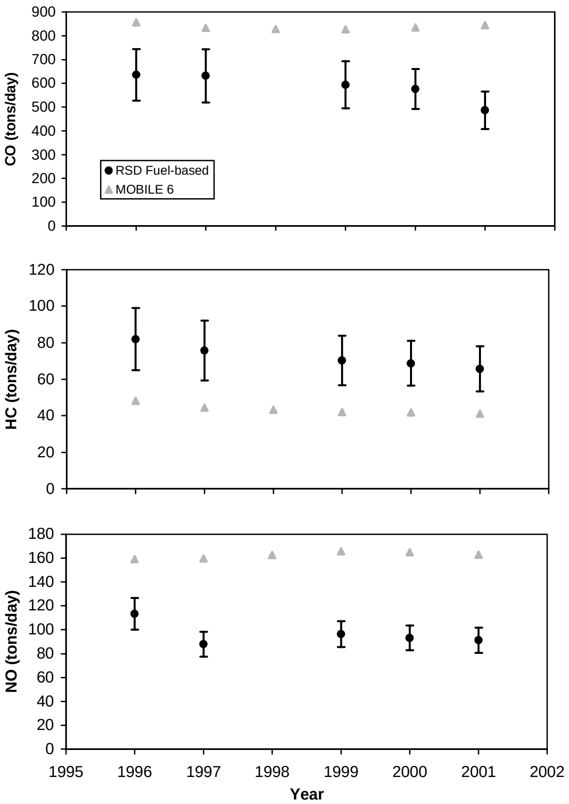

Fig. 2. Plots of emissions inventories for three pollutants during 5 years of study in the Denver area.

One caveat in comparing fuel-based inventories from remote sensing data and MOBILE6 inventories is that one has to convert NO values from the two methods into common units. MOBILE6 reports emissions factors in NO_2 g/mi equivalents (even though what is measured in automobile exhaust is mostly NO). RSD measurements are in grams of NO. Thus, the modeled NO_x inventories need to be adjusted by the molecular weight ratio of NO to NO_2 (30/46) to convert to NO equivalents.

4. Discussion

The trends seen in Fig. 2 with the fuel-based approach indicate decreasing emissions of all three pollutants from on-road vehicles during the years from 1996 to 2001. The general improvement in mobile source emissions over several years has been measured (Singer and Harley, 2000), and is not as clear in our MOBILE6 estimates. We believe that the improvement is the result of vehicle fleet turnover and emission control strategies. Even though on-road fuel use, and thus travel, is increasing (19% between 1996 and 2000), emission factors have been decreasing on a fuel basis. The emission factors have decreased enough so that the increased fuel use is more than offset. Even in the MOBILE model, the emission factors decrease with

Table 2
Fuel use and gasoline emission factor values used during the various years of study

Year	State gasoline use ($\times 10^4$) (gal/day)	State diesel use ($\times 10^4$) (gal/day)	CO emission factor (g/kg of fuel)	HC emission factor (g/kg of fuel)	NO emission factor (g/kg of fuel)
2001	567	127	53 ± 11	5.3 ± 1.5	6.3 ± 1.0
2000	565	125	65 ± 14	5.8 ± 1.7	6.5 ± 1.0
1999	542	108	69 ± 14	6.5 ± 1.9	7.9 ± 1.3
1997	510	97	80 ± 17	8.2 ± 2.4	8.3 ± 1.3
1996	488	91	84 ± 18	9.2 ± 2.7	11.6 ± 1.9

successive calendar year. It is the increasing total vehicle mileage that overwhelms the trend in the overall modeled emissions inventory.

Besides the trends seen with the two approaches, the absolute values differ somewhat. The modeled values are 30–70% higher, 40% lower and 40–80% higher than the calculated CO, HC and NO values, respectively. One cause for this discrepancy may be the elimination of all starting and evaporative emissions from the MOBILE factors. Only running exhaust emissions were modeled to correspond more closely to what is measured by remote sensing. According to the model, start emissions are 32% and evaporative emissions are 44% of total HC. Furthermore, start emissions contribute 50% and 27% to total CO and NO_x, respectively. Since the fractions attributed to non-running exhaust emissions are so large in the model, any measurement of starting and evaporative emissions by remote sensing would cause the fuel-based inventories to be higher than a model of exclusively hot running emissions.

Though the results of the two approaches disagree somewhat, both are able to produce estimates of regional on-road emissions inventories. If one is to use the modeling technique, emission factor and travel demand models must first be constructed. Collection of relevant data and compilation of the software for the models involve large investments. After models pertinent to the region in question have been constructed, further data specific to the region must be gathered, requiring additional investment.

With the fuel-based approach, a region that does not possess the resources required to construct computational models can obtain emission inventories. With two weeks of fieldwork, enough data can be gathered to calculate fuel-based emission factors and uncertainties. Fuel use data are easily available from tax records. With minimal further data analysis, emission inventories are calculated. These inventories have the added advantage of containing estimated uncertainties so that it is known how well the inventories reflect real-world values.

One consideration with the remote-sensing approach is the possibility of systematic misses of vehicles of a certain type. As was discussed in regards to Fig. 1 above, relative measurement frequency of the light-duty fleet reflected the national average for relative travel frequency included in MOBILE6. Thus, it can be assumed systematic errors do not occur in the measurement of the light-duty fleet. A portion of the heavy-duty fleet, on the other hand, may be missed if the equipment is optimized to capture exhaust plumes of the light-duty vehicles. In this study, heavy-duty emission factors were obtained from a study specifically of these vehicles. Thus, the heavy-duty fleet has also undergone a representative sampling. One class of vehicles that is not sampled significantly in this remote sensing study is motorcycles. Motorcycles, however, contribute only 0.6% to the total miles traveled and even less to the fuel consumed. Thus, the difference caused by unrepresentative sampling of motorcycles is not significant.

It must be noted that this fuel-based assessment has treated each day during the year as being equivalent since the reported model estimates also do so. However, emission during weekend days would be expected to be lower than during weekdays. Emission factors have been measured during weekend days, and these are well within the uncertainty in the factors. Fuel use, on the other hand, should be less during the weekend. Thus, the reported daily inventories are a weighted average of weekdays and weekend days.

Another result of treating all days equivalently is that seasonal variations in vehicle emissions and fuel composition are not considered explicitly. Atmospheric conditions, such as temperature and humidity, have known effects on emission rates (Cadle et al., 2001). Several groups have demonstrated seasonal effects on CO emissions (USEPA, 1998) caused by temperature, fuel Reid vapor pressure and fuel oxygenation, specifically in the Denver area (Bishop and Stedman, 1990). Again, however, emission measurements have been made during both winter and summer months to assess the uncertainty in the emission factors. Thus, seasonal variations are within the reported uncertainty.

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