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Fuel-based motor vehicle emission inventory for the metropolitan area of Mexico city

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Abstract

The level and nature of air pollution varies substantially from city to city. Hence, the first requirement is the creation of an adequate knowledge base on local air quality on which to develop an air quality policy. Because the availability of data used in traditional on-road mobile source estimation methodologies is limited in Mexico, an alternative methodology was implemented to estimate motor vehicle emissions. In the year 2000, on-road gasoline powered vehicle emissions in the Metropolitan Area of Mexico City (MAMC), were characterized using fuel sales as a measure of vehicle activity, and exhaust emissions factors from remote sensing measurements. In a similar way, remote sensing data obtained by researchers of the University of Denver back in 1991 and 1994 were used to estimate a fuel-based emission inventory for those years. Average emissions factors were estimated in 113.5 ± 13, 13.1 ± 1.9 and $9.84 \pm 2.3\ g\,l^{-1}$ for CO, hydrocarbons (HC) and nitrogen oxides (NO_x), respectively, based on remote sensing measurements of 42,800 vehicles. For year 2000 light and medium gasoline vehicles exhaust emissions contributed with 2065, 238, and 179 metric ton day-1 of CO, HC and NO_x, respectively. The inventory is 48% and 26% lower in CO and NO_x, respectively, than official inventory estimates for the year 1998 using travel-based models. Calculated CO reduction from 1991 to 1994 is approximately 46% while the atmospheric CO measurements, as indicator of mobile activity, in the same period decrease 51%. For the period 1994–2000 the reductions were 36% and 31%, respectively. The calculations indicate a continually decreasing inventory over the study period, and represents and ideal alternative for locations such as Mexican cities lacking the resources to develop an emissions model.

Keywords: Remote sensing; Fuel-based inventory; Mexico city

1. Introduction

As with many countries in the world, motor vehicle emissions are a significant element of the overall emissions inventory in Mexico. An example of the significance of motor vehicles emissions inventories can be seen in the 1998 emissions inventory developed for the Metropolitan Area of Mexico City (MAMC).

According to this inventory, motor vehicles contributed 40% of the total hydrocarbon (THC) emissions, 98% of the total carbon monoxide (CO) emissions, and 81% of the total nitrogen oxides (NO_x) emissions

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(Metropolitan Environmental Comission, 1998; Molina and Molina, 2002).

Those figures may be compared with those of the Environmental Protection Agency of the United States (EPA), where nationwide mobile sources are estimated to contribute more than half of the nitrogen oxides (NO_x) inventory; 42% of the volatile organic compounds (VOC) inventory; one-quarter of the particulate matter-10 (PM-10) inventory; and 80% of the carbon monoxide (CO) emissions (USEPA, 1993).

An accurate assessment of emissions from motor vehicles is crucial to understand the air quality of a given region. Traditional models used to estimate vehicle emissions combine travel-normalized emission factors (EF) with travel demand models (TDMs) to calculate total exhaust emissions.

Compared to the United States, the availability of motor vehicle activity data in Mexico is somewhat limited. In major US metropolitan areas, TDMs are typically used to estimate vehicle kilometers traveled (VKT).

In Mexico, however, VKT based upon some limited traffic count statistics, informal surveys, and anecdotal information, therefore TDMs are not widely used; and the development of TDMs for the entire country is not technically or economically feasible at this time.

Fuel sales data are sometimes used to check the reasonableness of motor vehicle emission estimates. These can also be used to estimate VKT in situations where other VKT estimates are not available, if assumptions regarding fuel efficiencies for various vehicle classifications are made.

EFs are measured primarily in the laboratory from recruited vehicles. Individual vehicles are tested under controlled environmental conditions. Exhaust emissions are normalized to travel and measured while each vehicle is driven through a standard dynamometer cycle intended to simulate on-road driving.

The current generation of EF models, including the EMFAC series of models used in California (CARB, 1997) the MOBILE series of models developed by EPA (USEPA, 1993), are based upon emissions data for selected driving cycles.

The MOBILE 5 model has frequently been used to estimate vehicle EFs for uncontrolled vehicles in such countries as Chile, Indonesia, and Mexico (Gwilliam et al., 2003).

The MOBILE 5 version applied in Mexico is a computer program that calculates the emissions of CO, HC and NO_x based on data obtained from American vehicles tested under the Federal Test Procedure, FTP-75 (Burnette et al., 2001; Office of Federal Register, 1993). Later the total emissions are obtained by multiplying the estimated emissions per kilometer for an average urban trip, times the estimated number of trip miles traveled in the area.

For the MAMC three emissions inventories have been issued in 1994, 1996 and 1998. Large discrepancy exists among the different inventories, i.e. in 1996 the NO_x contribution of transport was calculated as 85,000 metric ton yr^{-1}, while in the 1998 inventory it was reported that 165,800 metric tons yr^{-1} were emitted (Metropolitan Environmental Commission, 1998).

Regarding the Mexican inventory, Nobel prized Mario Molina have stated that the lack of consistence among the values is a consequence, among others, of the lack of reliable information on EFs and daily activity of the fleet (Molina and Molina, 2002).

On-road vehicle emissions can be measured directly by sampling the ventilation air in roadway tunnels (Pierson et al., 1996), and through the use of remote emissions sensors (Pierson et al., 1996; Bishop et al., 1989; Stephens, 1994).

On-road techniques allow for large numbers of vehicles to be sampled, including the entire population of vehicles at each sampling size. Emissions are normalized to fuel consumption and fleet average emissions factors are derived directly from on-road measurements (Singer, 1998).

Furthermore, remote sensing (RS) 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. Also with RS, a proportionate picture of the relative activity of sub-sets of vehicles is obtained since the frequency of measurements is the frequency of travel.

In the fuel-based approach, emission rates for individual vehicles are obtained directly from RS pollutant ratios, and later normalized to fuel consumption and expressed as gram of pollutant emitted per liter of gasoline burned instead of grams of pollutant per kilometer.

The factors of each subgroup are weighted by the fraction of total fuel used by that subgroup to obtain the overall fleet EF. Then, the fleet-average EF is multiplied by local fuel sales to compute pollutant emissions. The accuracy of a fuel-based model depends on how well the vehicles and driving modes, as well as age distribution, from which EF were measured; represent the entire area under study (Frey et al., 2002a, b).

One have to acknowledge that remote sensing is not capable of practical measurements at many locations of interest, because of difficulty in making measurements across multiple lanes of traffic and/or with very slow moving vehicles (when it is difficult to reliably measure background levels). Furthermore, remote sensing provides only a snapshot which may not be representative of emissions during a trip. The activity pattern at many remote sensing sites is not comparable to the activity pattern of a typical trip (e.g., based upon comparisons of the joint distribution of speed and acceleration (Frey et al., 2002b).

In this study, the evolution of the emission inventory base on remote sensing measurements is described. For that purpose data on emissions achieve back in year 1991 and 1994 by researchers of the University of Denver (Bishop et al., 1997; Beaton et al., 1992) were combined with those obtained by our group in year 2000 (Schifter et al., 2003) to obtain a revised motor vehicle emission inventory for the MAMC.

2. Method

2.1. MAMC vehicle distribution in year 2000

Several data sources were used to develop default assumptions about the fleet age distribution in Mexico in the year 2000. First the official inventory of the Metropolitan Environmental Commission (1998) provides the number of registered vehicles in the MAMC that was compared with the historical data of sales provided by the Mexican Manufacturers Association of Vehicles.

Data were extrapolated to year 2000 according to annual sales and scrappage rate. Scrappage is defined as the number of vehicles scrapped or otherwise removed from circulation in any given year, this value is reported as percentage of the number of vehicles in use.

To determine the effects of annual vehicle scrappage, we consulted rates reported recently by the Inspection/Maintenance Program of vehicles that failed to obtain the permission to circulate. We estimated a scrappage rate of 2.7% annually. The general vehicle count formula is

$$VC_x = (VC_{x-1} + Sales_x)(1 - SR_x), \quad (1)$$

where VC is the vehicle count, SR the % of in-use fleet that is scrapped, and x is the current calendar year.

Table 1 shows the vehicle distribution by model year and activity in year 2000. From a total fleet of 3,200,000 vehicles, 2,820,000 are light duty gasoline vehicles, 150,000 medium duty gasoline, and 174,000 medium and heavy-duty diesel trucks.

Of the total vehicles registered in year 2000, about 42% of them are more than 10 years old, and 68% of the vehicle fleet was manufactured before 1991 and thus it is not equipped with catalytic converters. It is worth to mention that the amount of taxis is among the highest is the world, and represent the 22% of the total kilometers traveled per year.

2.2. Fuel consumption

In order to calculate fuel consumption in the MAMC, we performed a detailed study of the gasoline distribution cycle in the area (Schifter et al., 2002). Gasoline in the entire country is produced by the state-owned oil company and marketed by franchised gasoline stations selling two type of gasoline: Regular and Premium gasoline. Regular gasoline represents 90% of the total sales.

The gasoline distribution comprises the storage and transfer of gasoline from refineries by pipeline to the

Table 1
Vehicle distribution by activity and technology in the year 2000

Type of vehicle	Model year	Technology	Number	Traveled km yr^{-1} (millions)
Private LDGV	2000	[a]MPFI, [a]TWC	222,342	3326.5
	1993–99	MPFI, [a]TWC	998,546	14,592.2
	1991–92	Carburetor, oxidative catalyst	319,283	4607.0
	<1990	Carburetor, no emission control	1,016,205	13,279.5
Commercial LDGV	2000	[a]MPFI, [a]TWC	2745	223.4
	1993–99	[a]MPFI, [a]TWC	88,055	7165.9
	1991–92	Carburetor, oxidative catalyst	12,335	10,003.8
	<1990	Carburetor, no emission control	6519	530.5
Pick up	2000	[a]MPFI, [a]TWC	12,755	184.4
	1993–99	[a]MPFI, [a]TWC	98,638	1426.4
	1991–92	Carburetor, oxidative catalyst	18,603	253.3
	<1990	Carburetor, no emission control	226,551	3084.6
Class 3	2000	[a]MPFI, no emission control	7586	131.8
	1993–99	[a]MPFI, no emission control	65,848	1143.9
	1991–92	Carburetor, no emission control	18,258	317.2
	<1990	Carburetor, no emission control	72,373	1257.2
Totals			3,186,642	61,527.6

[a]Multipoint injection. TWC, three way catalyst.

state jurisdiction bulk terminal with a storage capacity of 1.51 million barrels, and three marketing terminals with a total storage capacity of 415 million barrels. Monthly fuel sales statistics were obtained from the three terminals as well as the gasoline transferred from the refineries to the bulk plant.

According to the results, 18,342 million liters by day were sale in the MAMC during year 2000. Density of this fuel is equal to $732\,g\,l^{-1}$, and a carbon mass fraction was 0.866.

2.3. Remote sensing data

On-road remote sensing was developed in Denver, the methods and policy implications have been described elsewhere (Zhang et al., 1996). Measurements were carried out between June and September, 2000 with an AccuScan® system from Environmental Systems Products, CT, USA.

The instrument is made up of a non-dispersive infrared component for CO_2, CO and HC detection, and a dispersive ultraviolet spectrometer for NO measurement. The mobile unit included the equipment required to measure speed and acceleration, as well as license plate recognition.

The RSD is designed to generate and monitor a non-dispersive infrared and ultraviolet beam emitted and reflected approximately 10–18 inches above ground, preferably across a single lane road. Gasoline, diesel, or other fossil fuel powered vehicles drive through this beam, and the exhaust interferes with this transmission of the beam. Quantifying the interference enables the calculation of tailpipe concentrations of CO, HC, CO_2, and NO.

A camera simultaneously captures a digitized video image of the rear of the vehicle and its number-plate. HC measurements were expressed in their "*n*-hexane ppm equivalents", except where indicated (e.g. *n*-propane equivalents). The CO tolerance was 10% or 0.25% (whichever is greater) for all expected concentrations below 3.0%, and 15% for all CO expected concentrations above 3.0%.

In the case of HC, the tolerance was 150 ppm or 15% of the expected HC concentration (whichever was greater) throughout the range of HC concentrations.

The NO tolerance was 250 ppm or 15% of the expected NO concentration (whichever is greater) throughout the range of NO concentrations.

The unit was equipped with a speed and acceleration measurement system that uses low energy lasers to calculate the speed to within $\pm 0.8\,km\,h^{-1}$ and acceleration to within $\pm 0.5\,km\,h^{-1}\,s^{-2}$ at the moment exhaust plume is measured. The percent emissions can be directly converted into mass emissions per liter of consumed fuel.

2.4. Fuel economy

Fuel economies data for each subgroup of vehicles are not available in Mexico; therefore they were estimated from selected database available in our laboratories. The analysis includes vehicles with engine displacement ranging from. 1.7 to 5.8 liters, tested over the United States Federal Test Procedure (FTP-75) (Office of the Federal Register, 1993). HC, CO, NO_x, and CO_2 emissions were measured. The engine displacement group consisted of three sub-groupings (4 cylinders, 6 cylinders, and 8 cylinders). Fuel economies were derived using a carbon balance methodology and plotted versus engine displacement, as shown in Fig. 1, to have a range of data that wrap the engine displacements of the MAMC fleet.

2.5. Fuel-based emissions factors

By carbon balance, it is possible to relate the amount of pollutant measured by the RS to the amount of fuel burned if the molar exhaust concentrations of CO_2, and the three pollutants are measured (Singer and Harley, 2000).

An EF for pollutant E can be computed as follows:

$$E_p = \frac{(P)}{(\mathrm{CO}) + (\mathrm{CO_2}) + (\mathrm{HC})} \times \frac{w_c + \mathrm{MW}_p \times \rho_f}{12}, \qquad (2)$$

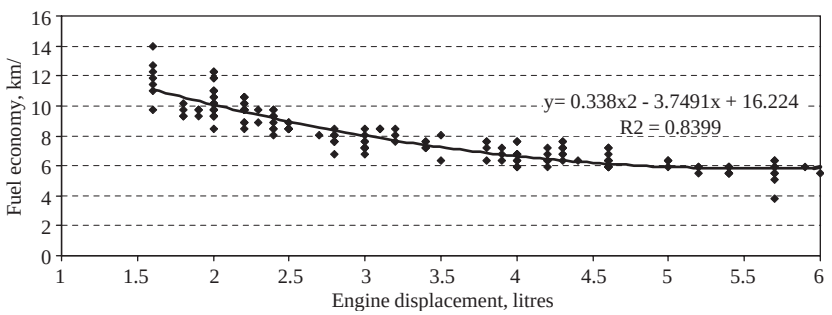

Fig. 1. Fuel economy (from FTP-75) versus engine displacement.

where E_p is in grams of pollutant P emitted per unit volume of fuel consumed, (P) is the exhaust concentration of pollutant P_f, w_c is the carbon weight fraction of the fuel, ρ_f is the fuel density, and MW_p is the molecular weight of P. The denominator of Eq. (2) represents a sum of carbon atoms in the exhaust; the factor of 12 is the atomic mass of carbon.

Fleet-average EFs were calculated using the fuel fractions and EF for each vehicle MY, and total stabilized exhaust emissions of each pollutant were calculated as the product of the fleet-average EFs and the total consumed gasoline by cars and light medium vehicles in the MAMC.

2.6. Sites of measurements

Twelve sites were chosen for the RS study within the MAMC, representative of all the on-road vehicle traffic. The routes were selected base on previous studies on traffic patterns performed by the TUV, Rheinland (Brosthaus, 1995). Data acquisition was done over diverse routes representing travel under congested conditions, on main freeways, arterial and collectors.

Measurements were distributed to represent different socioeconomic wards of the MAMC according to data provided by the environmental authorities and characteristic of main access routes to the city. These studies were complemented by us with 1500 people recalling their driving activities such as: trip length, time between trips, driving distance, number of trips per day, average daily driving time, among others.

All the sites were pre-qualified for single lane operation with space for the RSD equipment to be deployed without disrupting traffic flow (Schifter et al., 2003). The instrument measured exhaust emissions from vehicles typically traveling between 20 and 40 km h^{-1}. The fraction of total consumed fuel used by vehicles of each MY, was calculated according to (Singer and Harley, 2000), with the travel fraction of vehicles of each MY obtained from the RS measurements, and the average fuel economy of each MY vehicle.

3. Results and discussion

In total, 122,800 measurements were collected during the sampling periods at the 12 locations, of which 84,650

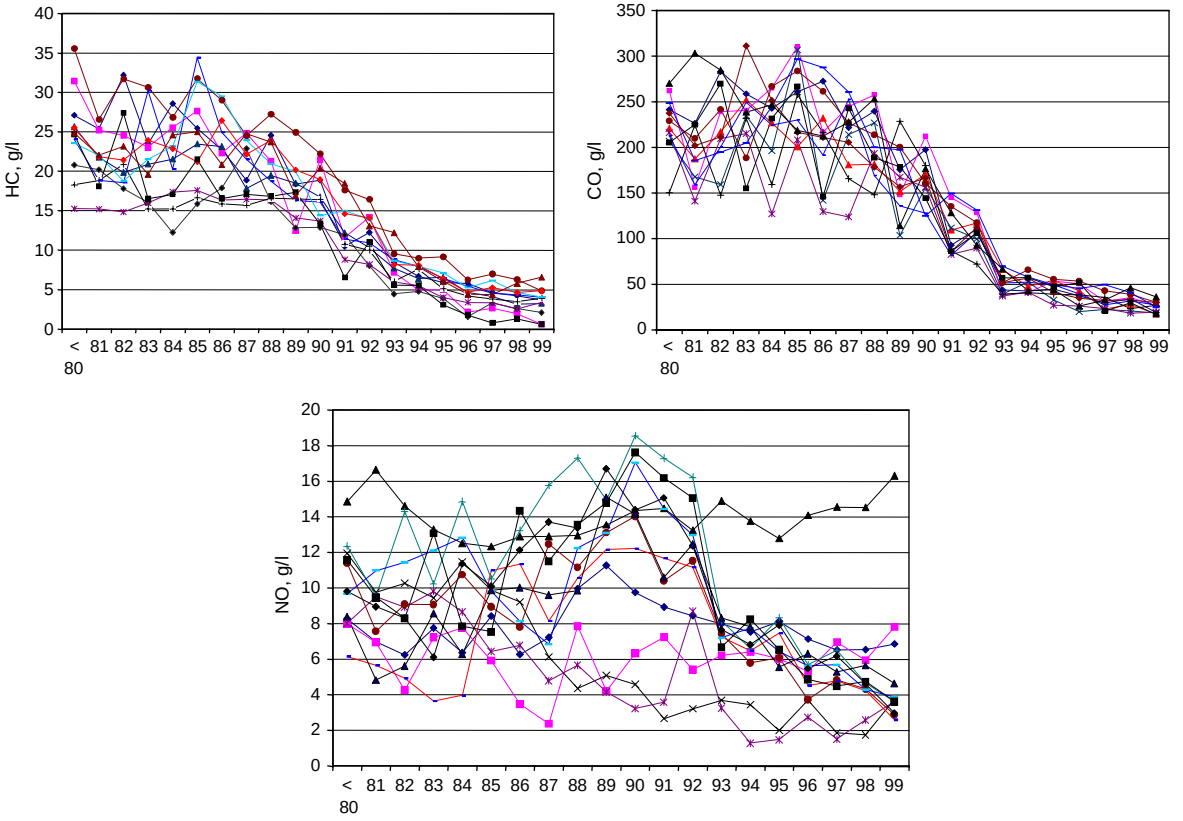

Fig. 2. Mean vehicle emissions per site illustrated as a function of model year.

(69%) records were valid for CO and CO_2 emissions measurements but only 72,269 had readable license plates. The valid attempts were reduced to 69%, 63% and 63% of the total attempted measurements of CO, HC and NO, respectively. The total number of matched plates was 42,822 records.

Fig. 2 shows mean vehicle emissions per site illustrated as a function of model year. Data shows that vehicle age distributions in Mexico are radically different than those in the United States.

On notice that early-model-year vehicles having carburetors and no catalyst (model year <1990) and those model year 91–92 with only oxidative catalysts, exhaust emissions exhibited much scatter, while exhaust emissions from fuel-injected, three way catalyst-equipped vehicles (1993 and newer vehicles) showed less scatter.

The scatter of data in the older vehicles could be related also with the small number of vehicles (8273 records) detected with the RS. Additionally, it is known that fuel based values are less sensitive to load than kilometer base ones. CO is almost independent of load, HC is more dependent, and NO is most dependent (Pokharel et al., 2002a).

Furthermore, our group has reported in the past that new vehicles sold in Mexico show high NO_x degradation rates in emissions that could be ascribed, among others to non-optimal catalyst formulation or high sulfur sensitivity (Schifter et al., 2004; Díaz et al., 2000).

Measured average emissions based on the twelve sites with 95% confidence interval were $113.5 \pm 13\,g\,l^{-1}$ CO, $13.1 \pm 1.9\,g\,l^{-1}$ HC, and $9.84 \pm 2.3\,g\,l^{-1}$ NO.

Speed and acceleration distributions of vehicles recorded with the RS equipment are shown in Fig. 3. These results indicate that vehicles were driven in average at an acceleration of $2\,km\,h^{-1}\,s^{-2}$. About 85% of the vehicles were driven in an acceleration mode, about 7% in a deceleration mode, and less than 7% were in cruising mode.

Average speed of the fleet was computed as $30\,km\,h^{-1}$. The resulting data shows that driving in the MAMC is relatively slow when compared to the cities in USA and Europe.

According to the license plate identification vehicles were grouped as follows: 34,737 were light duty passengers vehicles, 6032 light duty and utility vehicles (Pick-up and Vans), and 567 median duty gasoline vehicles.

Average EF, in grams of pollutant per liter, by model-year of light duty vehicles obtained in the RS study and 95% confidence intervals for each data point are shown in Fig. 4, and those of light and medium duty vehicles grouped according to the technological characteristics (pick-ups, vans and SUVs) are shown in Fig. 5.

In Fig. 6 a plot of the average fleet emissions of CO (as % volume) and HC (ppm, as propane) is display for

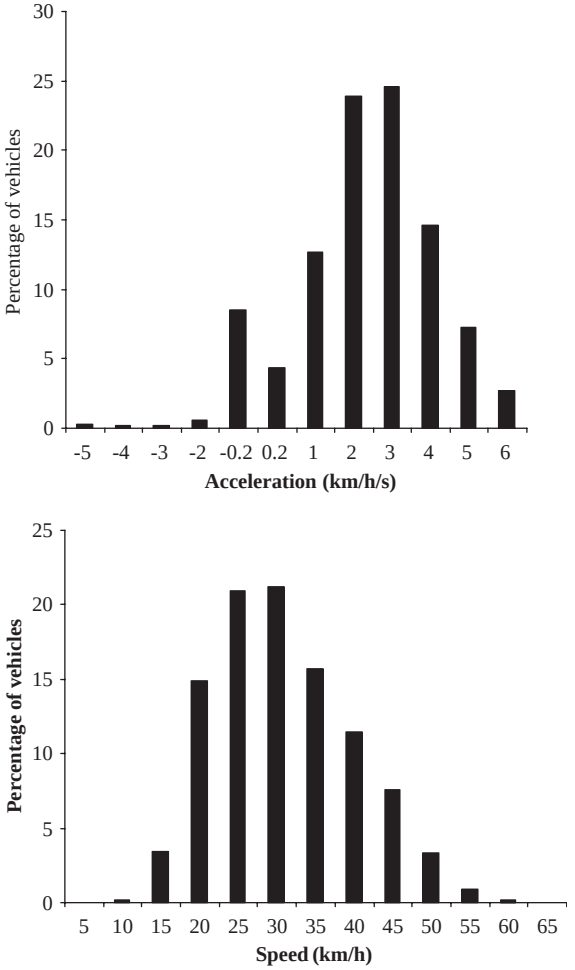

Fig. 3. Speed and acceleration distributions of vehicles recorded with the RS equipment.

the results of this work with those reported in the 1991 and 1994 studies. It is clear that the renewal of the fleet concomitant with more advanced emissions control technologies demonstrates a net amelioration in the exhaust emissions.

Additionally, average EFs among different urban areas in the United States with those of the MAMC and the city of Monterrey in Mexico, were compared in Table 2. It is worth to mention that Denver 2000 and Los Angeles 1997 data were calculated following the methodology of this work, while the rest are average RS data (Bishop et al., 1997; Pokharel et al., 2001a, b).

From the data of Table 2, it is evident that CO and HC emissions in the MAMC are still 2–4 times higher than those of the US cities, while NO_x emissions are from 2 to 3 times higher. The calculated mean MY in the case of the US cities is 1994.5 MY, while in the case of the MAMC the mean is 1991.8 ± 1.1 MY.

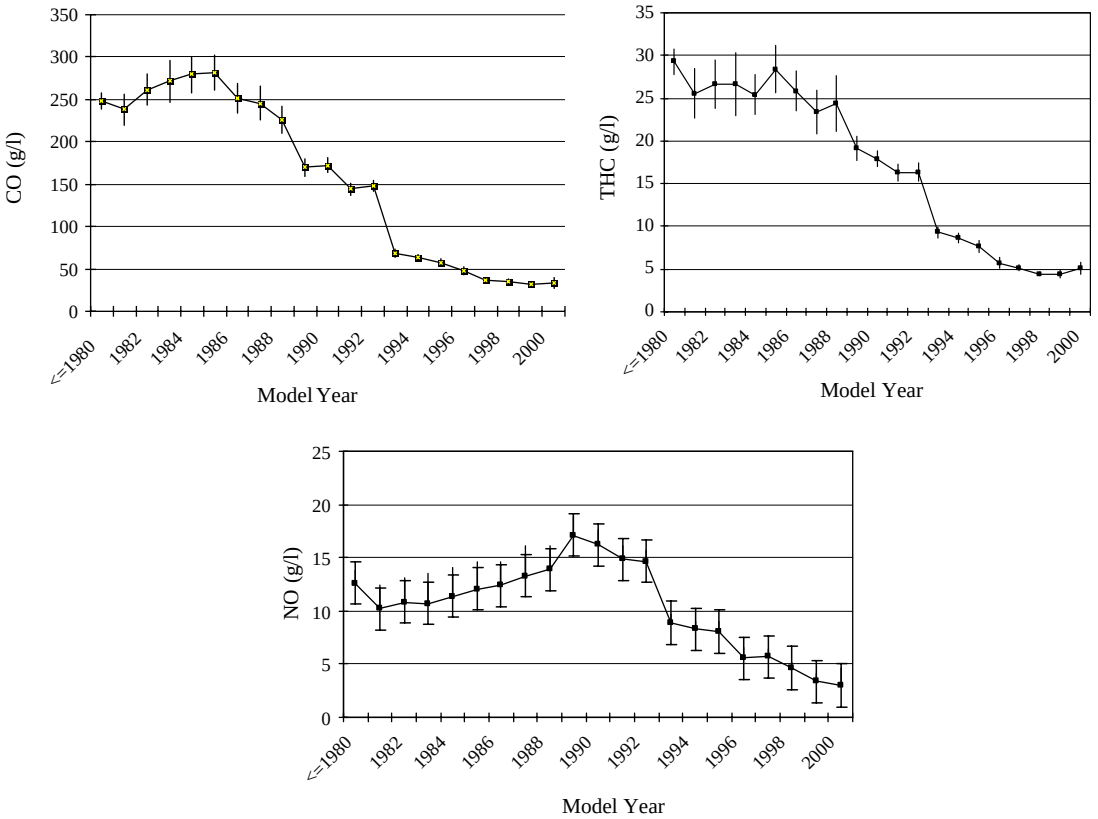

Fig. 4. Average CO, THC and NO EFs, by model-year of light-duty vehicles from the RS study, within 95% confidence intervals.

3.1. Emission inventory

Fuel-based emission inventory estimates was developed employing the EFs described in Figs. 4 and 5 combined with the year 2000 vehicle distribution by model year and activity.

The estimates developed in this study for the year 2000, and 95% confidence intervals, compared with the official inventory of 1998, are summarized in Table 3. Propagation of the errors led to overall uncertainties in tons of pollutant per year for CO, HC, and NO_x as 11.4%, 14.6%, and 23.4%, respectively.

The uncertainties include fuel economy estimations from the FTP tests and, for which the reproducibility depends on the automobile emissions systems rather than the test system (Berg, 1978), variability on-road remote sensing as discussed by Bishop et al. (1996).

Another source of uncertainty is the fuel sales used in this study. Using monthly data of fuel transferred from the refineries into the bulk, and later to the marketing installations an uncertainty of $\pm 2.8\%$ was calculated for gasoline fuel use.

Thus, the estimates are 48% and 26% lower in CO and NO_x, respectively, than those of 1998 using travel-based models. It is worth to point out that the inventory is 39% lower in HC, but the fuel-based inventory did not account for evaporative emissions.

Part of the discrepancies in absolute inventory values may be the result of differences in the treatment of cold start vehicles between model and fuel-based calculation. Accordingly, cold start does not contribute appreciably to the inventory since not much fuel is consumed during the short period of time when the vehicle is in "cold" mode. Moreover, Singer et al. (1999) have estimated that the fleet average EFs reported would be higher by only 6% than actual on-road stabilized emission levels.

3.2. Trends over years

We estimated an emission inventory with the data reported of emissions factors based on remote sensing measurements of CO and HC made in 1991 and 1994. Databases were obtained from the electronic page on Fuel Efficiency Automobile Test Data Center, University of Denver.

These data were used to obtain gasoline EFs and inventories for those years of measurement. This

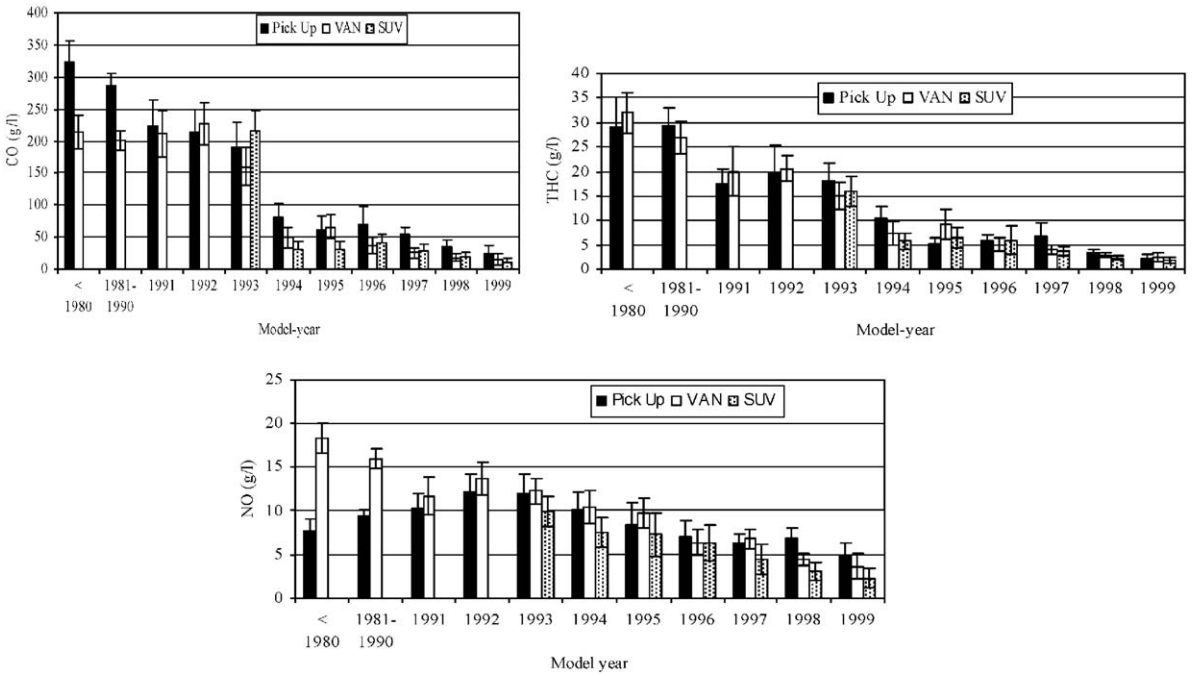

Fig. 5. Average CO, THC and NO EFs, by model-year of medium-duty vehicles (pick-ups, vans and SUVs) from the RS study, within 95% confidence intervals.

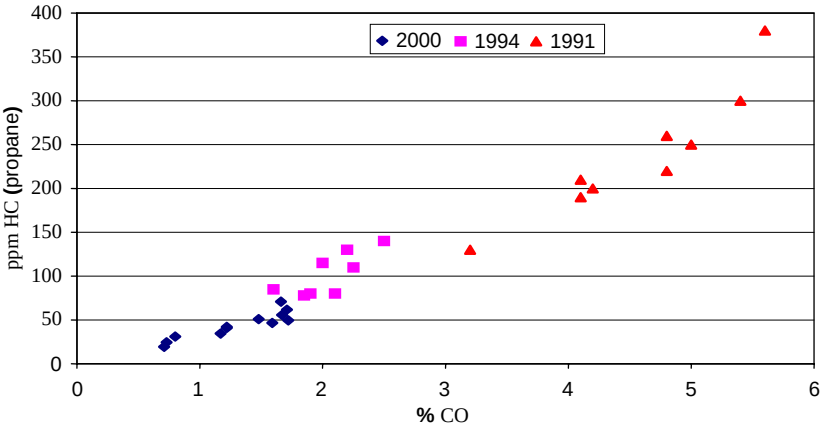

Fig. 6. Estimated average hydrocarbon and carbon monoxide concentrations per sites from RS studies in the years 1991, 1994 and 2000.

inventory analysis assumes that the outweighed average $g l^{-1}$ emissions can be good estimators, as a result of the large number of vehicles studied (Pokharel et al., 2001a).

The resulting emission inventories are summarized in Table 4 in which we include CO annual average concentration measured in the atmosphere of the MAMC as reported by the Metropolitan Automated Monitor System.

It is important to remark that the reduction in tons emitted of CO in 1994 with respect to those of 1991, amounts 46%. In the same period of comparison the decrease in measured CO in the atmosphere of the MAMC is in the order of 51%. For the 1994–2000 periods, the estimated reduction in tons of CO emitted was 36% while a 31% decrease was observed at the atmosphere.

Table 2
Comparison of vehicles EFs between US and Mexico cities

City (ref.)	# Measurements	CO ($g\,l^{-1}$)	HC ($g\,l^{-1}$)	NO ($g\,l^{-1}$)
Monterrey, Mex. 1994[a]	24,738	155	18.2	15
MAMC 2000	42,822	113.5 ± 13	13.1 ± 1.9	9.84 ± 2.3
Los Angeles 1997[b]	60,000	$80 + 7$	$9.3 + 1.5$	—
Phoenix 2000[c]	20,801	25	1.46	4.7
Los Angeles 2000[d]	23,303	45.4	3.5	4.4
Denver 2000[e]	22,986	48.3	6.6	5.1
Chicago 2000[f]	22,065	24	3	3.3

[a]Bishop et al. (1997).
[b]Pokharel et al. (2001d).
[c]Pokharel et al. (2002b).
[d]Pokharel et al. (2001c).
[e]Pokharel et al. (2001a).
[f]Pokharel et al. (2001b).

Table 3
Comparison of fuel-based total emissions in $ton\,yr^{-1}$ for year 2000, and the official inventory of 1998

Pollutant	Fuel-based 2000 (this work)	1998 emission inventory
CO	$753{,}848 \pm 86{,}140$	1,446,621
THC	$87{,}023 \pm 12{,}775$	143,320[a]
NO_x	$100{,}229 \pm 23{,}506$[b]	87,888

[a]Includes evaporative emissions.
[b]As NO.

Table 4
Evolution of fuel-based inventories for light and medium gasoline vehicles

Year (ref.)	Sold gasoline (million $liter\,day^{-1}$)	HC ($ton\,yr^{-1}$)	CO ($ton\,yr^{-1}$)	CO atmospheric concentration (ppm)[c]
1991[a]	17.16	318,316	2,180,437	5.9
1994[b]	18.26	160,512	1,187,388	2.9
2000	18.3	87,023	753,848	2

[a]Beaton et al. (1992).
[b]Bishop et al. (1997).
[c]Annual average.

These calculations indicate a continually decreasing inventory over the study period, and represents and ideal alternative for locations such as Mexican cities lacking the resources to develop an emissions model.

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