ON-ROAD MOTOR VEHICLE EMISSIONS FROM AROUND THE WORLD

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ABSTRACT

In 1993, on-road emissions in Continental Europe showed a pronounced South/North declining gradient for CO, HC and NO fuel specific emissions (gm/kg). Emissions in Hamburg and Rotterdam were comparable to emissions measured in 1993 in the U.S.A. with the same on-road instrument. Contrasts between emissions in the USA, the UK and Sweden demonstrated the importance both of modern catalysts and of good maintenance. The same contrast in 1998 shows the same picture but lower emissions at all three locations. In the decade from 1993-2003, U.S. emissions have declined markedly, apparently mainly as a result of lower emitting, better maintained, new vehicles. Results from Asia and South America will also be presented together with an analysis which suggests a correlation between average on-road vehicle emissions and the inverse of reported per-capita income. This latter relationship holds only up to a point. Beyond that point of increasing poverty, measured on-road emissions reach a plateau. Results from Mexico City in 1991 and 1994 show a dramatic decline in emissions, mainly caused by the imposition of modern control measures to the taxi cabs which, while only a small fraction of the registered fleet, contribute a large fraction of the vehicle miles traveled. Further improvement in Mexico City on-road emissions was observed by studies in 2000 [1] and 2003 [2].

INTRODUCTION

On-road emissions measurements have demonstrated skewed distributions in which relatively few vehicles are responsible for most of the pollution [3], and analysis has shown that identification and repair of these few vehicles is potentially more cost-effective than altering fuel chemistry, or scrapping vehicles just because they are older model years [4]. Further analysis given in this paper demonstrates that new vehicle emission standards which have been very successful in the past have reached a level of diminishing returns combined with escalating costs.

The remote sensor used in these studies was developed at the University of Denver for measuring the pollutants in motor vehicle exhaust, and has previously been described in the literature [5,6]. The instrument consists of a non-dispersive infrared (IR) component for detecting carbon monoxide, carbon dioxide (CO_2), and hydrocarbons, and a dispersive ultraviolet (UV) spectrometer for measuring nitric oxide. The source and detector units are positioned on opposite sides of the road in a bi-static arrangement. Collinear beams of IR and UV light are passed across the roadway into the IR detection unit, and are then focused onto a dichroic beam splitter, which serves to separate the beams into their IR and UV components. The IR light is then passed onto a spinning polygon mirror, which spreads the light across the four infrared detectors: CO, CO_2, HC and reference.

The UV light is reflected off the surface of the beam splitter and is focused into the end of a quartz fiber-optic cable, which transmits the light to a UV spectrometer. The UV unit is then capable of quantifying nitric oxide by measuring an absorbance band at 226.5 nm and comparing it to a calibration spectrum in the same region.

The exhaust plume path length and density of the observed plume are highly variable from vehicle to vehicle, and are dependent upon, among other things, the height of the vehicle's exhaust pipe, wind, and turbulence behind the vehicle. For these reasons, the remote sensor only directly measures ratios of CO, HC or NO to CO_2. The ratios of CO, HC, or NO to CO_2, termed Q, Q' and Q'' respectively, are constant for a given exhaust plume, and on their own are useful parameters for describing a hydrocarbon combustion system. The studies report measured emissions as %CO, %HC and %NO in the exhaust gas, corrected for water and excess oxygen not used in combustion. These are the data format found on the web site www.feat.biochem.du.edu. We recommend that users of the databases calculate the ratios of the reported data to CO_2 before further analysis. The %HC measurement is a factor of two smaller than an equivalent measurement by an FID instrument [7]. Thus, in order to calculate mass emissions as described below, the %HC values must first be multiplied by 2.0, assuming that the fuel used is regular gasoline. These percent emissions can be directly converted into mass emissions by the equations shown below.

$$\text{gm CO/gallon} = 5506 \bullet \%CO / (15 + 0.285 \bullet \%CO + 2(2.87 \bullet \%HC))$$
$$\text{gm HC/gallon} = 2(8644 \bullet \%HC) / (15 + 0.285 \bullet \%CO + 2(2.87 \bullet \%HC))$$
$$\text{gm NO/gallon} = 5900 \bullet \%NO / (15 + 0.285 \bullet \%CO + 2(2.87 \bullet \%HC))$$

These equations indicate that the relationship between concentrations of emissions to mass of emissions is quite linear, especially for CO and NO and at low concentrations for HC. Thus, the percent difference in emissions calculated from the concentrations of pollutants reported here is equivalent to a difference calculated from masses.

Another useful conversion is from percent emissions to grams pollutant per kilogram (g/kg) of fuel. This conversion is achieved directly by first converting the pollutant ratio readings to the moles of pollutant per mole of carbon in the exhaust from the following equation:

$$\frac{\text{moles pollutant}}{\text{moles C}} = \frac{\text{pollutant}}{CO + CO_2 + 3HC} = \frac{(\text{pollutant}/CO_2)}{(CO/CO_2) + 1 + 3(HC/CO_2)} = \frac{(Q, 2Q', Q'')}{Q + 1 + 6Q'}$$

Next, moles of pollutant are converted to grams by multiplying by molecular weight (e.g., 44 g/mole for HC since propane is measured), and the moles of carbon in the exhaust are converted to kilograms by multiplying (the denominator) by 0.014 kg of fuel per mole of carbon in fuel, assuming gasoline is stoichiometrically CH_2. Again, the HC/CO_2 ratio must use two times the reported HC (as above) because the equation depends upon carbon mass balance and the NDIR HC reading is about half a total carbon FID reading [7].

Quality assurance calibrations are performed twice daily in the field unless observed voltage readings or meteorological changes are judged to warrant more frequent calibrations. A puff

of gas containing certified amounts of CO, CO_2, propane and NO is released into the instrument's path, and the measured ratios from the instrument are then compared to those certified by the cylinder manufacturer (Praxair). These calibrations account for day-to-day variations in instrument sensitivity and variations in ambient CO_2 levels caused by local sources, atmospheric pressure and instrument path length. Since propane is used to calibrate the instrument, all hydrocarbon measurements reported by the remote sensor are as propane equivalents.

Studies sponsored by the California Air Resources Board and General Motors Research Laboratories have shown that the remote sensor is capable of CO measurements that are correct to within ±5% of the values reported by an on-board gas analyzer, and within ±15% for HC [8,9]. The NO channel used in this study has been extensively tested by the University of Denver. Tests involving a late-model low-emitting vehicle indicate a detection limit (3σ) of 25 ppm for exhaust NO, with an uncertainty of ±5% of the reading at higher concentrations.

The remote sensor is accompanied by a video system to record a freeze-frame image of the license plate of each vehicle measured. The emissions information for the vehicle, as well as a time and date stamp, is also recorded on the video image. The images are stored on videotape, so that license plate information may be incorporated into the emissions database during post-processing. A device to measure the speed and acceleration of vehicles driving past the remote sensor was also used in these studies. The system consists of a pair of infrared emitters and detectors (Banner Industries) which generate a pair of infrared beams passing across the road, six feet apart and approximately two feet above the surface. Vehicle speed is calculated from the time that passes between the front of the vehicle blocking the first and the second beam. To measure vehicle acceleration, a second speed is determined from the time that passes between the rear of the vehicle unblocking the first and the second beam. From these two speeds, and the time difference between the two, the acceleration is calculated.

An equation for determining the instantaneous power of an on-road vehicle has been proposed by Jimenez [10], which takes the form

$$\mathrm{VSP} = 4.39 \bullet \sin(slope) \bullet v + 0.22 \bullet v \bullet a + 0.0954 \bullet v + 0.0000272 \bullet v^3$$

where VSP is the vehicle specific power in kW/metric tonne, *slope* is the slope of the roadway (in degrees), v is vehicle speed in mph, and a is vehicle acceleration in mph/s. Derived from dynamometer studies, and necessarily an approximation, the first term represents the work required to climb the gradient, the second term is the $f = ma$ work to accelerate the vehicle, the third is an estimated friction term, and the fourth term represents aerodynamic resistance. Using this equation, vehicle specific power is calculated for all measurements in the databases. This equation, in common with all dynamometer studies, does not include any load effects arising from road curvature.

In reports to CRC (see web site www.feat.biochem.du.edu) vehicle emissions data are compared by selecting data in various VSP bins, or are corrected by using emissions per VSP bin from each measurement year and the numbers per VSP bin from a reference year.

RESULTS AND DISCUSSION

Figure 1 shows data obtained in 1992. There are several notable features. The newest vehicles in Sweden have lower emission than the newest vehicles in Los Angeles. The "cliff" in the 1988/89 period when modern emission controls were introduced is also very apparent in the Swedish data [11,12]. The low emitting new vehicles in Sweden could arise because they are built lower emitting or it could arise from a better maintenance, or a combination of both. One can see that maintenance is important from these data because the US vehicles with MY from 1975 to 1980 were originally built with catalysts while the Swedish vehicle were not, but have been maintained for 12-17 yrs by Swedes. After this passage of time it appears that good maintenance predominates over emission control technology. Measurements in 1994 from other regions of Europe indicated that The Netherlands and Germany had emissions comparable to Sweden while France, Italy and Portugal showed increases with the UK comparable to Portugal (see Table 2). Apparently the UK suffered from a combination of poor technology and poor maintenance. The UK did begin modern emission controls in the 1992 MY.

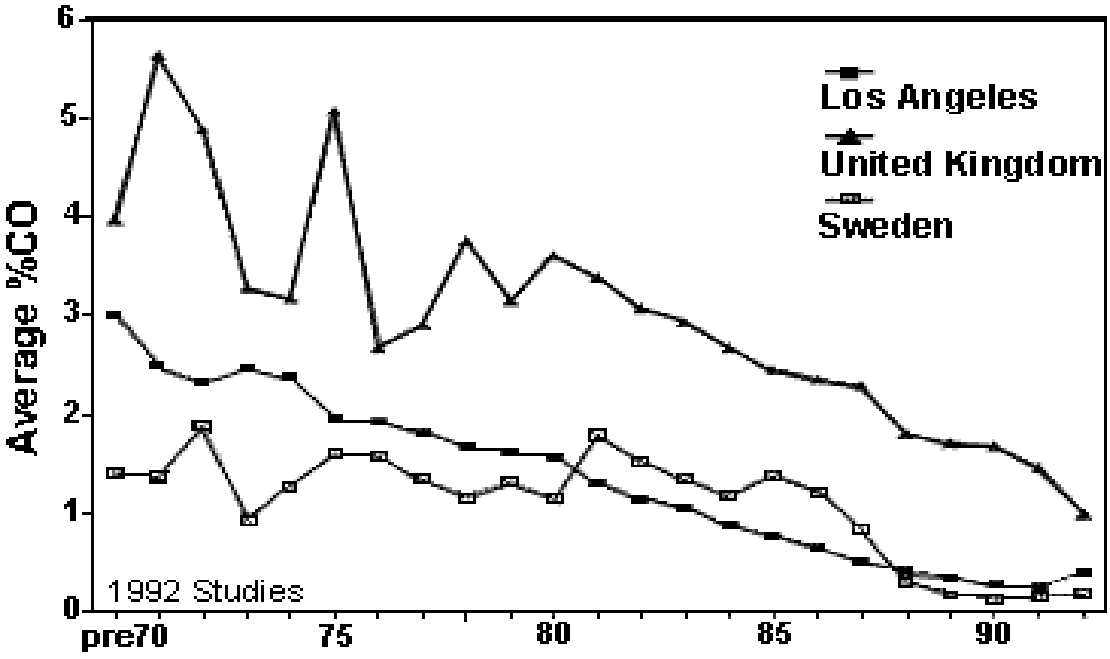

Figure 1. On-road percent CO emissions measurements collected in Los Angeles, Sweden and the United Kingdom in 1992.

Figure 2 shows the same comparison but with measurements carried out in 1998-1999 [13, 14, 15]. As before, the newest Swedish vehicles have uniformly lower emissions than the USA and the UK lags. We do not have UK data, but the US/Sweden comparison can be continued to 2001 data [15, 16]. Again, the Swedish measurements show emissions lower than the US across all of the post 1990 MY.

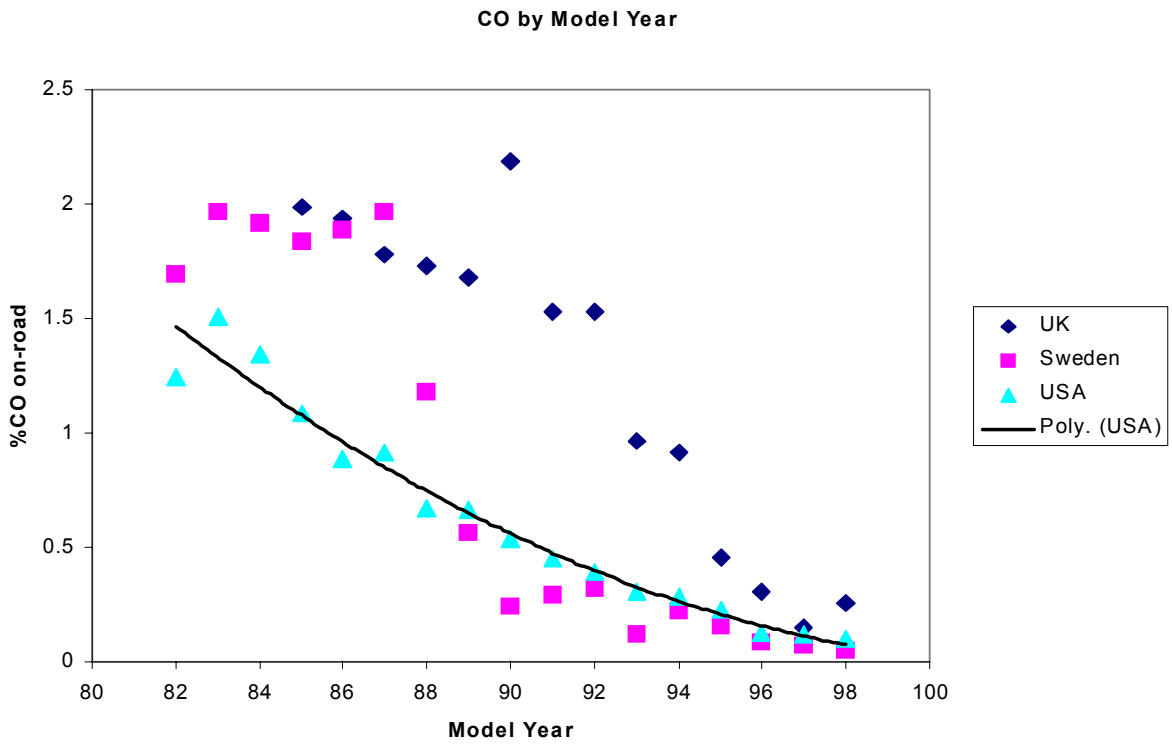

Figure 2. An updated version of Figure 1 with mean percent CO emissions plotted against model year for data from Sweden, United Kingdom and the United States. The USA data were collected in Denver, CO.

On-road emissions measured by means of remote sensing can be directly converted to emissions per gal or per kg of fuel as discussed. With the aid of fuel sales, often available from tax statistics, a fuel-based motor vehicle emissions inventory can be determined [16,17]. Bradley et al [18] compared emission measurements from several locations worldwide with a measure of poverty, the inverse of GDP per capita. She proposed a linear relationship stopping at a maximum (no emission controls, poor maintenance as observed in Bangkok and Katmandu in 1993 and Mexico City in 1991). The more recent data in Tables 1 and 2 confirm this result. We have also taken the data in Tables 1 and 2 and correlated on-road CO with on-road HC. The result is shown in Figure 3 where the slope is 0.3 and the correlation coefficient 0.49.

From Tables 1 and 2 several features stand out. Countries with low per capita income have higher emissions [18]. The highest HC emissions are dominated by large fractions of two-stroke engines. These engines produce HC/CO emission ratios which are much larger than even uncontrolled four-stroke engines. Countries with significant diesel fraction show NO > 15 g/kg. Overall there is a reasonable HC/CO correlation (Figure 3), not unexpected since both increase with increasing age and both increase absent emission controls.

Table 2 shows data from Mexico City in 1991 and 1994. Schifter et al has reported on-road emissions in 2000 [1]. Volkamer et al [2] report a rush hour CO/CO_2 ratio from which average

Table 1. FEAT data sets and mean emissions collected in the United States.

Location & Year	Measurements	Mean gCO/kg	Mean gHC/kg	Mean gNO/kg	Mean Model Year
Austin TX, 1998	17,265	70.2	13.2	4.1	1993.0
Baltimore MD, 1992	4,761	61.4	4.4	*	1987.8
Chicago IL, 1989	11,818	105	*	*	1983.5
Chicago IL, 1990	13,639	125	49	*	1985.3
Chicago IL, 1992	8,733	121	32.8	*	1986.0
Chicago IL, 1997	19,682	56	5.3	5.5	1992.7
Chicago IL, 1998	23,560	49	5.3	5.7	1993.6
Chicago IL, 1999	23,088	44	4.5	5.3	1994.3
Chicago IL, 2000	22,065	33	3.9	4.5	1994.9
Chicago IL, 2002	22,320	29	3.2	3.7	1997.4
Denver CO, 1989	4,909	92	*	*	1983.1
Denver CO, 1991	13,391	89	*	*	1985.9
Denver CO, 1992	40,019	84	17.4	*	1985.5
Denver CO, 1994	2,787	69	6.0	6.8	1987.0
Denver CO, 1995	3,176	64	5.6	10.7	1989.2
Denver CO, 1996	30,675	66	9.5	11.8	1989.2
Denver CO, 1997	46,120	61	10.1	8.2	1990.3
Denver CO, 1997	39,152	77	9.8	7.2	1990.4
Denver CO, 1997	33,561	85	8.3	5.6	1990.8
Denver CO, 1999	26,709	56	5.0	8.4	1992.4
Denver CO, 2000	22,986	54	4.6	7.2	1993.4
Denver CO, 2001	21,021	43	4.6	6.8	1994.6
Denver CO, 2002	10,025	33	2.6	5.7	1996.5
Denver CO, 2003	21,321	44	3.4	6.5	1996.4
El Paso TX, 1993	14,109	143	26	*	1986.1
Glenwood Springs CO, 2001	627	44	7.9	9.9	1995.2
Grand Junction CO, 2001	5,240	69	6.8	9.2	1993.6
La Brea CA, 2001	20,319	56	4.6	5.6	1994.4
La Brea CA, 2003	20,191	42	4.5	4.5	1996.5
Los Angeles CA, 1989	16,511	141	*	*	1981.8
Los Angeles CA, 1991	91,679	95	27.2	*	1984.7
Los Angeles CA, 1999	12,655	66	8.0	6.1	1991.6
Naperville IL, 1990	1,890	39	*	*	1986.3
Nevada, 1994	20,747	93	12	18	1987.1
Phoenix AZ, 1998	17,759	34	3.9	5.1	1993.3
Phoenix AZ, 1999	18,894	38	3.0	8.1	1994.0
Phoenix AZ, 2000	20,801	34	3.8	6.4	1995.3
Phoenix AZ, 2002	23,679	27	2.6	4.6	1997.4
Provo UT, 1992	17,442	119	57	*	1984.6
Riverside CA, 1999	18,752	67	3.0	5.2	1992.4
Riverside CA, 2000	23,303	62	2.6	6.0	1993.3
Riverside CA, 2001	19,800	48	2.0	5.6	1994.5
Sacramento CA, 1999	18,405	36	10.4	3.1	1992.9
San Jose CA, 1999	37,335	49	6.1	4.4	1992.5
Tucson AZ, 1994	14,051	121	27.6	*	1985.2
Ute Pass CO, 1989	4,541	188	*	*	1982.3

Table 2. FEAT data sets and mean emissions collected worldwide.

Locations & Year	Measurements	Mean gCO/kg	Mean gHC/kg	Mean gNO/kg	Mean Model Year
Bangkok THA, 1993	5,260	264	220	*	*
Edinburgh SCT, 1993	1,681	172	20	48	*
Hamburg DEU, 1994	11,128	69	14	18	*
Hong Kong, 1993	5,891	115	20	*	*
Kathmandu NPL, 1993	11,227	362	189	*	*
Leicester UK, 1993	901	213	6.0	56	*
Lisbon PRT, 1994	10,426	174	21.6	18	*
London UK, 1993	4,149	105	14.2	34	*
Lyon FRA, 1994	9,638	132	26	16	1989.1
Kuala Lumpur MAL, 1995	9,478	209	22	31	*
Mexico City MEX, 1991	31,838	465	64.3	*	*
Mexico City MEX, 1994	39,273	234	32.5	*	*
Milan ITA, 1994	13,546	157	22.8	17.4	1989.3
Monterrey MEX, 1995	10,654	203	23.8	20	1987.8
Rotterdam NLD, 1994	11,359	73	13.5	27	1989.2
Seoul KOR, 1993	3,104	100	14.7	*	*
Taipai TWN, 1993	12,062	180	23.4	*	*
Toronto CAN, 1990	2,809	71	*	*	1986.2
Tokyo JPN, 1995	3,881	67	7.6	*	*
Zurich CHE, 1994	11,298	100	9.5	*	*

CO emissions can be determined. Figure 4 shows that all four of these data sets indicate a steadily reducing CO emission level, albeit remaining well above US levels.

We have shown recently that emissions of newer vehicles in the USA have decreased dramatically as a result of new vehicle emission standards and the manufacturer's ability to meet them with increasingly robust hardware [20]. As a result, most of the on-road emissions arises from a very few broken vehicles. We illustrate this with the most recent CRC data from California [21]. Figure 5 shows three sets of deciles in which vehicle emissions for the three pollutants CO, HC and NO are plotted in such a way that the heights of the bars represents the emissions which a fleet of ten vehicles would have to match the 2002 on-road statistics.

From these graphs it is apparent that a very few vehicles are causing most of the emissions. In fact for these data, 50% of the emissions of CO, HC and NO are caused by 4.4%, 7.4% and 7.4% of the measurements respectively. Because so few vehicles are responsible for most of the emissions and these vehicles are broken, it is apparent that further tightening of new car standards will do little to improve average on-road emissions, despite their considerable expense. It is also apparent from the same data that altering the fuel chemistry in such a way as to cause relatively small percent reduction in emissions of all vehicles is not at all a cost effective means of reducing on-road emissions. These conclusions were demonstrated quantitatively by Beaton et al in 1995 [4] and have been reinforced from an economics standpoint by Rask [22] and the more recent data makes the conclusion even stronger because the emissions distribution in the USA is even more skewed now than it was then. The conclusion, then as now, is that identification and repair (or scrappage if appropriate) of the

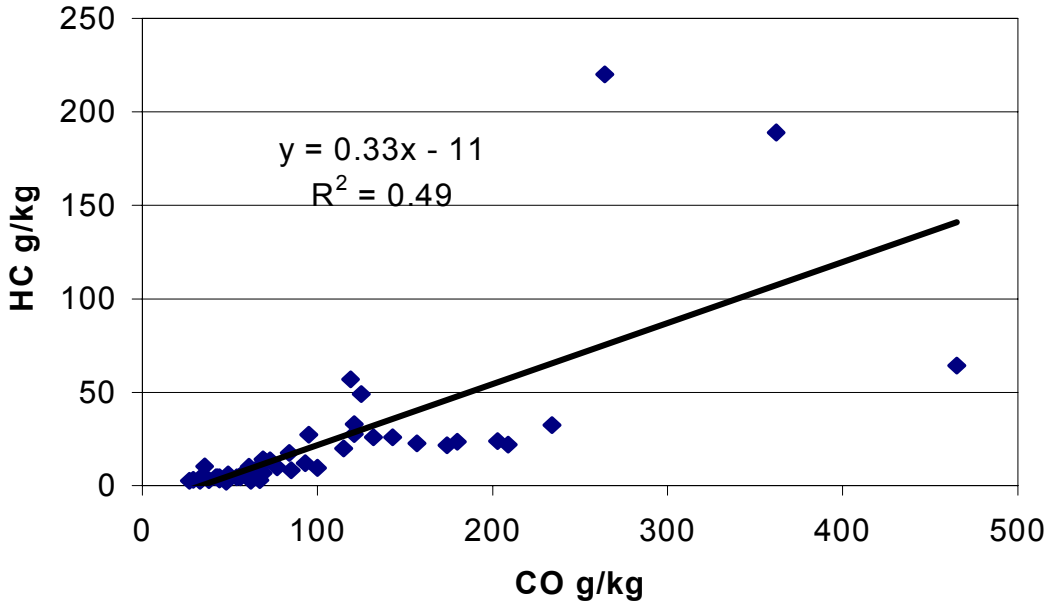

Figure 3. Correlation graph of HC versus CO for data sets in Tables 1 and 2 with more than 5000 measurements.

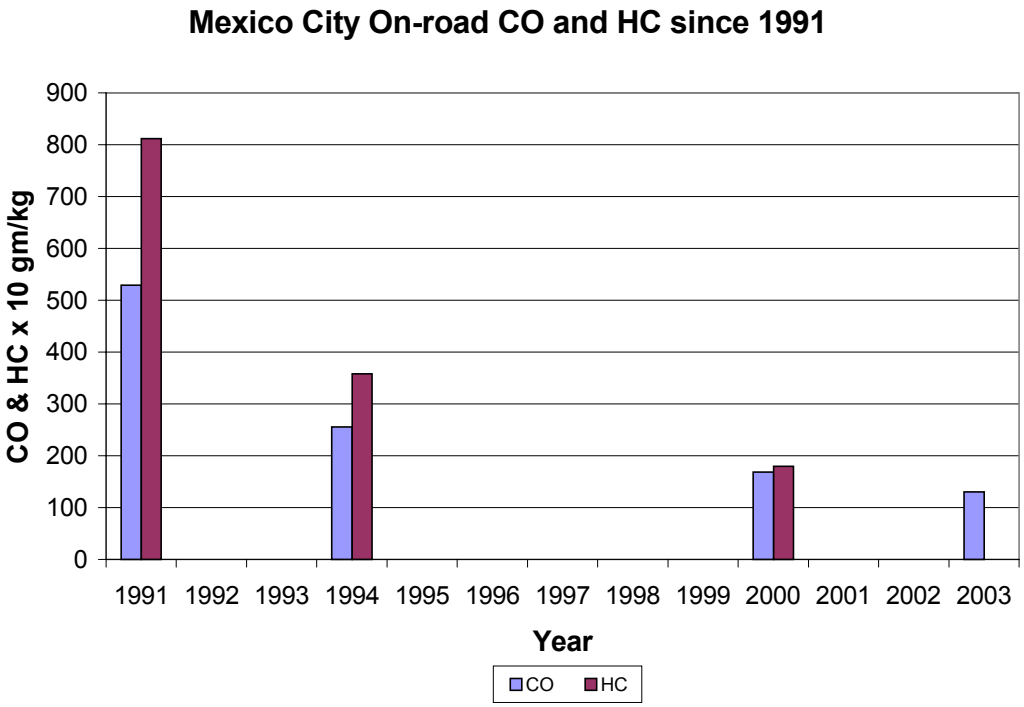

Figure 4. CO and HC emissions collected in Mexico city by different researchers. HC emissions are multiplied by 10 relative to CO.

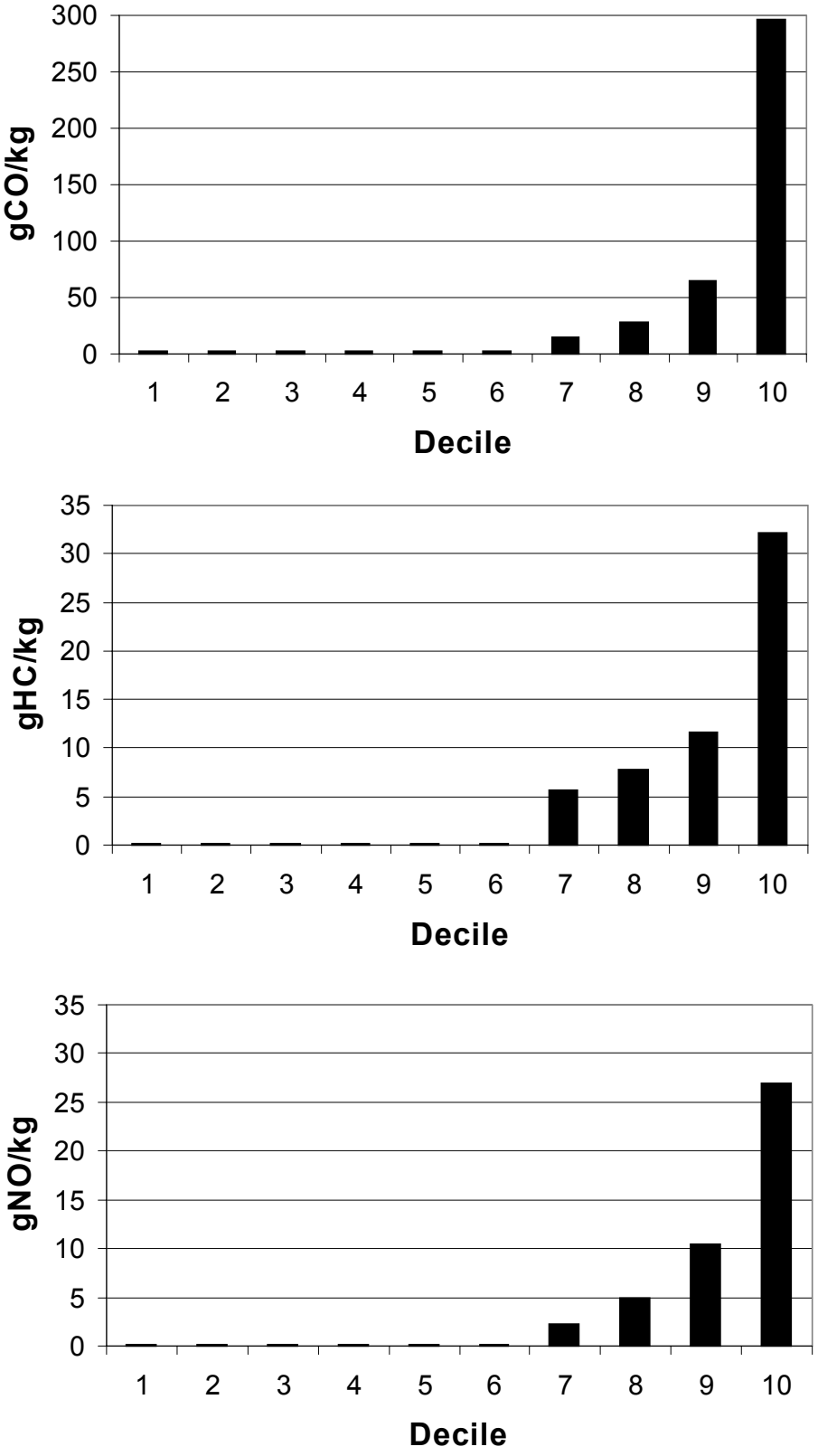

Figure 5. Emissions by decile for on-road emissions collected in Los Angeles CA, October 2003.

on-road gross emitters, is the most effective strategy, and that current Inspection and Maintenance programs are not as successful at achieving this goal as they are supposed to be [23,24].

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