On-Road Emissions in Asia Measured by Remote Sensing.

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ABSTRACT

On-road emissions are precisely the emissions which are impacting the air quality and which need to be controlled if air quality improvements are desired. Remote sensing measures mass emissions of CO, HC, NO and smoke per kg (or per L) of fuel burned. On-road emissions are not certification emissions for new vehicles, although new vehicles measured on-road most often provide remote sensing readings well within their certification values. On-road emissions are not the same as readings from scheduled emission tests, although on average the two correlate very well. The major difference between scheduled emission tests and on-road emissions readings is how emissions variability is treated. All emissions tests show low variability when testing new, well controlled vehicles and show very high variability when testing older, and especially older broken, vehicles. Because on-road remote sensing readings can be obtained with a throughput of 10,000 vehicles per day, the variability of readings taken day after day on the same vehicle is readily apparent. By contrast, scheduled emission tests tend to be performed only once for vehicles which pass the first time (most vehicles) and are only repeated for vehicles which fail the first time. It has been shown from analysis of US data that many of the vehicles which subsequently receive passing readings do so because their emissions naturally vary and happen on the first test to be high and the second test low. Roadside tests in California carried out by their Bureau of Automotive Repair show that vehicles identified as on-road gross emitters by means of remote sensing and pulled over immediately by a policeman have an 83-88% (depending on the pollutant) chance of failing a California emissions test performed on the spot. On road emissions measured in Asia show large geographic variability and everywhere that a few vehicles are responsible for most of the on-road emissions.

INTRODUCTION

On-road emissions measurements have demonstrated excellent correlation ($r2>0.96$ in all cases) for fleet average emissions by model year versus IM240 dynamometer tests. These results were obtained for CO, HC and NO in Chicago, Denver and Phoenix, USA where IM240 programs are carried out [1]. On-road emissions measurements have demonstrated excellent individual vehicle correlation with the California Bureau of Automotive Repair ASM tests, wherein high-emitting on-road vehicles had an 83-86% (depending on pollutant) chance of failing the California ASM test [2]. On-road emissions measurements everywhere demonstrate 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].

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 (diesel and LPG fuels are not very different). 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. Similar equations for emissions per kg of fuel used can be derived directly from the ratios Q, Q',Q". The equations are:

CO = 28xQx71.7/(1+Q+6Q’): HC = 2x44xQ’x71.7/() and NO= 30xQ”x71.7/()

Where in all cases the denominator is (1+Q+6Q’) and the results are in gm/kg of fuel.

Quality assurance calibrations are performed at least twice daily in the field unless 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. 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

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

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. Apparently the UK suffered from a combination of poor technology and poor maintenance. The UK did begin modern emission controls in the 1992 MY. There are similar geographic differences in Asia (Table 1) with results from Singapore and Tokyo

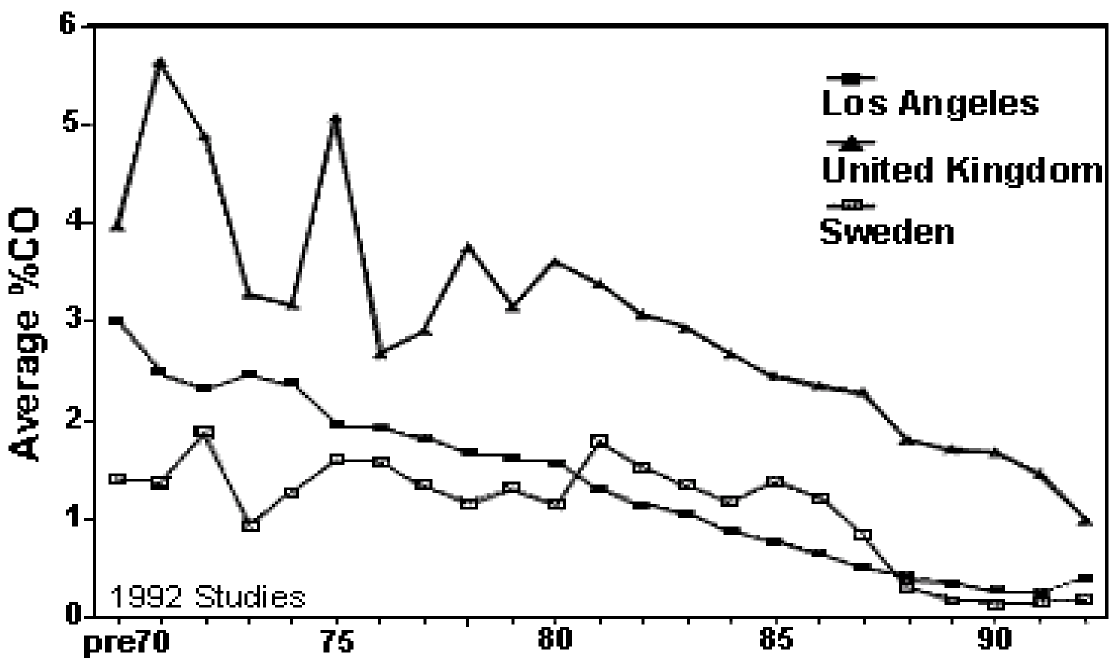

Figure 1. On-road percent CO emissions measurements collected in Los Angeles, Sweden and the United Kingdom in 1992.

standing out as evidence of low emissions, while Bangkok and Katmandu both appear in the opposite category.

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. We believe that these data emphasize the successful combination of good technology with good maintenance.

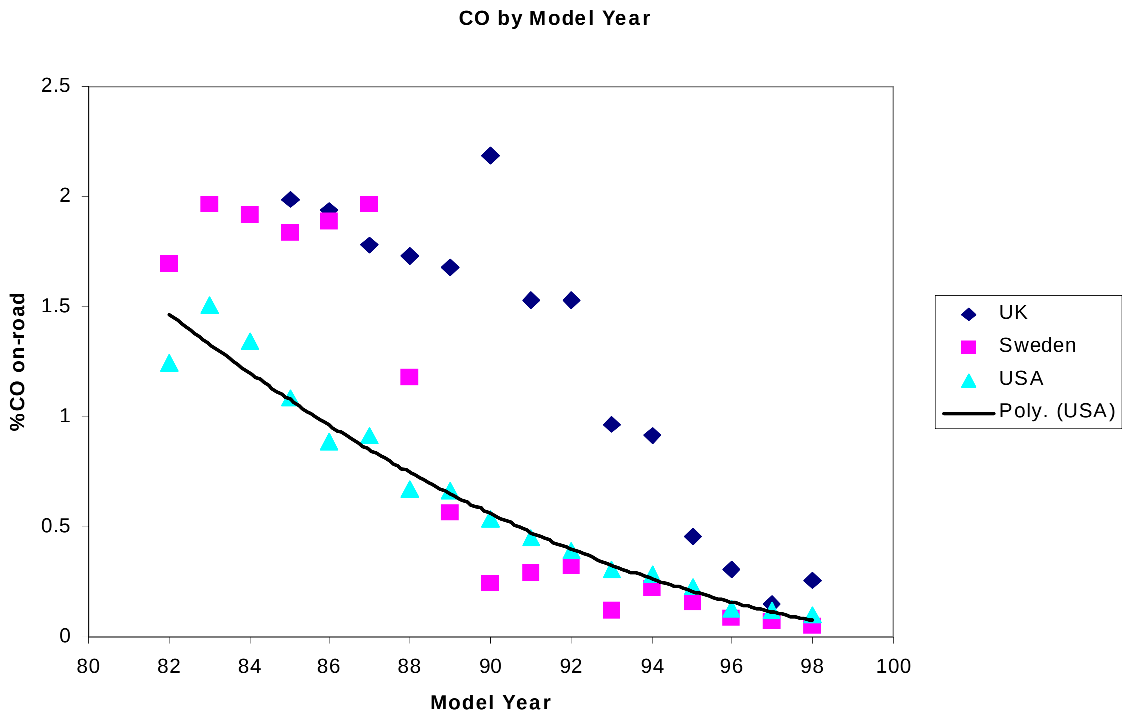

Figure 2. An updated version of Figure 1 with mean percent CO emissions plotted against model year for data from Sweden, United Kingdom and the United States. The USA data were collected in Denver, CO.

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). More recent data (in Tables 1 and 2 of reference [19]) confirm this result. We have also taken our worldwide data 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. Elimination of two very early HC RSD systems and the two highest HC points, (Bangkok and Kathmandu with many two-stroke motorcycles) leads to a more realistic correlation coefficient of 0.8 and a slope of 0.15.

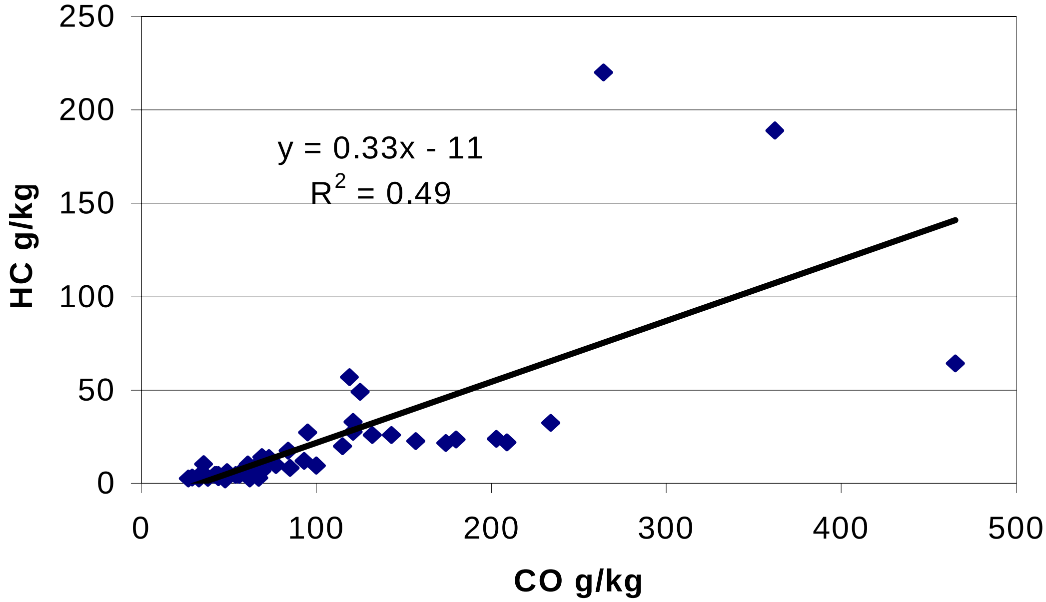

Figure 3. Correlation graph of HC versus CO for data sets in Tables 1 and 2 with more than 5000 measurements.

Table 1 in this paper shows on-road data measured in Asia with various RSD units and, in the last two rows, some of the oldest and some of the most recent US averaged data (Denver, Chicago and Los Angeles). It is apparent that some locations in Asia, Singapore in particular, come very close to demonstrating on-road emissions comparable to the current U.S. on-road fleet averages.

Averages however obscure sometimes as much as they illuminate. We find everywhere that the majority of the emissions come from a few on-road vehicles, the so-called gross emitters. We have also found that gross emitters can be any age, technology or model year, although there is a strong correlation between the incidence of gross emitters and the vehicle age, and a lesser correlation with the original vehicle purchase cost [20]. It has been suggested that the latter effect arises from a combination of influences. The lowest cost vehicles have the lowest cost emission control components and are purchased (and repurchased) by owners less financially capable of continuing proper maintenance [21].

Figure 4 illustrates just this kind of skewed distribution from recent studies in New Delhi, India. For the purpose of this figure we focus solely on vehicles identified by license plate as automobiles. In this presentation, the data set of more than 6000 cars has been divided into ten "vehicles" (deciles) in increasing order of their emissions (CO, HC and NO separately although there is considerable overlap). It is apparent from this diagram that there is a

Table 1. On-Road data sets and mean emissions collected in Asia: as recorded, no corrections for validity, fuel type or motorcycles. *US comparison below.*

Locations & Year	Measurements	Mean gCO/kg	Mean gHC/kg	Mean gNO/kg
Bangkok THA, 1993	5,260	264	220	*
Hong Kong, 1993	5,891	115	20	*
Kathmandu NPL, 1993	11,227	362	189	*
Kuala Lumpur MAL, 1995	9,478	209	22	31
Seoul KOR, 1993	3,104	100	14.7	*
Taipai TWN, 1993	12,062	180	23.4	*
Singapore 1995 2004	1,681	148 32	7 5	24 10
Tokyo JPN, 1995	3,881	67	7.6	*
Melbourne AUS, 1992	5,260	149	6.8	24
New Delhi, India, 2004	10,800	106	61	20
Aukland NZ, 2004	34,400	89	13	10
Average USA 1989-92	*34,000*	*113*	*26*	*
Average USA 2003	*63,000*	*39*	*3.5*	*4.9*

majority of cars with negligible emissions, while the average is dominated by a very small fraction of the fleet. Comparison of these data with recent on-road data from Los Angeles (Figure 5, mostly automobiles) shows that the on-road emissions in New Delhi have considerable room for improvement in both the lowest and highest emitting vehicles.

From Table 1 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_2 emission ratios which are much larger than even uncontrolled four-stroke engines. Countries with significant diesel fraction show larger NO readings. Overall there is a reasonable HC/CO correlation (Figure 3), not unexpected since both increase with increasing age and both increase absent emission controls.

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 [21]. 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 [22]. 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.

These graphs show that a very few vehicles are causing most of the emissions. 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 in the USA 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

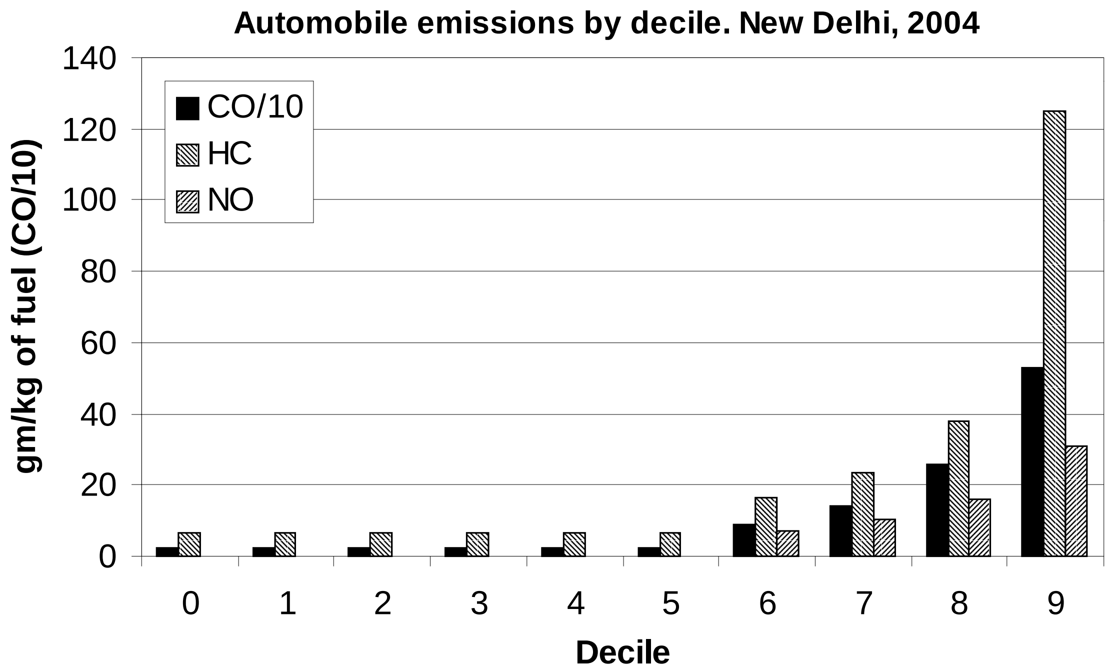

Figure 4. Emissions measurements from New Delhi India arranged in increasing order of emissions for each pollutant separately and shown as ten bars whose mean and distribution resembles the on-road fleet. The lowest emitting 60% have been plotted as equal because their small differences are not statistically greater than the instrument noise.

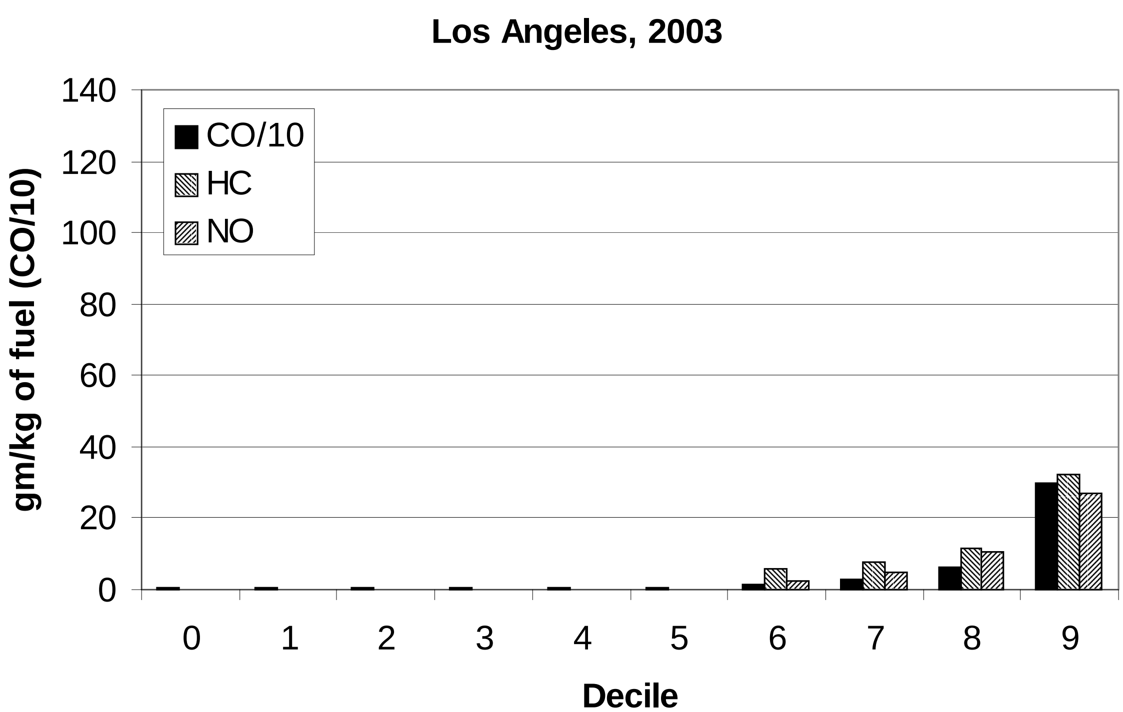

Figure 5. Emissions measurements from Los Angeles plotted identically to those from New Delhi presented in Figure 4.

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 [23] 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 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 [24,25].

Figure 6 shows four recent emissions comparisons from New Delhi in which the percentage contributions of the various members of the fleet are compared to their contributions to emissions. As one might have expected, for CO all fleet emissions contributions are more or less equal to their presence. For HC the four wheel cars and vans stand out as low and the three wheelers as disproportionately high. For NO the diesel and petrol autos stand out and for smoke the diesel and predominantly two-stroke, three wheeler vehicles obviously dominate the emissions picture.

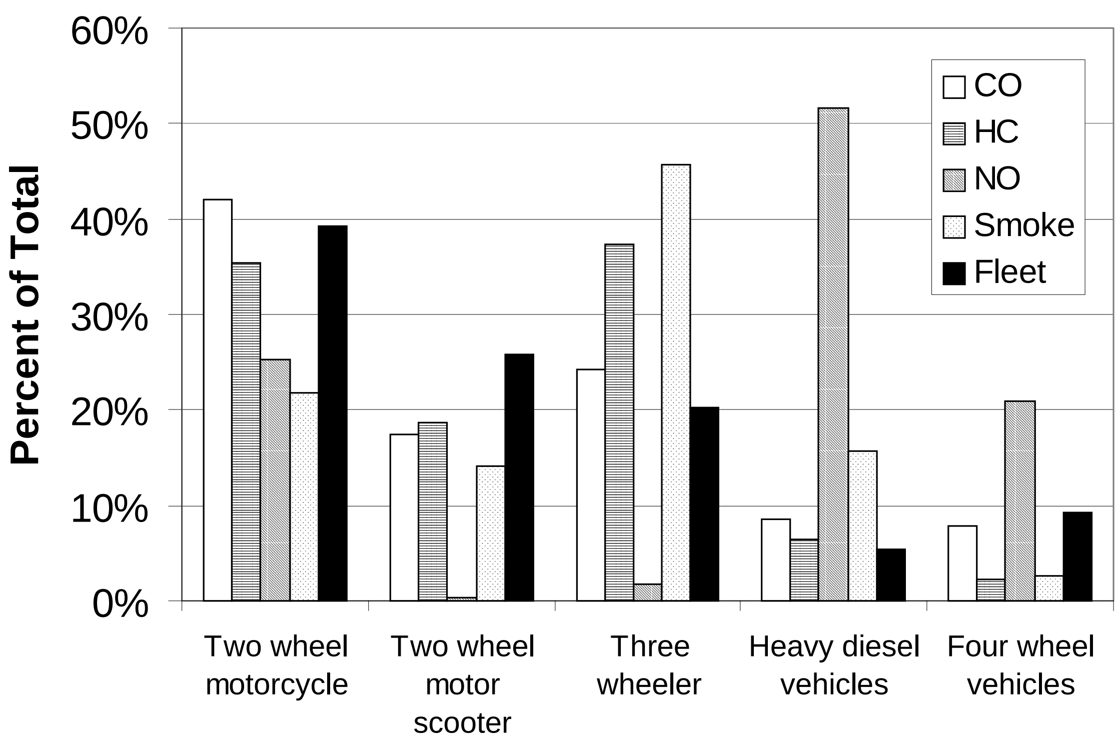

Figure 6. In black are shown the percent contributions of the various vehicle fleets to on-road mileage. In shaded bars as indicated are the percent contributions of their emissions.

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