Real World Emission Factors estimated from Street Canyons and Road Tunnel in Stockholm

Master's Thesis carried out in the International Master Program for Traffic Environment and Safety Management, Linköping University

by

Byeongsup Moon

Reg nr: LITH-ITN-EX-   -SE

Supervisors: Christer Johansson
Institute of Applied Environmental Research,
Stockholm University

CONTENTS

Introduction

Urban air pollutants arise from a variety of sources. Road traffic is an important source of key pollutants on global, regional and urban scales. Several attempts to estimate emission rates of traffic pollutants have been made as part of measures for controlling air pollution. As a result of continuous development of measurement techniques, it comes to light that the pattern of trips undertaken affects emission rates and that vehicle factors such as engine type and weight also affect emission rates as do driving conditions such as average speed, accelerations and decelerations.

Without regard to emission rates, pollutant concentrations can vary considerably on an hourly and daily basis. These variations are due not only to the source characteristics, but also to meteorological factors which give rise to dispersion, physical and chemical transformations which can alter the nature of the pollutant and removal processes. Buildings can, first of all, complicate dispersion patterns by deflecting the wind and causing greater turbulent activity in their wake. This may be of particular importance for the dispersion of vehicle exhausts where pollutants are released at ground level, and in urban areas, within so-called street canyons (Cloke *et al*., 1998).

It is important to understand dipersion processes and pollutant concentrations in street canyons as well as emission rates because all persons are living in urban areas and urban areas are largely composed of street canyons on which most of urban traffic pass through and dispersion processes are unique.

Moreover, to make traffic management schemes on air quality it is necessary to get the real world emission rates. Tests carried out in the laboratory do not accurately reflect emission rates encountered on the road, although laboratory conditions provide the best way to control repeatability. Actually, emission rates are dependent on the operation of the vehicle and this may not be adequately represented by standardised cycles. To assess the real adverse impact of the air pollution it is necessary to estimate directly emission rates in street canyons where air pollutants from vehicles are released.

The main aim of this thesis is to estimate emission rates under realistic driving conditions in a street canyon and to compare those with emission rates both estimated from tunnel measurement and calculated by the EVA model, the Swedish road traffic emission model. For the purpose this thesis starts by looking at the general principles in estimation of vehicle emissions and the methodology of combining the atmospheric pollution dispersion model with vehicle emissions. In chapter 3, the dispersion processes in a street canyon and the model capable of simulating dispersion processes are reviewed.

In chapter 4, based on the previous methodology the trends of emission factors of NOx, CO and PM from 1994 to 1999 in Stockholm are estimated. Emission factors of same pollutants based on tunnel measurements in 1998 and 1999 are also estimated in chapter 5. Real world emission factors based on street canyons and tunnel measurements are compared with those of the EVA model in chapter 6. Finally this thesis makes a conclusion by discussing the uncertainties of the different measurements.

Methodology to integrate Emission and dispersion model

Emission Model

General principles

The general principles in estimating pollutant emissions from road traffic are based on two steps: 1) determining a set of emission factors which specifies the rate at which emissions are generated (tailpipe, evaporative, or running loss emissions); 2) determining an estimate of vehicle activity as a function of vehicle class, time of day, location, speed and density. The emission inventory is then caluclated by multiplying the results of these two steps, as expressed by the following equation (Hickman *et al*., 1997):

$$Q_i = \sum_{j=1}^{n} \sum_{k=1}^{m} N_{j,k} \cdot e_{i,j,k} \qquad (1)$$

where, Q_i is the amount of pollutant i emitted

e is an emission factor
N is the amount of traffic
k identifies different types of vehicle
j identifies different types of vehicle operation

This expression shows the broad categories of data that are required in emission modelling, but it hides the large number of variables within each category. For example, there are hundreds of types of vehicles in service, and each will have different characteristics in terms of emissions. The categories are therefore usually sub-divided, according to the characteristics of vehicles and vehicle operation that exert and influence on emission rates. From this process one can obtain a time and spatially resolved emission inventory. This is usually based on laboratory measurements of predetermined driving conditions (Cloke *et al*., 1998).

Traditionally, modelling has concentrated on hot exhaust emissions. More recently procedures to estimate cold start and evaporative emissions have been developed. Hot emission models can be divided into three basic groups of increasing complexity: emission factor model, average speed model, and modal model.

Emission factor model

Emission factor models operate on the simplest level, with a single emission factor used to represent a particular type of vehicle and a particular type of driving condition (e.g. urban, rural, motorway). The emission factors are calculated as mean values of repeated measurements over a given drive cycle, and are usually stated in terms of the mass of pollutant emitted per unit distance.
These factors are useful on a large spatial scale, such as national and regional emissions inventories, where there is little detail on flows and operation. This approach has major disadvantages in terms of predicting emissions on a microscale such as traffic management scheme. Emission factors are based on average driving characteristics and, usually, unrepresentative drive cycles (Cloke *et al*., 1998).

Average speed model

Average speed emission models are at present those most commonly used. Emission rates are measured for a variety of trips, each with a different average speed, and this yields emission functions.

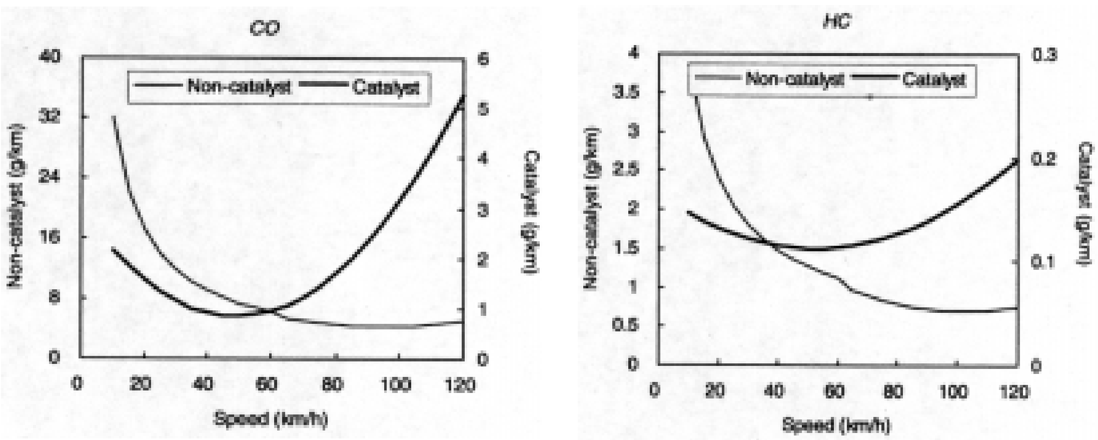


Figure 2.1 Emission factors with respect to average speed
Source: Joumard *et al*. (1995), Influence of instantaneous speed and acceleration on hot passenger car emissions and fuel consumption, SAE technical paper series 950928, pp. 9

This approach is most useful in compiling an inventory of emissions for a road network. Only limited variations in vehicle operation (e.g. changes in speed, cold starts) are accommodated in these types of models and the application to the microscale may not be appropriate (Cloke *et al*., 1998).

Modal emission model based on speed and acceleration

Modal emission models have been designed to provide an estimation technique on the microscale, and in this way complement the more simple models. Modal modelling improves on the average speed approach by relating the modes of vehicle operation encountered on a given trip in terms of the phases of steady speed, acceleration, deceleration, and idling, to the emissions produced during those modes.

The rate of acceleration of a vehicle is a direct measure of the variation in speed (Joumard *et al.*, 1995). For example at a given engine input, a slow moving vehicle will accelerate at a considerably higher rate than a faster one. The demand on an engine, which determines the rate at which pollutants are produced, is given by the product of speed and acceleration as instantaneous values. But speed-acceleration modal emission models cannot take into account yet other important variables such as road gradient. These models are barely more precise than average speed based models and cannot be used to assess the impact of slight changes in the driving pattern, sometimes leading to completely false conclusions (Joumard *et al.*, 1999).

Inverse Dispersion Model

The inverse dispersion model for estimating emission rates was first developed by Palmgren *et al*. (1999). This thesis has used the methodology to estimate emission factors in a street canyon.

Atmospheric pollution dispersion models are usually used for calculation of air quality based on known theoretical relationships between emissions, meteorology and air concentrations. On the other hand, combining model calculations with ambient pollution measurements allows *in-situ* estimations of traffic emissions.
Considering dispersion in street canyons of non-reactive or only slowly reactive vehicle exhaust gases, the chemical transformations can be disregarded, and we can formulate simply the relationships in the following way,

$$C = F(\text{meteorology}) \cdot Q + C_{background} \qquad (2)$$

where C is the concentration of a particular pollutant in the street, Q is the emission of pollutants exhausted from the traffic in the street and F(meteorology) is a function describing dispersion processes. $C_{background}$ is the contribution to pollutant concentrations in the street from all other sources than the traffic in the street.
The dispersion function, F(meteorology) is given by a street pollution model, in this thesis by the Street Canyon Model. The Street Canyon Model describes the dispersion in a street canyon based on meteorological parameters, mainly wind speed and direction above roof tops. In comprehensive tests on measurements from a number of monitoring sites, the Street Canyon Model has been shown to give a satisfactory description of the air pollutant dispersion in urban street canyons.
Eq.(2) can be used for calculations of hourly emissions from traffic, provided that both street and background concentrations are available on an hourly basis.

$$Q_h = \frac{C_h - C_{h,background}}{F_h(\text{meteorology})} \qquad (3)$$

where, the index h refers to a particular hour of the day

For a specific pollutant the hourly total emissions can be calculated by either Eq.(3) or Eq.(1) with the exception of a subscript of vehicle operations.

$$Q_h = \frac{C_h - C_{h,background}}{F_h(\text{meteorology})} = \sum_k N_{k,h} \cdot e_k \qquad (4)$$

Accordingly, we can estimate emission factors through the inverse dispersion model if street and urban background concentrations are measured with traffic counts of each vehicle category. Constructing this equation for each hour of the day forms a system of 24 equations with the emission factors as the unknown variables. Applying multiple regression analysis methods, the emission factors can be determined as a solution of this linear equation system by the best fit in the least square sense. However, this methodology cannot reflect the effects of operation factors such as average speed and instantaneous speed on emission factors.
A condition for the success of the method is that the different vehicle categories have different diurnal variations in the traffic flow. In the case of significant co-variation of the traffic, the system becomes badly conditioned and the solution will be uncertain. And the accuracy of the determination of the emission factors, e_k , depends also on the accuracy of the determination of the total emission for each hour, Q_h (Palmgren *et al*., 1999).

The Street Canyon Model

Basic Properties of Dispersion Processes in a Street Canyon

The main characteristics of the wind flow in a street canyon are well known. The special properties of wind flow and dispersion processes in a street canyon need to be taken into account for estimating emission factors. But traditional Gaussian line source models used for, for example, highways and roads in open areas are not applicable to a street canyon.

When the wind direction is perpendicular to the street direction a vortex is generated in a street canyon, whereby the wind flow at street level is opposite to the flow above roof level. Such wind circulation results in a characteristic dependence of pollution on wind direction in a street canyon. Concentrations on the windward side are much lower than on the leeward side because of the vortex as shown by Figure 3.1. The difference between the leeward and the windward concentrations is most pronounced at higher wind speeds (Berkowicz *et al.*, 1997).

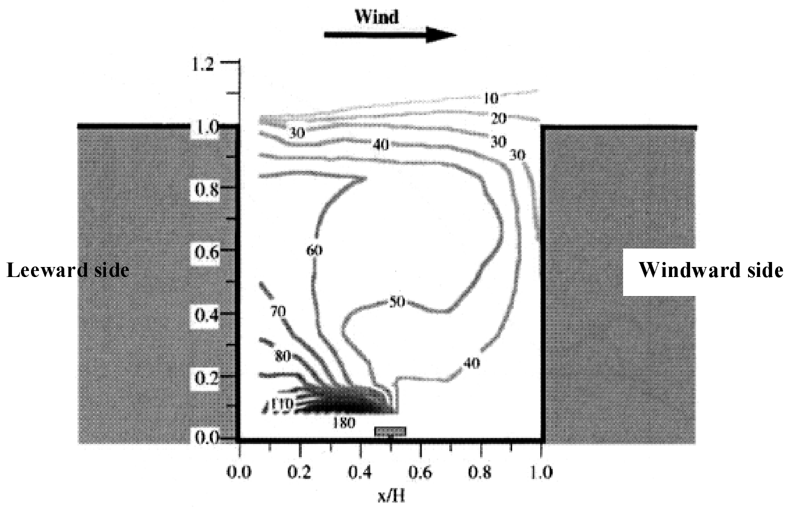


Figure 3.1 Spatial distribution of mean concentration in a street canyon

Source: Pavageau and Schatzmann (1999), Wind tunnel measurements of concentration fluctuations in an urban street canyon, Atmospheric Environment, Vol. 33, pp. 3966

Figure 3.1 shows the spatial distribution of concentrations of a pollutant emitted at the center of road. Interpolated contours depict well the main street vortex and the wrapping of the fluid around an axis parallel to the street direction and located approximately at two thirds of the building height. The parallel iso-concentration lines in the upper part of the canyon suggest that the approaching wind compells the main vortex, in average, to remain confined within the urban canopy (Pavageau and Schatzmann, 1999).

The ratio of the heights of the leeward and windward buildings has an influence on the magnitude of concentrations of both sides. In case of both buildings having the same height, the magnitude of concentrations on leeward side is about a factor of two higher than that on the windward side. This has been observed in a large number of ambient observational studies. In the case where the height of windward side is taller than that of leeward, concentrations of both sides are generally a factor of two lower than those of the even height. In the case where the height of leeward is, conversely, taller than windward, windward concerntrations are slightly higher than those of leeward. This feature is present for wind angles through 0 (i.e., perpendicular to the street) ~ 30 degrees. For wind angles of 50 ~ 90 degrees, the more common situation of higher leeward side concentration is observed. On the other hand, concentration magnitudes for the rectangular street are generally a factor of two higher than for the square street at which the height of buildings is the same length as the width of road (Hoydysh and Dabberdt, 1988).

Model Description

The Street Canyon Model is a small-scale empirical model that allows to simulate the street level concentrations on a single street that has a row of buildings on each side. The model takes into account the most essential features of pollution dispersion in street canyons.

The expressions of the Street Canyon Model have been tested and modified in a study in Scandinavia including the Stockholm (SNV, 1977). Expressions for CL (concentration of leeward side) and CW (concentration of windward side) are as follows:

$$CL = C_b + \frac{K \cdot Q}{(u + u_0) \cdot [(x^2 + z^2)^{1/2} + L_0]} \qquad (5)$$

$$CW = C_b + \frac{K \cdot Q \cdot (H - z)}{W \cdot (u + u_0) \cdot H} \qquad (6)$$

where, C_b the above-canyon background concentration in g/m^3

K is an empirical constant, set to 10
Q is the traffic emission in g/m,s
u is the wind component perpendicular to the street axis in m/s
u_0 is a minimal dilution parameter, set to 0.5 m/s
L_0 is an initial pollutant mixing length, set to 2 m
W is the width between the buildings in m
H is the typical building height in m
x, z are horizontal and vertical distances from street emission segments in m

The vertical section between the buildings is divided into a grid, on which the concentrations are evaluated. The road, i.e. the part of the road width (RW in Figure 3.2) where the emission takes place, is divided into a large number of road segments with equal emission rate.

The wind component is taken from the wind field calculation and evaluated at the street location. If the wind is blowing within 22.5 degrees from the principal street direction, the concentration of each side will be calculated as an average of CL and CW, being symmetric. For cases where a wind direction is more perpendicular to the street direction, an asymmetry with higher concentrations on the side of the street from which the roof wind blows (leeward side) and lower values on the opposite side (windward side) is found as discussed in previous chapter. The concentrations on windward side of the emitting road are calculated according to CW. At the other gridpoints, a summation of CL values takes place, implying a contribution from each emitting segment to the gridpoint situated on leeward side, in order to calculate the concentrations on leeward.

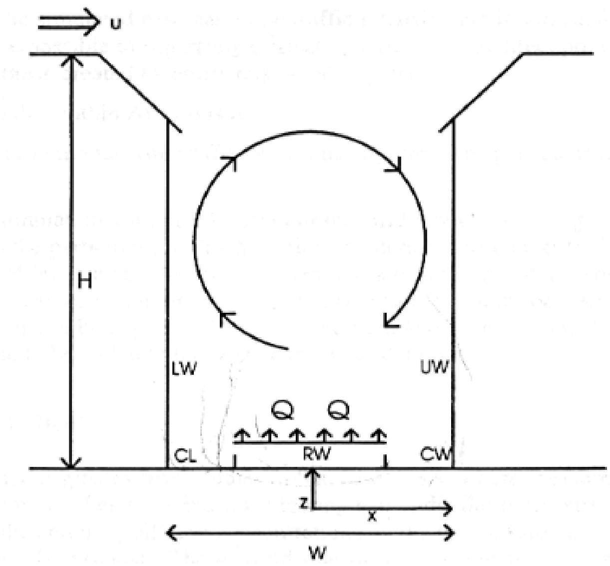


Figure 3.2 Vortex and concentration measurement in a street canyon

Sensitivity Analysis

Sensitivity analysis is a fundamental part of the evaluation of a model because it identifies its critical inputs and allows the evaluator to determine if nature displays the same sensitivity to these inputs as the model (Ermak and Merry, 1988; quoted in Bellasio, 1997). This thesis carried out sensitivity analysis with respect to some parameters and variables in order to find out the potential uncertainty of the Street Canyon Model.

Wind direction

If the wind blows within 22.5 degrees from the principal street direction, concentrations of CL and CW should be symmetric as explained in model description. It means that within 22.5 degrees from the street direction the vortex flow is non-existent or more sporadic existent (Yamartino and Wiegand, 1986).

The shadow areas in Figure 3.3 show absolute values of difference between CW and CL in Honsgatan street where NOx was measured in 1999 within 67.5 ~ 112.5 degrees and 247.5 ~ 292.5 degrees, at which the vortex is not created in theory. However, CL and CW are not the same in both areas. Only in 67.5 ~ 90 degrees two concentrations look to be a little symmetric. The poor representation of wind direction dependence is therefore a problem of the Street Canyon Model.

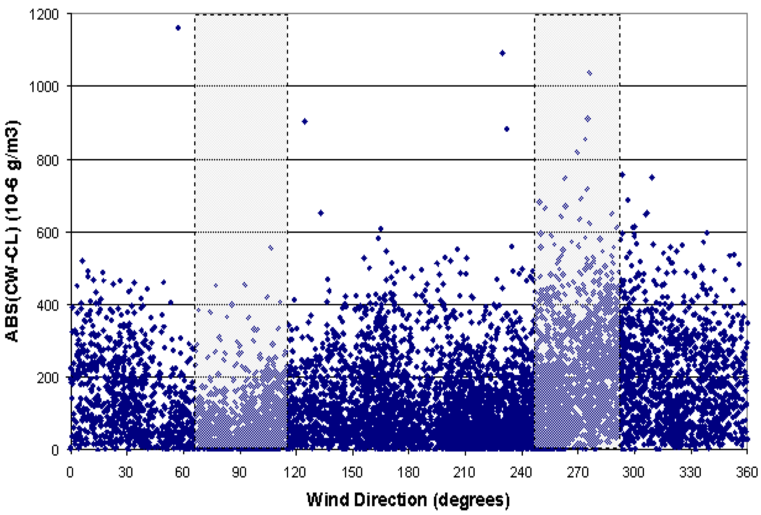


Figure 3.3 Absolute differences between *CW* and *CL* for NOx in Hornsgatan, 1999

Traffic emissions (Q_{CL} and Q_{CW})

Traffic emissions are derived from both Eq.(5) and Eq.(6).

$$Q_{CL} = \frac{(CL - C_b)\cdot(u + u_0)\cdot[(x^2 + z^2)^{1/2} + L_0]}{K} \qquad (7)$$

$$Q_{CW} = \frac{(CW - C_b)\cdot W \cdot (u + u_0)\cdot H}{K \cdot (H - z)} \qquad (8)$$

where, Q_C is the traffic emissions on leeward side in g/m,s

Q_{CW} is the traffic emissions on windward side

Theoretically, two values of Q_C and Q_{CW} within 22.5 degrees from the street direction should be the same because traffic emissions are released from the single emitter, vehicles, on the middle of the road although concentrations of leeward and windward side could be slightly different. However, two values of traffic emission are always not the same.

Figure 3.4 explains the relationships of Q_C and Q_{CW} as all variables except C_b and CW are constant. Only on the diagonal line two values are the same. In the areas above the diagonal line Q_C is higher than Q_{CW} and *vice versa*. It implies that under only limited conditions the equations for dispersion processes in the Street Canyon Model can be approved mathematically.

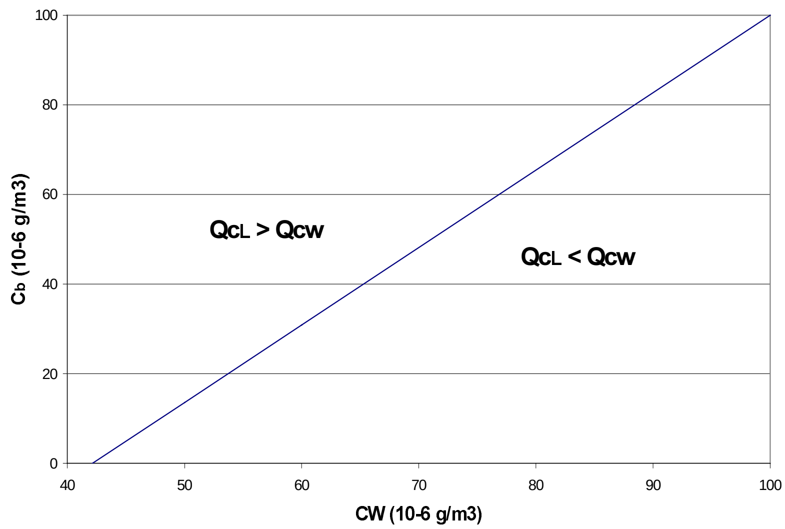


Figure 3.4 Relationships of Q_C and Q_{CW} for NOx as CL=100, u=2

Figure 3.5 explains graphically the difference between Q_C and Q_{CW} under the same conditions as Figure 3.4. The farther from the diagonal line, the larger uniformly the difference.

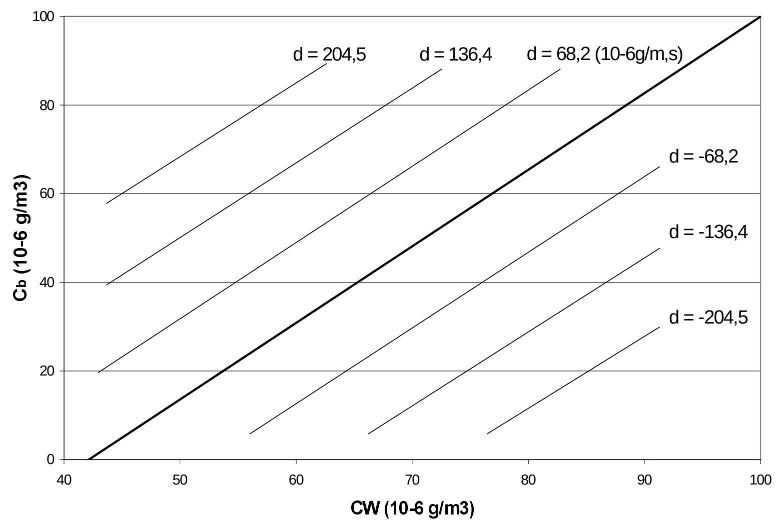


Figure 3.5 Differences between Q_C and Q_{CW}, $d = Q_{CL} - Q_{CW}$

L_0 factor

L_0 is normally set to 2m for an initial pollutant mixing length or a length of the individual car. However, It is very difficult to find out the reason why it is 2m or it is needed.

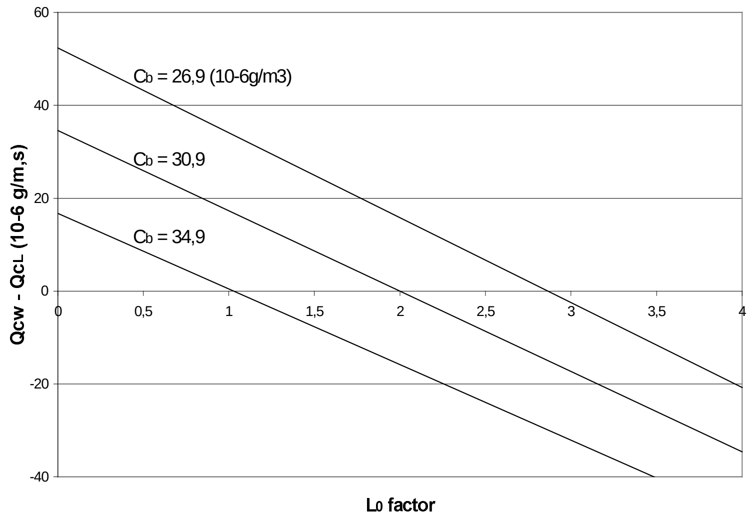
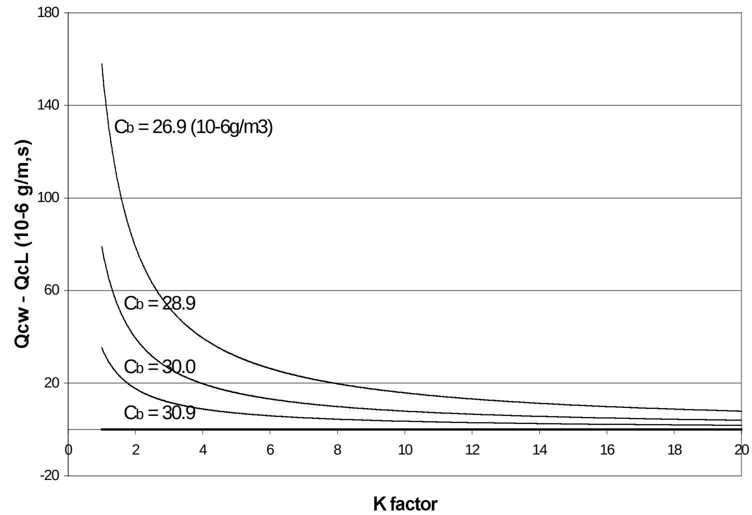


Figure 3.6 Differences with repect to L_0 as *CL*=100, *CW*=60, *u*=2

It seems to be a complementary constant to make the difference between Q_{CL} and Q_{CW} to be zero. The diagonal center line of Figure 3.6 is drawn by one condition (i.e. *CL*=100, *CW*=60, C_b =30.9, *u*=2) of cases in which the diagonal line of Figure 3.4 is made, in short, the difference is to be zero. The difference is zero at only 2 values of L_0 under the condition. Other conditions (i.e. C_b =26.9 and 34.9) of Figure 3.6 cannot make the difference zero and for the purpose of it the value of L_0 should be set to about 1 or 3. In the result, the Street Canyon Model can make the difference zero only when L_0 is set to 2.



K factor

Figure 3.7 Differences with repect to *K* as *CL*=100, *CW*=60, *u*=2

The Street Canyon Model has a main problem with K factor. The lack of a sound theoretical basis for the value of K has inhibited the model's acceptance and transportability to other canyon geometries (Yamartino and Wiegand, 1986). As one can see in Figure 3.7, the difference between Q_{CL} and Q_{CW} is zero when the value of K is infinite or conditions are the same as the diagonal line in Figure 3.4. Although for K factor this thesis sets 10 and other studies like San Jose study set 7 based on empirical experiments, there is the limitation that the difference is never to be zero.

The complexity of the wind flow within the urban street canyons prevents the development of the model capable of simulating well the unique dispersion processes. The Street Canyon Model is a simpler model compared with recently developed models such as the Canyon Plume Box Model (Yamartino and Wiegand, 1986) and the Operational Street Pollution Model (Hertel and Berkowicz, 1989). The Street Canyon Model describes merely the most essential features of pollutant dispersion, so the Street Canyon Model involves some uncertainties in estimating emission factors as well as pollutant concentrations as examined in sensitivity analysis.

Emission Factors from Street canyons

Street Measurement

Air pollution has been monitored in Stockholm since the middle of the 1960's. Originally SO_2 and black smoke were studied. Today there is a network of monitoring stations that include measurements of e.g. NOx, CO, SO_2, PAH and a number of meteorological parameters. The monitoring network includes both roof level and street level measurements and several different methods are employed. They provide hourly data of a number of parameters. In addition to the monitoring stations mentioned above there are several stations outside the city of Stockholm. These include air pollutants and meteorological parameters and provide important information on the background levels and input to air quality dispersion models (Luftvårdsförbundet, 1997; quoted in Johansson *et al*, 1999).

Figure 4.1 Air pollution monitoring sites in Stockholm

The two streets at which pollutant concentrations and traffic counts have been measured simultaneously in Stockholm are the Hornsgatan and the Sveavägen. In Hornsgatan NOx, CO and PM were measured hourly at both roof level and street level. This thesis has used concentration data of NOx and CO from 1994 to 1999 and PM in 1999 for estimating emission factors respectively. In Sveavägen NOx and CO were measured hourly at roof and street level. Concentration data of both pollutants from 1995 to 1999 are used. Automatic hourly traffic counts with respect to three vehicle categories were also performed at two streets.

Table 4.1 Street data

Street	Width/height (m)	Orientation (degree)	Pollutants	Remarks
Hornsgatan	24/25	90	NOx,CO,PM	Measurements on both sides & roof
Sveavägen	33/25	0	NOx,CO	Measurements on both sides & roof

Reference: orientation of the streets is given with respect to North

An additional monitoring site for PM is situated in the Rosenlundsgatan close to the Hornsgatan. Measurements from this site serve as estimates of urban background concentrations of $PM_{2.5}$ and PM_{10}. Meteorological measurements such as wind speed, wind direction, temperature and relative humidity were performed at the station placed in the Högdalen district, southern part of Stockholm.

To calculate emission factors of a specific pollutant with respect to light-duty vehicle (LDV, length between wheelaxles ≤ 5.5m) and heavy-duty vehicle (HDV), Eq.(4) is divided by total traffic volumes of each hour and Eq.(9) is derived as follows:

$$\begin{aligned}\frac{Q_h}{N_{tot,h}} &= \frac{N_{LDV,h}}{N_{tot,h}} \cdot e_{LDV} + \frac{N_{HDV,h}}{N_{tot,h}} \cdot e_{HDV} \\ &= \frac{N_{LDV,h}}{N_{tot,h}} \cdot e_{LDV} + \frac{N_{tot,h} - N_{LDV,h}}{N_{tot,h}} \cdot e_{HDV} \\ &= \frac{N_{LDV,h}}{N_{tot,h}} \cdot (e_{LDV} - e_{HDV}) + e_{HDV} \end{aligned} \tag{9}$$

where, Q_h is total emission rates for a pollutant at a specific hour in g/m,s

$N_{tot,h}$ is total traffic volumes in veh/h

$N_{LDV,h}$ is traffic volumes of light-duty vehicle in veh/h

$N_{HDV,h}$ is traffic volumes of heavy-duty vehicle in veh/h

e_{LDV} is emission factor with respect to light-duty vehicle in g/km

e_{HDV} is emission factor with respect to heavy-duty vehicle in g/km

Emission Factors for NOx

Hornsgatan

In 1994

In 1994 concentrations of NOx were measured in only Hornsgatan. Average annual traffic volumes of both directions at every hour in each day of the week are displayed in Figure 4.2. The peak hours are 8~9 hours and 16~17 hours. In afternoon peak hours traffic volumes of both directions are about 3000 veh/h.

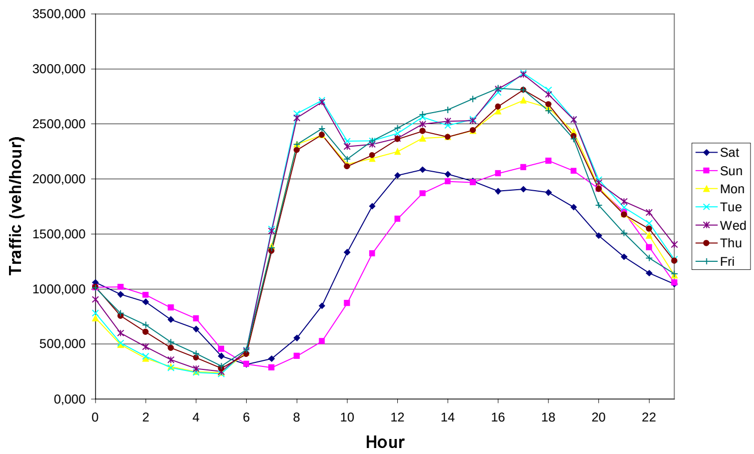


Figure 4.2 Diurnal variation of traffic volumes at Hornsgatan in 1994

Figure 4.3 shows average annual fractions of light-duty vehicle in both directions. The fraction of heavy-duty vehicle is the highest at 5 o'clock and gradually decreasing to 4% even though the fractions at 7~9 hours are relatively lower compared with immediately before and after hours.

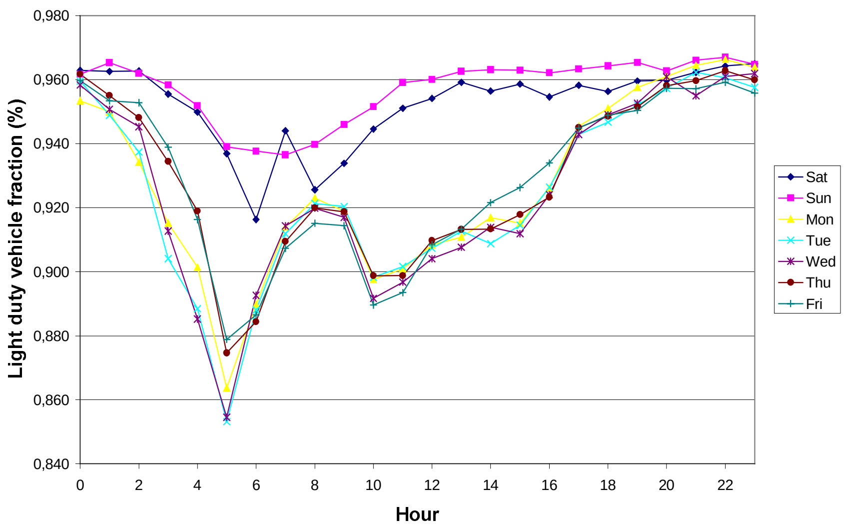


Figure 4.3 Diurnal variation of light-duty vehicle fractions at Hornsgatan in 1994

Table 4.2 Average daily traffic volumes at Hornsgatan in 1994

	Total(veh/day)	LDV(veh/day)	HDV(veh/day)	Fraction of LDV
1994	38 176	35 701	2 474	0.93

Figure 4.4 shows average annual emission rates of NOx. The trends of NOx are similar to the trends of traffic volumes in Figure 4.2. NOx emission rates in working hours are held approximately 1100 μg/m,s.

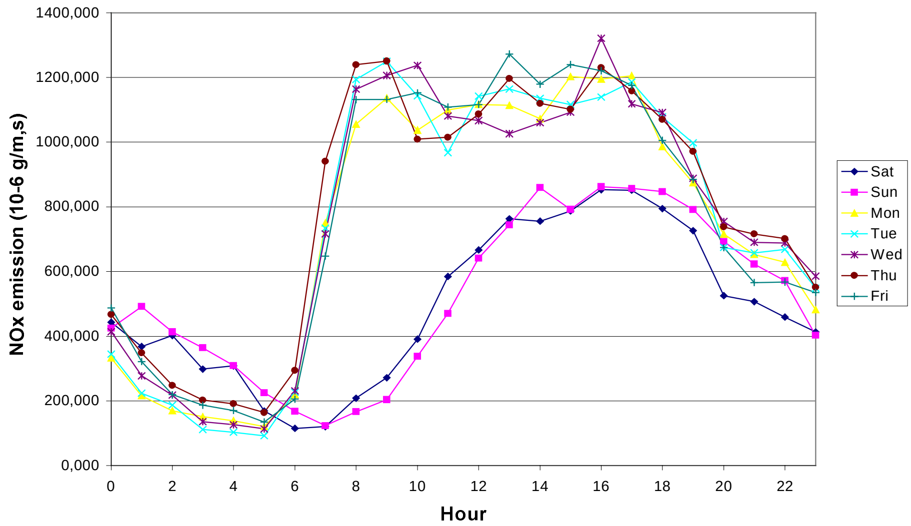


Figure 4.4 Diurnal variation of emission rates of NOx at Hornsgatan in 1994

Applying the linear regression method to Eq.(9), emission factors of NOx in 1994 are calculated as shown in Table 4.3.

Table 4.3 Estimated emission factors for NOx at Hornsgatan in 1994

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	5.66	0.48	11.69	0.0001
LDV	1.28	0.04	35.08	0.0001

In 1995

The trends of average annual traffic volumes in 1995 are very similar to those in 1994. The trends of light-duty vehicle fractions are also similar to those in 1994 as shown in Figure 4.5. Average fraction of light-duty vehicle in 1995 is 0.92 and 1% lower than that of light-duty vehicle in 1994.

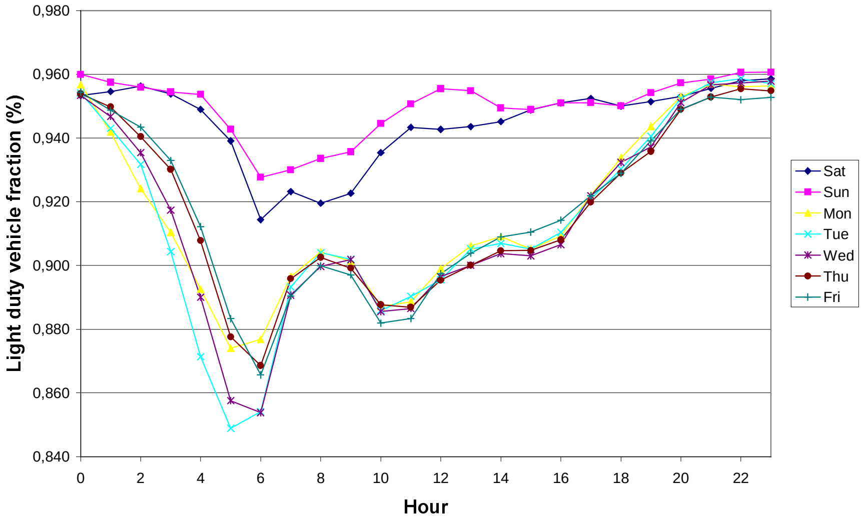


Figure 4.5 Diurnal variation of light-duty vehicle fractions at Hornsgatan in 1995

Table 4.4 Average daily traffic volumes at Hornsgatan in 1995

	Total(veh/day)	LDV(veh/day)	HDV(veh/day)	Fraction of LDV
1995	35 111	32 443	2 668	0.92

Average daily traffic volume of heavy-duty vehicle is increased and that of light-duty vehicle is decreased compared with average daily traffic volumes in 1994. In accordance with, emission factor of NOx with respect to heavy-duty vehicle in 1995 is increased, 9.84 g/km, and for light-duty vehicle is decreased, 0.78 g/km.

Table 4.5 Estimated emission factors for NOx at Hornsgatan in 1995

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	9.84	0.50	19.87	0.0001
LDV	0.78	0.04	18.26	0.0001

In 1996

The trends of average annual traffic volumes in 1996 are also very similar to those in 1994 and 1995. However, average fraction of light-duty vehicle in 1996 is 1% lower than that of light-duty vehicle in 1995 and 2% lower than in 1994 as shown in Table 4.6.

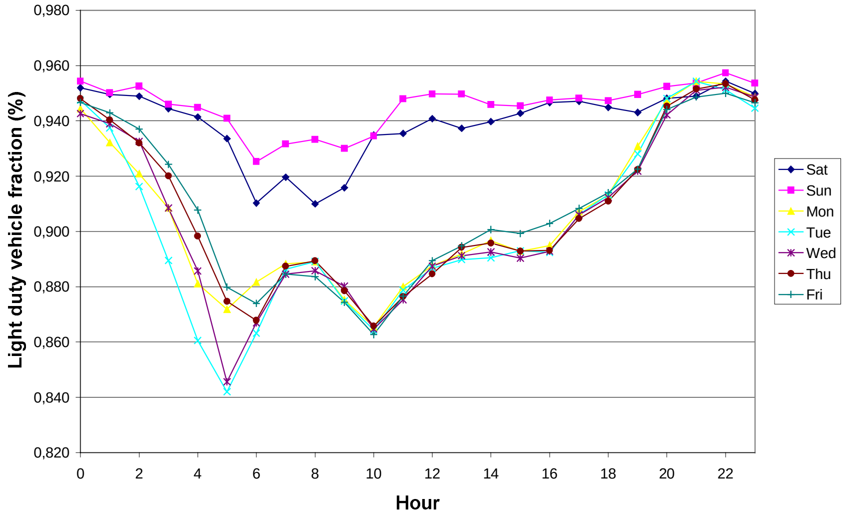


Figure 4.6 Diurnal variation of light-duty vehicle fractions at Hornsgatan in 1996

Table 4.6 Average daily traffic volumes at Hornsgatan in 1996

	Total(veh/day)	LDV(veh/day)	HDV(veh/day)	Fraction of LDV
1996	34 946	31 905	3 041	0.91

Emission factor of NOx with respect to heavy-duty vehicle is 6.08 g/km and emission factor for light-duty vehicle is 1.08 g/km.

Table 4.7 Estimated emission factors for NOx at Hornsgatan in 1996

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	6.08	0.37	16.23	0.0001
LDV	1.08	0.04	29.59	0.0001

In 1997

The trends of average annual traffic volumes in 1997 are not changed greatly compared with those in 1994, 1995 and 1996. Average fraction of light-duty vehicle in 1997 is increased again and 1% higher than that in 1996.

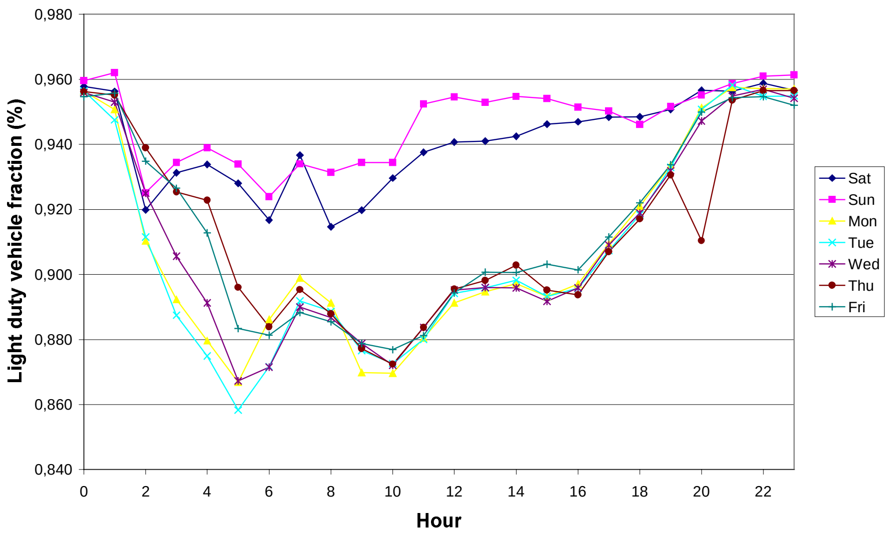


Figure 4.7 Diurnal variation of light-duty vehicle fractions at Hornsgatan in 1997

Table 4.8 Average daily traffic volumes at Hornsgatan in 1997

	Total(veh/day)	LDV(veh/day)	HDV(veh/day)	Fraction of LDV
1997	37 965	34 832	3 133	0.92

Emission factors of NOx with respect to light- and heavy-duty vehicle are 0.93 g/km and 5.95 g/km respectively. Two values are decreased compared with those in 1996 although all average traffic volumes are increased in Table 4.8.

Table 4.9 Estimated emission factors for NOx at Hornsgatan in 1997

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	5.95	0.34	17.32	0.0001
LDV	0.93	0.03	29.22	0.0001

In 1998

Average daily traffic volumes in 1998 are decreased compared with those in 1997. But average fraction of light-duty vehicle, 0.93, is increased.

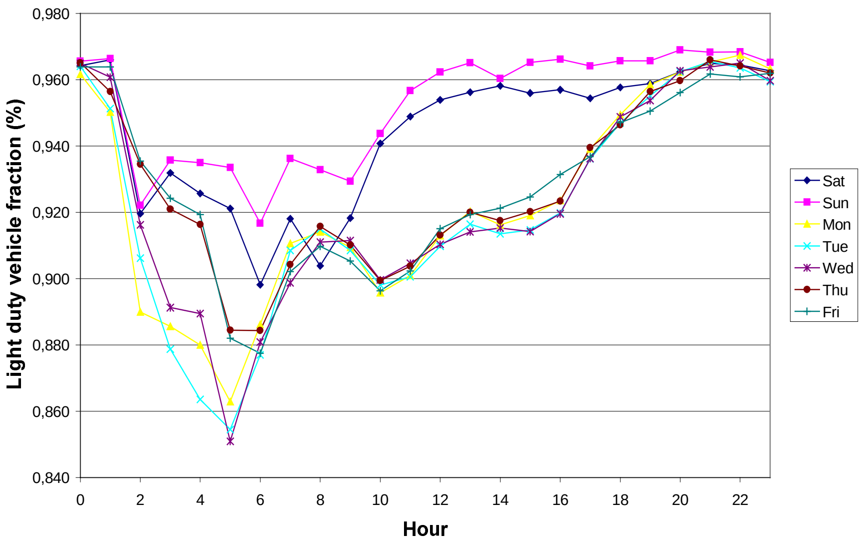


Figure 4.8 Diurnal variation of light-duty vehicle fractions at Hornsgatan in 1998

Table 4.10 Average daily traffic volumes at Hornsgatan in 1998

	Total(veh/day)	LDV(veh/day)	HDV(veh/day)	Fraction of LDV
1998	34 769	32 481	2 288	0.93

Emission factor of NOx with respect to light-duty vehicle is 0.90 g/km and lower than that in 1997. And emission factor with respect to heavy-duty vehicle is 7.10 g/km and higher than that in 1997

Table 4.11 Estimated emission factors for NOx at Hornsgatan in 1998

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	7.10	0.38	18.87	0.0001
LDV	0.90	0.03	29.99	0.0001

In 1999

Although average daily total traffic volumes in 1999 are increased, average fraction of light-duty vehicle is 0.91 and lower than that in 1998.

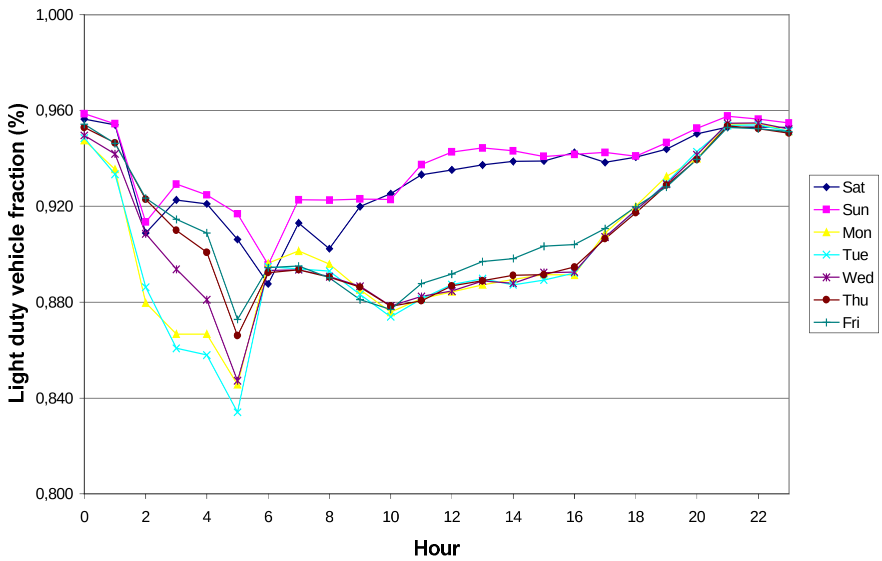


Figure 4.9 Diurnal variation of light-duty vehicle fractions at Hornsgatan in 1999

Table 4.12 Average daily traffic volumes at Hornsgatan in 1999

	Total(veh/day)	LDV(veh/day)	HDV(veh/day)	Fraction of LDV
1999	36 993	33 819	3 174	0.91

Emission factor of NOx with respect to light-duty vehicle is 0.72 g/km. And emission factor with respect to heavy-duty vehicle is 8.32 g/km.

Table 4.13 Estimated emission factors for NOx at Hornsgatan in 1999

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	8.32	0.40	20.85	0.0001
LDV	0.72	0.04	18.31	0.0001

Annual trend

Figure 4.10 shows the annual trends of average daily traffic volumes in Hornsgatan. The trend of average daily volumes of heavy-duty vehicle is changed greatly but has a tendency to be increased generally. On the contrary, the trend of average daily volumes of total vehicles and light-duty vehicle is a little decreased.

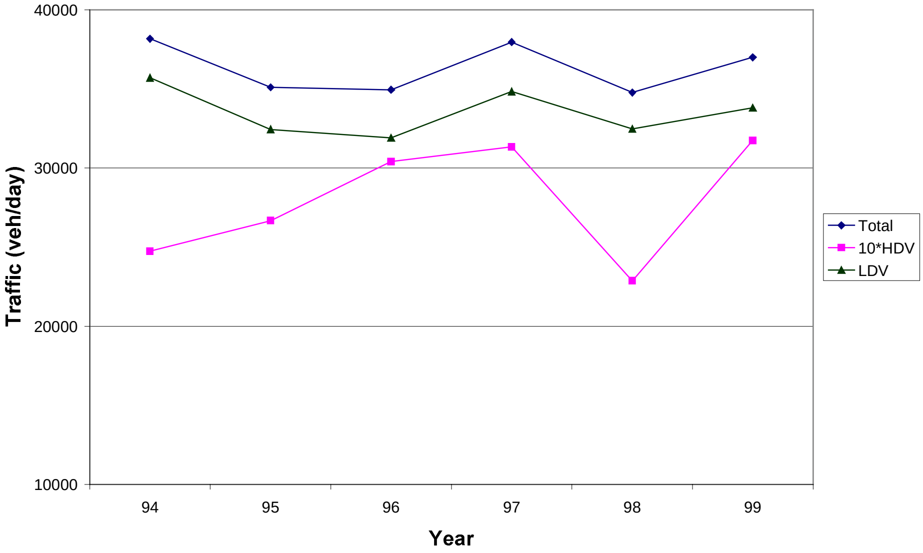


Figure 4.10 The annual trends of average daily traffic volumes in Hornsgatan

The trend of emission factors with respect to light-duty vehicle is steady and a little decreased. But it is difficult to find out the tendecty of trend with respect to heavy-duty vehicle because it is fluctuated greatly.

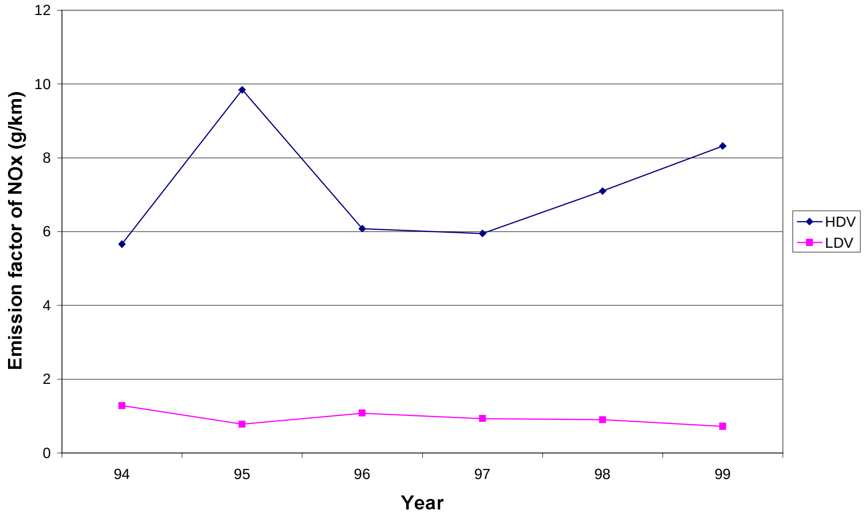


Figure 4.11 The annual trends of emission factors of NOx in Hornsgatan

Sveavägen

In 1995

Average annual traffic volumes at every hour in each day of the week are displayed in Figure 4.12. The peak hours in Sveavägen are the same hours, 8~9 hours and 16~17 hours, as in Hornsgatan. However, the maximum traffic volume in peak hours and the light-duty vehicle fraction are lower than those in Hornsgatan.

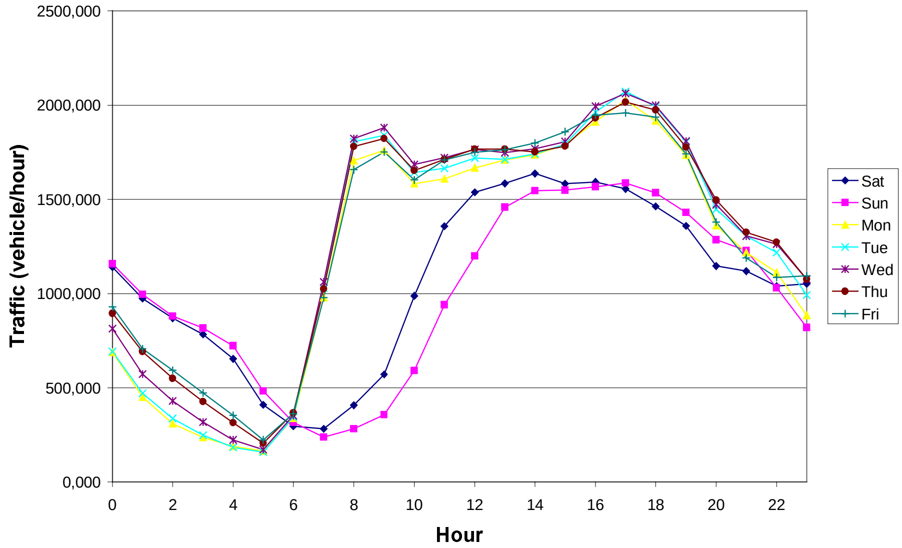


Figure 4.12 Diurnal vairation of traffic volumes at Sveavägen in 1995

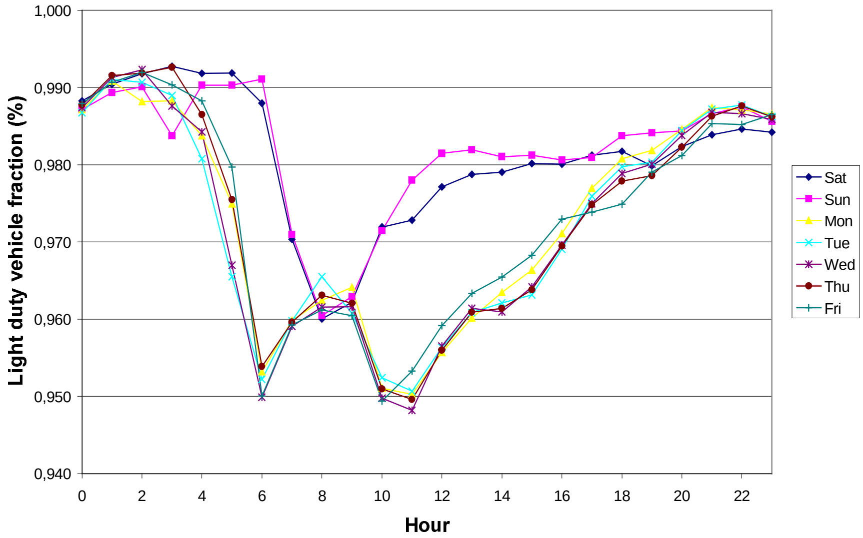


Figure 4.13 Diurnal variation of light-duty vehicle fractions at Sveavägen in 1995

Figure 4.13 shows average fractions of light-duty vehicle. The trends are greatly different from those in Hornsgatan. The fraction of heavy-duty vehicle is the highest at 11 o'clock. The fraction of heavy-duty vehicle at 8 o'clock in weekend is higher than that in weekdays.

Table 4.14 Average daily traffic volumes at Sveavägen in 1995

	Total(veh/day)	LDV(veh/day)	HDV(veh/day)	Fraction of LDV
1995	28 864	28 084	780	0.97

Figure 4.14 shows average annual emission rates of NOx. Emission rates in 12~14 hours are the highest unlike Hornsgatan and maximum rates are lower than those in Hornsgatan.

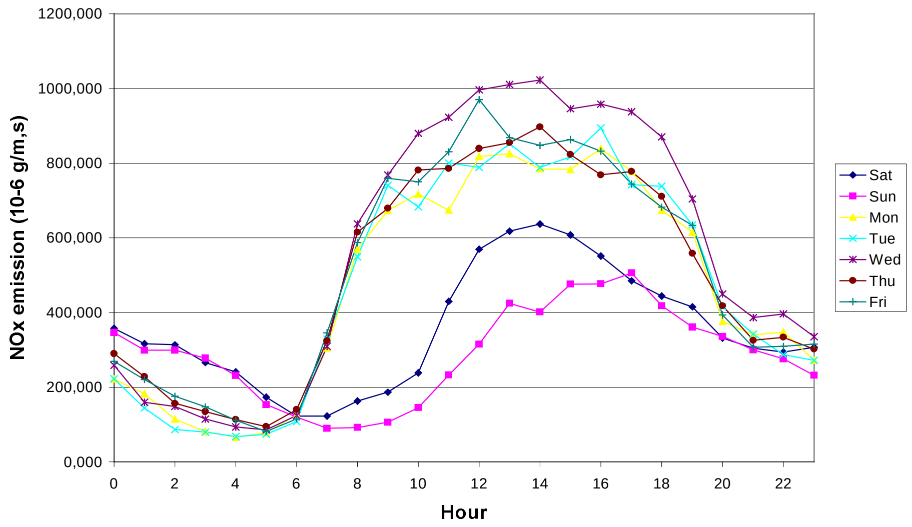


Figure 4.14 Diurnal variation of emission rates of NOx at Sveavägen in 1995

Emission factors with respect to light- and heavy-duty vehicle are 0.97 g/km and 14.50 g/km respectively.

Table 4.15 Estimated emission factors for NOx at Sveavägen in 1995

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	14.50	1.31	11.09	0.0001
LDV	0.97	0.04	25.89	0.0001

In 1996

The trends of average annual traffic volumes and light-duty vehicle fractions are similar to those in 1995. Average daily traffic volumes are also slightly different as shown in Table 4.16. Figure 4.15 shows average annual emission rates. The trends are a little decreased compared with those in Figure 4.14.

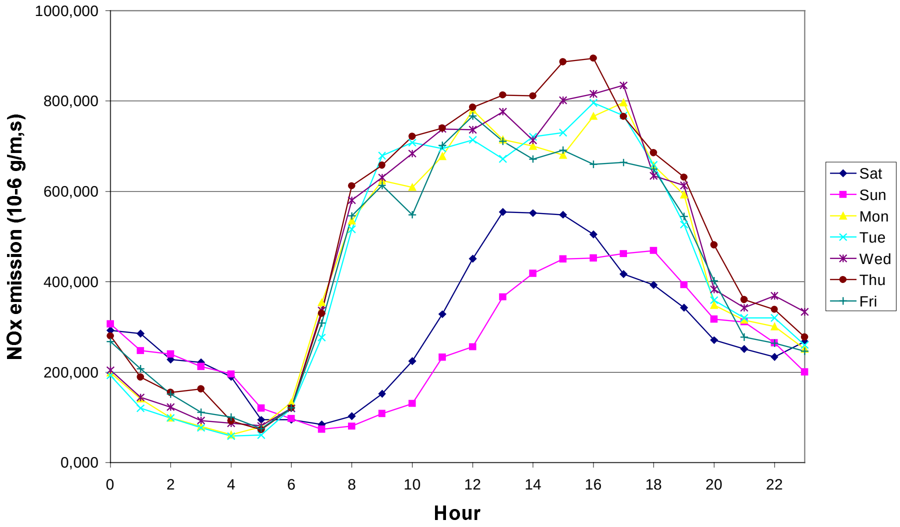


Figure 4.15 Diurnal variation of emission rates of NOx at Sveavägen in 1996

Table 4.16 Average daily traffic volumes at Sveavägen in 1996

	Total(veh/day)	LDV(veh/day)	HDV(veh/day)	Fraction of LDV
1996	28 815	28 058	757	0.97

Emission factors of NOx with respect to light- and heavy-duty vehicle are also not greatly different but slightly decreased compared with those in 1995.

Table 4.17 Estimated emission factors for NOx at Sveavägen in 1996

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	13.37	0.99	13.51	0.0001
LDV	0.86	0.03	31.29	0.0001

In 1997

Average daily traffic volumes in 1997 are slightly different and the fraction of light-duty vehicle is constant, 0.97, compared with previous years.

Table 4.18 Average daily traffic volumes at Sveavägen in 1997

	Total(veh/day)	LDV(veh/day)	HDV(veh/day)	Fraction of LDV
1997	29 352	28 526	826	0.97

Emission factor of NOx with respect to light-duty vehicle is 0.87 g/km and with respect to heavy-duty vehicle is 11.35 g/km.

Table 4.19 Estimated emission factors for NOx at Sveavägen in 1997

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	11.35	1.16	9.77	0.0001
LDV	0.87	0.03	25.52	0.0001

In 1998

All trends of traffic in 1998 are also very similar to those in 1995, 1996 and 1997. However, emission factors are different. With respect to light- and heavy-duty vehicle emission factors in 1998 are 0.90 g/km and 11.55 g/km, respectively.

Table 4.20 Average daily traffic volumes at Sveavägen in 1998

	Total(veh/day)	LDV(veh/day)	HDV(veh/day)	Fraction of LDV
1998	29 162	28 313	848	0.97

Table 4.21 Estimated emission factors for NOx at Sveavägen in 1998

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	11.55	1.20	9.59	0.0001
LDV	0.90	0.04	24.77	0.0001

In 1999

Although all trends of traffic data in 1999 are still similar to those in previous years, emission factors in 1999 are different from previous years.

Table 4.22 Average daily traffic volumes at Sveavägen in 1999

	Total(veh/day)	LDV(veh/day)	HDV(veh/day)	Fraction of LDV
1999	29 482	28 603	879	0.97

Table 4.23 Estimated emission factors for NOx at Sveavägen in 1999

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	10.57	1.22	8.64	0.0001
LDV	0.96	0.04	24.99	0.0001

Annual trend

The annual trends of average daily traffic volumes in Sveavägen are settled and a little increased unlike in Hornsgatan.

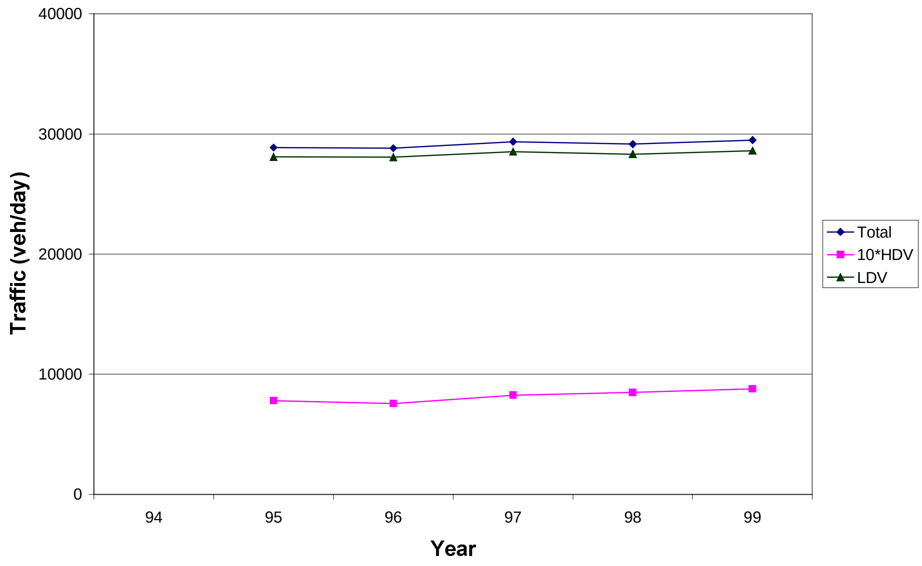


Figure 4.16 The annual trends of average daily traffic volumes in Sveavägen

Since all trends of traffic data are steady in Sveavägen we can analyse the variation of emission factors caused by other factors. The trend of emission factors for heavy-duty vehicle is decreased continuously, which is mainly caused by the increased rates of vehicles equipped catalyst converter. But the trend for light-duty vehicle is steady regardless of the catalyst vehicle rates.

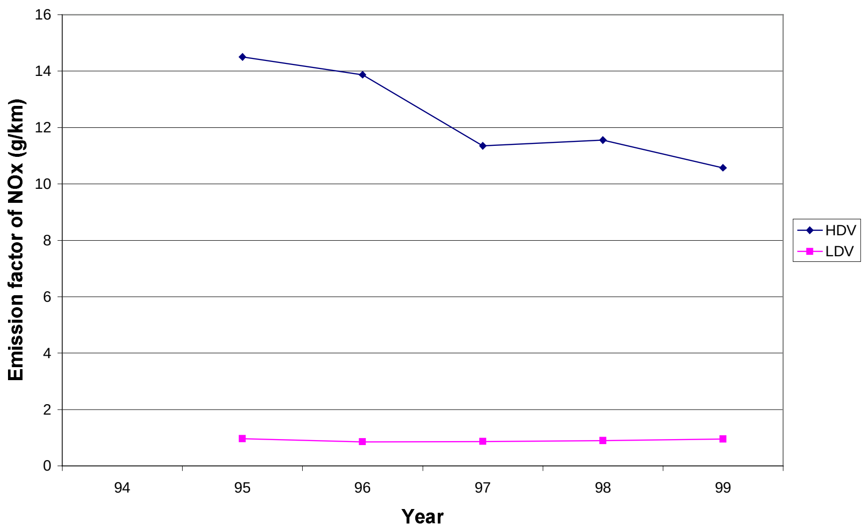


Figure 4.17 The annual trends of emission factors of NOx in Sveavägen

Emission Factors for CO

Hornsgatan

In 1994

All traffic data at Hornsgatan in 1994 are diplayed in Figure 4.2 and Table 4.2. Figure 4.18 shows average annual emission rates of CO. The trends in 7~17 hours at week days are similar to trends of light-duty vehicle fractions in Figure 4.3. It implys that CO emission rates are correlated with light-duty vehicle volumes.

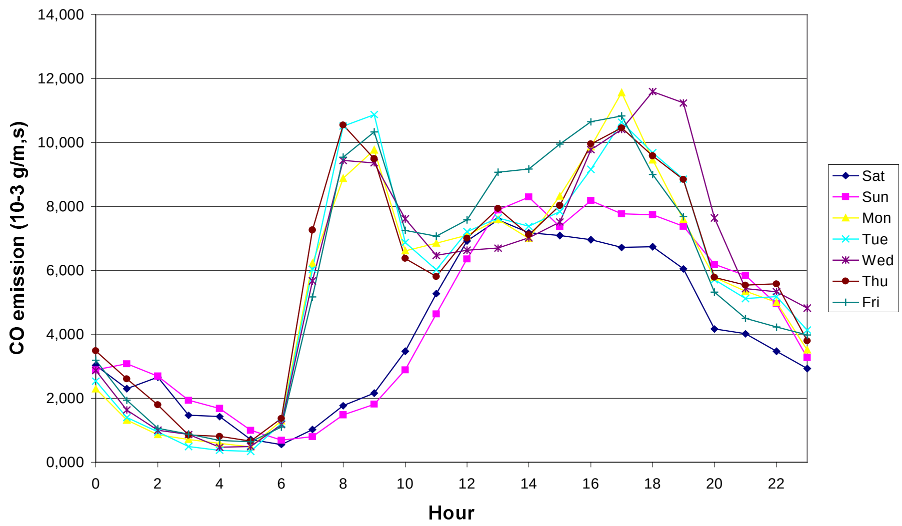


Figure 4.18 Diurnal variation of emission rates of CO at Hornsgatan in 1994

By using Eq.(9) emission factor of CO for light-duty vehicle is calculated, 12.78 g/km. However, emission factor for heavy-duty vehicle has a problem that t-value is not statistically significant in 95% confidence. And R^2 value is very low, 0.07.

Table 4.24 Estimated emission factors for CO at Hornsgatan in 1994

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	-10.28	6.22	-1.65	0.1004
LDV	12.78	0.47	27.17	0.0001

In 1995

Average annual emission rates in Figure 4.19 are slightly different and lower than those in 1994. It seems to be caused by the reduction of average daily traffic volumes as shown in Table 4.4. Emission factors with respect to light- and heavy-duty vehicle are 8.63 g/km and 38.17 g/km, respectively. These values are statistically significant but R^2, 0.07, is very low.

In Table 4.25 emission factor with respect to heavy-duty vehicle is higher than that with respect to light. It is contray to results of other studies (Palmgren *et al*, 1999, John *et al*, 1999) that emission factor of CO with respect to light-duty vehicle is generally higher than that with respect to heavy.

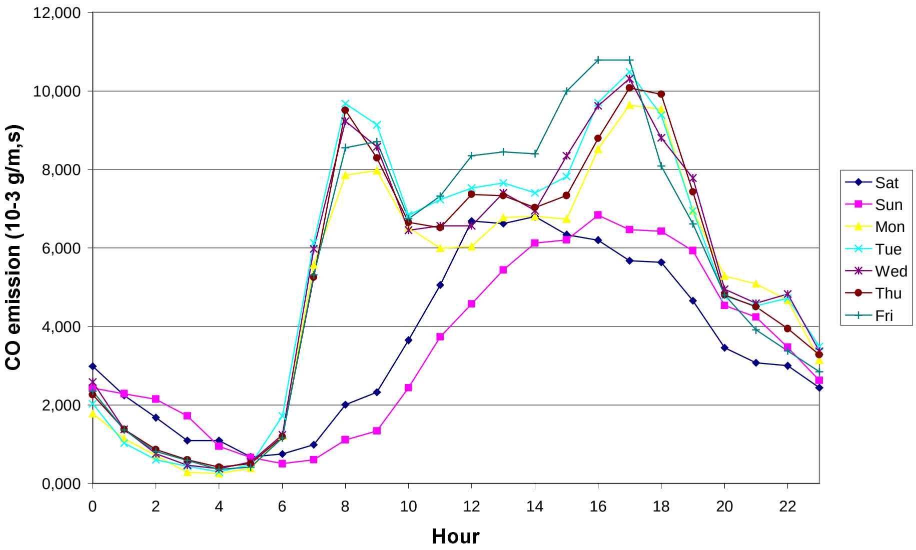


Figure 4.19 Diurnal variation of emission rates of CO at Hornsgatan in 1995

Table 4.25 Estimated emission factors for CO at Hornsgatan in 1995

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	38.17	7.53	5.07	0.0001
LDV	8.63	0.65	13.21	0.0001

In 1996

The trends of average annual emission rates are similar to those in 1995. Emission factors with respect to light- and heavy-duty vehicle are 10.29 g/km and 17.49 g/km respectively. They are statistically significant in t-test but R^2, 0.01, is very low. And emission factor with respect to heavy-duty vehicle is still higher than that with respect to light-duty vehicle.

Table 4.26 Estimated emission factors for CO at Hornsgatan in 1996

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	17.49	5.35	3.27	0.0013
LDV	10.29	0.52	19.80	0.0001

In 1997

The trends of average annual emission rates are also similar to those in 1995. Emission factors with respect to light- and heavy-duty vehicle are 8.17 g/km and 18.29 g/km respectively. But R^2, 0.02, is very low. And emission factor with respect to heavy-duty vehicle is still higher.

Table 4.27 Estimated emission factors for CO at Hornsgatan in 1997

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	18.29	5.37	3.41	0.0008
LDV	8.17	0.50	16.41	0.0001

In 1998

The trends of average annual emission rates in 1998 are different and totally reduced compared with previous years. Emission factor with respect to light-duty vehicle is 8.71 g/km. This value is no problem in t-test and identical with values in previous years. However, emission factor with respect to heavy-duty vehicle is 4.30 g/km and not statistically significant in 95% confidence. Moreover, R^2 value, 0.004, is lower than 0.01.

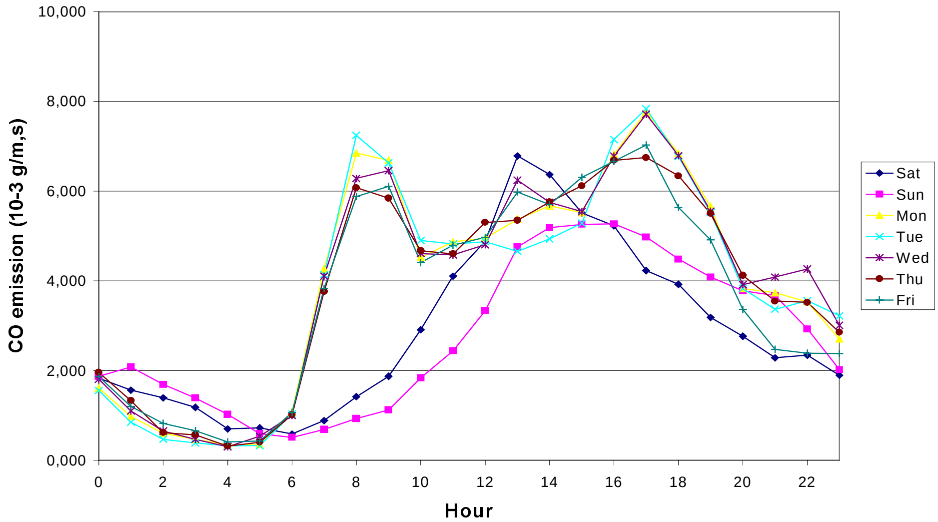


Figure 4.20 Diurnal variation of emission rates of CO at Hornsgatan in 1998

Table 4.28 Estimated emission factors for CO at Hornsgatan in 1998

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	4.30	4.84	0.89	0.3753
LDV	8.71	0.39	22.47	0.0001

In 1999

Emission factors with respect to light- and heavy-duty vehicle are 7.54 g/km and 9.07 g/km, respectively. These values are no problem in t-test. However, R^2 value, 0.0007, is too much low.

Table 4.29 Estimated emission factors for CO at Hornsgatan in 1999

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	9.07	3.96	2.29	0.0233
LDV	7.54	0.39	19.33	0.0001

Annual trend

The trends of emission factors with respect to light- and heavy-duty vehicle are decreased even although R^2 is very low every year and some values have problems in t-test. In particular, the whole trend with heavy-duty vehicle is contrary to the trend of average daily volumes of heavy-duty vehicle which is increased.

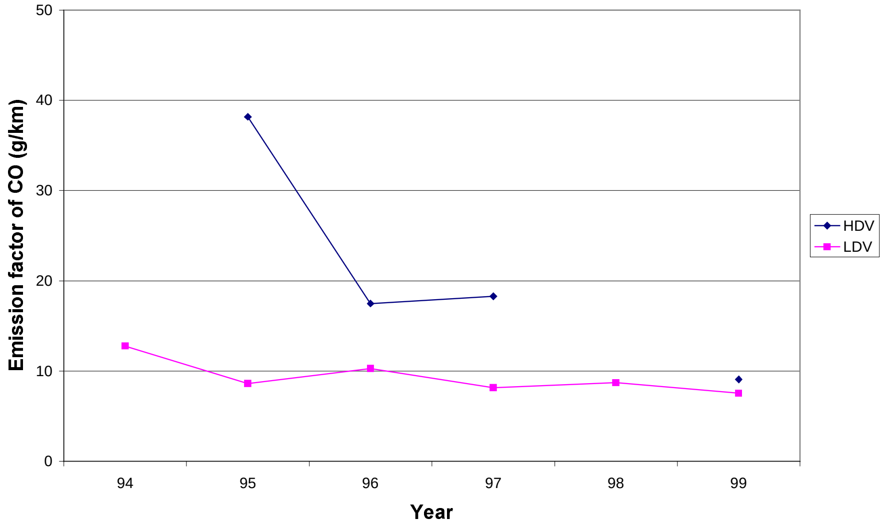


Figure 4.21 The annual trends of emission factors of CO in Hornsgatan

Sveavägen

In 1995

Average annual emission rates in Figure 4.22 are greatly different from those in Hornsgatan. Emission rates in work hours are continuously increased until afternoon peak hours.

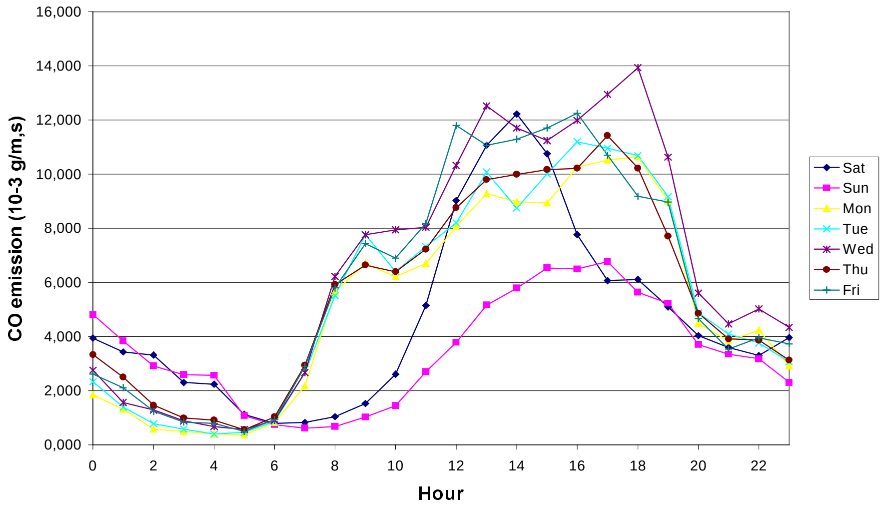


Figure 4.22 Diurnal variation of emission rates of CO at Sveavägen in 1995

Emission factors with respect to light- and heavy-duty vehicle are 11.19 g/km and 122.65 g/km, respectively. Emission factor for heavy-duty vehicle is higher than for light-duty vehicle like Hornsgatan and R^2 value, 0.10, is not high.

Table 4.30 Estimated emission factors for CO at Sveavägen in 1995

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	122.65	25.78	4.76	0.0001
LDV	11.19	0.74	15.14	0.0001

In 1996

CO emission rates in Figure 4.23 are reduced compared with those in 1995.

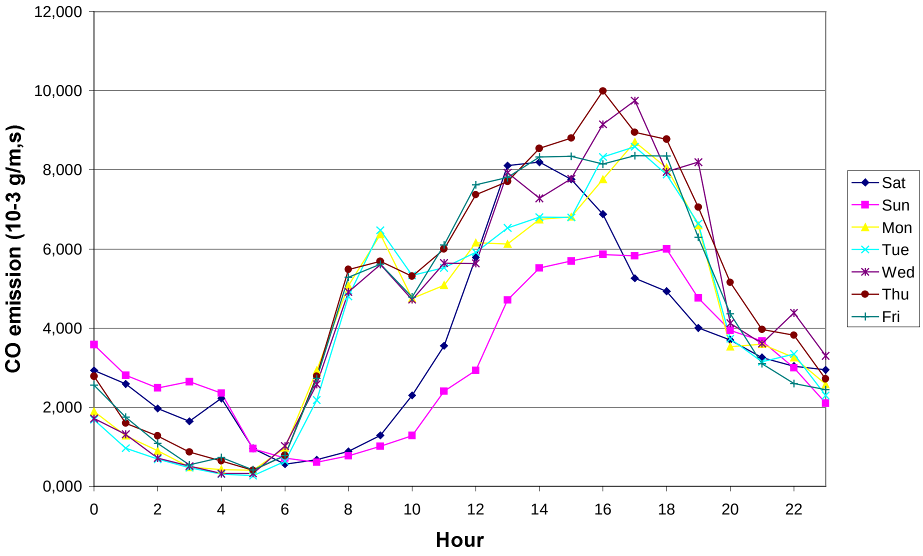


Figure 4.23 Diurnal variation of emission rates of CO at Sveavägen in 1996

Emission factors with respect to light- and heavy-duty vehicle are 9.50 g/km and 78.08 g/km respectively. They are lower than those in 1995. And emission factor for heavy-duty vehicle is also higher than that for light and R^2 value is 0.09.

Table 4.31 Estimated emission factors for CO at Sveavägen in 1996

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	78.08	17.04	4.58	0.0001
LDV	9.50	0.48	19.99	0.0001

In 1997

The trends of average annual emission rates in 1997 are similar to those in 1996 like traffic data. Emission factors with respect to light- and heavy-duty vehicle are 8.58 g/km and 93.17 g/km, respectively. And R^2 value, 0.16, is more increased.

Table 4.32 Estimated emission factors for CO at Sveavägen in 1997

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	93.17	14.83	6.28	0.0001
LDV	8.58	0.44	19.67	0.0001

In 1998

The trends of average annual emission rates and traffic data in 1998 are similar to those in previous years. Emission factors with respect to light- and heavy-duty vehicle are 9.29 g/km and 52.43 g/km, respectively. And R^2 value is 0.16.

Table 4.33 Estimated emission factors for CO at Sveavägen in 1998

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	52.43	12.13	4.32	0.0001
LDV	9.29	0.36	25.46	0.0001

In 1999

Traffic data from 1995 to 1999 in Sveavägen are not greatly changed. However, average emission rates in 1999 are slightly decreased compared with previous years as shown in Figure 4.24. And emission factors with respect to light- and heavy-duty vehcle are lowest. R^2 value is not high, 0.04.

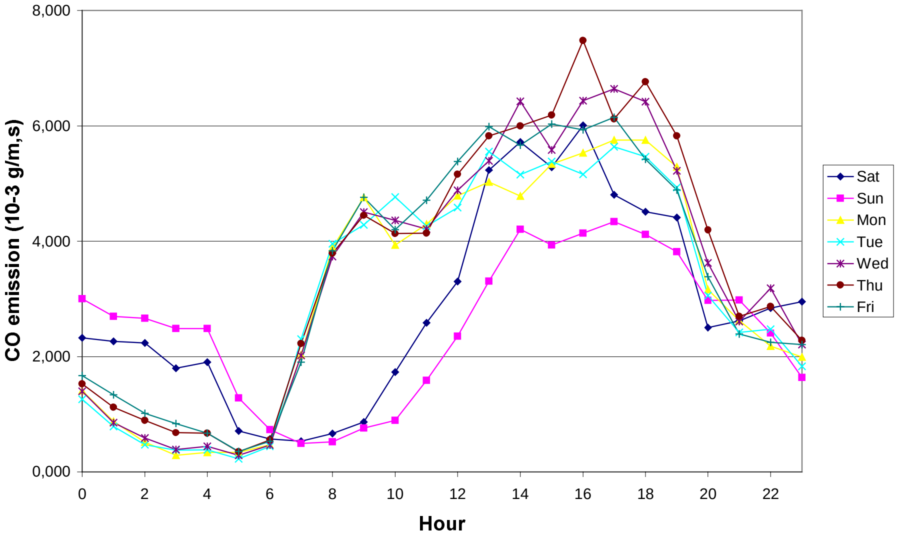


Figure 4.24 Diurnal variation of emission rates of CO at Sveavägen in 1999

Table 4.34 Estimated emission factors for CO at Sveavägen in 1999

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	33.92	10.36	3.27	0.0013
LDV	7.79	0.33	23.95	0.0001

Annual trend

The trends of emission factors with respect to light- and heavy-duty vehicle are decreased. But the whole trend with heavy-duty vehicle is decreased suddenly, which is mainly caused by small number of heavy-duty vehicle, only 3%, in each year.

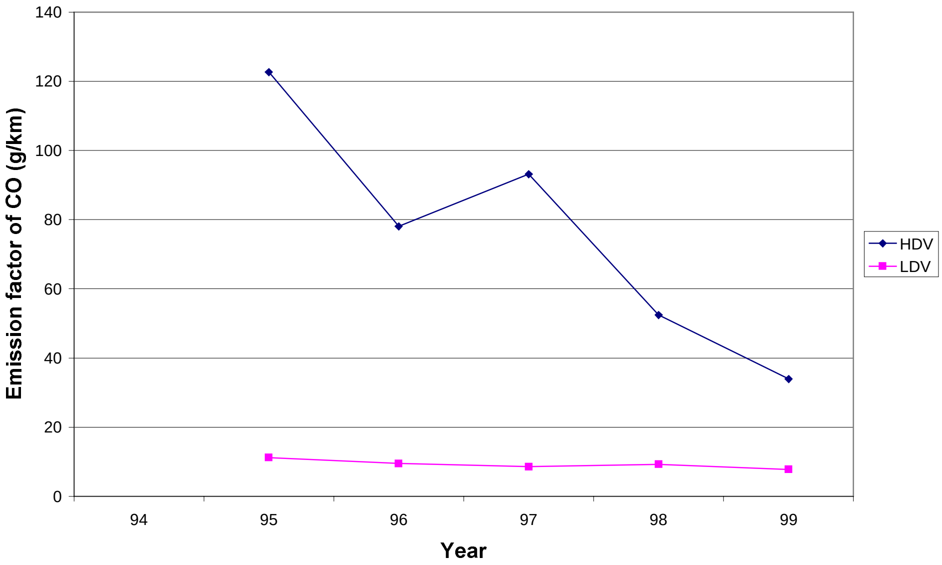


Figure 4.25 The annual trends of emission factors of CO in Sveavägen

Emission Factors for PM

Hornsgatan

PM_{10}

PM_{10} concentrations of street level were measured at Hornsgatan in 1999. And PM_{10} concentrations of roof level were measured in Rosenlundsgatan for urban backgroud concentrations. Figure 4.26 shows average emission rates of PM_{10}. Although emission rates of each day are irregular, the whole trends are similar to average annual traffic volumes. It is confirmed by other study (Namdeo, *et al*, 1999) that the high traffic volume is responsible for the high proportion of coarse particles during the day.

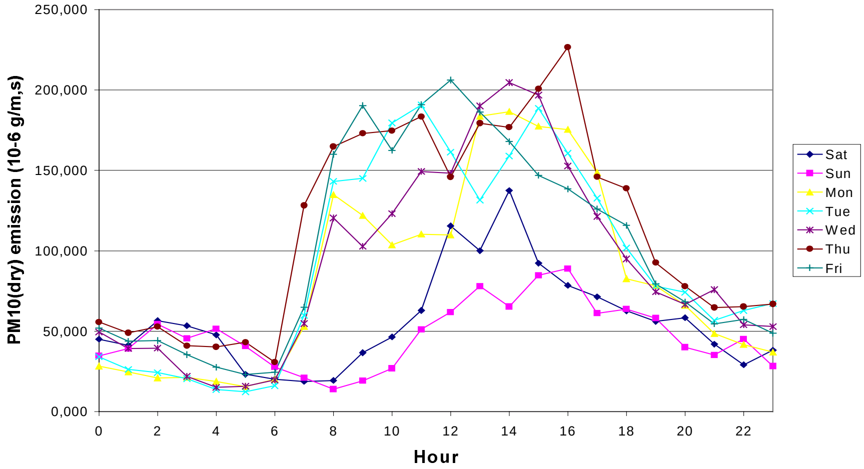


Figure 4.26 Diurnal variation of emission rates of PM_{10} at Hornsgatan in 1999

Emission factors may be divided into two categories, dry and wet conditions, according to the precipitation because PM_{10} concentrations are influenced greatly by humidity. When precipitation is higher than 2mm, emission factors are classified to wet conditions in this thesis.
In dry condtions emission factors with respect to light- and heavy-duty vehicle are 0.06 g/km and 1.66 g/km respectively. However, emission factors in wet conditions are not statistically significant in t-test because the number of sample in wet conditions is not large.

Table 4.35 Estimated emission factors for PM_{10} at Hornsgatan in 1999

Condition	Vehicle	Emission factor	Standard deviation	t value	Probability
Dry	HDV	1.66	0.13	12.64	0.0001
	LDV	0.06	0.01	4.83	0.0001
Wet (p≥2mm)	HDV	0.24	0.44	0.56	0.5772
	LDV	0.06	0.04	1.35	0.1824

$PM_{2.5}$

$PM_{2.5}$ concentrations were also measured at stret level in Horsgantan and roof level in Rosenlundsgatan in 1999. Average emission rates in Figure 4.27 is different from emission rates of PM_{10} in Figure 4.26. It has a tendency that emission rates in morning peak hours are the highest and then gradually decreased.

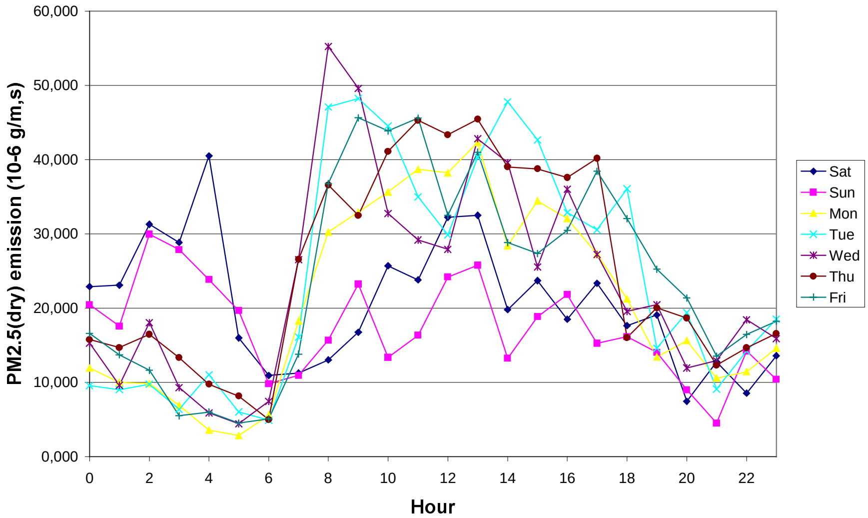


Figure 4.27 Diurnal variation of emission rates of $PM_{2.5}$ at Hornsgatan in 1999

In dry conditions emission factor with respect to heavy-duty vehicle is higher than that with respect to light-duty vehicle in Table 4.36. Emission factors in wet conditions have the same statistical problems as PM_{10} emission factors in wet.

Table 4.36 Estimated emission factors for $PM_{2.5}$ at Hornsgatan in 1999

Condition	Vehicle	Emission factor	Standard deviation	t value	Probability
Dry	HDV	0.40	0.07	5.72	0.0001
	LDV	0.03	0.01	4.19	0.0001
Wet (p≥2mm)	HDV	0.07	0.59	0.11	0.9120
	LDV	0.03	0.06	0.49	0.6332

Emission Factors from Road Tunnel

Tunnel Measurement

NOx, CO and PM concentrations were measured at Söderleds tunnel situated under the Södermalm at Stockholm from 1998 to 1999. The tunnel is about 1.5 km long and composed of two tubes which consist of two lanes and carry traffic in one direction. One tube for north direction is linked with one on-ramp and one off-ramp on route. The other for south is linked with one on-ramp and two off-ramps.

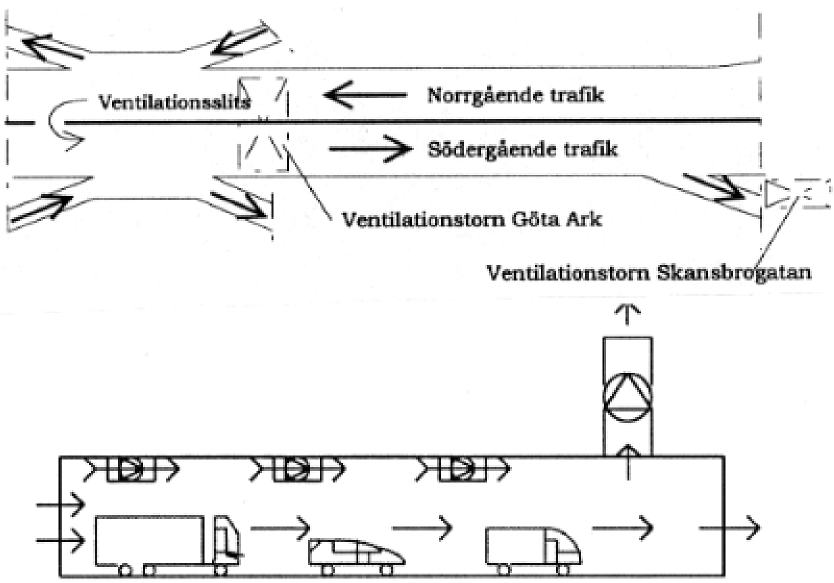


Figure 5.1 Cross-sectional view and ventilation in Söderleds tunnel

The measurements were taken during both 15 days in 1998 (Dec. 4th ~ Dec. 18th) and 25 days in 1999 (Jan. 18th ~ Feb. 11th). All measurements were performed in only south direction tube. The pollutants of NOx, CO and PM were simultaneously sampled on the middle of tunnel without ramp. The distance between two monitoring sites (called entrance and exit) is 595 m. Traffic measurements were also performed hourly at the same tube.

Table 5.1 tunnel measurements

Year	Date	Pollutants	Traffic	Other
1998	4th Dec. 01:00 ~18th Dec. 14:00	NOx, CO, PM_{10}	Volume and speed for light-duty vehicle, truck, bus and motorcycle	Wind speed, ventilation flow, temperature and relative humidity
1999	18th Jan. 14:00 ~11th Feb. 00:00	NOx, CO, $PM_{2.5}$		

The hourly total emissions of a specific pollutant exhausted from the entire traffic passing the tunnel are calculated by

$$Q_h = \frac{(C_h^{exit} - C_h^{entrance}) \cdot V_h}{L} = \frac{(C_h^{exit} - C_h^{entrance}) \cdot w_h \cdot A}{L} \qquad (10)$$

where, C_h^{exit} is concentration at tunnel exit in g/m^3

$C_h^{entrance}$ is concentration at tunnel entrance in g/m^3

V_h is ventilation flow through the tunnel in m^3/s

L is distance between two measurement sites in km

w_h is wind speed through the tunnel in m/s
A is cross-section area of the tunnel in m^2

Since Eq.(10) is equal to Eq.(4), we can estimate the emission factor of a specific pollutant for light- and heavy-duty vehicle after dividing Eq.(10) by hourly total traffic volumes.

Emission Factors for NOx

In 1998

The trends of NOx concentrations at both sampling sites look to be influenced by the trend of traffic volumes in Figure 5.2. Concentrations are the highest at 9 o'clock because of commuting traffic in the morning and the lowest at dawn when traffic is reduced. Concentrations of exit site are higher than concentrations of entrance site because of NOx emissions exhausted from traffic passing under the tunnel. The average fraction of light-duty vehicle is 0.95 and average speed is 70.17 km/h during measurement periods.

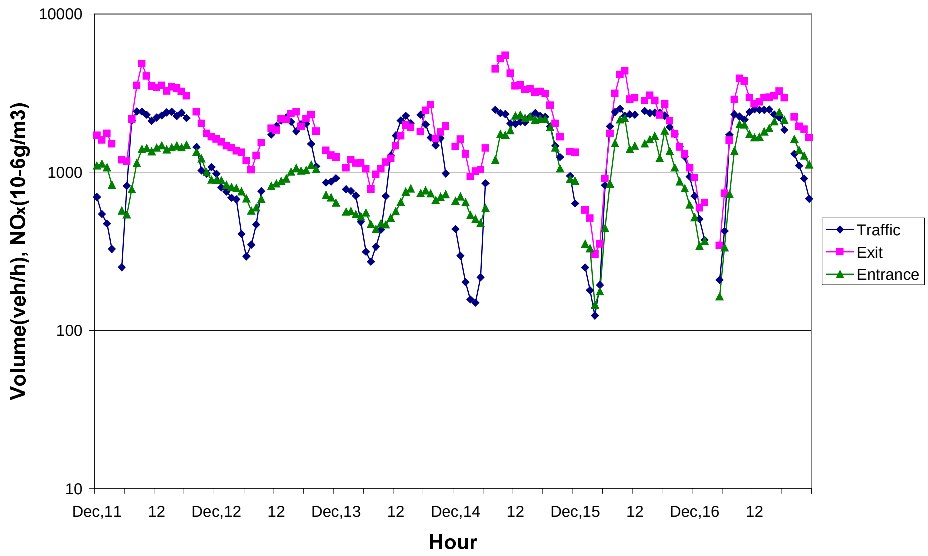


Figure 5.2 Traffic volume and NOx concentrations at Söderleds tunnel in 1998

Emission factors of NOx with respect to light- and heavy-duty vehicle are 1.11 g/km and 7.37 g/km, respectively. R^2 value, 0.14, is not high even although concentrations look to be correlated with traffic volumes in Figure 5.2. However, emission factors have statistically significant.

Table 5.2 Estimated emission factors for NOx at Söderleds tunnel in 1998

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	7.37	1.26	5.84	0.0001
LDV	1.11	0.07	15.47	0.0001

In 1999

The trends of NOx concentrations in 1999 are similar to those in 1998 because they are also influenced by the variation of traffic volumes which are not different from traffic in 1998. Concentrations are the highest at morning peak hours and the lowest at dawn as shown in Figure 5.3. The average fraction of light-duty vehicle, 0.95, is the same value as in 1998, but average speed is higher, 74.13 km/h.

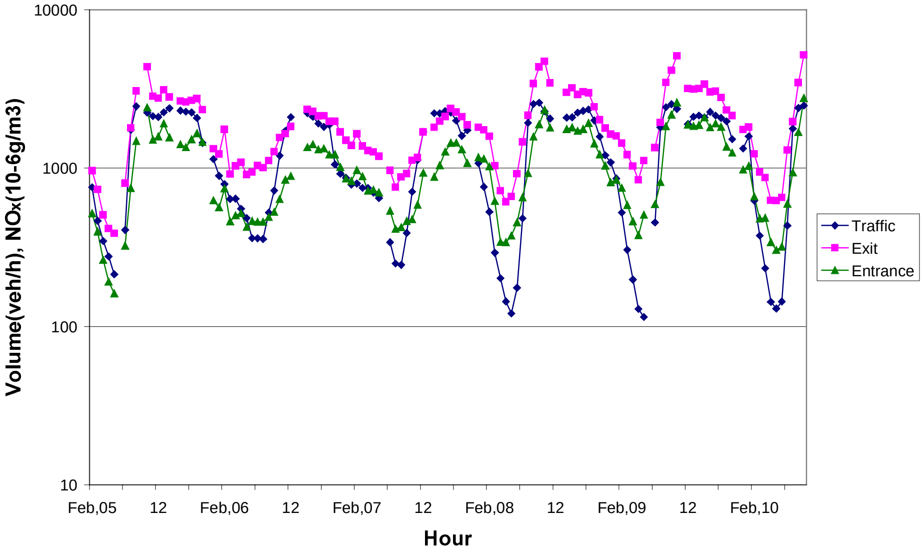


Figure 5.3 Traffic volume and NOx concentrations at Söderleds tunnel in 1999

Although all trends are identical with those in 1998 and intervals between two measuring periods are not large, emission factors are different. Emission factor for light-duty vehicle is 0.81 g/km and lower than emission factor in 1998. Emission factor for heavy-duty vehicle is 9.08 g/km and higher than in 1998. And R^2 value, 0.37, is not high but increased. In t-test two emission factors have statistical significance in 95% confidence.

Table 5.3 Estimated emission factors for NOx at Söderleds tunnel in 1999

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	9.08	0.77	11.79	0.0001
LDV	0.81	0.05	16.52	0.0001

Emission Factors for CO

In 1998

CO concentrations of two sampling sites under Söderleds tunnel are changed according to the fluctuation of traffic volumes passed the tunnel like NOx concentrations. On the whole, concentrations of exit site are the highest in morning peak hours. Concentrations of entrance site are lower but sometimes higher than concentrations of exit site.

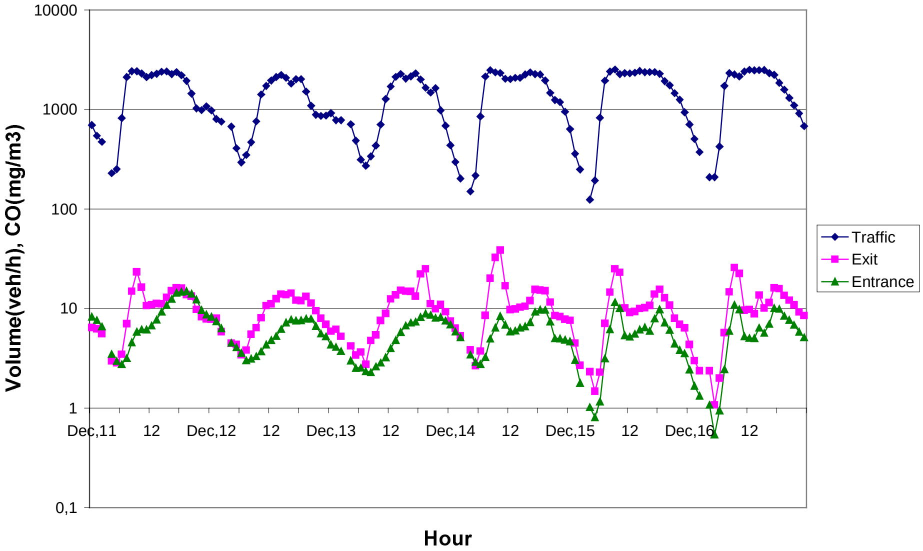


Figure 5.4 Traffic volume and CO concentrations at Söderleds tunnel in 1998

Emission factor with respect to heavy-duty vehicle is not statistically significant in t-test. And R^2 value, 0.01, is very low.

Table 5.4 Estimated emission factors for CO at Söderleds tunnel in 1998

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	14.48	9.89	1.46	0.1453
LDV	4.07	0.55	7.40	0.0001

In 1999

CO concentrations of two sampling sites in Figure 5.5 are also influened by traffic volumes.

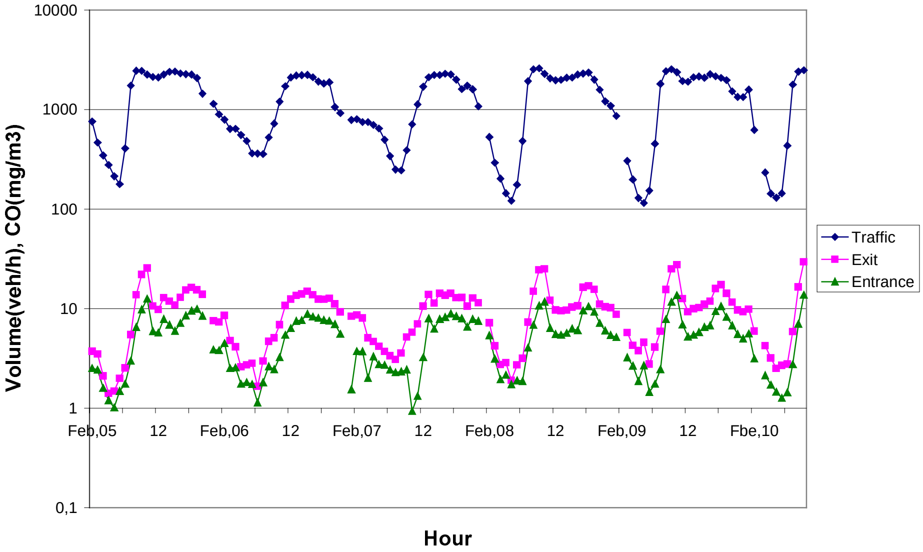


Figure 5.5 Traffic volume and CO concentrations at Söderleds tunnel in 1999

Emission factor with respect to light-duty vehicle is increased compared with in 1998. However, Emission factor with respect to heavy-duty vehicle is still not statistically significant in t-test. And R^2 value is 0.00.

Table 5.5 Estimated emission factors for CO at Söderleds tunnel in 1999

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	3.22	4.40	0.73	0.4658
LDV	5.54	0.29	18.97	0.0001

Emission Factors for PM

PM_{10} in 1998

The trends of PM_{10} concentrations under Söderleds tunnel are very different from trends of NOx and CO. Traffic volumes in everyday are not changed greatly, but concentrations of two sites are not constant, uncorrelated with traffic volumes. They are, although concentrations looks to be influenced partly by exhausted emissions from traffic, changed mainly according to other factors in Figure 5.6. It is different from trends of PM_{10} measured in street canyons.

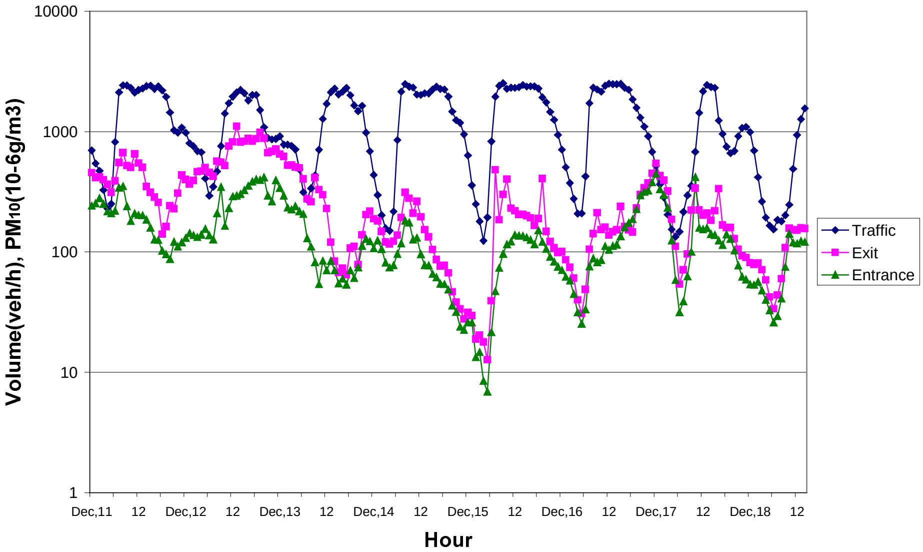


Figure 5.6 Traffic volume and PM_{10} concentrations at Söderleds tunnel in 1998

It is difficult to estimate emission factors of PM_{10} because emission factor with respect to heavy-duty vehicle has a minus value in Table 5.6 and even R^2 value is 0.00.

Table 5.6 Estimated emission factors for PM_{10} at Söderleds tunnel in 1998

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	-0.05	0.61	-0.08	0.9391
LDV	0.24	0.03	7.62	0.0001

$PM_{2.5}$ in 1999

$PM_{2.5}$ concentrations under the tunnel are different from PM_{10} concentrations in 1998. $PM_{2.5}$ concentrations unlike PM_{10} are influenced in some degree by traffic volumes as shown in Figure 5.7.

Emission factor with respect to light-duty vehicle is 0.04 g/km and with respect to heavy-duty vehicle is 0.57 g/km. They are statistically significant in t-test. And R^2 value, 0.10, is higher than that of PM_{10} in 1998.

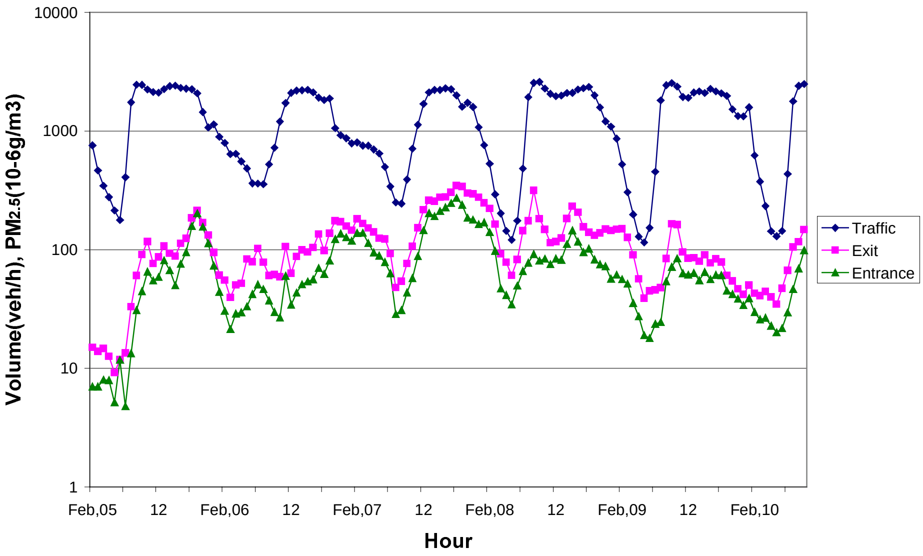


Figure 5.7 Traffic volume and $PM_{2.5}$ concentrations at Söderleds tunnel in 1999

Table 5.7 Estimated emission factors for $PM_{2.5}$ at Söderleds tunnel in 1999

Vehicle	Emission factor	Standard deviation	t value	Probability
HDV	0.57	0.12	4.69	0.0001
LDV	0.04	0.01	5.18	0.0001

Comparison between Measured and Calculated Emission Factors

Emission Factors for NOx

The comparison of the emission factors for NOx reveals several general features as shown in Table 6.1. Looking at emission factors in different measurements the confidence intervals for heavy-duty vehicles are larger than for light-duty vehicles. One important reason is that the number of heavy-duty vehicles is small (John *et al*., 1999), heavy-duty vehicles in the street canyon and in the tunnel were merely on average 3~9% of all vehicles.
The emission factor for heavy-duty vehicles is 7~15 times higher than the corresponding factor for light-duty vehicles in Table 6.1. It means that NOx is exhausted much more from heavy-duty vehicles than light-duty vehicles. Actually, in the city of Stockholm the exhaust emission of NOx from heavy-duty vehicles is about 40% of the total exhaust emission from all vehicles (Johansson *et al*., 1999).

Table 6.1 Estimated emission factors (mean ± 95% confidence interval, g/km) of NOx in different measurements

Year	Hornsgatan		Sveavägen		Tunnel		EVA	
	HDV	LDV	HDV	LDV	HDV	LDV	HDV	LDV
1994	5.7±1.0	1.3±0.1	-	-	-	0.9±0.04[a]	5.53	0.82
1995	9.8±1.0	0.8±0.1	14.5±2.6	1.0±0.1	-	1.3±0.03[a]	5.33	0.79
1996	6.1±0.7	1.1±0.1	13.4±2.0	0.9±0.1	-	-	5.06	0.75
1997	5.9±0.7	0.9±0.1	11.3±2.3	0.9±0.1	-	-	4.81	0.69
1998	7.1±0.7	0.9±0.1	11.5±2.4	0.9±0.1	7.4±2.5	1.1±0.1	4.50	0.64
1999	8.3±0.8	0.7±0.1	10.6±2.4	1.0±0.1	9.1±1.5	0.8±0.1	4.19	0.60

[a]Two values are quoted from other study (Bylin *et al*., 1999) which includes emission factors estimated from past measurements for the same tunnel

Emission factors estimated from street canyons are in good agreement with emission factors from tunnel measurement as shown in Table 6.1. Real world emission factors for heavy-duty vehicles, estimated from both street canyons and tunnel, are higher than emission factors calculated by the EVA model which is based on the chassis dynamometer tests. The real world emission factors except for Sveavägen do not indicate a decrease as expected from the EVA calculation according to the assumed increasing rate of vehicles equipped the catalyst converter.

The main uncertainties in street canyons are due to uncertainties in traffic counts and fraction of heavy-duty vehicles and light-duty vehicles without catalysts. These two vehicle types constitute about 50% of the total emission from road traffic, which are the dominant sources of NOx close to the monitoring stations in central Stockholm. And a closer examination of the traffic counts close to the measurement stations at Hornsgatan and Sveavägen has shown that the traffic count of heavy-duty vehicles is periodically connected with substantial uncertainty. Comparison with manual counting has shown that at Hornsgatan the heavy-duty vehicles count is somewhat too high and at Sveavägen it is too low (Johansson *et al*., 1999).

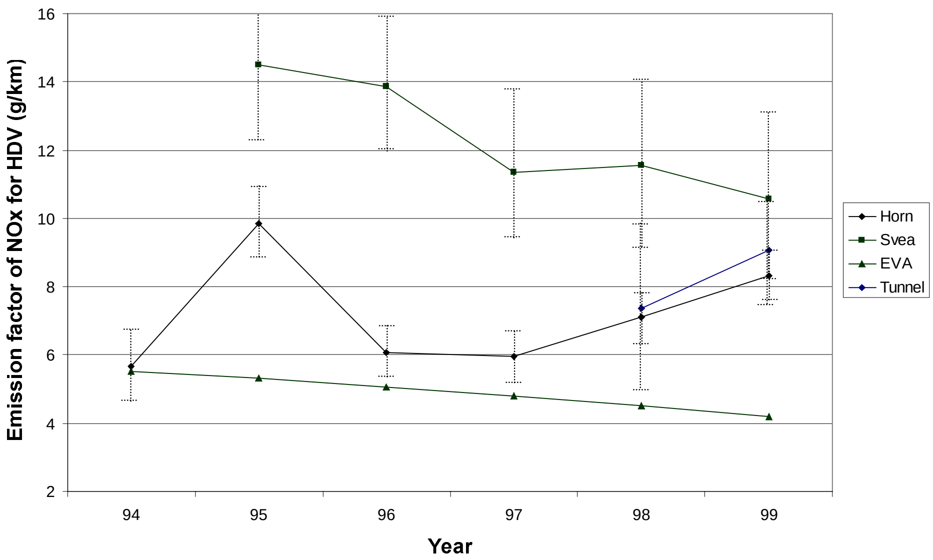


Figure 6.1 Comparison of annual emission factors of NOx for heavy-duty vehicles in different measurements

Emission factors of NOx for light-duty vehicles calculated by the EVA are close to the confidence intervals of both street canyons and tunnel measurements as shown in Figure 6.2. And all real world emission factors are gradually decreased like the corresponding factors in EVA.

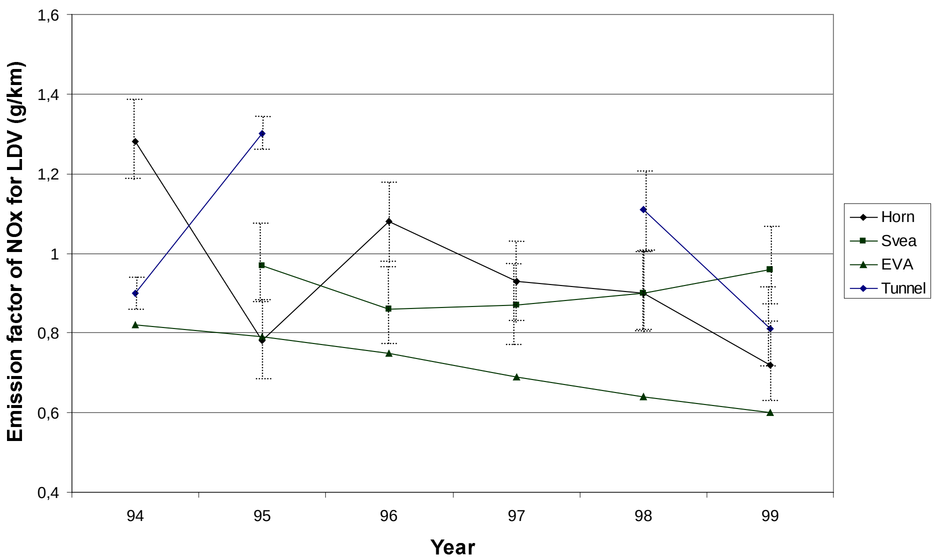


Figure 6.2 Comparison of annual emission factors of NOx for light-duty vehicles in different measurements

Emission Factors for CO

The 95% confidence intervals of CO emission factors for heavy-duty vehicles are also larger than light-duty vehicles because the fraction of heavy-duty vehicles is very small causing a small contribution to the total concentrations of CO in street canyons and in the traffic tunnel. The emission factors for heavy-duty vehicles are much higher than the corresponding factors for light-duty vehicles in Table 6.2. This is completely different from emission factors calculated by the EVA model and other studies (Palmgren *et al*., 1999, John *et al*., 1999).

A major uncertainty results from the fact that the fractions of heavy-duty vehicles which is lower emitter than light-duty vehicles are very small in Hornsgatan and Sveavägen and that the Street Canyon model cannot estimate accurately emission rates of CO exhausted from a few of heavy-duty vehicles. This uncertainty is confirmed when looking at the comparison of annual emission factors with respect to heavy-duty vehicles in Figure 6.3. Emission factors for heavy-duty vehicles in Sveavägen where the fraction of heavy-duty vehicles is 3% are relatively much higher than emission factors in Hornsgatan where the fraction is 7~9% compared with emission factors in EVA.

Table 6.2 Estimated emission factors (mean ± 95% confidence interval, g/km) of CO in different measurements

Year	Hornsgatan		Sveavägen		Tunnel		EVA	
	HDV	LDV	HDV	LDV	HDV	LDV	HDV	LDV
1994	-	12.8±0.9	-	-	-	7.0±0.2[a]	3.41	20.84
1995	38.2±14.9	8.6±1.3	122.7±50.9	11.2±1.5	-	5.2±0.2[a]	3.13	19.95
1996	17.5±10.6	10.3±1.0	78.1±33.6	9.5±0.9	-	-	2.78	18.08
1997	18.3±10.6	8.2±1.0	93.2±29.3	8.6±0.9	-	-	2.51	16.65
1998	-	8.7±0.8	52.4±23.9	9.3±0.7	-	4.1±1.1	2.10	15.27
1999	9.1±7.8	7.5±0.8	33.9±20.5	7.8±0.6	-	5.5±0.6	1.73	14.28

[a]Two values are quoted from other study (Bylin *et al*., 1999) which includes emission factors estimated from past measurements for the same tunnel

Emission factors estimated from street canyons are 27~56% higher than emission factors from tunnel. The real world emission factors for heavy-duty vehicles are higher than emission factors by the EVA model although the uncertainty caused by the small number of heavy-duty vehicles is included. On the contrary, real world emission factors for light-duty vehicles are lower than emission factors by the EVA as shown in Figure 6.4.

There are some studies of comparison between emission factors based on dynamometer tests and real world emission factors in Sweden. Sjödin and Lenner (1995) estimated emission factors of CO through the remote sensing measurement and concluded that estimates based on laboratory test largely underestimated the real world emission factors, especially for catalyst cars, because laboratory test did not take into account high-emitting cars with malfunctioning catalyst converter. Sjödin *et al* (1998) estimated emission factors from tunnel measurement and concluded that real world emission factors of CO for light-duty vehicles were in fairly good agreement with emission factors derived from dynamometer tests within ±10~20% on the assumption that light-duty vehicles were composed of 40% non-catalyst vehicles and 60% catalyst-equipped vehicles.

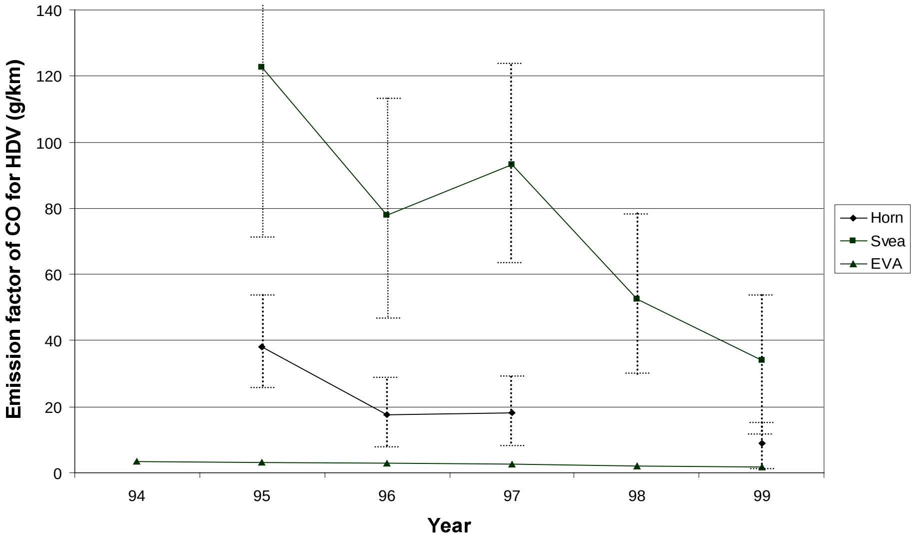


Figure 6.3 Comparison of annual emission factors of CO for heavy-duty vehicles in different measurements

It seems that the EVA model overestimates the real world emission factors for light-duty vehicles since emission factors in both Hornsgatan and Sveavägen are almost same and the number of light-duty vehicles in both streets is enough large to make the statistic estimation more reliable every year. Also emission factors estimated from tunnel measurements are lower than those in EVA. As already discussed, the lack of more detail information on rates of non-catalyst vehicles leads to a substantial uncertainty.

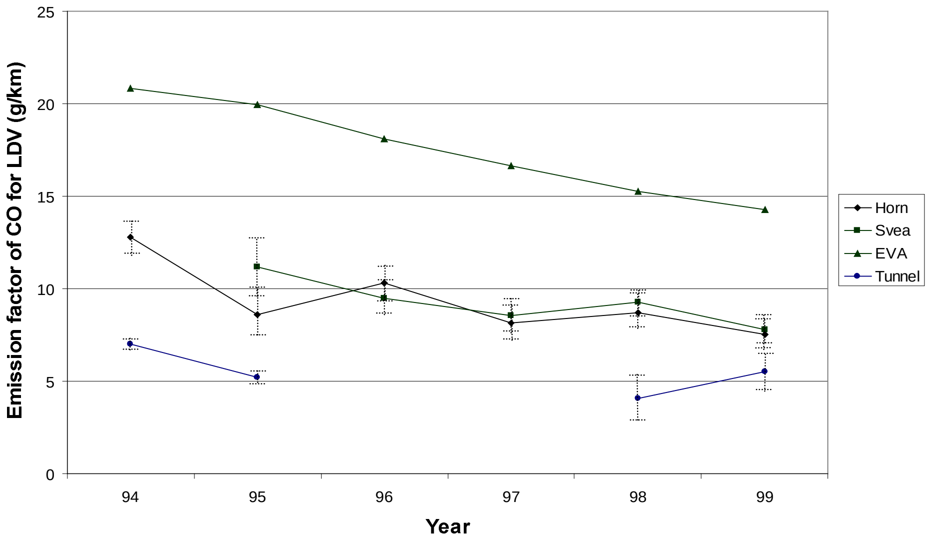


Figure 6.4 Comparison of annual emission factors of CO for light-duty vehicles in different measurements

Emission Factors for PM

The emission factors for heavy-duty vehicles are higher than the corresponding factors for light-duty vehicles, implying that heavy-duty vehicles is higher emitter than light-duty vehicles. In fact, in the city of Stockholm the exhaust emission of PM from heavy-duty vehicles has been estimated to be about 60% of the total exhaust emission from all vehicles. (Johansson *et al.*, 1999).

Table 6.3 Estimated emission factors (mean ± 95% confidence interval, g/km) of PM in different measurements

Year	Hornsgatan				Tunnel[a]		EVA[b]	
	PM_{10}(dry)		$PM_{2.5}$(dry)					
	HDV	LDV	HDV	LDV	HDV	LDV	HDV	LDV
1998					-	0.24±0.06	0.13	0.01
1999	1.66±0.26	0.06±0.03	0.40±0.14	0.03±0.01	0.57±0.24	0.04±0.02	0.11	0.01

[a]In tunnel measurement, emission factors in 1998 are estimated for PM_{10} and emission factors in 1999 are for $PM_{2.5}$
[b]In the EVA model, emission factors are calculated for PM not divided into PM_{10} and $PM_{2.5}$

Emission factors of $PM_{2.5}$ estimated from street canyons are 33~43% lower than emission factors from tunnel measurement in Table 6.3. In both PM_{10} at Figure 6.5 and $PM_{2.5}$ at Figure 6.6 emission factors from street canyons are 67~92% higher than emission factors by the EVA model. The main reason for these differences is that traffic on road, unlike a vehicle in laboratory, contribute to increased levels of coarse particle concentrations because of increased turbulence and re-suspension of coarse particles from road surface and tyre wear and tear (Namdeo *et al.*, 1999).

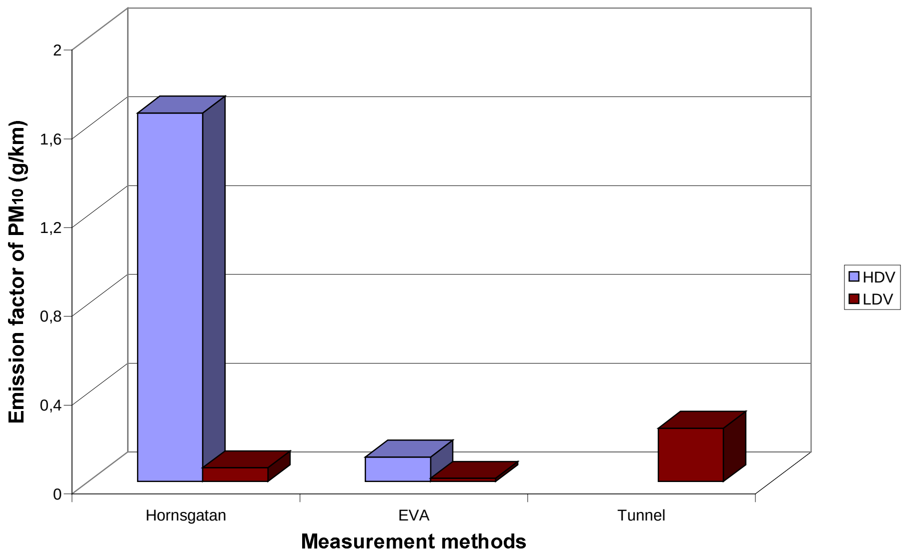


Figure 6.5 Comparison of emission factors of PM_{10} in different measurements

A major contribution to particulate pollution in urban areas is believed to be attributed to traffic, and especially, to emissions from diesel powered vehicles (Vignati *et al*, 1999). Actually, for PM the specific emission factors are about 10 times higher than petrol powered vehicles without catalysts (Johansson *et al*, 1999). The fact that background concentrations of PM_{10} and $PM_{2.5}$ were measured at not Hornsgatan but Rosenlundsgatan results in some uncertainty in PM_{10} and $PM_{2.5}$. The contribution of secondary particulate matter formed through physical and chemical reactions is a small source of additional uncertainty.

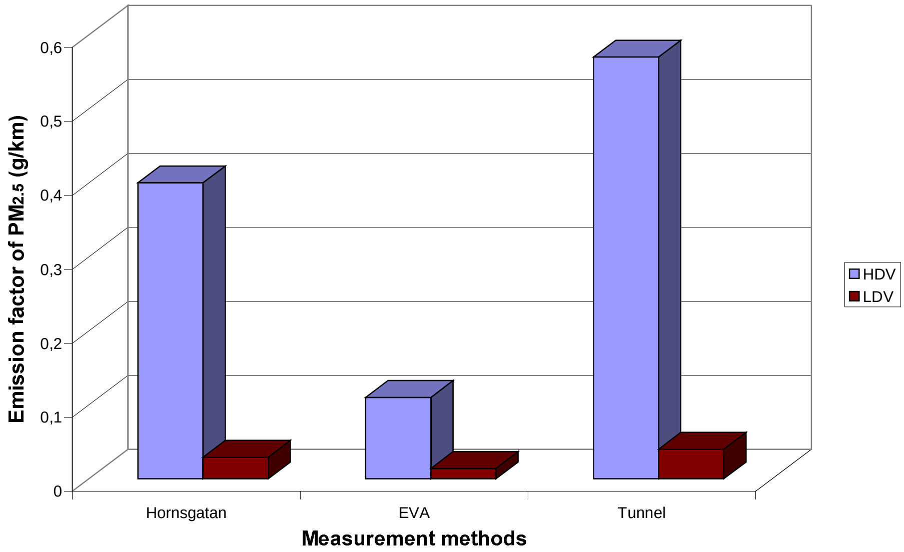


Figure 6.6 Comparison of emission factors of $PM_{2.5}$ in different measurements

An attempt was made to separate the PM data between dry and wet periods in order to distinguish the contribution of re-suspended PM from the total PM emission. However, this analysis did not give reasonable results, possibly due to the small number of samples left for statistical calculations.

Conclusions

To determine emission factors under realistic driving conditions, an inverse street air quality model developed by Palmgren *et al* (1999) is applied to air quality measurements and traffic counts at Hornsgatan and Sveavägen in Stockholm from 1994 to 1999. Based on inverse calculations using a Street Canyon Model, average emission factors of NOx, CO, PM_{10} and $PM_{2.5}$ with respect to light- and heavy-duty vehicles are estimated. And the emission factors estimated from street canyons are compared with the corresponding factors estimated from tunnel measurements as well as calculated by the EVA model of the National Road and Traffic Administration.

All emission factors estimated from street canyons are in good agreement within ±56% with emission factors estimated from tunnel measurements. With the exception of the emission factors of NOx and CO for light-duty vehicles, the emission factors from street canyons are higher than emission factors by the EVA model which is derived from dynamometer test. For the emission factors of NOx with respect to light-duty vehicles, a good agreement is found within the expected range of uncertainty. However, emission factors of CO for light-duty vehicles are lower than emission factors by the EVA model in contradiction to other emission factors and general expectations.

In a recent study (Sjödin and Lenner, 1995) results of a large number of measurements obtained with the Infra Red based remote sensing technique presented the importance of determining the share of different technologies in the fleet. The remote sensing measurement revealed that the strongly skewed distribution of CO was mainly caused by the large differences of old vehicles without exhaust gas reduction in contradiction to new vehicles with controlled catalytic converters.

It can be concluded that emission factors estimated through the inverse street air quality modelling agree with emission factors from tunnel measurements and that with the exception of NOx emission factors for light-duty vehicles they are not in accordance with the emission factors calculated by the EVA model. And it should be pointed out that the conclusions are only valid for highway driving. No information for stop and go traffic and cold start and evaporative emissions can be gathered from street canyons and tunnel measurements.

References

Abbott, P.G., Hartley, S., Hickman, A.J., Layfield, R.E., McCrae, I.S., Nelson, P.M., Phillips, S.M. and Wilson J.L. (1995), The environmental assessment of traffic management schemes: a literature review, TRL Report 174

Andre, M., Hammarström, U. and Reynaud, I. (1999), Driving statistics for the assessment of pollutant emissions from road transport, INRETS Report LTE 9906

Bellasio, R. (1997), "Modelling traffic air pollution in road tunnels", *Atmospheric Environment*, Vol. 31, No. 10, pp. 1539-1551

Berkowicz, R., Palmgren, F., Hertel, O. and Vignati, E. (1996), "Using measurements of air pollution in streets for evaluation of urban air quality – meteorological analysis and model calculations", *The Science of the Total Environment*, Vol. 189/190, pp. 259-265

Boulter, P.G. (1999), Remote sensing of vehicle emissions near two traffic calming measures in Gloucester, TRL Report 423

Bylin, G., Bellander, T., Bjornback, M., Bostrom, C-E., Johansson, C., Jungnelius, S., Svartengren, M. and Westerlund, K-G., (1999), Impact on health of exposure in raod tunnels, Presented at PIARC conference in Kuala Lumpur

Cloke, J., Boulter, P., Davies, G.P., Hickman, A.J., Layfield, R.E., McCrae, I.S. and Nelson, P.M. (1998), Traffic management and air quality research program, TRL Report 327

Edwards, H. (1998), Estimation of excess cold start emissions on links in traffic networks, KFB & VTI forskning/research 25A

Eggleston, H.S., Gorissen, N., Joumard, R., Rijkeboer, R.C., Samaras, Z. and Zierock, K.H. (1991), CORINAIR working group on emission factors for calculating 1990 emissions from road traffic, volume 1: methodology and emission factors, Final report, Commission of the European Communities

Eggleston, S., Gorisen, N., Hassel. D., Hickman, A.J., Joumard, R., Rijkeboer, R. and Zierock, K.H. (1999), COPERT III: computer programme to calculate emissions from road transport, European environment agency and European topic centre on air emission

Ericsson, E. (1999), Variability in exhaust emission and fuel consumption in urban driving, Proceedings of the conference of urban transport systems, Lund

Hammarström, U. (1995), Exhaust emissions from road traffic – description of driving patterns by means of simulation models, Pre-proceedings of the workshop Estimation of pollutant emissions from transport, COST 319, Belgium, VTI särtryck 272, 1997

Hegobert, P. (1995), *Automobiles and pollution*, Society of Automotive Enigneers, INC

Hertel, O. and Berkowicz, R. (1989), Modelling pollution from traffic in a street canyon: evaluation of data and model development, National environmental research institute, Report A129

Hickman A.J. and Colwill, D.M. (1982), The estimation of air pollution concentrations from road traffic, TRRL Laboratory Report 1052

Hickman, A.J., Andre, M., Hammarström, U. and Kyriakis, N. (1998), Road traffic characteristics for estimating pollutant emissions, TRL Project Report SE/432/98

Hoglund, P.G. and Niittymaki, J. (1999), Estimating vehicle emissions and air pollution related to driving patterns and traffic calming, Proceedings of the conference of urban transport systems, Lund

Hoydysh, W.G. and Dabberdt, W.F. (1988), "Dispersion characteristics of flows in asymmetric street canyons and sensitivity to block shape", *Transportation Research Record* 1176, pp. 78-86

John, C., Friedrich, R., Staehelin, J., Schläpfer, K. and Stahel, W.A. (1999), "Comparison of emission factors for road traffic from a tunnel study (Gubrist tunnel, Switzerland) and from emission modelling", *Atmospheric Environment*, Vol. 33, pp. 3367-3376

Johansson, C., Johansson, P.Å. and Burman, L. (1996), Emissioer av kväveoxider och kolmonoxid från trafik, Institute of applied environmental research, Report 49 (in Swedish)

Johansson, C., Hadenius, A., Johansson, P.Å. and Jonson, T. (1999), The Stockholm study on health effects of air pollution and their economic consequences, Part I: NO_2 and particulate matter in Stockholm, Swedish national road Administration No. 1999:41

Joumard, R., Jost, P. and Hickman, A.J. (1995), Influence of instantaneous speed and acceleration on hot passenger car emissions and fuel consumption, SAE technical paper series 950928

Joumard, R., Philippe, F. and Vidon, R. (1999), "Reliability of the current models of instantaneous pollutant emissions", *The Science of the Total Environment*, Vol. 235, pp. 133-142

LeBlanc et al. (1995), "Driving pattern variability and impacts on vehicle carbon monoxide emissions", *Transportation Research Record* 1472, pp. 45-52

Lenner, M. (1995), Measurement by on board apparatus, of passenger cars' real-world exhaust emissions and fuel consumption, VTI meddelande, No. 771A

Namdeo, A.K., Colls, J.J. and Baker, C.J. (1999), "Dispersion and re-suspension of fine and coarse particulates in an urban street canyon", *The Science of the Total Environment*, Vol. 235, pp. 3-13

Palmgren, F., Berkowicz, R., Ziv, A. and Hertel, O. (1999), "Actual car fleet emissions estimated from urban air quality measurements and street pollution models", *The Science of the Total Environment*, Vol. 235, pp. 101-109

Pavageau, M. and Schatzmann, M. (1999), "Wind tunnel measurements of concentration fluctuations in an urban street canyon", *Atmospheric Environment*, Vol. 33, pp. 3961-3971

Rosqvist, L.S. (1999), Vehicular emissions and fuel consumption for street characteristics in residential areas, Proceedings of the conference of urban transport systems, Lund

Sawyer, R.F., Harley, R.A., Cadle, S.H., Norbeck, J.M., Slott, R. and Bravo, H.A. (2000), "Mobile sources critical review: 1998 NARSTO assessment", *Atmospheric Environment*, Vol. 34, pp. 2161-2181

Sjödin, Å., Cooper, D.A. and Andreasson, K. (1995a), "Estimations of real-world N_2O emissions from road vehicles by means of measurements in a traffic tunnel", *Journal of Air & Waste Management Association*, Vol. 45, No. 3, pp. 186-190

Sjödin, Å. and Lenner, M. (1995b), "On-road measurements of single vehicle pollutant emissions, speed and acceleration for large fleets of vehicles in different traffic environments", *The Science of the Total Environment*, Vol. 169, No. 8, pp. 157-165

Sjödin, Å., Persson, K., Andreasson, K., Arlander, B. and Galle, B. (1998), "On-road emission factors derived from measurements in a traffic tunnel", *International Journal of Vehicle Design*, Vol. 20, No. 1-4(special issue), pp. 147-158

Sjödin, Å. (2000), On-road/real-world emission measurements as a means of estimating, evaluating and controlling road vehicle emissions, Lecture note at MTC

Staehelin, J., Schläpfer, K., Burgin, T., Steinemann, U., Schneider, S., Brunner, D., Bäumle, M., Meier, M., Zahner, C., Keiser, S., Stahel, W and Keller, C. (1995), "Emission factors from road traffic from a tunnel study

(Gubrist tunnel, Switzerland). Part I: concept and first results", *The Science of the Total Environment*, Vol 169, pp. 141-147

Staehelin, J., Keller, C., Stahel, W., Schläpfer, K., Steinemann, U., Burgin, T. and Schneider, S. (1997), "Modelling emission factors of road traffic from a tunnel study", *Environmetrics*, Vol. 8, pp. 219-239

Staehelin, J., Keller, C., Stahel, W., Schläpfer, K. and Wunderli, S. (1998), "Emission factors from road traffic from a tunnel study (Gubrist tunnel, Switzerland). Part III: results of organic compounds, SO_2 and speciation of organic exhaust emission", *Atmospheric Environment*, Vol. 32, No. 6, pp. 999-1009

Touaty, M. and Bonsang, B. (2000), "Hydrocarbon emissions in a highway tunnel in the Paris area", *Atmospheric Environment*, Vol. 34, pp. 985-996

Vignati, E., Berkowicz, R., Palmgren, F., Lyck, E. and Hummelshoj, P. (1999), "Transformation of size distributions of emitted particles in streets", *The Science of the Total Environment*, Vol. 235, pp. 37-49

Watkins, L.H. (1991), Air pollution from road vehicles, TRRL, State of the Art Review 1

Yamartino, R.J. and Wiegand, G. (1986), "Development and evaluation of simple models for the flow, turbulence and pollutant concentration fields within an urban street canyon", *Atmospheric Environment*, Vol. 20, No. 11, pp. 2137-2156

Weingartner, E., Keller, C., Stahel, W.A., Burtscher, H. and Baltensperger, U. (1997), "Aerosol emission in a road tunnel", *Atmospheric Environment*, Vol. 31, No. 3, pp. 451-462