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PM10 non-exhaust emission factors from road tunnel measurements considering deposition and resuspension

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Abstract

An improved understanding of PM non-exhaust sources from abrasion and resuspension processes is important for ambient air quality management and tunnel air management. Traffic tunnels are often used to assess real world air pollutant emissions. In contrast to ambient air quality measurements, flow and dilution conditions are well defined, vehicle speeds are typically constant. The aim of this work was to study the role of traffic volume on emission factors and to examine if PM10 deposition as opposed to resuspension needs consideration in road tunnel emission measurements. Data from two monitoring campaigns in a 10 km long tunnel with different distances between the air quality monitoring stations (3380 m/7180 m) have been re-analyzed. Neglecting potential prevailing sink processes, non-exhaust emission factors approach 0 with high traffic volumes or even non-physical negative values resulted. Significantly increased non-exhaust emission factors resulted by using an effective deposition velocity as a sink term accounting for the interplay of wall deposition losses and resuspension of PM10. Average light vehicle (LV) PM10 non-exhaust EF ranged between 0.015 g/km/LV (35000 LV/day) to 0.060 g/km/LV (6500 LV/day). Average heavy duty vehicle (HV) PM10 non-exhaust EF ranged between 0.080 g/km/HV (6000 HV/day) to 0.300 g/km/HV (1500 HV/day).

Keywords: PM10 non-exhaust emission factor, tunnel measurements, deposition, resuspension

1 Introduction

Road traffic non-exhaust emissions are a major particulate matter (PM) source in

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Europe. In contrast to PM exhaust emissions, non-exhaust emission factors (EF) are difficult to characterize and have large uncertainties. An improved understanding of PM non-exhaust sources is important for ambient air quality management but also tunnel air and waste management.
In contrast to exhaust pipe particles, mechanically produced particles from road traffic abrasion and resuspension processes are still poorly characterised, detailed quantitative and process specific information is missing. These non-exhaust or often termed diffuse road traffic emissions consist of brake, tire and road wear generated particles, as well as vehicle induced resuspension of deposited road dust. Wet road surface conditions may retain particles, under dry conditions particles get resuspended [1]. Measurements to characterize non-exhaust PM emissions in ambient air are influenced by complex atmospheric processes such as flow, turbulence, rain, snowfall acting upon road conditions. Long road tunnels are less impacted by ambient atmospheric conditions apart from the entry portals. Non-exhaust emissions may also be influenced by different speed, acceleration or deceleration. Therefore, unidirectional traffic tunnels have been used for real world emission measurements as flow and dilution conditions are mainly controlled by traffic and the tunnel geometry. Emission factors for gaseous species and PM2.5/PM10 have been assessed by Kristensson et al. (2004) [2], Meier et al., (2015) [3], and Hinterhofer et al., (2015) [4] using approaches as described by Equation 1. Average fleet emission factors ($EF_{fl\text{-}avg}$) were determined by monitoring the concentration difference ΔC $(= C_{down} - C_{up})$ at the traffic related upstream (C_{up}) and downstream (C_{down}) locations with distance ΔL apart and monitoring the number of vehicles (N_{veh}) passing by in a certain time interval and by assessing the airflow in the tunnel by the longitudinal velocity (V_L) multiplied by the tunnel cross section (A) as illustrated in Figure 1.

$$EF_{fl-avg} = \frac{\Delta C \cdot V_L \cdot A}{\Delta L} \cdot \frac{\Delta t}{N_{veh}} \tag{1}$$

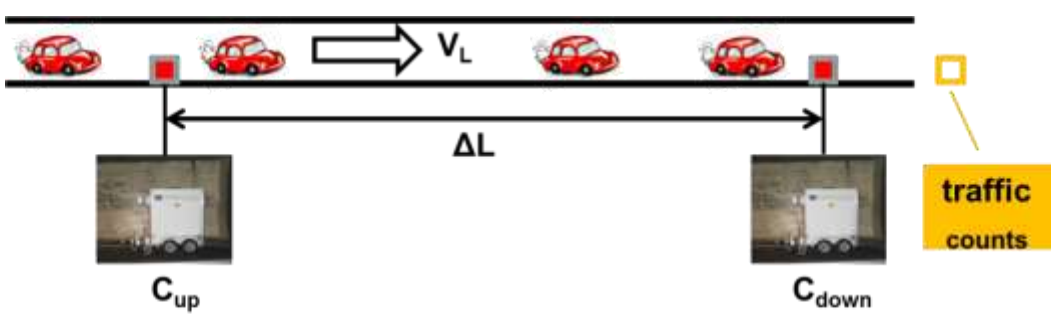

Figure 1: Concept of the tunnel emission monitoring set-up.

Multiple linear regression is typically used to split fleet average (*fl-avg*) total PM emissions between light vehicles (*LV*, i.e. cars and vans) and heavy duty vehicles (*HV*, i.e. coaches, trucks, tractor-trailer), see Equation 2. There, $Emis_{tot}$ denotes the total amount of emissions, N_{LV} and N_{HV} the number of light and heavy duty vehicles, respectively. Generally, considerable uncertainty may be introduced by splitting the fleet emission factors into vehicle type specific emission factors [5].

$$Emis_{tot} = EF_{LV} \cdot N_{LV} + EF_{HV} \cdot N_{HV} \tag{2}$$

These total PM10 emission factors for light vehicles and heavy duty vehicles are usually further split into exhaust and non-exhaust emission by subtracting the exhaust emissions from total PM10 vehicle emissions. The exhaust part is frequently specified by using emission models or emission data bases, e.g. HBEFA (2010) [6].

Table 1 presents a compilation of non-exhaust emission factors from tunnel monitoring studies and studies based on ambient air monitoring. The vehicle speed conditions of the cited studies are similar to the present study. The non-exhaust PM10 EF analysed in long tunnels appear to be significantly lower than non-exhaust EF from ambient air monitoring. In the almost 5 km long Islisberg tunnel in Switzerland air pollutants were measured at a distance $\Delta L = 4756$ m apart. Interestingly, the PM10 non-exhaust EF for LV analysed are actually negative -0.0025 g/km. The PM10 non-exhaust EF for HV is 0.003 mg/km (Meier et al., 2015) and is the lowest value in Table 1. Hinterhofer et al., (2015) obtained as well very low EF based on monitoring within the 10 km long Plabutsch tunnel near Graz. Differences in the non-exhaust emissions are related to the different emission models used to compute exhaust emission factors. In this study, the HBEFA [6] and the Network Emission model NEMO [7] were employed. With HBEFA lower exhaust emissions resulted and by subtracting from total PM10 EF, slightly higher non-exhaust emissions were computed (Table 1). The differences between the NEMO and HBEFA exhaust emission results are related to the assignment of a single traffic situation and single emission factor (HEBFA) versus emission mapping from cycle averaged engine power (NEMO).

Table 1: Compilation of PM10 non-exhaust emission factors (EF) for motorway, speed ~100 km/h conditions, LV are cars, vans and HV are buses, trucks, and tractor-trailer.

Study /Reference	Location/Conditions	Methodology	EF LV g/km/LV	EF HV g/km/HV
Meier et al. (2015) [3]	Islisberg-Tunnel (CH), 2014, ΔL = 4756, 100 km/h	Monitoring, exhaust HBEFA3.1	-0.003	0.003
Hinterhofer et al. (2015) [4]	Plabutsch-Tunnel (A) 2012, ΔL = 3380 m, 100 km/h	Monitoring, exhaust NEMO	0.007	0.044
		exhaust HBEFA 3.1	0.015	0.063
	Plabutsch-Tunnel (A) 2013, ΔL = 7180m, 100 km/h	Monitoring, exhaust NEMO	0.002	0.019
		exhaust HBEFA 3.1	0.010	0.050
Kristensson et al. (2004) [2]	Söderledstunnel Stockholm (S) ΔL = 595 m, 75-90 km/h, 5% HV; studded tires	Monitoring PM10 - PM0.9	fleet-avg	0.268
Bukowiecki et al. (2010) [5]	Reiden (CH), motorway, ambient, 120 km/h	monitoring down/upwind, HBEFA 2.1 AB_120 [8]	0.030 ±0.014	0.169 ±0.082
Gehrig et al. (2004) [9]	Humlikon (CH), motorway, ambient, 85 km/h	monitoring down/upwind PM10-PM1	0.022	0.144
	Birrhard (CH), motorway, ambient, 120 km/h	monitoring down/upwind	0.047	0.074
Düring et al. (2004) [10]	Analysis monitoring stations in Saxony (D), motorway, ambient	Monitoring & NOx tracer method/modelling, HBEFA, motorway AB_100	0.030	0.200
Schmidt et al. (2011) [11]	Analysis monitoring stations in Germany (D), motorway, ambient	Monitoring & NOx tracer method/modelling, HBEFA, motorway AB_100	0.022	0.130
	Analysis monitoring in tunnels (D)	Monitoring and adaptation towards HBEFA AB_100	0.010	0.200
	A1 Hamburg motorway (D)	Monitoring & NOx tracer method/modelling	0.010	0.264
	B10 Karlsruhe (D)	Monitoring & NOx tracer method/modelling	0.030	0.081

Generally, there is a big difference in the EFs illustrated in Table 1 between the first two tunnel studies evaluated EFs and the EFs evaluated from ambient down/upwind measurements and combined monitoring/modelling approaches. For the comparatively short Söderleds tunnel in Stockholm (Table 1) high fleet averaged emission factors were reported. However, these results were also affected by increased road wear due to studded tires [2].

On the one hand, traffic flow in road tunnels is relatively constant which may minimize brake, tyre and road wear. On the other hand, dry deposition velocities for coarse PM (2.5 to 10 µm) are high (typically 0.01 m/s [12][13]) compared to gaseous species such as NOx (typically 0.00016 m/s [12]. In tunnels the wall surface area is large and at close distance related to the PM emission sources which leads to high PM impingement at all surfaces. Consequently, PM is deposited at the surfaces and large fractions of the deposited material may be redistributed by traffic induced momentum and turbulence. A balance between deposition and resuspension may result, and a net sink process may prevail due to deposition. In that case the "idealized" concentration gradient within a long tunnel as assumed for the evaluation of EFs would not be constant as shown by the blue dotted line in Figure 2. Instead, the gradient would decline with increasing tunnel length as indicated by the green profile in Figure 2. Consequently, applying Equation 1 would result in underestimated EFs.

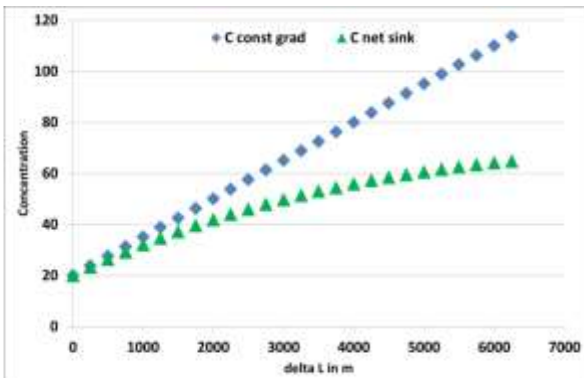

Figure 2: Schematic of the "idealized" concentration profile due to traffic emissions within a tunnel section in the absence of sink processes (blue diamonds) and an "idealized" concentration profile in the presence of a sink process (green triangles).

2 Objectives

The aim of this work is to analyse non-exhaust emissions as a function of mean traffic volumes and to what extent PM10 deposition as opposed to resuspension may play a role in the evaluation of emission factors.

3 Methodology

In this work, two data sets from air quality monitoring within the Plabutsch tunnel near Graz in Austria (Hinterhofer et al., 2014, Hinterhofer et al., 2015) were re-analyzed. The monitoring set-ups from both campaigns 2012 and 2013 are shown in Figure 3. In 2012 the air quality monitoring equipment was placed at two break-down bays located 3380 m apart, and in 2013 the equipment of the upstream location was moved further up, a distance ΔL of 7180 m resulted. In both campaigns the monitoring locations near the end of the tunnel were identical. The 2012 campaign took place from 24.08.12 to 07.09.12 and the 2013 campaign from 30.08.13 to 16.09.13 The Plabutsch tunnel is part of the A9 motorway, which bypasses Graz at its western periphery underneath the Plabutsch hill range. Both tunnel tubes are about 10 km long with unidirectional traffic. The monitoring took place in the eastern bore (direction towards Linz) using break-down bays to place the monitoring equipment. Approximately 20 000 to 30 000 vehicles pass through each tunnel bore per day. The tunnel has a gentle roof profile with slopes of 1 %. For the analysis, data were only used when the complex ventilation system was not operated, meaning that the vehicles pushed the air forward within the tunnel tube. The resulting vehicle induced longitudinal velocity (V_L) was monitored. Traffic volumes are operationally monitored outside the tunnel, 332 m after the north portal (Figure 3). All monitored data were recorded with one minute time resolution.

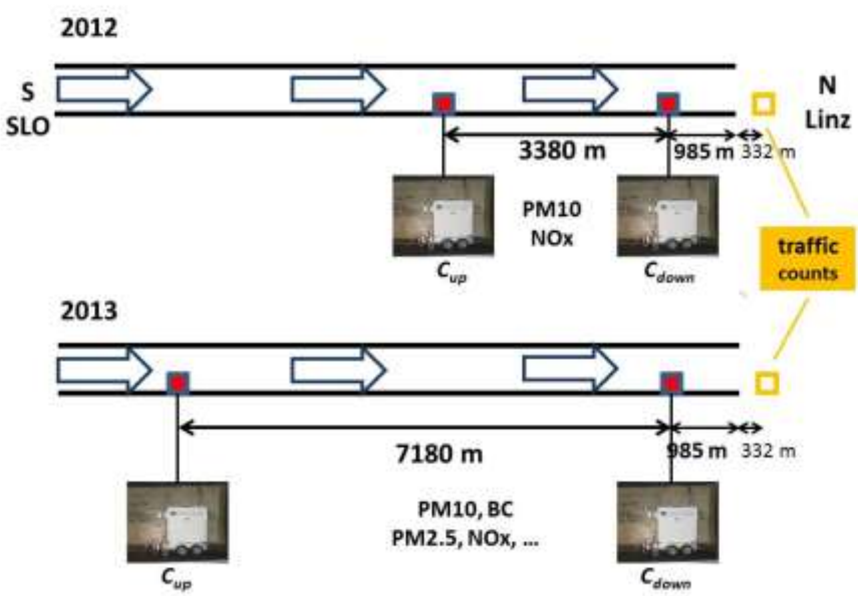

Figure 3: Schematic illustration of the monitoring set-ups used 2012 (top) and 2013 (bottom).

The basic approach is to bring together both data sets by grouping both of them into different traffic volume levels to complement the aforementioned approach (Equation 1) with an additional term accounting for coexisting PM wall deposition losses and resuspension from road surface, tunnel side walls and ceiling. Therefore, Equation 1 may be modified by a simplified approach to account for PM wall deposition and resuspension using an exponential term which describes the interplay between deposition and resuspension by using an "effective" wall deposition velocity V_{Deff} (Equation 3). If V_{Deff} is 0, all deposited PM10 is immediately resuspended and Equation 3 is identical to Equation 1. If V_{Deff} is > 0 depositional losses predominate and consequently the emission factor derived from the monitored difference between two monitoring locations would increase due to wall losses. W is the tunnel width and H the tunnel height.

$$EF_{fl-avg} = \left(\frac{C_{down}}{\exp\left(-2V_{Deff}\,(1/W + 1/H) \cdot \Delta L / V_L\right)} - C_{up} \right) \cdot \frac{V_L \cdot W \cdot H \cdot \Delta t}{\Delta L \cdot N_{veh}} \qquad (3)$$

According to Equation 3, the higher the effective deposition velocity or the higher emissions and concentration levels within the tunnel and the longer the distance between monitoring stations the stronger is the deposition effect.

4 Main Results and Discussion

In order to use the monitoring results of the two tunnel data sets, traffic counts were classified into 9 different daily traffic volume (DTV) ranges using 5 000 vehicles per day (veh/day) as step size (Table 2). For the chosen DTV ranges, mean DTV, mean measured concentrations and mean V_L were used for further emission analysis. In order to obtain reliable results only those data were taken into account, which fulfilled the following conditions: $\Delta NO_x > 100$ µg/m³, $\Delta PM10 > 5$ µg/m³, *number of LV passing the traffic counter ≥ 2 vehicles per minute*, and $V_L > 1.5$ m/s.

Table 2: Daily traffic (DTV) volume based classification, number of valid data and related mean parameters from Plabutsch campaigns ΔL = 7180 m (2013, top) and ΔL = 3380 m (2012, bottom) and NO_x was calculated as NO_2.

DTV range veh/day	N data [1]	DTV veh/day	%-HV	ΔNO_x µg/m³	ΔPM10 µg/m³	V_L [m/s]
< 5000	515	3574	26%	2309	62.0	4.2
5000 to 10000	1015	7726	17%	2859	94.8	4.9
10000 to 15000	1138	12291	14%	3225	116.1	5.3
15000 to 20000	673	17565	15%	3908	147.0	6.1
20000 to 25000	1015	22771	17%	4518	149.8	7.0
25000 to 30000	1755	27595	17%	5025	164.5	7.4
30000 to 35000	1831	32615	15%	5123	175.8	7.5
35000 to 40000	1457	37157	14%	5292	185.6	7.6
> 40000	712	42929	14%	5558	198.5	7.8

DTV range veh/day	N data [1]	DTV veh/day	%-HV	ΔNO_x µg/m³	ΔPM10 µg/m³	V_L [m/s]
< 5000	988	3151	29%	1318	10.5	3.9
5000 to 10000	1632	7532	15%	1510	32.8	4.5
10000 to 15000	1005	12187	13%	1585	33.2	5.1
15000 to 20000	540	17525	14%	1901	47.0	6.1
20000 to 25000	462	22833	17%	2179	55.7	7.1
25000 to 30000	712	27694	16%	2431	57.5	7.3
30000 to 35000	787	32288	14%	2432	62.6	7.3
35000 to 40000	323	36925	14%	2516	67.3	7.6
> 40000	114	42841	14%	2622	69.3	7.8

Figure 4 shows the distribution of light and heavy duty vehicles of the two data sets. Generally, the difference between the two data sets in DTV for all vehicles as well as DTV light vehicles is small (within ± 2.5 %) except for the smallest class (< 5 000 veh/day). The 2013 data set (ΔL = 7180 m) shows generally slightly higher numbers of heavy duty vehicles compared to the 2012 data (ΔL = 3380 m). Figure 5 shows the relative DTV differences of the two data sets. Significant differences (> 5%) in DTV and fleet composition are at the DTV ranges < 10 000 veh/day and 25 000 to 30 000 veh/day. Due to the lower number of total vehicles counted, comparatively small differences in DTV light vehicles and DTV heavy vehicles have a greater weight in the determination of emission factors. Particularly the results below 10 000 veh/day must be interpreted with caution.

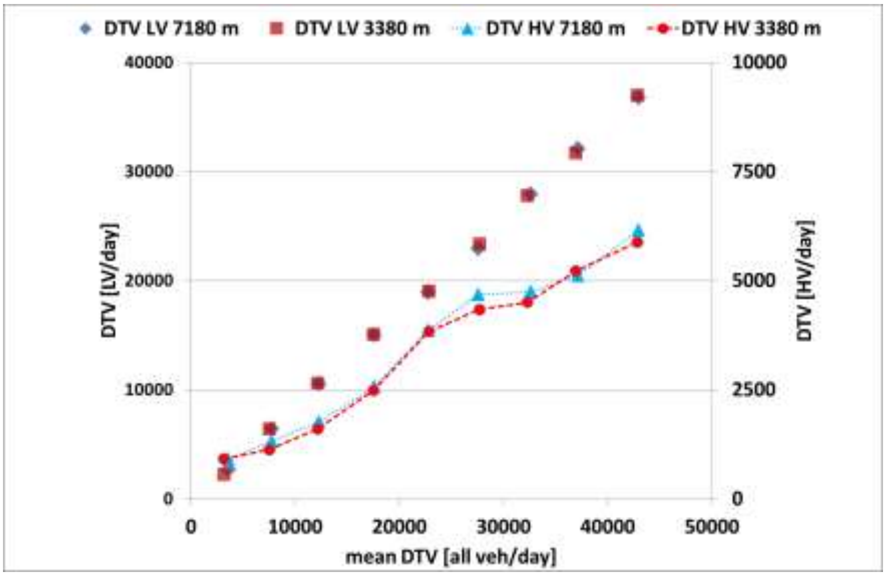

Figure 4: Distribution of daily traffic volumes (DTV) from the two data sets for light vehicles (LV, left y-axis) and heavy duty vehicles (HV, right y-axis) versus DTV all vehicles.

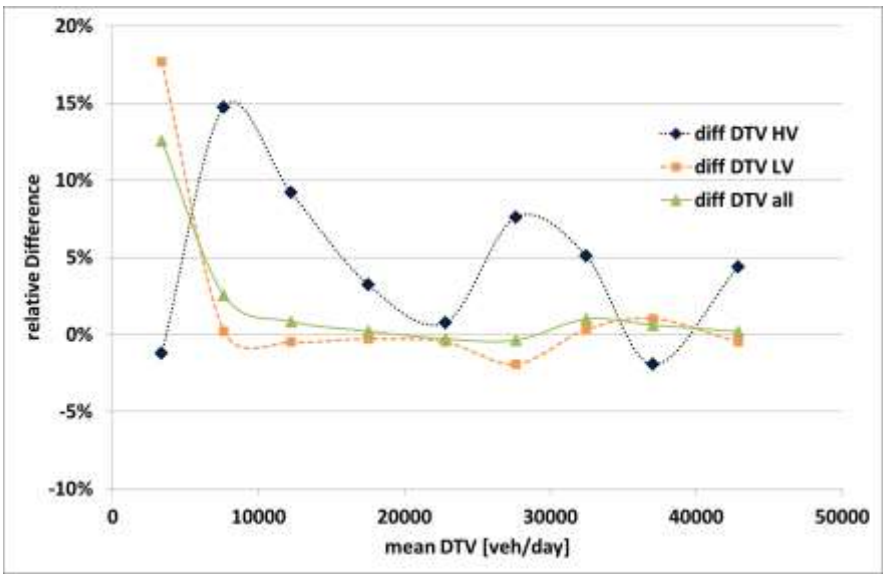

Figure 5: Relative differences (7180 m vs. 3380 m) LV, HV and all vehicles.

Figure 6 shows the airflow within the tunnel (V_L) due to the piston effect versus traffic volume. A strong functional relation was obtained with both data sets.
In conclusion, the traffic densities as well as V_L of the two classified data sets

match well and enable a combined analysis of EFs.

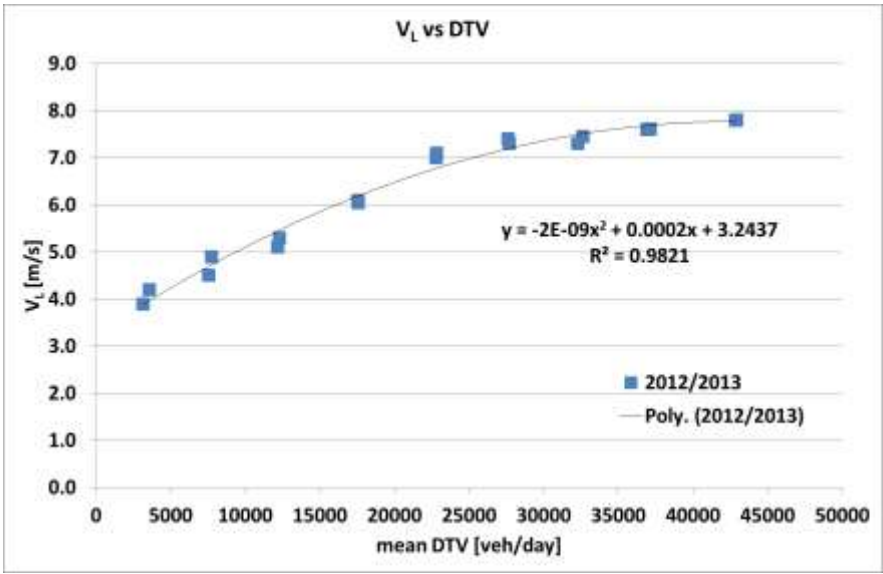

Figure 6: Averaged measured longitudinal velocity V_L within the tunnel versus traffic volume.

Mean NO_x and PM10 longitudinal concentration gradients were computed for the different traffic volume ranges of both campaigns and are shown in Figure 7.

The averaged NO_x gradients from both monitoring campaigns compare well. Averaged monitored NO_x concentrations gradients increase rather linearly with increasing traffic volume for both data sets. Deviations and the large intercept below 10 000 veh/day DTV may be attributed to differences in traffic volumes, fleet composition and may be related to higher dynamic driving behaviour (less constant) at low traffic densities.

Averaged monitored PM10 concentrations gradients show a strong increase towards low traffic volumes. Towards large traffic volumes the gradients increase only moderately. However, in contrast to the NO_x gradients, in tunnel PM10 concentration gradients for the two monitoring distances differ significantly. The concentration gradients derived for the shorter distance ($\Delta L = 3380$ m) near the rear end of the tunnel are weaker than the gradients derived for the long distance ($\Delta L = 7180$ m) indicating that with progressing tunnel length a net sink process has an effect on PM10 concentration profiles (cf. Figure 2).

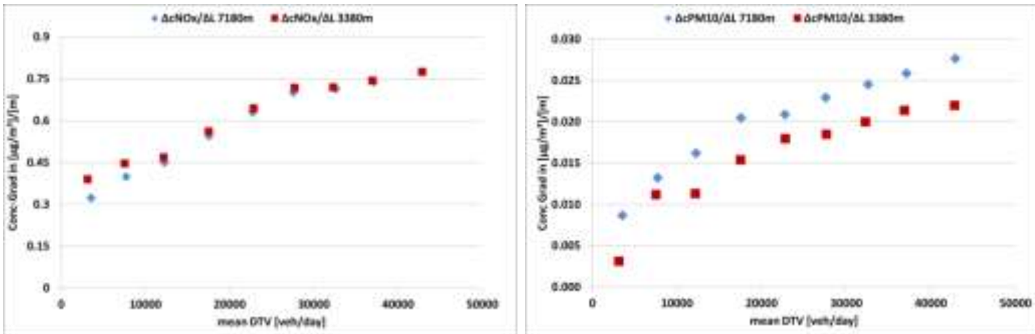

Figure 7: Mean longitudinal concentration gradients for NO_x (left) and PM10 (right) between the monitoring stations 7180 m and 3340 m apart.

Subsequently, EF results are presented and discussed for NO_x and PM10 neglecting deposition and resuspension (Equation 1) and accounting for deposition and resuspension (Equation 3).

4.1 Emission factors neglecting deposition and resuspension

Figure 8 shows fleet average NO_x and PM10 emission factors for both campaigns using the approach of Equation 1.

Daily traffic volume (DTV) related NO_x EFs from both campaigns match very well. Generally, NO_x EFs decrease with increasing traffic volume, highest EFs (1.05 g/km/veh in the 5 000 to 10 000 veh/day range and up to 2 g/km/veh below 5 000 veh/day) and the highest variability is related to low traffic volumes. For DTV greater than 10 000 veh/day total fleet average NO_x EFs range between 0.6 to 0.8 g/km and vehicle. The results obtained compare well with fleet average emission factors from HBEFA [6] 0.59 g/km/veh and NEMO [7] 0.90 g/km/veh.

The total PM10 fleet average EFs shown in Figure 8 (right) are low compared to cited emission factors from Kristensson et al., (2004)[2], Bukowiecki et al. (2010)[5], Gehrig et al., (2004)[9] and Schmidt et al., (2011) [11]. Emission factors determined for $\Delta L = 3380$ m are lower than those for $\Delta L = 7180$ m using Equation 1 which is related to the weaker gradients (Figure 7).

The total PM10 fleet average vehicle emissions (Figure 8) can be further specified by using multiple linear regression to split between light vehicles (LV, i.e. cars and vans) and heavy duty vehicles (HV, i.e. coaches, trucks, tractor-trailer). Here, the factors from the analysis by Hinterhofer et al., (2014) [14] were used. These factors relate to the analysis of the entire data set (no classification into different DTV levels). Therefore, the data for DTV < 5 000 vehicles/day must be scrutinized because the percentage of heavy duty vehicles deviates significantly from the average (cf. Table 2).

Total PM10 (exhaust and non-exhaust) emission factors for light vehicles are shown in Figure 9 (left) and heavy duty vehicles (right). Also modelled exhaust emission factors are shown. NEMO [7] was used for these computations. By subtracting modelled exhaust EFs from total PM10 LV and HV emission factors, finally non-exhaust emission factors for LV and HV can be obtained (not shown here) which are extremely low. Several data points are below the modelled ones which would imply negative non-exhaust emission factors.

Within the 2013 campaign, black carbon was additionally monitored over three days. Black carbon can be regarded as a well suited indicator for exhaust emissions. On average, 18 % of the total PM10 sampled consisted of black carbon [14]. As a consequence non-exhaust EFs which are expected to be composed predominantly by mineral PM components should exhibit much higher PM10 per km to be in line with the results presented so far.

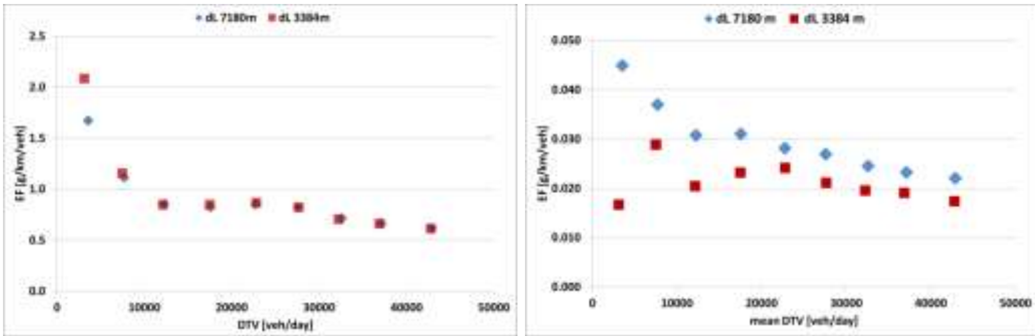

Figure 8: Fleet average EF NO_x (left) and total EF PM10 (right) using Equation 1.

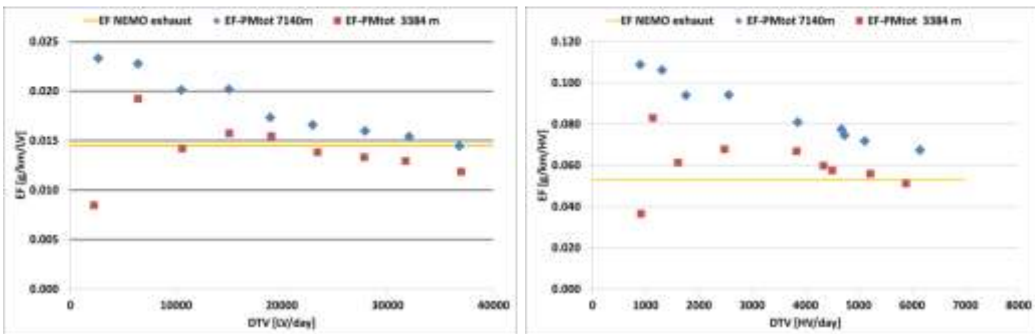

Figure 9: PM10 total EF for light vehicles (LV, left) and heavy duty vehicles (HV, right) versus LV/HV traffic volume using Equation 1. The NEMO exhaust EF is also shown.

4.2 Emission factors accounting for deposition and resuspension

Here, Equation 3 was employed and the effective deposition velocity V_{Deff} was adjusted to obtain a match of the fleet average emission factors of both data sets (Figure 10). An effective deposition velocity V_{Deff} of 0.001 m/s was used for all traffic volume ranges of both data sets (ΔL = 7180 m in 2013, ΔL = 3380 m in 2012). The emissions factors per DTV range agree well for both data sets. The derived emission factors decrease with increasing traffic volume. The total PM10

fleet average emission factors shown in Figure 10 are significantly increased compared to the corresponding EFs (Figure 8 right) for which deposition and resuspension was neglected.
The adjusted effective deposition velocity is small. For particles, deposition velocities vary with particle diameter [12],[13], for PM10 typically a value of 0.01 m/s is frequently applied [12]. Compared to the deposition velocity of NO 0.00016 m/s [12] the adjusted effective deposition velocity of 0.001 m/s is considerably higher. Neglecting resuspension processes and setting V_{Deff} to 0.01 m/s would result in extremely large i.e., unrealistic emission factors. Moreover, the match of the EFs of both data sets within the DTV ranges (Figure 10) would disappear.

At low traffic volumes (< 10 000 veh/day DTV) high fleet averaged total PM10 EFs are obtained. At low traffic volumes e.g. during night time, there is more time for PM fractions larger than 10 µm for gravitational settling. Hence, the dust loading at the road surface would be increased compared to conditions with high traffic volumes (> 20 000 veh/day DTV). Moreover, there is more time for PM10 to deposit. Therefore, the higher EFs found at lower traffic volumes may be explained by increased abrasion and resuspension per passing vehicle.
However, at the lowest traffic density (< 5 000 veh/day) the shares of heavy duty vehicles are significantly higher than within the other traffic volume ranges and there is significant difference in DTV light vehicles (Figure 5). Due to the low traffic volume, comparably low mean longitudinal velocities with high variation result. Hence, the uncertainties can be expected as larger.
The emission factors presented relate to dry conditions. Due to the length of the Plabutsch Tunnel, only near the entry portal wet conditions within the tunnel may be encountered with heavy rainfall. The tunnel section where the monitoring took place remained practically always dry.

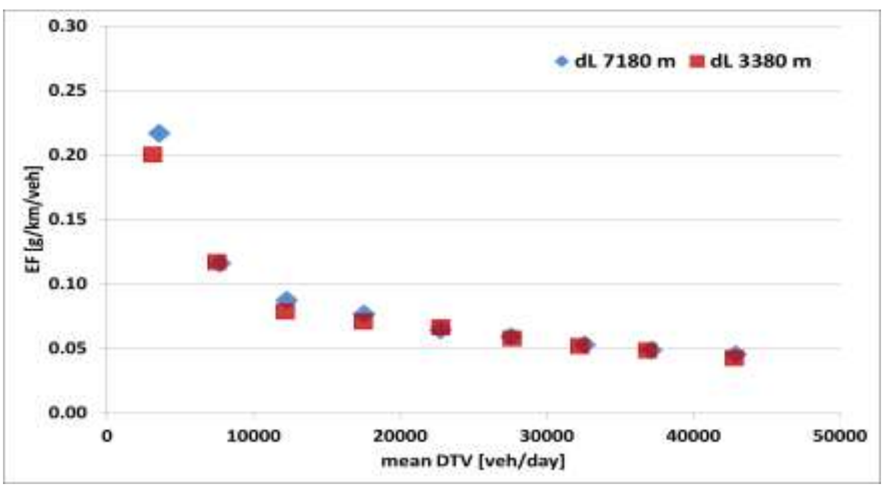

Figure 10: Deposition and resuspension corrected total PM10 fleet average vehicle EF versus traffic volume.

The total PM10 fleet average vehicle emissions (Figure 10) can be further specified to split between light vehicles (LV, i.e. cars and vans) and heavy duty vehicles (HV, i.e. coaches, trucks, tractor-trailer) using multiple linear regression analysis ((Equation 2). NEMO [7] was used to compute PM10 exhaust emission factors which were subtracted from total PM10 LV and HV emission factors. The PM10 non-exhaust emission factors for LV and HV are shown in Figure 11.

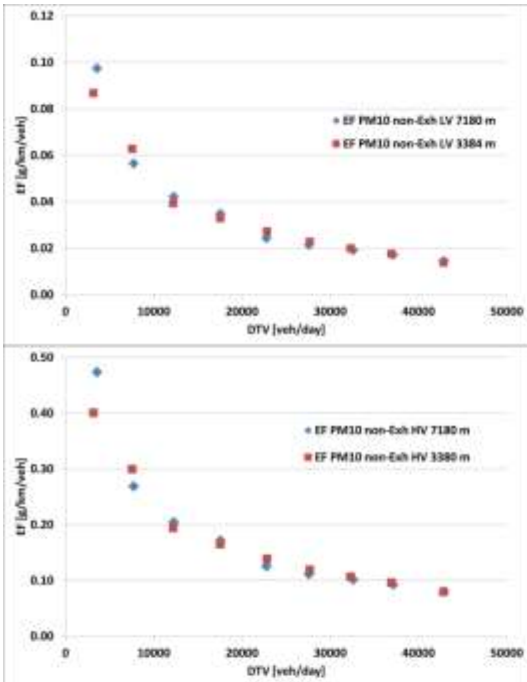

Figure 11: PM10 non-exhaust EF for LV (left) and HV (right) versus traffic volume.

PM10 non-exhaust EF for light vehicles (LV) range from 0.014 g/km/veh (at > 40 000 DTV to 0.063 g/km/vehicle (~7500 DTV). Average PM10 non-exhaust EF for heavy duty vehicles range from 0.079 g/km/HV (> 40 000 DTV) to 0.30 g/km/HV (~7500 DTV). Higher EFs are obtained for DTVs below 5 000 veh/day. However, at low traffic volumes the uncertainties are larger.

These results can be also related to LV and HV traffic densities and can be compared to PM10 non-exhaust EFs obtained from other studies with comparable traffic situations. Figure 12 shows the PM10 non-exhaust EFs for light vehicles obtained from this study, and studies cited in Table 1. Figure 13 shows the corresponding results for heavy vehicles. PM10 non-exhaust EF-LV range from 0.015 g/km/LV (~35 000 LV/day) to 0.060 g/km/vehicle (~6 500 LV/day). Average PM10 non-exhaust EF for heavy duty vehicles range from 0.080 g/km/HV (~6 000 HV/day) to 0.300 g/km/HV (~1 500 HV/day). The determined PM10 non-exhaust EF for LV and HV increase with decreasing traffic volume.

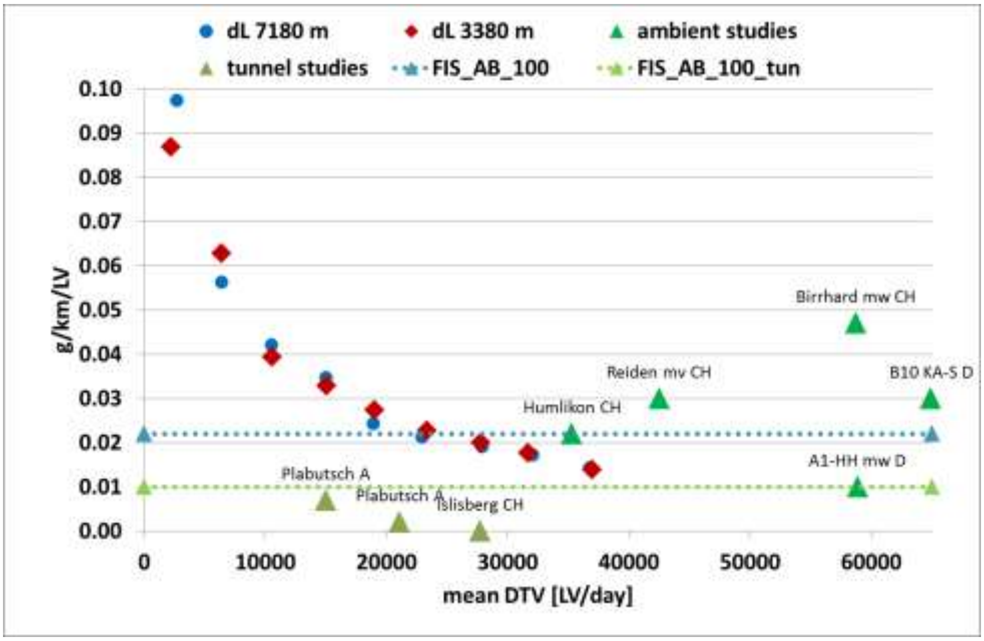

Figure 12: Compilation of PM10 non-exhaust EF for LV, from this study (indicated by dL 7180 m and dL 3380 m), tunnel studies Islisberg [3], Plabutsch [4], ambient studies Reiden motorway [5], Humlikon and Birrhard motorway [9], A1 motorway Hamburg, B10 expressway Karlsruhe [11], FIS_AB_100 and FIS_AB_100_tun are EFs based on HBEFA traffic classification for motorway with speed 100 km/h (AB_100) and motorway tunnel with speed 100 km/h [11].

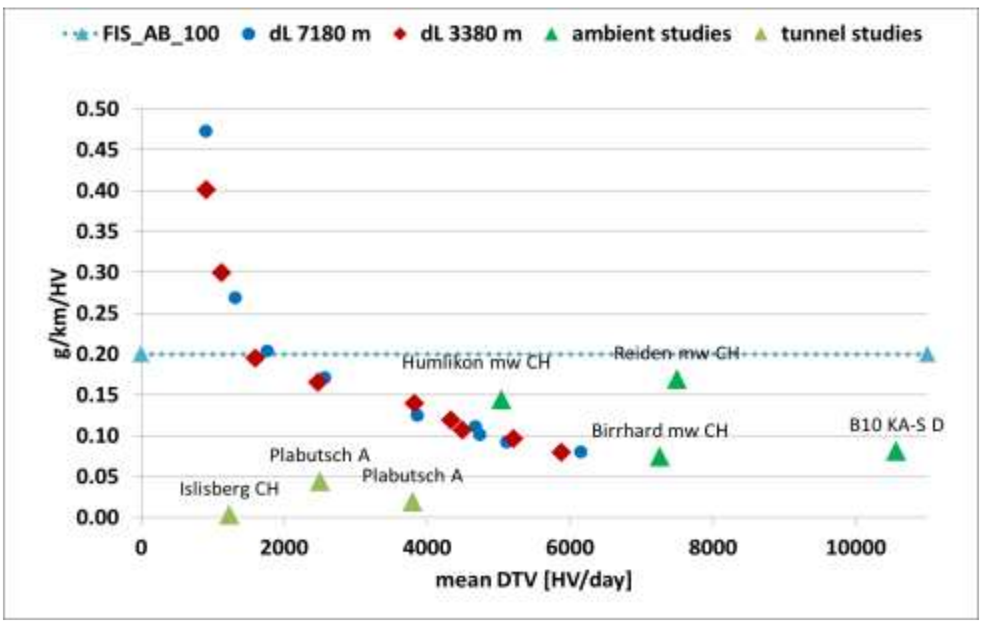

Figure 13: Compilation of PM10 non-exhaust EF for heavy vehicle (HV) for this study and various other studies, see Figure 12 and Table 1. The emission factors for motorway and tunnels with speed limit 100 km/h are identical (FIS_AB_100, 0.2 g/km/HV).

The results for the LV non-exhaust EF obtained in this work are from 15 000 LV/day onwards on a comparable level with results from various ambient studies as illustrated in Figure 12. Non-exhaust emission factors for light vehicles below 10000 LV/day are comparable to levels found near Birrhard at a Swiss motorway (0.047 g/km/LV, [9]) or a very busy road intersection in Zürich (0.045 g/km/LV, [15]). HV Non-exhaust EF for traffic volumes larger than 2 000 HV/day are on a comparable level with results from various ambient studies as illustrated in Figure 13, i.e. the studies cited in Table 1 EF results for heavy duty vehicles around 1 000 HV/day are on the same level as analysed for a street canyon in Zürich (0.498 g/km/HV) by Buckowiecki et al., (2010) [5].

5 Conclusion

Two data sets from two air pollutant tunnel measurement campaigns in a 10 km long tunnel have been analysed to study the potential role of traffic volume and PM10 deposition interacting with resuspension on non-exhaust emission factors.

Ideally, in order to measure in tunnels non-exhaust emission factors and to account for simultaneous PM deposition and resuspension effects, at least three monitoring locations equipped with identical monitoring devices should be operated at the same time in a tunnel. Here, data from two monitoring campaigns in a 10 km long tunnel with different distances between the air quality monitoring stations (3380 m/7180 m) have been re-analysed. In order to use the monitoring results of the two tunnel measurements campaigns, traffic counts were classified into different DTV ranges and all measured parameters were averaged.

Neglecting sink processes in the determination of non-exhaust emission factors resulted in unrealistically low PM10 non-exhaust emission factors. In-tunnel PM10 concentration gradients from two different campaigns in the same tunnel indicate stronger PM10 gradients at the front section compared to the rear section at comparable traffic volumes. In contrast, NO_x gradients matched well from both campaigns and NO_x EFs compared well with HBEFA and NEMO computed ones.

Therefore, the impact of a correction for mass losses due to wall deposition was hypothesised using an effective deposition velocity. A small effective deposition velocity of 0.001 m/s, i.e. approximately one-tenth of the PM10 deposition velocity found in literature would significantly increase the non-exhaust emission factors derived from two monitoring stations in tunnels with long distances apart. This correction would imply different concentration gradients for PM10 and a larger share of non-exhaust PM10. Moreover, a better agreement with black carbon measurements as an indicator for the PM10 exhaust fraction is obtained. On average, 18 % of the total PM10 sampled consisted of black carbon.

The determined PM10 non-exhaust EFs for LV and HV decrease with increasing traffic volume. PM10 non-exhaust EFs for light vehicles range from 0.015 g/km/LV (~35 000 LV/day) to 0.060 g/km/LV (~6 500 LV/day). Average PM10

non-exhaust EFs for heavy duty vehicles range from 0.080 g/km/HV (~6 000 HV/day) to 0.300 g/km/HV (at ~1 500 HV/day). At low traffic volumes e.g. during night time, there is more time for PM fractions larger than 10 μm for gravitational settling and particularly for coarse PM (2.5 to 10 μm) to deposit. Hence, the dust loading at the road surface would be increased compared to conditions with high traffic volumes (> 20 000 DV). Therefore, emissions due to increased abrasion and resuspension per vehicle may result.
Finally, a correction term to account for deposition and resuspension of PM10 in tunnel EF studies is proposed.

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