**No. C 429**
June 2019

# Improving cabin air quality in road vehicles

## *Vägar till förbättrad luftkvalitet i fordon*

Jana Moldanová, Rasmus Parsmo, Sarka Langer, Aleksandra Sebastian, Maria Bernander, Andreas Andersson, Anna Dahlman-Höglund, Bo Strandberg, Håkan Sahlberg, Sara Jutterström, Anne-Marie Rydström

**In cooperation with** Volvo Cars, AB Volvo and Occupational and Environmental Medicine, University of Gothenburg

**Author:** Jana Moldanová, Rasmus Parsmo, Sarka Langer, Håkan Sahlberg, Sara Jutterström, Anne-Marie Rydström, IVL; Aleksandra Sebastian, Maria Bernander, Andreas Andersson, Volvo Cars; Anna Dahlman-Höglund, Bo Strandberg, Occupational and Environmental Medicine, University of Gothenburg
**Funded by:** Energimyndigheten, forskningsprogram Fordonsstrategisk Forskning och Innovation (FFI), delprogram Energi och miljö
**Report number** C 429
**ISBN** 978-91-7883-083-1
**Edition** Only available as PDF for individual printing

IVL Swedish Environmental Research Institute Ltd.
P.O Box 210 60, S-100 31 Stockholm, Sweden
Phone +46-(0)10-7886500 // www.ivl.se

# Final report

Date
2019-06-xx

Dnr
2014-007235

Projectnr
40255-1

| | |
|---|---|
| Energimyndighetens titel på projektet – svenska<br>Vägar till förbättrad luftkvalitet i fordon (AQIFOR) | |
| Energimyndighetens titel på projektet – engelska<br>Improving cabin air quality in road vehicles (AQIFOR) | |
| Universitet/högskola/företag<br>IVL, Svenska Miljöinstitutet | Avdelning/institution |
| Adress<br>Box 530 21, SE-400 14 Göteborg, Sweden | |
| Namn på projektledare<br>Jana Moldanová | |
| Namn på ev. övriga projektdeltagare<br>Rasmus Parsmo, Sarka Langer, Håkan Sahlberg, Sara Jutterström, Anne-Marie Rydström (IVL), Maria Bernander, Aleksandra Sebastian, Andreas Andersson (Volvo Cars), Olof Willstrand (AB Volvo) | |
| Nyckelord:<br>Air pollution in vehicle, air cleaning, real-drive testing methodology | |

# Preface

The project "Improving cabin air quality in road vehicles (AQIFOR)" has been financed by the Swedish Energy Agency, the research program Strategic vehicle research and innovation (FFI), Energy and Environment, which is a partnership programme run jointly by the Swedish state and Swedish automotive industry. The project consortium involved research partners from IVL, Swedish Environmental Research Institute and Occupational and Environmental Medicine at the University of Gothenburg and industrial partners from Volvo Cars and AB Volvo.
The industrial partners funded their part with both funding from the Energy Agency and with in-kind. The project was supported by an advisory board with representatives from the partnering organisations management groups and from external organisations: Karin Sjöberg, IVL, Gunilla Karlsson, Volvo Cars, Lars Mårtensson, AB Volvo, Martin Juneholm/Hung Nguyen, Swedish Transport Administration, Christer Johansson, Department of Environmental Science and Analytical Chemistry (ACES) and Carin Ström, Environmental board (Miljöförvaltningen), City of Gothenburg, from 2017 IVL.

# Table of contents

# Summary

The purpose of the project was to understand the most important sources and amount of air pollution in vehicle compartments, with focus on particles. Along with investigating how the design of the vehicle's ventilation system and air purification equipment affects the air quality in the vehicles and to develop a robust and effective methodology for measurement in traffic environments. Within five extensive full-scale and a number of smaller, scaled-down measurement campaigns, levels of air pollution were measured both inside and outside the vehicle cabin, along with comfort parameters while driving in real traffic. The experiments were mainly carried out on a route in Gothenburg, which contained both city driving, highway driving and stationary in a road tunnel. The measurements were carried out with online instruments and with pumped samples on filters or absorbents. Online measurements were used for measuring concentrations of size-resolved particles, $NO_2$, NO and $CO_2$, temperature and humidity, along with pumped samples were for $PM_{2.5}$ mass, VOC and PAH.

The project showed, in accordance with previous studies, that the most important parameters affecting the air quality in vehicles are the air filters and air quality outside the vehicle. The experiments showed repeatable differences between the reduction efficiency of the different filters used in the ventilation systems. Additional parameters that affected the reduction efficiency were the ventilation rate and the degree of recirculation of the air in the vehicles. The outside concentration affected the absolute concentrations inside the vehicles, however did not show any great influence on the reduction efficiency. On the other hand, at very low concentrations outside the vehicle, sources and resuspension of particles inside the vehicle contributed significantly and the reduction efficiency decreased.

Different air pollutants were affected differently by the ventilation and the filter used. All tested filters reduced the number of particles; reduction efficiency measured for $PM_{2.5}$ was from 40% to over 80% for the different filters. Testing without filter showed no statistically significant reduction efficiency. The experiments showed that particles in the size range of 10 - 600 nm were more difficult to remove than the larger particle sizes, and filters with higher efficiency also showed higher efficiency in the small size range. Filters containing activated carbon also purified the air from $NO_2$, reduction efficiency showed similar patterns to particle reduction efficiency. Sampling for VOC and PAH showed that very volatile substances were not removed by filters, however the less volatile were effectively reduced by these carbon filters.

In addition to the driving experiments so-called recirculation experiments were carried out, measuring the reduction of air pollution levels in the vehicle during recirculation of the air. These experiments were carried out while standing in the tunnel in order to get stable and high outside concentrations of air pollutions. From the decay curves, decay rate constants were calculated these relate to the reduction efficiency and can e.g. be used in developing a dynamic model that describes air quality in the vehicle. Decay constants calculated from the recirculation experiments were comparable to the reduction efficiency measured during driving experiments.

The project also investigated how professional drivers were exposed to air pollution in both passenger cars and trucks. Exposure measurements were performed on 21 drivers measuring PAH, VOC and $PM_{2.5}$ during a full working session. The exposure was measured using pumped samples. The results showed that the drivers' exposure to benzo (a) pyrene, 1,3-butadiene and benzene was low, well below the occupational exposure limits.

# Sammanfattning

Syftet med projektet var att förstå de viktigaste källorna till och omfattningen av luftföroreningar i fordonskupéer med betoning på partiklar, att undersöka hur utformningen av fordonets ventilation och luftreningsutrustning påverkar luftkvaliteten i fordon och att utveckla robusta och effektiva verklighetsbaserade testmetoder för mätning i trafikmiljöer. Inom fem omfattande fullskaliga och ett antal mindre, nerskalade mätkampanjer har vi parallellt mätt halter av luftföroreningar och komfortparametrar i och utanför fordonet under körning i verklig trafik. Körexperimenten utfördes huvudsakligen på en körslinga i Göteborg vilken innehöll både stadskörning, körning på motorväg och perioder med stillastående i en vägtunnel. Mätningarna utfördes med online instrument och med pumpad provtagning på filter eller absorbent. Online-mätningar gjordes för storleksupplösta partikelkoncentrationer, $NO_2$, NO och $CO_2$ -halter, temperatur och luftfuktighet; pumpade prover samlades för $PM_{2.5}$, VOC och PAH.

Projektet visade, i överenstämmelse med tidigare studier, att de viktigaste parametrar som påverkar luftkvaliteten i fordon är val av kupéluftfilter samt luftkvaliteten utanför fordonet. Experimenten visade repeterbara skillnader mellan reduktionseffektivitet av ventilationssystemens olika luftfilter. Ytterligare parametrar som påverkade reduktionseffektiviteten för luftföroreningar i fordonet var ventilationshastighet samt grad av recirkulation av luften i fordonen. Ute-koncentrationen påverkade de absoluta koncentrationerna inne men hade inte någon större påverkan på reduktionseffektiviteten. Vid väldigt låga koncentrationer ute, då källor i fordonet samt resuspension av partiklar utgjorde signifikanta bidrag, minskade dock reduktionseffektiviteten.

Olika luftföroreningar påverkas olika av ventilationen och filtret. Alla testade filter renade luft från partiklar; reduktionseffektiviteten för $PM_{2.5}$ var från 40% till över 80% för olika filter. Tester utan luftfilter visade ingen statistisk signifikant reduktion. Experimenten visade att partiklar i storleksområdet 10 – 600 nm var svårare att rena än större partiklar och filter med högre effektivitet utmärkte sig med högre effektivitet just i det storleksområdet. Filter som innehöll aktivt kol renade även luften från $NO_2$; reduktionseffektivitet visade på liknande mönster som reduktionseffektivitet för partiklar. Provtagning för VOC och PAH visade att flyktiga ämnen inte renas av filter men de mindre volatila reduceras effektivt av luftfilter.

Körexperimenten kompletterades med s.k. recirkulationsexperiment där man mätte minskning av halter av luftföroreningar i fordonet under recirkulation av ventilationsluft. Dessa experiment utfördes stående i en tunnel. Från avklingningskurvor beräknades avklingningshastighetskonstanter som relaterar till reduktionseffektivitet och kan t.ex. användas i en dynamisk modell för att beskriva luftkvaliteten i fordonet. Hastighetskonstanter beräknade från recirkulationsexperiment var jämförbara med reduktionseffektiviteten uppmätta under körexperimenten.

Projektet undersökte också exponering för luftföroreningar av yrkeschaufförer i både personbilar och lastbilar. Exponeringsmätningar utfördes på 21 chaufförer för PAH, VOC och $PM_{2.5}$ under ett fullt arbetspass. Exponeringen mättes med hjälp av pumpade prover. Resultaten visade att chaufförernas exponering för benzo(a)pyren, 1,3-butadien och bensen var låga, långt under gränsvärden för arbetsmiljö.

# 1. AQIFOR background

Sweden has for a long time been one of the countries that lead the technical development in vehicle safety. Over the past decade awareness has increased regarding the adverse effects of air pollution on human health among researchers, legislators and the general public. Consequently, the technological development in the automotive industry faces new challenges in the form of increased demand for solutions that improves air quality inside the vehicle. The rapid development in this area involves development of both technical solutions for air cleaning along with methods to verify it.

The indoor air quality in vehicles is mainly associated with the air quality outside the vehicle. The air pollutants in question are mostly particles and particulate compounds, elemental carbon, organic carbon, polycyclic aromatic hydrocarbons (PAH) and metals along with volatile organic compounds (VOCs, e.g. benzene, 1,3-butadiene, aldehydes) and in some cities also carbon monoxide (CO).

The amount of time people spend in their cars differs from professional drivers spending significant time in the car, to people not using their car for work having a daily average driving time of 1-2 hours in Europe (Pasaoglu at al., 2012) and 5% of time, corresponding to 1.2 hours daily, in the US (Klepeis et al., 2001). The levels of air pollutants in a vehicle cabin are affected by several factors. In modern vehicles the air enters from the outside into the interior through active ventilation. Therefore, the type of filter solution in the ventilation system and the way it is operated are the most important parameters affecting the air quality. Design and technology in the ventilation system are therefore of great importance for the air quality in the vehicle. Other important factors are emissions from materials in the vehicle as well as chemical and physical processes, such as photochemical reactions in the gas phase and on surfaces as well as deposition within the cabin.

Today, ventilation components are tested in a laboratory environment by a standardized bench-test using air pollutants such as mineral dust and salt particles along with some gases. Field tests of the ventilation systems involve real-life driving at different locations leading to tests being performed in a wide range of different mixtures and concentration levels of air pollutants which provides a valuable complement to the laboratory testing.

The aims of the AQIFOR project were to

- Increase the understanding of:
    - Processes that affect concentrations of air pollutants in a vehicle
    - Exposure of drivers and passengers to these pollutants and their impact
- Develop a robust measurement methodology for evaluation of functionality of the air cleaning equipment and to utilize the knowledge gained in the project in technological innovations that can be used in the vehicles.

# 2. Current state of the knowledge

Exposure to air pollutants occurs mainly indoors where people spend majority of their time (87%). That environment is most often associated with dwellings, workplaces and schools. However, one important microenvironment is transportation – motor vehicles. The time people spent in vehicles is approximately 5%, varying somewhat between different countries (Klepeis et al, 2001). The indoor air pollution in vehicles is mainly associated with the air pollutants originating from outdoor air, such as carbon monoxide (CO), oxides of nitrogen NO and $NO_2$, volatile organic compounds (VOC), polycyclic aromatic hydrocarbons (PAH) and particles. Interior materials are sources of VOC, semi-volatile VOC (SVOC) and carbonyl compounds such as formaldehyde. Passengers/drivers contribute with $CO_2$ and VOC (e.g. isoprene), originating from exhaled air.  A recent review article by Xu et al. (2018) summarizes the occurrence of the above-mentioned pollutants inside vehicles, also including microbes. Recirculation of air and a use of high efficiency filters have been identified by the authors as the most effective measure to lower the air pollutant concentrations inside vehicles. The article also concludes that future work should focus on investigating the health risks of exposure to various air pollutants inside different vehicles and further development of advanced air filters to improve the in-cabin air quality.

A study that took the health effects into consideration is Bekö et al. (2015). The article describes the contribution of various microenvironments to daily exposure of ultrafine particles (UFP). The total exposure is a combination of time spent in various environments and the respective concentrations of the air pollutant in question. Although the highest concentrations of UFP were measured in vehicles, only 5% of the daily exposure time, i.e. 1.2 hours, occurred during the passive transportation.

To further understand how different parameters influences the air quality inside a vehicle, certain parameters needs to be discussed. Descriptions of such parameters can be found below along with the state-of-art knowledge on their typical ranges in vehicle cabins and relevant effects.

### Air exchange rate

One important factor that influences the air quality in a vehicle cabin is the Air Exchange Rate (AER), which reflects the number of times the cabin air volume is replaced by hour as controlled by the ventilation setting. Ventilation serves to control the thermal comfort such as temperature and relative humidity along with mist formation, it cleans the incoming air from outside pollution along with removing substances emitted from the interior and drivers/passengers by the air flow. The AER strongly affects indoor-to outdoor (I/O) ratios of air pollutants in the vehicle cabins (Knibbs et al., 2010; Hudda et al., 2011.)

The AER depends mainly on ventilation mode, but also on the vehicle's speed and age. In modern vehicles with windows closed, the air enters from the outside into the interior through active ventilation. However, besides the ventilation system, the outside air pollutants may also enter, mainly in older, less air-tight vehicles by air leakage (Knibbs et al., 2009a). AER in cars has been experimentally determined using concentration decay of trace gases such as CO, $SF_6$ or passenger exhaled $CO_2$ (Ott et al., 2008; Knibbs et al., 2009a; Fruin et al., 2011). The AER in passenger vehicles reported in the literature vary between $< 1$ $h^{-1}$ and 145 $h^{-1}$. For comparison, the building codes of many countries require the nominal air flow of fresh air in l/s per $m^2$ of floor area which in most dwellings corresponds to an AER of approximately 0.5 $h^{-1}$ (BBR 21, 2014).

## Main constituents of air pollution in vehicles:

### Carbon monoxide

Carbon monoxide is a product of incomplete combustion of fuels and may be present in elevated concentration in the vehicle exhaust gases, leading to the possibility of it entering the vehicle cabin. Carbon monoxide has been proven to be toxic to human health in concentrations of 50 ppm and higher. In literature, reported CO concentrations inside vehicles span from 1 ppm to 38 ppm, with the variation mainly reflecting the outside air concentration (Abi-Esber and El-Fadel, 2013). Intrusion from the engine combustion and/or exhaust return is a likely explanation to elevated CO concentrations in cars (Abi-Esber and El-Fadel, 2008; Harik et al., 2017). The WHO (2010) guidelines for indoor air quality specify the time averaged value for CO as 35 mg/m$^3$ = 30 ppm (1-hour average), 10 mg/m$^3$ = 9 ppm (8-hour average) and 7 mg/m$^3$ = 6 ppm (24-hour average). Neither modern petrol cars with 3-way catalyst nor modern diesel cars are expected to emit significant amounts of CO from their exhaust.

### Carbon dioxide

Carbon dioxide is a pollutant with several different sources, both natural and man-made. Among the man-made major sources are breathing, cement production, deforestation and burning of fossil fuels like coal, oil and natural gas. The major input to in-vehicle concentration of carbon dioxide is primarily the ventilation settings (outside air/recalculation) and the number of passengers (Silvergren et al., 2013). With the ventilation in the outside air mode, $CO_2$ concentration in a vehicle cabin reflects the outside air concentration (approx. 400 – 600 ppm in vehicle on road and elevated in tunnels). With recirculation mode, $CO_2$ from the exhaled air from the driver and passengers will stay inside the vehicle. Recirculation mode should not be used for prolonged periods of time to prevent $CO_2$ build up inside vehicles (Jung et al., 2017; Hudda and Fruin, 2018). Elevated carbon dioxide ($CO_2$) levels (above 5 000 ppm) evoke a decline in driver's concentration (Gladyszewska-Fiedoruk, 2011).

### NO and $NO_2$

The largest sources of emissions to air of nitrogen oxides are road transport and energy production. Nitrogen dioxide effects respiratory health; at levels about 400 µg/m$^3$ asthmatics exhibit small pulmonary function decrements (WHO, 2010). In the combustion processes NO and $NO_2$ (NOx) are formed, NO being the largely dominating species. Both are thus present in vehicle exhaust. When entering the ambient atmosphere, NO is in a fast rate oxidized by ozone to $NO_2$ which increases the $NO_2$/NO ratio in the ambient air comparing to that in the exhaust. Reduction of emissions of NOx has been the main target of emission legislation formulated in the later Euro classes (corresponding Tier classes in the US) and currently typical emissions of Euro 6 passenger car is 60-80 mg per vehicle kilometre (petrol and diesel passenger cars, respectively). The actual concentrations in vehicles depend on the surrounding environment, filters used ($NO_2$ can be captured if a carbon filter is used), and ventilation settings; the air recirculation mode are slightly protective towards elevated levels of both NO and $NO_2$.

The reported concentrations of NO and $NO_2$ inside vehicles were in the range of tens to hundreds of ppb (Chan et al., 1999; Zagury et al., 2000; Chan and Chung, 2003; Riediker at al., 2003; Johansson et al., 2013). Data from urban background monitoring stations cannot be used for estimation of the $NO_x$ concentrations inside vehicles as concentrations in the street canyons are much higher and the NO to $NO_2$ ratio is completely different (Zagury et al., 2000). The WHO (2010) indoor air quality guideline for $NO_2$ is 40 µg/m$^3$ (annual average).

### Particles: $PM_{10}$ and $PM_{2.5}$

Particles originate from both natural and man-made sources, among the man-made sources are combustion processes, construction and agriculture. Along with that particles can be formed in the atmosphere as secondary aerosol. Association between the mass of particulate matter $PM_{10}$ and $PM_{2.5}$ and negative health effects (morbidity and mortality, increased risk for myocardial infarction, lung inflammation) is well established in the literature and recognized by WHO (WHO 2016 and references there in). A decrease in annual mean concentration of $PM_{2.5}$ of 10 µg/m³ can decrease the risk of heart disease by 6%.

The reported $PM_{10}$ concentrations in vehicles were in the range of several to tens of µg/m³ (Lewné et at., 2006; 2007; Briggs et al., 2008; Gulliver and Briggs, 2004). Concentration of $PM_{2.5}$ inside vehicles were reported in the range of tens to hundreds µg/m³ (Adams et al, 2001a; 2001b; Boogaard et al., 2009; Knibbs and de Dear, 2010; Riediker et al., 2003). The actual concentrations in vehicles depend on the surrounding environment, filters used and ventilation settings; Recirculation mode reduces the amount of PM entering from the outside (Chuang et al., 2013). The WHO (2005) guideline for PM concentrations in the indoor environment is 50 µg/m³ and 25 µg/m³ for $PM_{10}$ and $P_{2.5}$, respectively as 24-h mean. For annual means the guideline value for indoor air is 10 and 20 µg/m³ for $PM_{2.5}$ and $PM_{10}$, respectively, the ambient air quality standard for annual mean values for $PM_{2.5}$ and $PM_{10}$ are 25 and 40 µg/m³, respectively.

### Ultrafine particles (UFP)

The sources of UFP (< 100 nm) are combustion along with emissions of condensable, semivolatile compounds, for example organics, having numerous sources, such as road traffic exhaust and evaporative emissions, solvent use or oil refinery. A large part of UFP is secondary PM. The UFP penetrate deep into the lungs and comes with significant health risks (WHO 2005). Particle number concentration of UFP in the air close to roads are in the range of 50 000 to 500 000 particle/cm³. Inside vehicles, typical concentrations in the order of $10^4$ particles/cm³ have been reported (Knibbs et al., 2019b; Zhu et al, 2002a; 2002b). Levels similar to those in the outside air can occur at occasions inside a vehicle, either if cleaning of the incoming air is poor, or as occasional maxima e.g. in tunnels (Zhu et al, 2007; Weichenthal et al., 2008; Kaur and Nieuwenhuijsen, 2009; Knibbs and de Dear, 2010, Hudda et al., 2012; Ragettli et al., 2013). UFP concentration in outside air, ventilation settings, leakage airflow rate, cabin air filter quality and driving speed were identified by previous studies as key parameters that could determine the in-cabin UFP exposure levels. Both in-cabin and outdoor UFP size distributions are mostly bimodal; the primary peak occurring at 10 – 30 nm and the secondary at 50 – 70 nm (Zhu et al., 2007; Joodatnia et al., 2013).

The primary determinant for the UFP concentration in a vehicle is the ventilation condition (Hudda et al., 2011; Bigazzi and Figliozzi, 2012; Lee et al., 2015). Low AER results in higher PN removal from the incoming air (Knibbs et al., 2010) and recirculation mode decreases PN concentrations exponentially (Zhu et al, 2007). The UFP number concentration inside vehicles can be significantly reduced by use of efficient filters in the outside air intake (Tartakovsky et al., 2013; Lee and Zhu, 2014; Hudda et al., 2011; Yu et al., 2017; Mayer et al., 2018)

### Polycyclic aromatic hydrocarbons (PAH)

Sources of PAH are for example, industrial processes and combustion processes. PAH are associated with different types of cancer depending on the exposure route. A number of PAHs are mutagenic and genotoxic; lung cancer is the most serious health risk from exposure to PAHs in indoor air. Benzo(a)pyrene is one of the most potent carcinogens among the known PAHs (WHO, 2020). They exist both as gaseous and particle-bound. Particle-bound PAH have been measured in the transportation microenvironments in the concentrations of tens to hundreds of ng/m³; though without any speciation of individual PAH species (Riediker et al., 2003; Houston et al., 2013; Li et al, 2013; Hudda and Fruin; 2013). Benzo(a)pyrene (B(a)P) is the only PAH compounds with a health-based target value. The lifetime exposure to B[a]P for excess lifetime cancer risks is 1/10 000, 1/100 000 and 1/1 000 000 corresponding to concentrations approximately 1.2, 0.12 and 0.012 ng/m3, respectively (WHO, 2010).

### Volatile Organic Compounds (VOC)

VOC sources are mainly connected to solvent use, fuel production as well as incomplete combustion. VOC are mainly associated with perceived air quality, like perception of odor. Some individual VOC such as benzene have negative health effects, i.e. leukemia. The concentrations of airborne benzene associated with an excess lifetime risk of 1/10 000, 1/100 000 and 1/1000 000 are 17, 1.7 and 0.17 $\mu g/m^3$, respectively (WHO, 2010). The major groups of VOC compounds contributing to air pollution in vehicle cabins are aliphatic and aromatic hydrocarbons. Among the aromatics benzene, toluene, ethylbenzene and xylenes (BTEX), present in fuels and exhaust gases, reach the highest concentrations. The VOC are often measured as Total Volatile organic Compounds (TVOC) defined as sum of all compounds in the range of $C_6 – C_{16}$. The recommended guideline value for TVOC in general indoor environments (long-term value) is 300 $\mu g/m^3$ (UBA, 2019).

The VOC measured inside a vehicle cabin are coming from emissions from the interior materials and by transportation from the outside air through ventilation system. The reported concentrations of VOC and TVOC were in the range of several to thousand $\mu g/m^3$ (Yoshida and Matsunaga, 2006; Yoshida et al. 2006; You et al., 2007; Chen et al., 2014; Xu et al., 2016; Bakhtiari et al. 2018; Xiong et al., 2019).

### Formaldehyde

Formaldehyde is classified as a known human cancerogenic. It is emitted from the interior materials (Salthammer et al., 2010). The reported mean concentrations in cars ranged from several ppb (Geiss et al., 2009) to tens ppb (Yoshida T, Matsunaga I, 2006; Yoshida et al., 2006; Zhang et al., 2008; Xu et al., 2016) and even hundreds of ppb (Bakhtiari et al, 2018). The WHO (2010) indoor air quality guideline for formaldehyde is 100 $\mu g/m^3$ (24-hour average).

More details information regarding the current state-of-art knowledge on air pollution in road vehicle cabins are presented in the internal report by Langer (2015).

# 3. Methods

The main goal of the project was to develop a methodology framework for investigation of air quality parameters and linking that to exposure of occupants in road vehicles. To prevent outdoor air pollution from entering the vehicle, newly produced vehicles are usually air-tight with a slight overpressure inside the cabin created by the climate system. The air intake is placed under the hood, in case of a passenger car close to the windshield, and from there the air enters the cabin through the ventilation system. The ventilation system is usually fitted with a cabin air filter that prevents particles, gases and odours from entering the cabin. The vehicle can be equipped with sensors controlling outside air quality and closing the recirculation flap in polluted environment, for example in tunnels. The core of the project consists of a series of extensive measurement campaigns where a set of air pollutants were measured in parallel in the cabin and outside, while driving a well-defined loop. This was carried out to investigate the impact of the air cleaning system and operation on concentrations inside the vehicle. These campaigns were used to build basic knowledge on processes influencing the in-cabin concentrations of air pollutants, exposure of these to passengers, as well as the relation between the indoor and outdoor concentrations of the investigated pollutants. The campaigns were designed to cover both passenger cars and trucks during both summer and winter conditions. Additionally, a method for dynamic testing of the efficiency of air cleaning components with help of air recirculation experiments was developed and applied during the campaigns. Several downscaled campaigns were designed and performed as well; these are suitable for routine use by the vehicle producers. The project also included testing of measurement equipment and sensors. An overview of campaigns and tests carried out during the project is shown in Table A1 in Appendix 1.

Large effort was spent on building a database containing the results from the measurement campaigns and on evaluating the data, this is described in sections 3.3 and 3.4. The database is an important source of background knowledge for further development of air cleaning techniques in road vehicles.

## 3.1. Design of the core campaigns

The experiments of the core measurement campaigns aim to investigate effectiveness of certain air cleaning components (cabin air filter or other) or different modes in the ventilation system during different conditions, both considering the ambient air pollution levels and type of road/ driving pattern. To cover this in a systematic manner the tests were performed while driving a defined loop covering different driving scenarios and air pollution levels, i.e. motorway/trunk road, city driving and very high air pollution level environment which was represented by tunnels.

The tests were primarily performed with ventilation in outside air mode only, without automatic recirculation of the cabin air, unless performance of the automatic ventilation system (used by costumers) was the subject of the testing. Preparation of the vehicles for the campaign involved installation of the 230 AC source for the campaign instrumentation (inverter from the on-board battery), installation of the computer system for logging of the vehicle parameters, such as ventilation fan velocity, air distribution and cabin temperature, and for manual control of ventilation, and installation of sample inlets for measurement instruments and pumps. Since the ventilation velocity has strong influence on the air cleaning efficiency of filters, the preferred method is to control the ventilation fan velocity manually from the control computer. This was achieved in later campaigns, ‘Core campaign PC summer’ and ‘Demonstration campaign T summer’.

A map with the driving loop for Gothenburg is shown in Figure 3.1. The loop covers three different driving scenarios, motorway (green in Figure 3.1), city driving (blue in Figure 3.1) and tunnels (the largest Lundby tunnel shown in pink in Figure 3.1). These scenarios are also associated with different air pollution levels, the tunnels representing very high air pollution levels and the city typically having higher air pollution levels than the highway. In the Lundby tunnel the vehicle is stopped for 15 min. Driving one loop takes about one hour, depending on traffic density. Due to the traffic restrictions the driving loop for the trucks differed from the driving loop of the passenger cars (dotted line for car measurements in the city, while trucks are solid line, see Figure 3.1). Typically, several cabin air filters differing by quality or age were tested during different operation modes in the ventilation system during one measurement campaign.

The tests were designed in a way that different scenarios of driving, air pollution levels and ventilation system operation were covered in sufficient time-intervals, allowing analysis of variance of the measured parameters and in the case of cumulative pumped samples also collection of samples with detectable quantities. For test of one ventilation system component (e.g. filter) the typical sampling scheme is shown in Table 3.1. The component was tested under 'normal' ventilation rate with outside air supply only, which was the base setup, and under low and high ventilation rates. The base setup was tested in four loops to obtain duplicates of the cumulatively pumped samples (PAH sampling needs 2 h driving, i.e. two loops), the high and low ventilation rates were tested in two loops. The component can optionally also be tested in the automatic ventilation, i.e. customer drive, allowing the automatic setting of recirculation and determination of fan-speed by the car. The system was also tested without the filter installed under 'normal' ventilation to investigate losses of air pollutants in other parts of the ventilation system and in the vehicle cabin.

Table A1.2 in Appendix 1 shows the testing protocols. Test protocol and Test log were used to keep track of the experiments conducted and to couple data to the different experiments. In addition, protocols for pumped sampling need to be filled for each sample to keep track of the sampled volumes and sampling times as well as to couple the samples to the experiments throughout the analytical procedures (Protocol for sampling point).

**Table 3.1. Sampling scheme for a test of cabin air filter in the core campaign. The 'No filter' test was performed only once.**

| Test | Ventilation setup | Number of loops | Time need (min) | Measured parameters | | Number of pumped samples |
|---|---|---|---|---|---|---|
| Filter X | 'normal' | 8 | 480 | Online PM, NOx | PM.5$^{*}$, PAH$^{\dagger}$, VOC$^{**}$ | 2+2+2 |
| | 'low' | 4 | 240 | Online PM, NOx | PM.5$^{*}$, PAH$^{\dagger}$, VOC$^{**}$ | 1+1+1 |
| | 'high' | 4 | 240 | Online PM, NOx | PM.5$^{*}$, PAH$^{\dagger}$, VOC$^{**}$ | 1+1+1 |
| | Customer drive | 4 | 240 | Online PM, NOx | PM.5$^{*}$, PAH$^{\dagger}$, VOC$^{**}$ | 1+1+1 |
| No filter | 'normal' | 4 | 480 | Online PM, NOx | PM.5$^{*}$, PAH$^{\dagger}$, VOC$^{**}$ | 1+1+1 |
| Total | | 24 | 1 440 | | | 6+6+6 |

* pumped sample cumulative over 4 loops

[†] pumped sample cumulative over 2 loops

** pumped sample cumulative over 1 loop

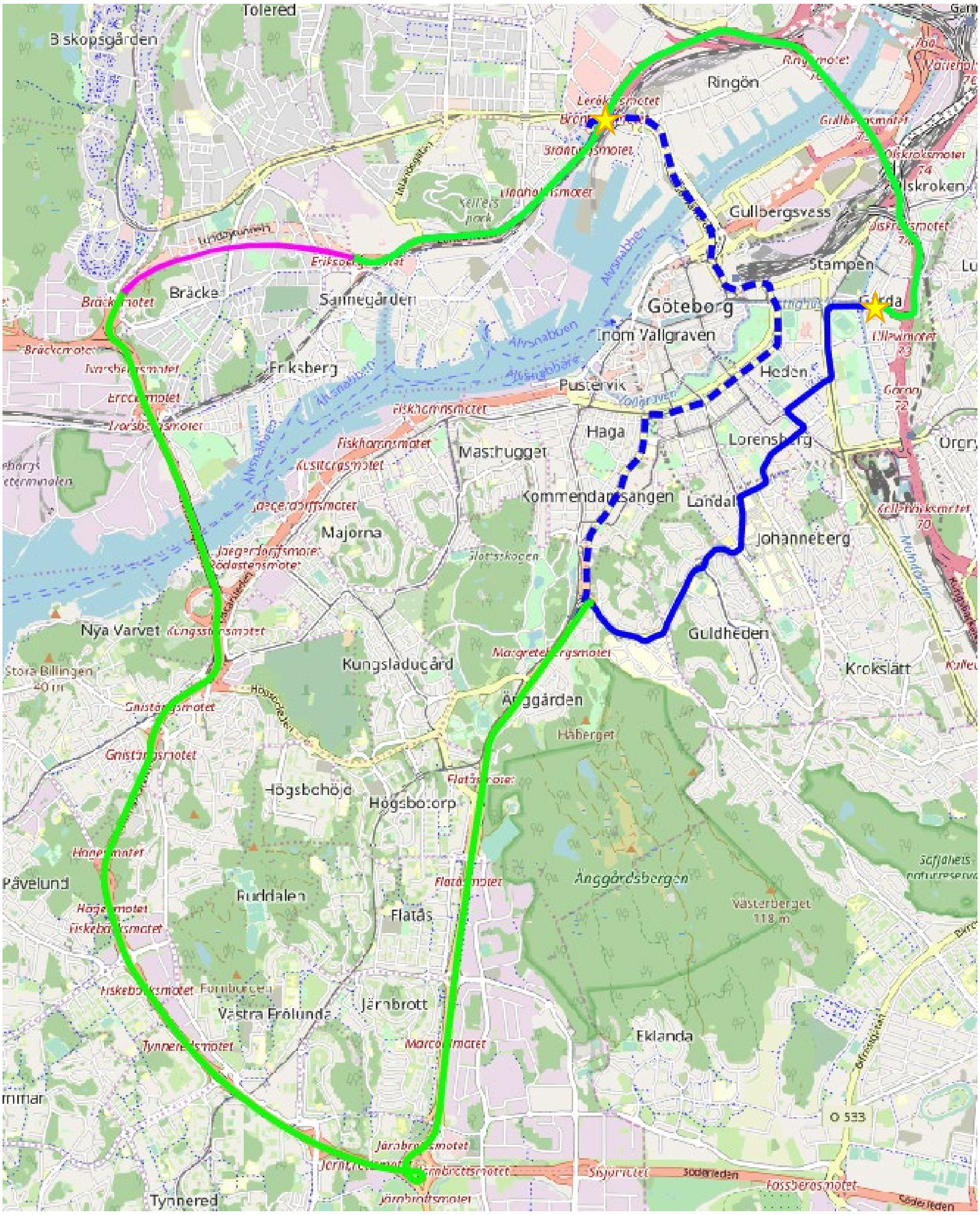


Figure 3.1: Illustrates the driving cycle in Gothenburg. The blue line represents the city driving conditions, the dashed line is for passenger cars, solid line for trucks. The green line represents the highway driving conditions, used by both car and truck, section between the yellow stars (between Gårda and Ringön) is for trucks only. The pink area is the Lundby tunnel where the vehicles were standing still for 15 minutes.

The main parameters investigated were continuous measurements of size-resolved particulate matter, $CO_2$, temperature and relative humidity, optionally of NO and $NO_2$, and pumped cumulative samples of non-methane volatile organic compounds (NMVOCs) and polyaromatic hydrocarbons (PAH).

All parameters were measures in parallel outside and inside the vehicle. In the truck the outside air inlet was placed in the front under the bonnet, in front of the air inlet area for the ventilation system. In the passenger car the inlet for online particle instruments was mounted above air inlet area (above plenum cover) under the hood. Since temperature under the passenger car's hood can reach temperatures up to 70°C when the vehicle was not moving, the outside air sampling devices for the pumped sampling were moved to the car roof to avoid desorption of volatile and semi-volatile material from the samples during the occasional high-temperature events. The outside air inlets are shown in Figure A1 in Appendix 1. The inside sample inlet was in the cab at the level of driver's head. For the particle instruments the inside and outside sampling lines were made of conductive material and of the same length to have minimum particle losses in the sampling system and the same conditions for both sampling points.

### 3.1.1. Online measurements

Online instruments with high time resolution were used for measurements of size-resolved particulate matter (PM), NO, $NO_2$ and $CO_2$. Two different instrumentation setups were used for PM measurements:

1. Grimm Mini Wide Range Aerosol Spectrometers (Mini-WRAS 1.371), which is equipped with a Faraday cup electrometer, measuring particle sizes of diameter from 10 to 200 nm and with an optical aerosol spectrometer part using laser and scattering light method, measuring particle sizes from 200 nm to 35 µm;
2. Combination of TSI instruments NanoScan Nanoparticle Sizer (3910) which is a scanning mobility particle sizer instrument, measuring particles in range 10 – 360 nm and Optical Particle Sizer (3330), an optical aerosol spectrometer using laser and scattering light method measuring in the size range 300 nm – 10 µm.

In some campaigns, aerosol spectrometers GRIMM 1–108 were used, this spectrometer is identical with the one mounted in Mini-WRAS. The maximum time resolution of Mini-WRAS is 60s, NanoScan is 60s, OPS is 1s and GRIMM 1-108 is 6s. $CO_2$ was measured together with temperature and relative humidity with a Woehler portable instrument. NO and $NO_2$ were measured with chemiluminescence instrument. The sampling frequency used for these instruments during the driving experiments was set to 1 min, for recirculation experiments a 1 s resolution was used where possible.

### 3.1.2. Pumped samples

In addition to the online measurements, cumulative samples were collected by pumping the examined air trough filters and/or absorbents which were later analyzed in the laboratory. These samples do not have high temporal resolution and were taken during one or more complete driving loops, depending on sampling time needed to collect samples with examined species in quantities above detection limits of the laboratory analyses. In Appendix A Table A3 gives an overview of number of pumped samples taken during the core campaigns. Table 3.1 shows an example of number of loops and time needed for testing of one cabin air filter.

### Filter samples for $PM_{2.5}$ mass and its components

Mass of the fine particulate matter $PM_{2.5}$ was collected on high-volume pumped filter samples which were analyzed with a gravimetric method at the IVL laboratory. These samples can also be further analyzed for chemical composition of the collected PM, e.g. for the content of light absorbing black carbon (BC, corresponding to the content of soot) and UV-light absorbing PM (UVPM, corresponds roughly to organic PM), for PM-bound PAHs with GCMS, or for elemental content with ED XRF method. These BC, UVPM and PAH analyses were performed by the laboratory of AMM, the ED XRF analyses by Cooper Environmental Services' laboratory, U.S.A. Further details of the sampling are presented in Appendix A.

### Combined filter and absorbent samples for polycyclic aromatic hydrocarbons (PAH)

The sampling line for PAH includes two compartments, an open-face filter cassette collecting total particulate matter and particulate associated PAHs, followed by a solid adsorbent XAD-2 tube (120 mg) collecting gas phase PAHs. The filters were composed of PTFE, diameter used were 37 mm, pore size 2 µm (Teflo; Pall Corporation, Port Washington, N.Y). This PAH sampler was coupled with a PTFE tubing to a battery-driven sampling pump (SKC, GilAir and Gilian) with a sampling flow of 2 l/min.

The following laboratory analyses of PAH were performed at AMM laboratory. After the gravimetric analysis the filters with the corresponding XAD-2 adsorbent were combined and analyzed for their PAH contents. Before extraction, an internal standard mixture containing the 16 US EPA priority PAHs were added to the samples. The target compounds were cleaned through a column filled with silica-gel and finally analyzed using high-resolution gas chromatography/low-resolution mass spectrometry (GC/MS). The target compounds were the 16 US EPA PAHs and 16 alkylated PAHs. Blank filters and XAD-2 blank tubes were analyzed in parallel with the samples. Some minor PAH residues were found in the blanks, but in no case were the blanks greater than 5% of the amount found in any sample. All PAH results were corrected to the blanks. The limits of detection (LODs) were calculated as three times the SD of the blanks. Moreover, a certified reference material (SRM 1649a urban dust) was used as a quality control sample (Bohlin et al., 2010). The measured levels of 12 PAHs never deviated more than 15% from the certified levels.

In data analyses the 32 PAH compounds were summed into several groups:

- Sum of 32 PAH
- 16 US EPA PAH
- Particulate PAH
- EPA 7 carcinogenic PAH
- EC 6 reported PAH
- EC 4 reported PAH
- Benzo(a)pyrene (B(a)P) equivalents, measure using Toxic EQuivalents of individual PAH species relative to that of B(a)P (TEQ).

An overview of the quantified PAH species, the groups they belong to and their TEQ is presented in Table A4 in Appendix A.

### Absorbent samples for volatile organic compounds (VOC)

Sampling of NMHCs was carried out on absorbent tubes which were connected with tubing to the battery-driven low volume sampling pumps. The sampled volume was 8-18 L, using a sampling air flow of 150-200 mL/min. The tubes contain adsorbents suitable to quantitatively collect $C_4$ – $C_9$ hydrocarbons including BTEX compounds (benzene, toluene, ethylbenzene and xylenes) and 1,3-butadiene. The samples were analyzed by gas chromatography with a flame ionization detector (GC-FID) or gas chromatography with a mass selective detector (GC–MS) as described in more detail in Appendix A. An overview of the methods used for VOC analyses in the different campaigns is presented in Table A5 in Appendix A.

The chemical identity of the individual hydrocarbons is presented in Table A6 in Appendix A. For further data analysis, the organic compounds (hydrocarbons) were divided into two groups:

1. **very volatile organic compounds (VVOC)** including butanes, butenes, pentanes, pentenes, 1,3-butadiene and methylated pentanes and
2. **volatile organic compounds (VOC)** in the context of indoor air quality, defined by an analytical ISO standard (ISO 16000-6, 2012) as compounds determined by a specified analytical procedure (GMS-MS) with a slightly polar column, eluting within the retention range $C_6$ - $C_{16}$ (n-hexane to hexadecane).

In this project the analyzed VOCs were compounds between (and including) $C_6$ (n-hexane) and $C_9$ (nonane, trimethyl benzenes) and is called tot-VOC in the data analyses. The unsaturated compounds from the VVOC group (butenes, pentenes) are part of so-called ozone precursor compounds (together with ethene and propene) important especially in the ambient air quality context. The VOC group contains the **BTEX** compounds which form the third VOC group used in the data analysis, these are regularly used to assess the urban load of VOC. An overview of the quantified VOCs and their classification into the above mentioned 3 groups is presented in Table A6 in Appendix A.

## 3.1.3. Dynamic recirculation tests

Each campaign also involves a set of dynamic recirculation experiments in the Lundby tunnel, which correspond to each driving tests of the campaigns. This was made by switching the ventilation mode from outside air to the recirculation mode. The cabin air was then recirculated though the cabin air filter and an exponential decay of particles and gases captured by the filter was observed. This decay can be fitted with an exponential function

$$C_x(t) = A * e^{(-k*t)} + B \qquad 3.1$$

Where $C_x(t)$ is the concentration of the examined species at time t, e is the natural number and A, k and B are the fitted parameters where k corresponds to the zero-order decay rate and B is the residual concentration to which the concentration is asymptotically closing in to. Inverse of k corresponds to the lifetime of the pollutant in the recirculation experiment.

To avoid bias from influence of the different initial concentrations in these experiments, which are actually dependent on the efficiency of the air cleaning system, the cabin air was polluted with the outside tunnel air prior to each experiment. A set of experiments with removed air filters were carried out during each campaign to evaluate the reduction efficiency of the other parts of the ventilation system and the car cabin.

Advantages of these dynamic tests are that they are relatively short, do not involve extensive driving and that the results can be directly used in the numerical model of the air quality in the vehicle. These experiments demand high time-resolution instruments (preferably higher resolution than 1 min), on the other side only one set of instrumentation is needed and the tests are thus suitable when more advanced instrumentation is involved. During the AQIFOR campaigns the recirculation tests involved measurements of volatile and non-volatile particles with thermodenuder and measurements of light absorbing black carbon.

## 3.2. Personal exposure campaigns

Two separate campaigns investigating personal exposure of professional drivers to air pollutants were performed within the project, one examining passenger cars and one trucks. To assess the contribution of in-vehicle exposure, each driver was equipped with two sets of sampling kits, one carried on the body and one to be placed in the cabin next to the driver. The air pollutants examined were PAH and VOC, using the pumped-sample methodology similar to that applied the core campaigns. Personal full-shift air samples (6–8 h) and stationary samples (same time interval) were taken in the vehicles. The sampling module for collecting PAHs and total particles, consisted of two compartments, an open-face filter cassette collecting total particulate matter and particulate associated PAHs, followed by a solid adsorbent XAD-2 tube (120 mg) collecting gas phase PAHs. These were connected to a pump operated with a flow rate of ca 2 L/min. This module was the same as used in the core campaigns. The sampling procedure and following laboratory analyses of VOC were also corresponding to those used in the core campaign, the only difference was that the sampling flow was adjusted to collect 10-l sample during an 8-h shift.

The campaign involving passenger cars took place November/December 2016 and involved 11 drivers working with delivery service and driving various routes in the city of Gothenburg, mixed driving and only country road. The campaign involving truck drivers took place in Mars 2017 and involved 10 drivers working with the delivery service and driving short distances on Hisingen, Gothenburg.

## 3.3. Reduced measurement campaigns and instrument tests

The project included a number of tests comparing the instruments or measurement methodologies, both in the field and in the laboratory, as well as several reduced measurement campaigns aiming to develop simple and robust testing methodology. Overview of the instrument comparison tests is presented together with all other field and laboratory tests in Table A1 in Appendix 1.

Initially the two Grimm Mini-WRAS instruments involved in AQIFOR and owned by Volvo Cars were compared with IVL-owned GRIMM 1-108 stand-alone aerosol spectrometers (4 pc.), with a simple particle condensation counter TSI P-trak and with TSI Engine Exhaust Particle Sizer Spectrometer 3090 (EEPS). The instruments were compared in laboratory conditions analyzing aerosols generated by an Aerosol Generator 3079A and during a driving test using the AQIFOR driving loop (EEPS instrument not involved in the driving test). The tests showed reasonable agreement between the instruments of the same kind and expected differences between Mini-WRAS and Grimm spectrometer which can be explained by the fact that an important part of the measured particle mass and dominant part of the particle number are in the ultrafine range which is not covered by the Grimm spectrometer. The tests also shown the importance of particle losses in the tubing connecting the sample inlet to the instrument. The campaign gave a good overview of the advantages of the different instrument and provided opportunity for practice and improvement of the measurement methodology. More details are presented in the report Borre and Nilsson Rabbaa (2015).

Two tests aiming at comparison of the methods used for determination of particulate matter mass in AQIFOR with standard method used at urban air quality monitoring stations for measurement of $PM_{2.5}$ were carried out. The first test was performed in winter 2017 when the Mini-WRAS, three parallel pumped $PM_{2.5}$ samples and IVL TSI SMPS instruments were placed at the Gothenburg urban background monitoring station Femman and was compared to the TEOM instrument monitoring $PM_{2.5}$ at the station. The second test was performed in spring 2018 at the urban monitoring station Gårda where the Mini-WRAS, small aerosol spectrometer Fidas Frog, TSI instruments NanoScan was compared with the OPS TEOM operating at the station. The comparison is showed in Figure A3, Appendix A.

An important part of the project was development and testing of the measurement methodology, especially relatively simple methods that could be used by the vehicle producers for testing the air cleaning equipment at different places around the world. A series of short campaigns was performed by Volvo Cars in Shanghai, China. All these campaigns involved the well-established online measurements of size-resolved particulate matter with two Mini-WRAS instruments. Different methodologies for measurements of other air pollutants were tested: The first campaign, early in the project, tested the possibility of using passive samplers for measurements of NO, $NO_2$, ozone, PAH, VOC and aldehydes. This methodology was found not suitable for the purpose of the driving campaigns, since the passive sampling demands many hours of sampling which is not possible to be harmonised with well-defined driving experiments. The second campaign tested the pumped sampling of PAH and VOC with small low-volume battery-driven pumps, as described in section 3.1.2, which proved to be successful and were later used in the core measurement campaigns. The third standalone short campaign in China took place in January 2017 and its main purpose was to test the reduced and robust methodology developed in the AQIFOR core campaigns in Chinese conditions, including testing the feasibility of transfer of the equipment to China. This involved VOC sampling with low volume pumps and combined sampling of $PM_{2.5}$ mass and PAH performed with high-volume sampling of $PM_{2.5}$ on filter followed by column with absorbent collecting gas-phase PAH. Details of this campaign are presented in a separate report: Moldanová et al. (2017).

## 3.4. Evaluating the results

The project campaigns generated a large amount of data. An important part of the project was building a database that organises the actual measurement data, campaign characteristic and auxiliary data together with the meta-data. The database is a Microsoft Access Database called: *AQIFOR.accdb*. The database is described in Appendix C.

To get a general overview of the campaigns and for first-hand evaluation of the measurements, time plots of the indoor and outdoor concentrations measured with the online instruments completed with lines indicating the driving regime and potential pumped sampling are used, example is shown in Figure 3.2. To analyze the measurement campaigns further some type of aggregation needs to be carried out to quantify the results and to compare tests with different filters, weather conditions, vehicles, ventilations speeds, ambient temperatures etc.

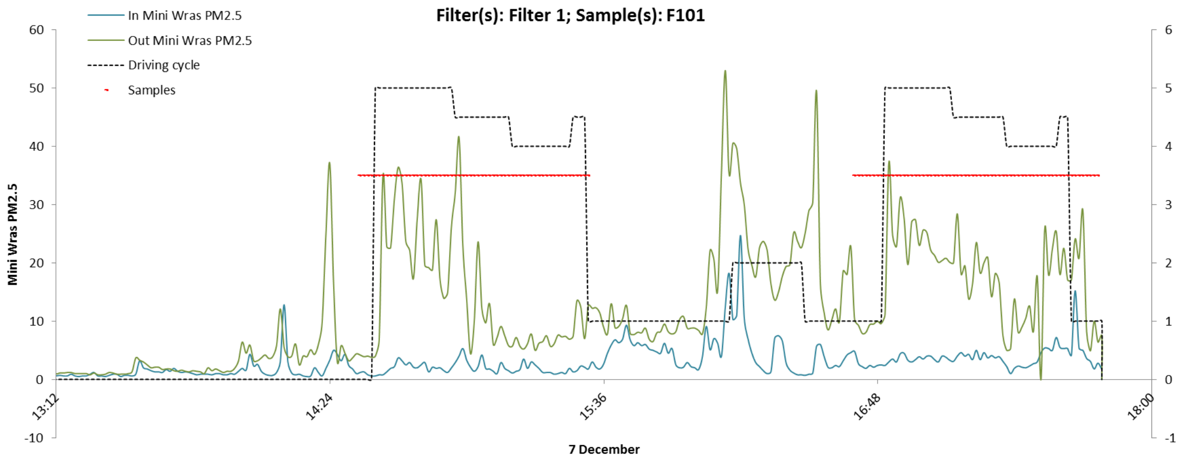


**Figure 3.2: Example of indoor and outdoor PM2.5 concentrations measured with Mini-WRAS. The dotted line represents different stages in the driving cycle: standing still in tunnel (5), highway driving (4.5), city driving (4), recirculation experiments (2) and driving outside the test loop (1). The red line represents periods when pumped samples were taken.**

The AQIFOR campaigns consist of tests, where one test represents e.g. filter A tested at medium ventilation rate, outside air only. One test consists of several experiments where each experiment was covered by one driving loop as shown in Figure 3.1. Each experiment can be further disaggregated to the different driving scenarios involved: city driving, motorway driving and standing or driving in the tunnel. The variability of the data measured with online instrument were used to calculate standard deviation of the mean for each experiment. Variability between the means of different experiments in one test (experiment replicates) can be used to calculate standard deviation of the mean of the test. Additionally, each campaign involves a set of dynamic recirculation experiments in the tunnel, which correspond to the driving tests of the campaign. Along with online instrument measurements, pumped sampling on filters and/or absorbents were taken in parallel.

In the AQIFOR-project we choose to evaluate the relation between the indoor and outdoor concentration as *system reduction efficiency*. In this project it was defined as the difference between the outdoor and indoor concentrations divided by the outdoor concentration. This relative difference may be calculated as follows as shown in an example for fine particulate matter:

$$\phi = \frac{PM2.5_{out} - PM2.5_{in}}{PM2.5_{out}} \qquad 3.2$$

where:

$PM2.5_{in}$: Indoor PM2.5 concentration

$PM2.5_{out}$: Outdoor PM2.5 concentration

$\phi$: The *system reduction efficiency* (use of *system* and not *filter* is acknowledging that reduction of PM2.5 takes place in the entire ventilation system as well as in the cabin)

The other measured parameters NOx, PAH and VOC as well as other quantities characterizing particulate matter were evaluated in the same manner and the system reduction efficiency was calculated for the corresponding pairs of indoor and outdoor samples.

### 3.4.1. Data from the online instruments

The mean system reduction efficiency in an experiment could either be calculated from experiment means, i.e. the *efficiency-of-means* or as the *mean-of-instant-efficiencies*. The *efficiency-of-means* and the *mean-of-instant-efficiencies* are further explained in Appendix B. Depending on what the purpose of the results were, either of the two different methods could be used. Since there was a delay between the change of the outside concentration and corresponding change inside, caused by time needed for the outside air to enter the cabin, and this time delay varies with ventilation velocity, we have used the *efficiency-of-means* for evaluation of the campaigns. However, there are some short experimental sequences where only the data under stable conditions were selected and in these cases the *mean-of-instant-efficiencies* were used. More detailed discussion on use of the *efficiency-of-means* is presented in Appendix A.

The results for an experiment are shown with a standard error for average indoor or outdoor concentration during a measurement session (e.g. Figures 4.2, 4.5, 4.12 and 4.13). When the average system reduction efficiency was evaluated, the uncertainty of this variable was assessed with help of the so-called error propagation method which takes into account covariation between the outdoor and indoor concentration. As we do not compensate for the time lag between the indoor and outdoor concentrations, the time lag decreases the correlation and thus increases the standard error of the experiments. Equations used and more details on uncertainty calculations are presented in Appendix A. When means of experiments performed within one test are presented, standard deviation of this mean is shown if 3 or more experiments are available.

### 3.4.2. Data from the pumped samples of $PM_{2.5}$, PAH and VOC

The pumped samples were taken during one entire experiment or several replicates as required by the detection limits of the consequent laboratory analyses. The pumped samples therefore represent average of one or several experiments with the same settings.

The uncertainty of the pumped $PM_{2.5}$, PAH and VOC measurements were assessed for the individual concentrations and for the system reduction efficiency. For the individual concentrations the uncertainty was associated with uncertainties associated with the laboratory analyses. Details of the uncertainty estimates for the different samples are presented in Appendix A.

### 3.4.3. Recirculation experiments

The concentration time series measured by the online instruments ($PM_{2.5}$, particle number concentrations in particle size classes) starting at the time each individual recirculation experiment begins (windows/doors were closed) to the time the baseline concentration is reached is fitted to an exponential function of the form 3.1 ( $C(x) = Ae^{-kt} + B$ ) (Figure 3.3).

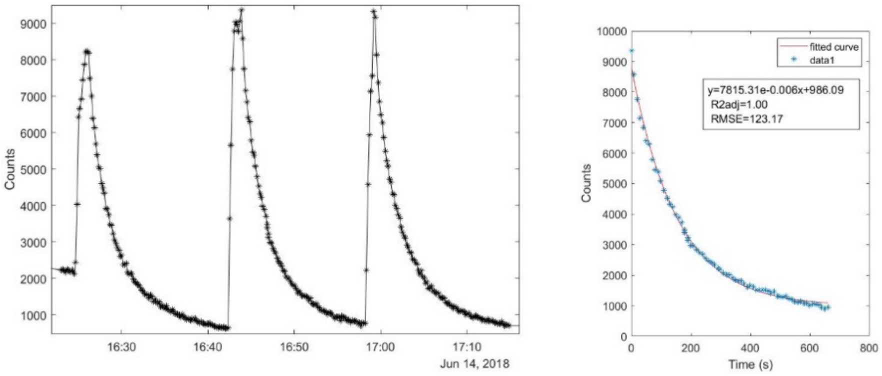


**Figure 3.3. Counts of particles in the accumulation mode measured during 3 recirculation experiments with the OPS instrument (left). Data extracted from the middle peak and fitted to an exponential function (right).**

The fit-function in MATLAB was used for the regression. For each fit, a plot shown to the right in Figure 3.3 was created and inspected together with the adjusted R2 and RMSE (root mean square error). In the cases where the fit was poor and if the cause seemed likely to be due to an unstable baseline, too few measurements or too short measuring time after the doors/windows had been closed, that test was removed. In all, less than 10% of the tests were removed for the particle measurements and less than 20% were removed for the $NO_2$ measurements. If no such explanation could be found, the test was kept. For the purpose of evaluating the efficiency of the filters, the regression constants together with 95% confidence intervals for each constant were exported.

### 3.4.4. Auxiliary parameters used for evaluation

A number of parameters such as meteorological data or parameters associated with vehicle operation, were collected. These data are used for further evaluation of parameters influencing the indoor concentrations and system reduction efficiencies.

Data on pressure (p), relative humidity (RH) and precipitation in Gothenburg were collected from the SMHI database, data on urban background concentrations of NOx, $PM_{2.5}$ and ozone were obtained from Gothenburg city environmental board, from measurement station Femman.

The vehicle operation data were logged by the manufacturers control system during the driving experiments.

# 4. Results from the measurement campaigns

During the core campaigns, processes affecting concentrations of particulate matter, NO, $NO_2$, PAH and VOC in the vehicle cabin were investigated. The particulate matter was investigated with online instruments and pumped filter sampling of $PM_{2.5}$, the later combined with analyses of light absorbing black carbon and particle-bound PAH, in one campaign also with analysis elemental composition. The focus was on air pollutants relevant for the vehicle interior that have known negative health effects and existing concentration limits and target levels (see Appendix B).

In last decades the focus has been on fine particulate matter, $PM_{2.5}$, which causes many premature deaths and morbidities in areas with air pollution problems around the world (WHO 2016). There is scientific evidence that it is even smaller particles than $PM_{2.5}$ that are the most dangerous ones and that the particle number can be a relevant to measure for negative health impacts of PM. Further, some compounds of PM are known or suspected to have larger impacts than others. One compound with known risk is elemental carbon, i.e. soot (Jansen et al., 2011). Hence, different PM metrics are evaluated to increase understanding of the physics and chemistry of PM pollution in vehicles and possibilities of its efficient mitigation.

## 4.1. Particulate matter

### 4.1.1. $PM_{2.5}$ reduction with online instruments

In this section, $PM_{2.5}$ is calculated from data measured by the Mini-WRAS or combination of OPS and NanoScan instruments. Both instruments combine two different measurement principles and cover approximately the same size range. Intercomparison of the two instruments have shown reasonable agreement, especially in terms of reduction efficiency which is a relative difference between measurements of two instruments of the same kind. An overview of the results of mean system reduction efficiency for $PM_{2.5}$ at medium ventilation rate from all core campaigns is illustrated in Figure 4.1. Each point represents one experiment and the colors represent different filter types. The filters are anonymized, however, experiments without any filter are "F0_x". The mean system reduction efficiency (y-axis), explained in Chapter 3.4, is plotted against the average outdoor concentration to illustrate which type of air pollution conditions the vehicle was driven in during the experiment. From the results it was possible to see that the outdoor concentration in general were higher when the vehicle were standing still in the tunnel compared to when it was driven around. The figure shows a span of reduction efficiencies for $PM_{2.5}$ between 30% and >90%, with everything from no filter to a completely new filter and even additional cleaning techniques, without any clear correlation with the outdoor concentration. The no-filter experiments have in general low average system reduction efficiency with high variability between the experiments. It is also possible to conclude that the mean system reduction efficiency was dependent on the cabin air filter type.

As can been seen in the example in Figure 3.2, the outdoor concentrations may vary a lot during one experiment. To illustrate this, in Figure 4.2 the standard deviation for the outdoor concentrations in one entire experiment or a driving case of one experiment is illustrated with error bars along the x-axis while the standard deviation for the mean system reduction efficiency, as described in section

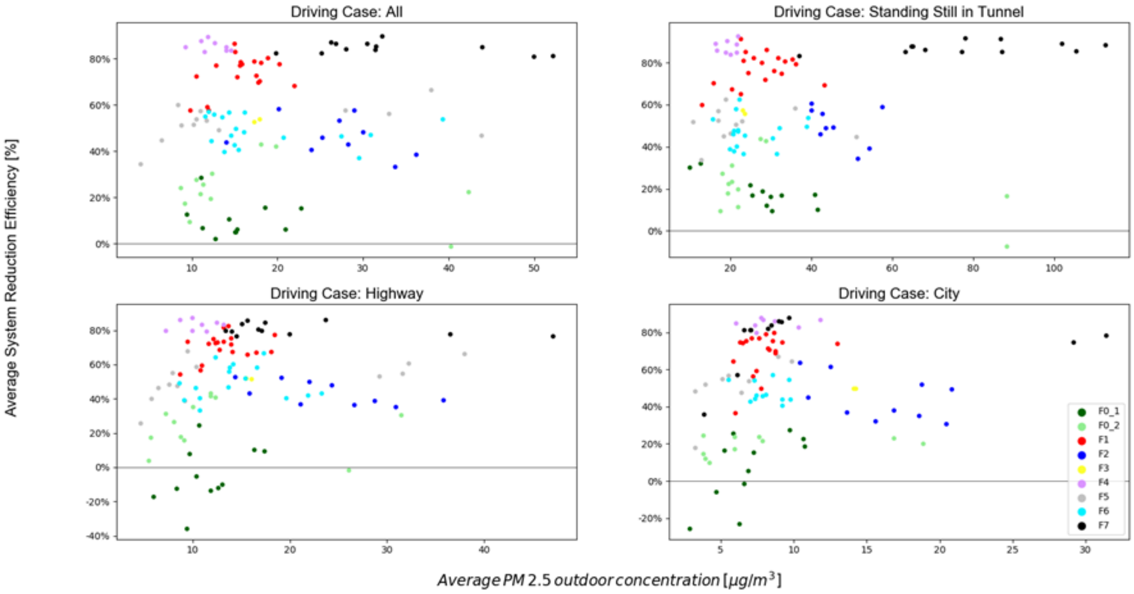


**Figure 4.1: Mean system reduction efficiencies for PM2.5 for all experiments with medium ventilation rate. The different types of cabin air filters tested are distinguished by different colors. The four plots show mean efficiencies for different driving cases in each experiment (explained in section 3.1), driving case 'All' represents mean value for entire experiment (driving loop), without weighing. Each point represents one experiment.**

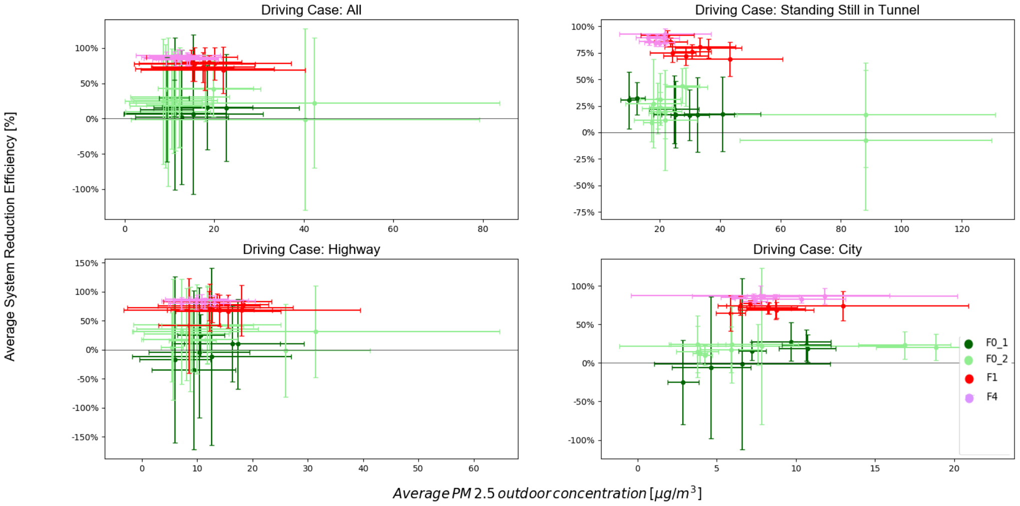


**Figure 4.2: The mean system reduction efficiencies for PM2.5 for two different filters with a higher reduction (F1 and F4) compared to a no-filter scenario (F0_1 and F0_2, two different types of vehicles), plotted with error bars. The error bars represent one standard deviation of the outdoor concentration (x-axis) and one standard deviation for the system reduction efficiency (y-axis). Each point represents one experiment.**

3.4, is illustrated with error bars along the y-axis. The variability of outdoor concentration had no connection to the type of filters, the error bars along the y-axis, on the other hand, reflects variability of the reduction efficiency which was connected to cabin air filters. The variability of the reduction efficiency was smaller for filters with high reduction efficiency compared to those with low efficiency or to no-filter experiments. The mean reduction efficiency of the no-filter experiments was not significantly different from zero. When comparing different driving modes of the same experiment, the variability of the reduction efficiency was comparably small for the tunnel experiment compared to other driving modes.

Further aggregation of the results by means of the experiment replicates gives an easy comparison of e.g. different cabin air filters tested. The uncertainty of the mean system reduction efficiencies can then be expressed as the standard deviation of the aggregated experimental means. To calculate the standard deviation the experiment type needs to have at least 3 replicates. Figure 4.3 shows the experimental means from Figure 4.1, only cases with at least 3 replicates are shown. The figure shows that the comparison of efficiencies for the different cabin air filters are consistent between different driving cases. The best performing filters have mean reduction efficiencies for $PM_{2.5}$ slightly above 80% on average, the least efficient ones around 40%. Those on the low end include an aged filter and a simple particle filter. The no-filter experiments have reduction efficiency between 0 and 20%, significantly lower than the other experiments and often not significantly different from 0%.

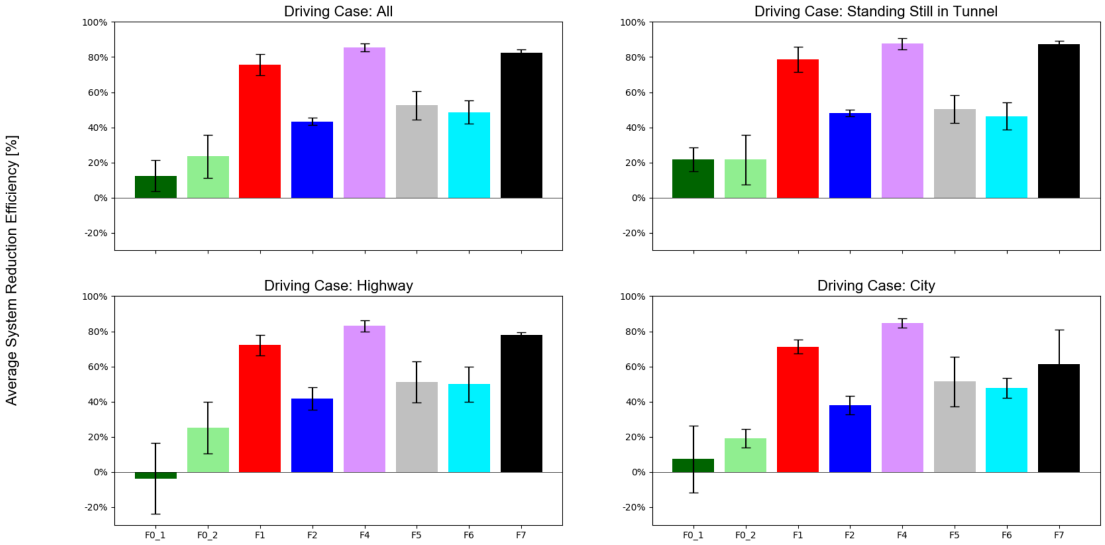


**Figure 4.3: Mean system reduction efficiency for $PM_{2.5}$ for different types of filters at medium ventilation rate. Driving case 'All' represents mean value for entire experiment (driving loop), without weighing.**

### 4.1.2. System reduction efficiency for different particle size categories

The particle number size distribution was typically dominated by particles in the ultrafine size-range (up to 100 nm) while the mass size distribution was to a much higher degree influenced by particles in the accumulation (100 nm - 1 µm) or coarse (1-10 µm) size ranges. Particles also have different physical and chemical properties and the different types of particles are typically associated to certain size-ranges (e.g. nucleation and soot particles ultrafine or road-dust particles coarse size range). Particles of different sizes and properties are removed from the air in different ways, both when filtration and deposition on surfaces are considered. This implies that the mean system reduction efficiency for $PM_{2.5}$, presented above, may not give the whole picture, reduction efficiency for $PM_{2.5}$ dominated by coarse particles can be very different from efficiency for $PM_{2.5}$ dominated by ultrafine particles. Since particles in the small size ranges have strong relevance for the negative health effects and are at the same time more difficult to remove by filters, it is of great importance to investigate concentrations and reduction efficiencies for particles in the different size ranges. The PM data were analysed both by comparing all size channels measured by the instrument and by aggregating the particles into 4 size categories:

- The ultrafine particles (10nm-100nm)
- Particles in the lower range of the accumulation mode (Accumulation mode 1) (≈300nm-600nm)
- Particles the upper range of the accumulation mode (Accumulation mode 2) (≈600nm-1000nm)
- Coarse particles (≈1µm-2.5µm)

These categories were based on the observed reduction efficiencies for individual particle-size channels, choosing intervals where these did not show large variation. At the same time, these categories correspond to the well-established categorisation of particles in aerosol physics, which takes both the removal processes and common origin of particles into consideration. The gap between 100 and 300 nm is related to instrumental issues – even though both the Mini-WRAS and TSI instruments cover this range with the nanoparticle part of their instrumentations, none of them performs well in this area and the data were associated with large uncertainties. While Mini-WRAS has a built-in data processing routine overbridging this area, the TSI measurement data can be fitted manually, the latter being a preferable way of overbridging the gap which would give better insight into the instrument performances and actual concentrations in this size range. This work, however, has not been done as the TSI instruments became available only in the later stage of the project.

The experiments have shown that cabin air filters typically have the lowest removal efficiency for particles in the ultrafine size-range, the more efficient filters were also more efficient in this range. Figure 4.4 illustrates this as an example of the average system reduction efficiencies for particles in all size channels measured by the instruments for filters F1 and F4, those belonging to more efficient filters examined.

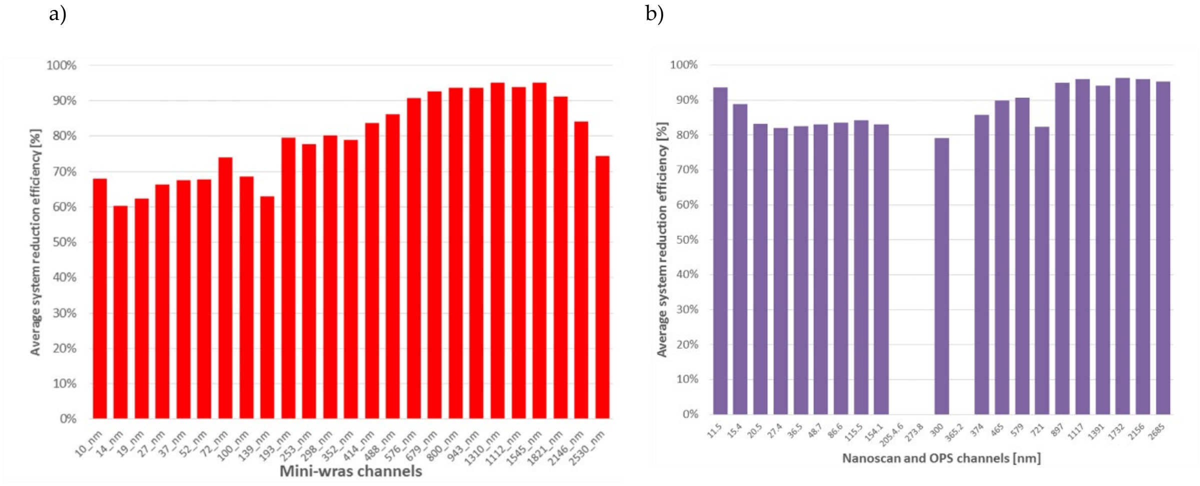


**Figure 4.4: Comparison of the mean filter reduction efficiencies for Filter F1 measured with Mini-Wras (a) and F4 measured with NanoScan/OPS (b) for data from individual instrument channels. The experiments were carried out standing still in a tunnel at medium ventilation rate mode. The missing channels in (b) are due to the low sensitivity of the NanoScan instrument in this size range.**

Reduction efficiencies for the four particle-size categories plotted with error bars representing one standard deviation, using tests of filters F1 and F4 is presented in Figure 4.5. One can see that the system reduction efficiency increases for the larger particle size-categories. The standard deviation of the system reduction efficiency indicates that the variation of for the ultra-fine particles is much larger than for the larger particle size categories (note that the y-axes have been cut to make it easier to see the results). The figure also indicates that filter F4 generally have a higher average system reduction efficiency than filter F1 for the ultra-fine particles, even if some experiments show results that are in the same range. For the other particle size ranges the system reduction efficiencies are in the same range for both filters.

The system reduction efficiency and the uncertainty limits for the individual no-filter experiments is shown in Appendix A, Figure A4. The figure shows that while reduction efficiency for the coarse and accumulation-mode-2 particles is 40-50% and ~60%, respectively, for the ultra-fine and accumulation-mode-1 particles the reduction efficiency is not statistically significant.

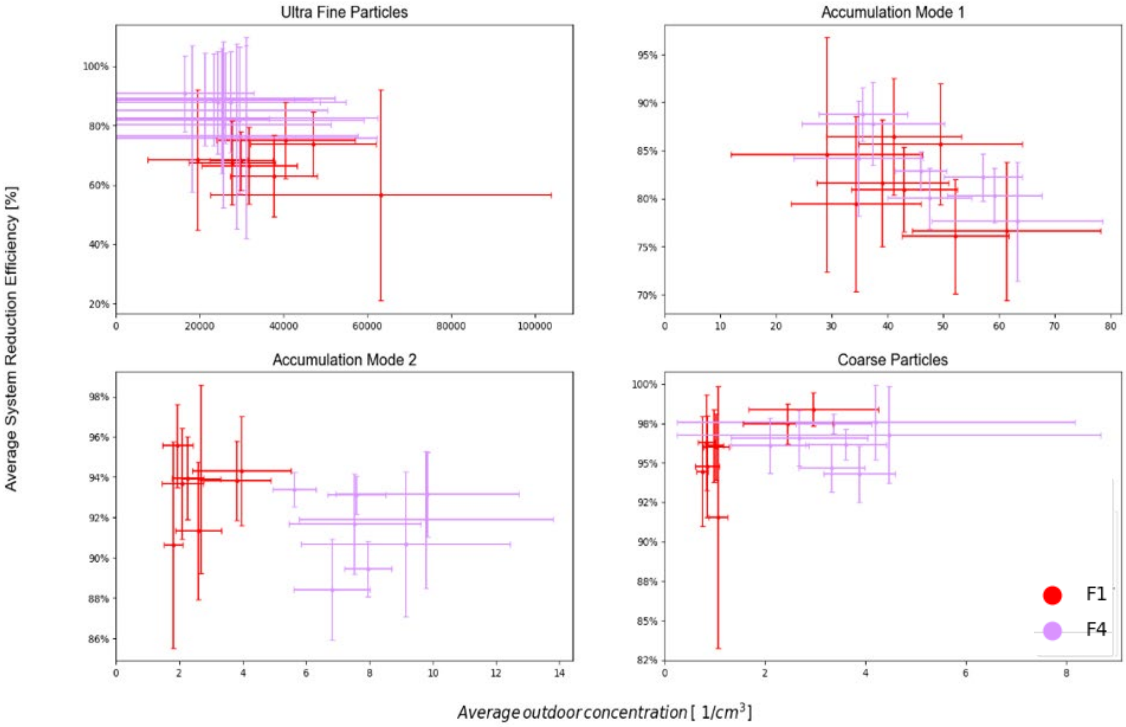


**Figure 4.5: Average system reduction for Filter F1 and F4 are plotted for four different types of particle sizes. The experiments are conducted when standing still in a tunnel at medium speed ventilation. The error bars represent one standard deviation of the outdoor concentration (x-axis) and one standard deviation for the indoor/outdoor-ratio (y-axis). Each point represents one experiment. Note that the y-axis has been adjusted and are different in the four graphs.**

### 4.1.3. Effects of the operation of the ventilation system

The air quality in the vehicle is dependent on operation of the ventilation system. A modern car or truck is usually equipped with sensors controlling outside air quality and closing the recirculation flap in polluted environment, for example in tunnels. The vehicles in this project were equipped with such sensors. The sensor system is working when the vehicle is set to the automatic mode, however, this can be over-turned by the driver. When driving in the automatic mode, the vehicle-fan speed was also regulated automatically by the vehicle to reach the requested climate comfort. Off course the driver can choose fan speed partly manual as well. An important factor was that the reduction efficiencies of the cabin air system for particles and affected gases depend on the ventilation rate. To distinguish between impact of the ventilation rate and impact of recirculation the standard experiments were always performed with no recirculation, using only outside air and with manual ventilation rate setup. Three different ventilation rates were examined for all filters tested, apart from 'medium' ventilation rate, one 'low' and one 'high' ventilation rate were tested. These tests were completed with test of the automatic mode with recirculation and ventilation rate in automatic operation, as a typical costumer would use the system. After the first campaigns it was found that even with manual ventilation setup the ventilation rate varies, and some recirculation can occur due to energy optimization of the vehicles, therefore in later campaigns the ventilation rate was manipulated from an external steering unit and set to a fixed rate, along with degree of recirculation and outside air intake flow.

The results for different ventilation rates show typically the pattern: “*reduced ventilation rate imply higher system reduction efficiency for* $PM_{2.5}$”. Figure 4.6 shows results for filter F1 where the ventilation rate was controlled externally. The pattern was in this case coherent for all driving cases, even though other parameters seem to influence the results as well. As the tests took place in real-life driving environments and took several hours, impacts from the not controlled parameters such as meteorological conditions or ambient concentrations can be large and hide the effect of the controlled parameter, in this case ventilation rate. Figure 4.7 shows such an example when sudden change in ambient concentration levels had larger impact on the observed reduction efficiencies than the change of the ventilation rate. For the medium ventilation rate the outside $PM_{2.5}$ concentrations dropped to very low levels (4-7 $\mu g/m^3$) which caused a drop in the observed reduction efficiency since with such low concentrations, interior sources of particles such as e.g. resuspension from cabin and ventilation system surfaces or emissions from persons in the vehicle become relatively more important. For the tunnel case where the $PM_{2.5}$ concentrations were high during all experiments, the pattern of change in the reduction efficiencies with ventilation rates was maintained. Both figures indicate that reduction efficiency in the automatic ventilation mode corresponds mostly to the reduction in low ventilation mode.

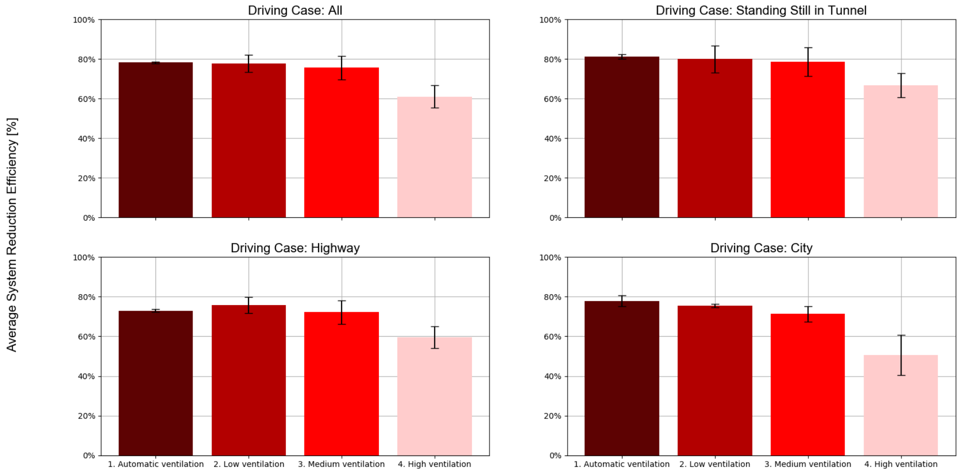


**Figure 4.6: System reduction efficiencies for $PM_{2.5}$ with different fixed ventilation rates, using Filter F1. The error bars represent one standard deviation. The mediums rate ventilation bars are based on six experiments and the rest of the bars are based on three experiments each.**

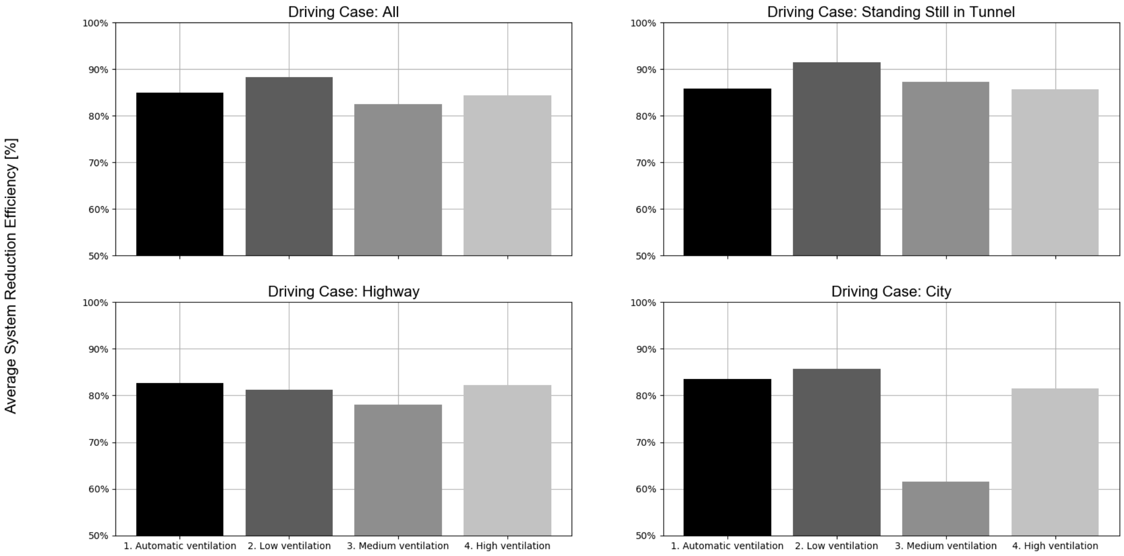


**Figure 4.7: System reduction efficiencies for PM$_{2.5}$ with different fixed ventilation rates, using Filter 7. The y-axes have been cut at 50 %.**

Comparing the reduction efficiencies for all particle size channels between 10 nm and 2.5 µm for the different ventilation rates, one can see that mainly the channels 10-139 nm are affected (Figure 4.8).

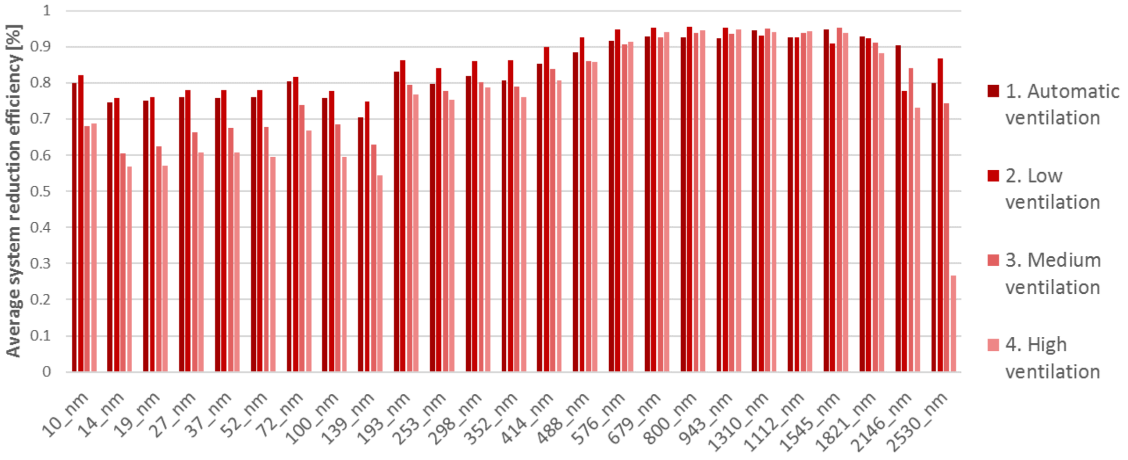


**Figure 4.8: Average system reduction efficiency for Filter F1 for individual particle size channels of the Mini-WRAS instrument, employed in the experiments, for the automatic ventilation mode and high, medium and low ventilation rates.**

The no-filter tests were performed with the different ventilation rates as well. Variability of the reduction efficiency in these experiments was high, not giving any conclusive answer on the relationship between the reduction efficiency and the ventilation rate (Figure A5 in Appendix A). This was an expected result as the element which is assumed to be driving the relation between the ventilation rate and removal efficiency is the filter.

The second ventilation parameter influencing the indoor air quality and the system reduction efficiency is the recirculation of air. Impact of partial recirculation on system reduction efficiency was tested in a series of experiments when the vehicle was in idle in quite high and stable levels of outdoor pollution and the recirculated air in ventilation was changed stepwise. Each test lasted 13-18 min allowing stabilization of the concentrations of gases and particles inside the cabin after the change of the recirculation rate and then the next experiment was started. In Figure 4.9 the system reduction efficiencies for the steady-state part of the experiments are shown for particle numbers in all size channels measured by the Mini-WRAS instrument. The values shown are averages of 3 measurement points (3 min) of the stabilized concentration part of the experiment and the reduction efficiencies are calculated as means of immediate reduction efficiencies.

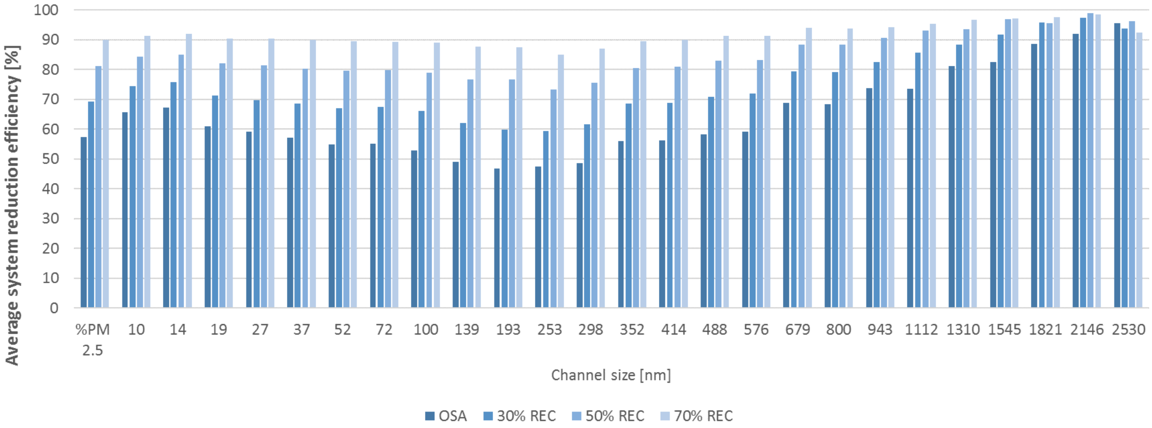


**Figure 4.9: Average of immediate system reduction efficiencies for size-resolved particle number concentrations for an aged filter with varying partial recirculation: OSA – outside air, 30%, 50% and 70% of recirculation air. Each experiment is an average of 3 immediate reduction efficiencies (3 min). Test performed in China, idle conditions.**

Volvo Cars has developed a new system with pre-ionisation to further improve the filtering efficiency for particles. This technique was developed within Volvo Cars, but was tested in this project. This new development, called Advanced Air Cleaning (AAC) system, improves especially filter reduction efficiency for particles in the nano- and accumulation-particles size range. Figure 4.10 shows reduction efficiencies for particles for Filter F1 and F2 (new and aged filters of the same quality) with and without the AAC system in operation. The tests were performed in China under city driving conditions. The system reduction efficiency for $PM_{2.5}$ was increased from 90% to 96% in case of the new filter and from 49% to 73% in case of the aged filter.

Figure 4.10: Average of immediate system reduction efficiencies for filter F1 (a) and F2 (b) with and without Advanced Air Cleaning system in operation. Experiments carried out in China under 30’ in city driving conditions.

### 4.1.4. Aged cabin air filter

Filters F1 and F2 shown Figure 4.3 and Figure A5 are the same filters, but filter F1 was new, while filter F2 have been aged for 500 hours (length of the service interval). Comparing these two filters in these two figures already reveal that the average filter reduction efficiency drops by the end of the filter’s lifetime. This can be explained, for example, by the decrease in electrostatic forces within the filter over time. An overview of the comparison between the two filters can be seen in Figure 4.1 and Figure 4.3 above. The results for different size channels are plotted next to each other in Figure 4.11, and as can be seen in the figure, the reduction decreases more for the smaller particles as filter ages, this is in line with the results shown in Figure 4.5.

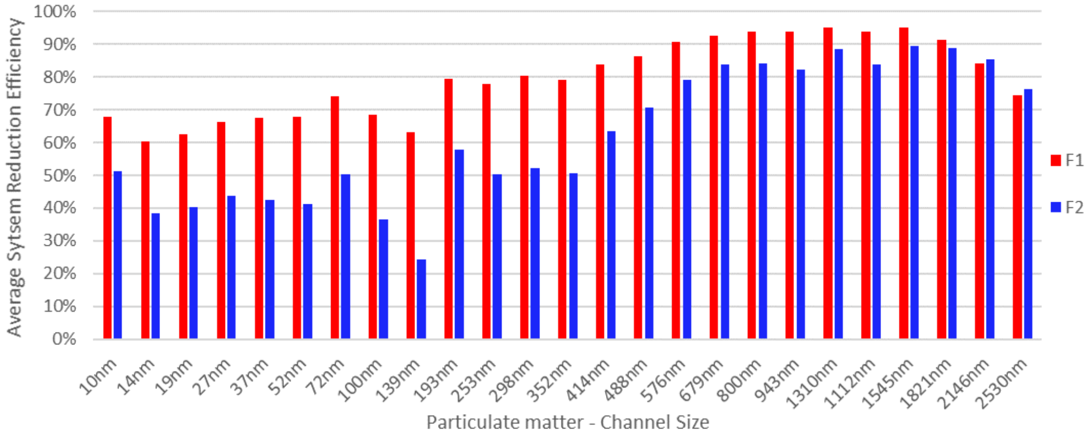


**Figure 4.11 Comparison of average system reduction efficiency for size-resolved particle number concentrations for a new (F1) and aged filter (F2), medium ventilation rate, measured with the Mini-WRAS.**

### 4.1.5. Impact of weather conditions and outdoor levels of air pollutants

The measured system reduction efficiencies were examined on influence of the ambient parameters, such as temperature and outdoor levels of air pollutants. Campaigns were intentionally conducted during dry weather conditions as measurements of PM during wet conditions are not possible. In general, plots of mean system reduction efficiencies for particle number concentration against the mean outdoor temperature during the experiment in question, show no correlation. In winter campaigns, some tests show positive correlation between the temperature and system reduction efficiency. Since these experiments have not used the external steering of the ventilation rate, we assume that the correlation is rather caused by the ventilation system operations at lower temperatures due to the heating of the cabin rather than a direct influence of the temperature on the system reduction efficiency.

Impact of the outdoor concentration levels on system reductions efficiencies has been studied and can be seen e.g. in Figure 4.1, Figure 4.2 and Figure 4.5. No clear trend can be found for $PM_{2.5}$, however, plots of efficiencies for particle-size categories reveal that at very low outdoor concentration levels the reduction efficiencies decreases for accumulation-2 and coarse mode particles. This is not caused by more air pollutants passing through the ventilation system but rather a result of the fact that at very low concentrations interior sources of particles such as resuspension from cabin and ventilation system surfaces or emissions from persons in the vehicle become relatively more important, as already discussed in section 4.1.3.

### 4.1.6. $PM_{2.5}$ measured gravimetrically on pumped filter samples

Pumped filter samples of $PM_{2.5}$ were taken in all core campaigns, however, this sampling experienced several experimental issues, which resulted in failure of the methodology in the first three campaigns. The first issue was occasionally too high temperature at the sampling place causing evaporation of the semi-volatile part of the sampled PM and the second issue was that the indoor

PM samples were below the level of detection of the gravimetric analysis. Reliable gravimetric data are hence available only from the Core campaign T summer 2 (trucks only). When comparing the absolute concentrations measured with the pumped samples with those measured with the online instruments, agreement is rather poor. This is not too surprising as the physical principles of these 2 methodologies are different. Comparison of the reduction efficiencies for the campaign with successful sampling was, however rather good (Figure A7, Appendix A).

The pumped $PM_{2.5}$ filter samples were further analysed for elemental content with energy dispersive X-ray fluorescence (ED XRF). The elemental analysis was performed only experimentally on few samples. The elemental composition can be related to the origin of the particles collected in the sample. The elemental analysis is not dependent of the gravimetric analysis, has lower detection limit and as the detected elements are not volatile the samples were assumed successful even if they were not successful for the gravimetric determination of $PM_{2.5}$. Figure A8 in Appendix A shows the reduction efficiencies for elements in the collected $PM_{2.5}$.

## 4.2. System reduction efficiency for $NO_2$

The chemiluminescence instrument for measurements of NO and $NO_2$ operates on the principle that the instrument measures NO and switches between direct measurement of NO in the sample and measurement of sample where $NO_2$ was converted into NO, i.e. measurement of NOx. $NO_2$ concentration is then calculated in the instrument by subtracting $NO_x$ with NO. Since the NO and NOx measurement is not simultaneous, the $NO_2$ concentrations tend to oscillate and occasionally become negative when concentrations have fast variation, as was the case with a driving car. Therefore, all NO and $NO_2$ measurements were resampled to one-minute temporal resolution before their mean and standard deviation have been calculated, see example of resampling in Figure A9, Appendix A.

The investigated filters can be divided into two categories, filters that don't reduces the $NO_2$ concentration (no-filter scenarios and filter F5 which is particle filter only), and filters that reduce the $NO_2$ concentration. To analyse the variation, the results are divided into two plots: filters F1, and F4 in Figure 4.12 and the no filter scenarios and filter F5 in Figure 4.13. The figures for filters F1 and F4 show reduction efficiencies for $NO_2$ of around 80%. In the city driving case, where the lowest outdoor concentrations occurred, one can see a decreasing trend in the system reduction efficiency at outdoor concentration ~15 ppb (does not correspond to an increase in indoor concentrations). For the tests without filter or with filter not affecting $NO_2$ the reduction efficiency decreases and turns negative at low outside concentrations. This phenomenon was likely caused by the fact that at low NOx concentrations some ozone was present in the outdoor air (at high concentrations it is completely depleted through fast reaction with NO emitted by the car exhaust) which reacts with NO to form $NO_2$ first after having been transported to the car interior through ventilation. NO is not affected by any filters.

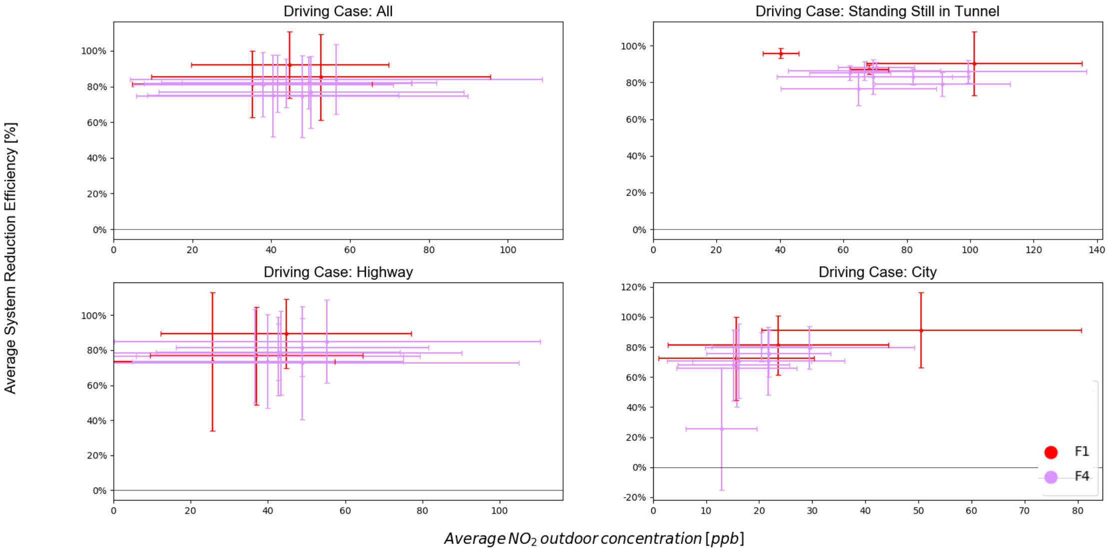


Figure 4.12: Average System reduction efficiencies for NO₂ for different driving cases plotted against outdoor concentrations, filter F1, F4 at medium speed ventilation. The error bars represent one standard deviation of the outdoor concentration (x-axis) and one standard deviation for the system reduction efficiency (y-axis). Each point represents one experiment.

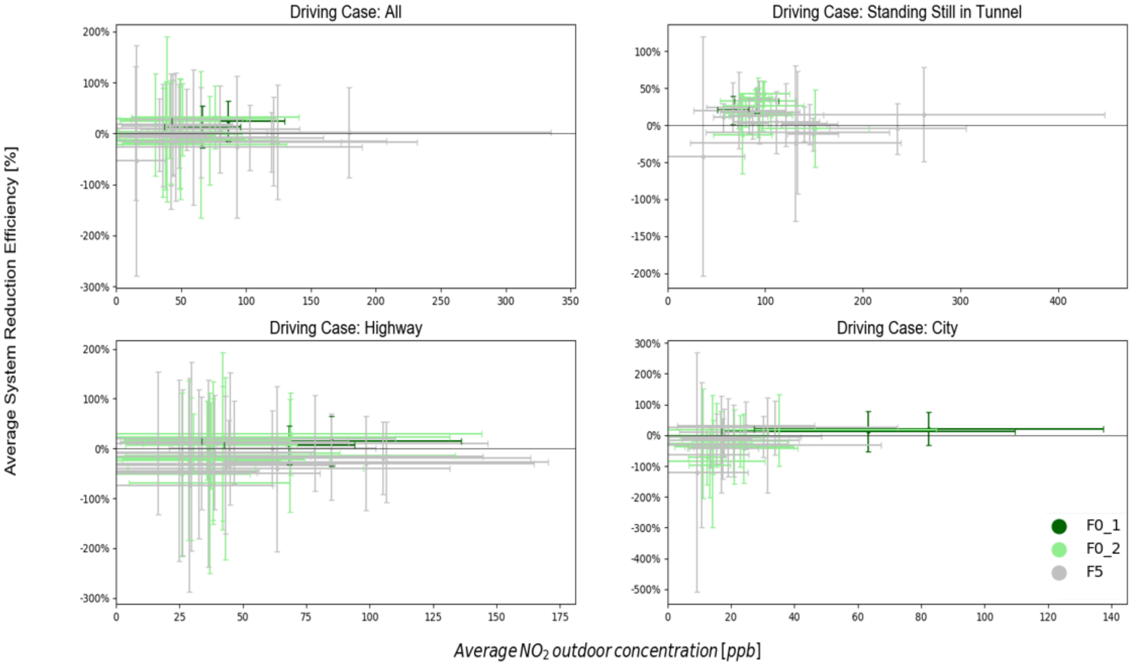


Figure 4.13: Average System reduction efficiencies for NO₂ for different outdoor concentration, no filter scenarios and filter F5, at medium speed ventilation. The error bars represent one standard deviation of the outdoor concentration (x-axis) and one standard deviation for the system reduction efficiency (y-axis). Each point represents one experiment. The Y-axis has been cut at ±150%.

The overall results are presented in Figure 4.14. The low reduction and the high standard deviation for filters F2 and F6 in the city driving case were due to the low outdoor $NO_2$ concentration when these experiments were carried out, as discussed above.

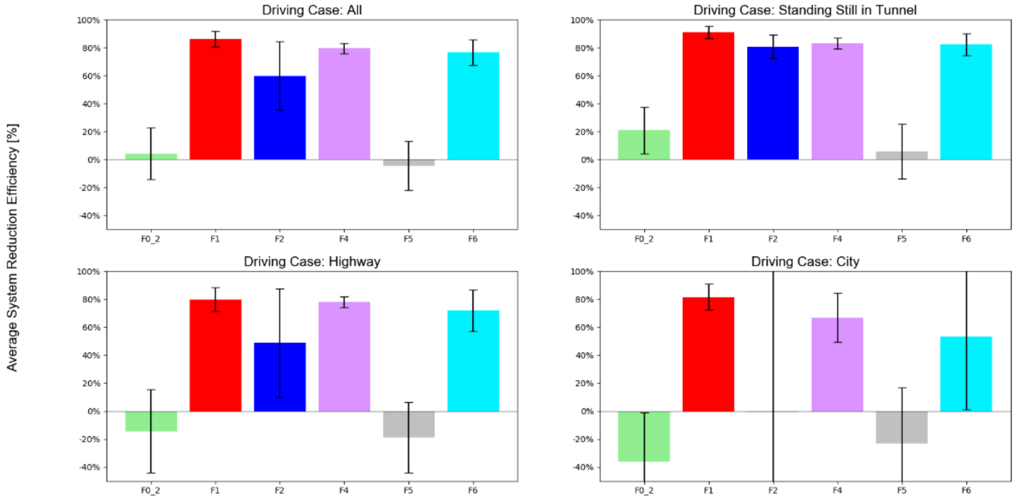


**Figure 4.14: Summary of system reduction efficiencies for $NO_2$ at medium ventilation rate. Only filters that have more than three measurements have been included in the graph. The y-axis has been cut at -50% and 100%. The error bars represent one standard deviation.**

## 4.3. PAH

System reduction efficiencies for gaseous and semi-volatile PAH, 16 US EPA PAH, particulate PAH and benzo(a)pyrene toxic equivalents are shown in Figure 4.15 together with their uncertainties. There was almost no effect on gas-phase PAH and US EPA PAH 16 while the particle-bound PAH were reduced in experiments employing filters. This indicates that the filters do not reduce the gas-phase PAH while they do reduce the particulate matter. US EPA PAH 16 covers also the particle-bound species, however, these species were typically three orders of magnitude lower in ambient concentrations than the most abundant gas-phase species, i.e. naphthalene. The B(a)P toxic equivalents contain only particle-bound species. Figure A10 in Appendix A shows reduction efficiencies for different ventilation rates. The particle-bound groups show similar pattern as found for the particulate matter.

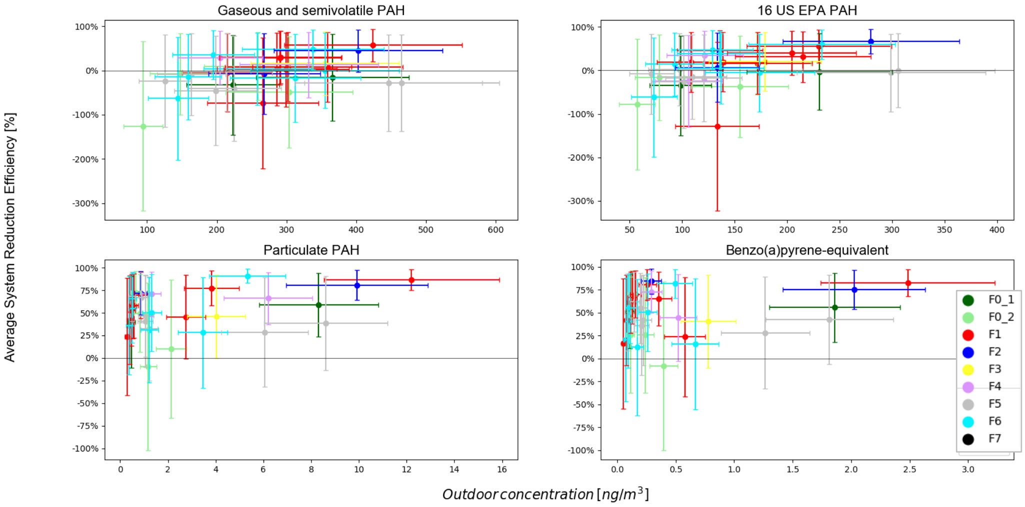


**Figure 4.15: System reduction efficiencies for the gaseous and semivolatile PAH, 16 US EPA PAH, particulate PAH and B(a)P toxic equivalents in all experiments, plotted against the outdoor concentrations. The error bars represent one standard deviation of the outdoor concentration (x-axis) and one standard deviation for the system reduction efficiency (y-axis). Each point represents mean of 1 test.**

## 4.4. VOC

The system reduction efficiencies vs. the outdoor concentrations for the sum of VVOC, sum of VOC (tot-VOC) and BTEX, with error bars, are presented in Figure 4.16.

The means in the VVOC system reduction efficiency plot is randomly distributed along the zero-line, neither the positive nor the negative values are significantly different from zero (the y-axis error bars are crossing the zero line). The in-vehicle concentrations mirror the outside concentrations so the VVOC inside can be determined from the outdoor values regardless the vehicles ventilation setting and filters.

For VOC and BTEX the system reduction efficiencies of 40-70 % were found for tests with filters F1, F4 and F6.

The system reduction efficiencies for benzene and 1,3-butadiene are presented in Figure 4.17, plotted vs. the outdoor concentrations. Reduction efficiency for 1,3-butadiene, belonging to the VVOC category, was not significantly different from zero, as indicated by the error bars in y-axes direction. The high standard deviations are results of the very low 1,3-butadiene concentrations measured. The concentration of 1,3-butadiene inside the vehicles follows its ambient concentrations. The concentration of benzene is reduced, especially by filters F1 and F4 but also by other filters such as F2 and F6.

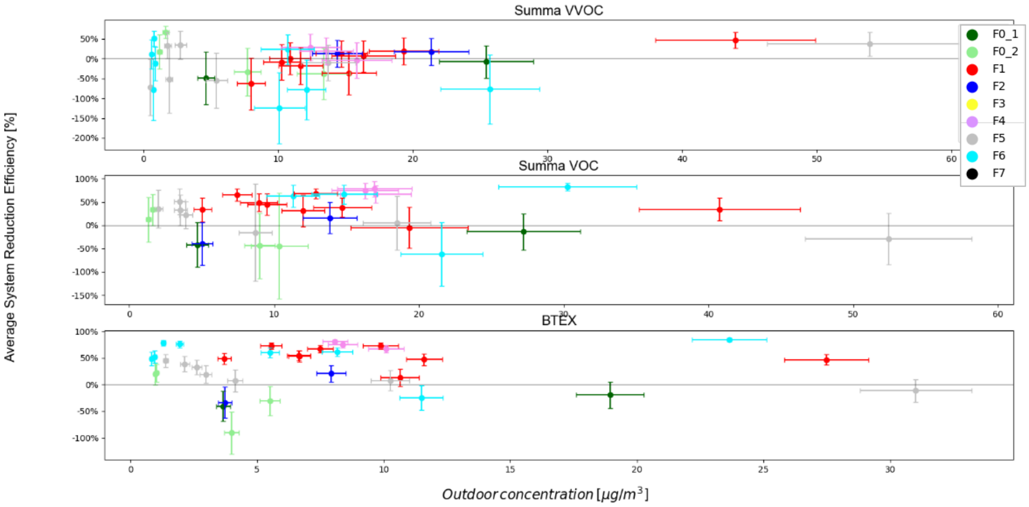


**Figure 4.16: System reduction efficiencies for VVOC, VOC and BTEX plotted with error bars. The error bars represent 1 standard deviation of the outdoor concentration (x-axes) and of the system reduction efficiency (Y-axes). The error bars represent one standard deviation of the outdoor concentration (x-axis) and one standard deviation for the system reduction efficiency (y-axis). Each point represents mean of 1 test.**

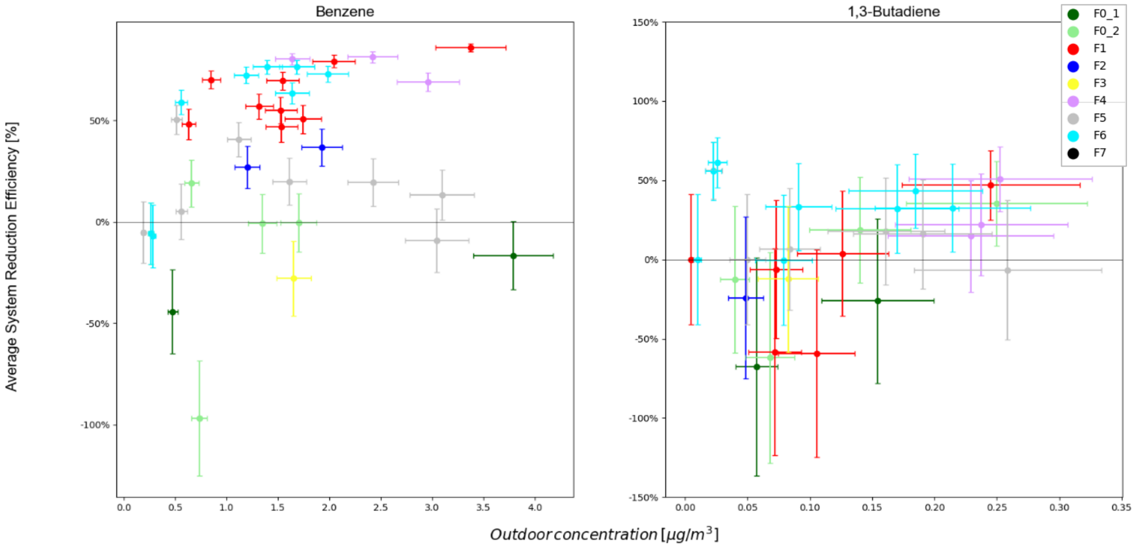


**Figure 4.17: System reduction efficiencies for benzene and 1,3-butadiene, plotted with error bars. The error bars represent 1 standard deviation of the outdoor concentration (x-axes) and of the system reduction efficiency (Y-axes). Each point represents mean of 1 test.**

Those VOC which are affected by the filter show similar trend of the system reduction efficiency change with changing ventilation rate as particulate matter. Figure 4.18 shows how the system reduction efficiency of filter F1 varies at different setting of the ventilation fan. Automatic and low-speed setting show higher reduction efficiencies than medium- and high-speed ventilation.

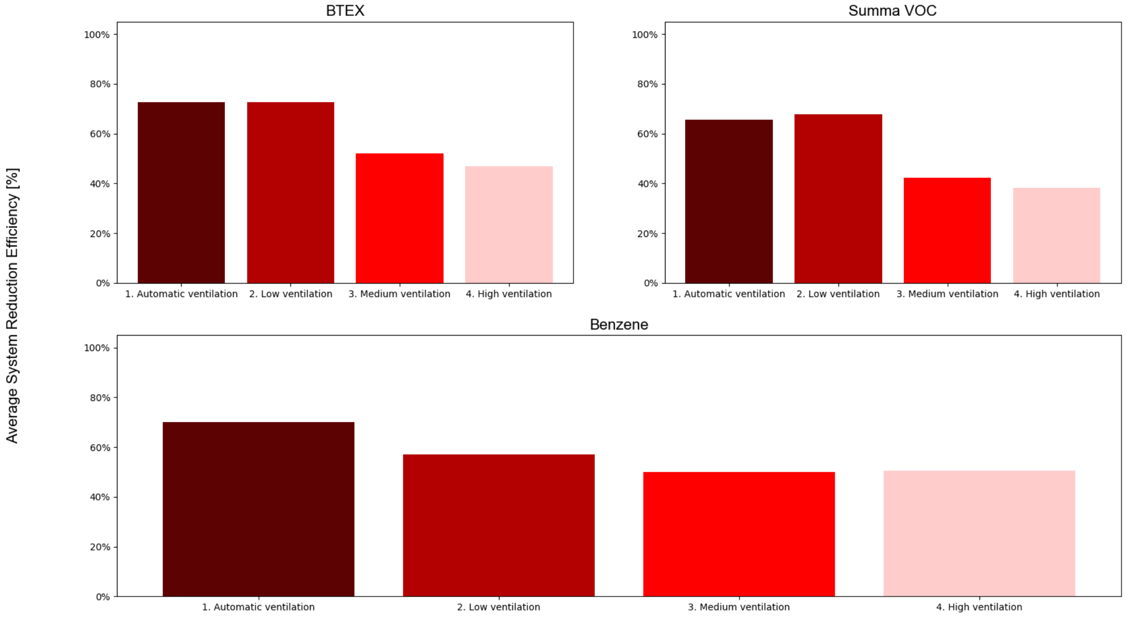


**Figure 4.18: Summary of system reduction efficiencies for BTEX, VOC and benzene as a function of ventilation setting; filter F1.**

## 4.5. Recirculation experiments

The ventilation rate has dual impact on how fast the particle concentration decreases since at higher rate the filtration rate of the cabin air through the cabin air filter is faster and at the same time the filter becomes less effective as shown in section 4.1.3. This makes the ventilation rate an important factor to consider when comparing the rate constants. The tests in each campaign that were run with the same filters and the same ventilation rates were sorted into groups. The mean of the reduction rate constants in each group was calculated and the standard deviation for each mean was calculated by taking the squared root of the mean variance of the reduction rate constants.

Figure 4.19 shows the results for the rate constants for F4 for the counts of the particles in the accumulation mode for three different ventilation speeds (L: 30 l $s^{-1}$, M: 40 l $s^{-1}$, H: 60 l $s^{-1}$) from the Core T summer 2018 campaign and a marked difference can be seen between the average rates depending on ventilation speed. After normalizing the reduction rate constants to compensate for the difference in ventilation rate, the resulting rates are very similar and even if the difference is not statistically significant one can see the same trend for the normalized rate constants for low, medium and high ventilation rate as seen in the driving experiments in section 4.1.3 (Figure 4.19, right). The normalized reduction rate constants for each filter were grouped for each campaign, and averages and standard deviations were calculated as previously described (Figure 4.20).

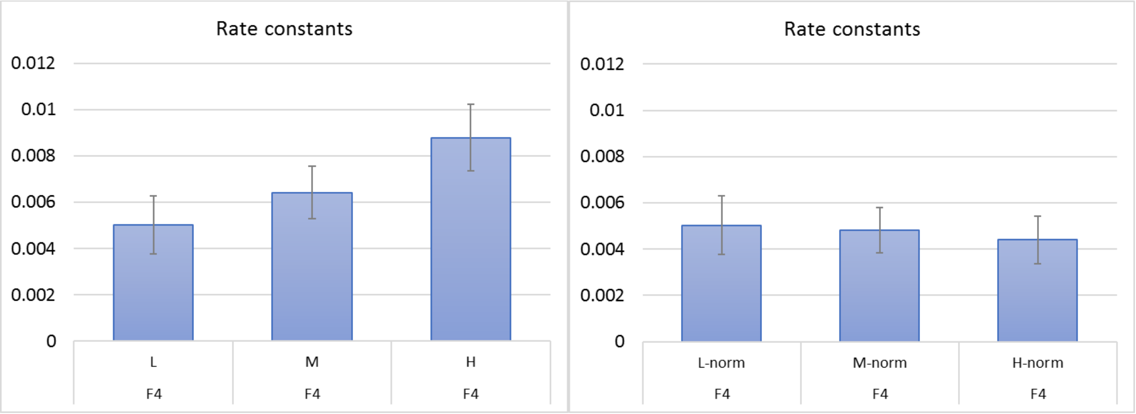


**Figure 4.19: The average reduction rate constants and their standard deviations for the accumulation mode particles (particle number concentration) calculated from recirculation experiments employing filter F4 and three different ventilation rates conducted during the 2018 core campaign T Summer 2 (left). The same reduction rate constants normalized by the relative differences of the respective ventilation rates from the low ventilation rate and their standard deviations (right).**

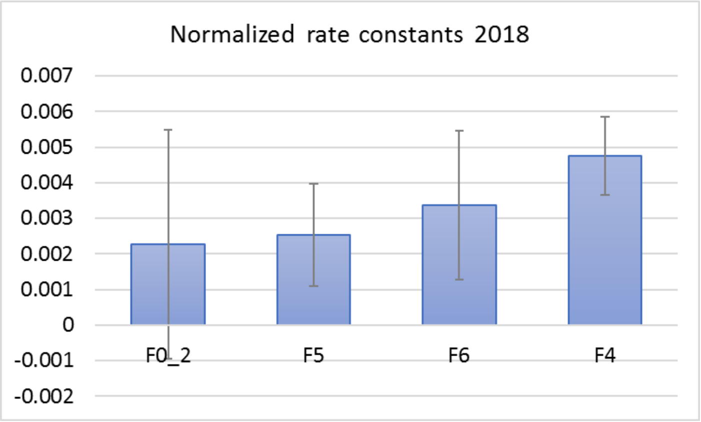


**Figure 4.20: The average reduction rate constants normalized by the relative differences of the ventilation rates in the experiments and their standard deviations for the accumulation mode particles (particle number concentration) calculated from recirculation experiments employing filter F0_2, F5, F6 and F4.**

For the 2018 campaign, a one-way analysis of variance (ANOVA) was run. The ANOVA- test indicated that there was a statistically significant difference among the groups and the subsequent Tukey-test of significance showed that the statistically significant difference was between F4 and F0_2, F4 and F5 as well as F4 and F6 but not the other combinations. There is also a difference between the rate constants for accumulation mode 1, accumulation mode 2 and course particles for the same filter as can be seen in Figure 4.21. The Filters seem to be more efficient for the larger particles.

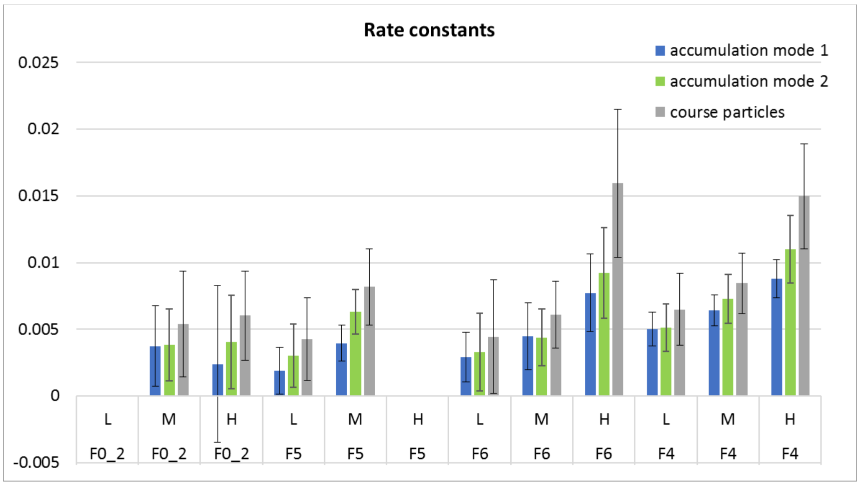


**Figure 4.21: Rate constants (not normalized) for different filters and ventilation speeds for accumulation mode 1 (blue), accumulation mode 2 (green) and course particles (grey).**

## 4.6. Personal exposures

During the passenger car drivers' personal exposure campaign 11 personal full-shift air samples were collected: three drivers drove various runs in the city (4, 9, 10), four drivers mixed driving (3, 5, 6, 8) and four drivers only country road (1, 2, 7, 11). During the truck drivers' personal exposure campaign additional 10 personal full-shift air samples were collected. The drivers drove short distances on Hisingen, however, due to linguistic hinders for communication, completed forms with driving schedule were not received and therefore we cannot specify the driving patterns in further detail. In both campaigns two different samples were collected for each driver: one with a personal carried sampling kit and one with a sampling kit placed inside the vehicle, close to the driver to see potential differences in total and in-car exposures.

Figure 4.22 shows full-shift exposures to B(a)P as mean exposure concentrations. The occupational exposure limit (OELs) for B(a)P is 2 µg/m$^3$ as 8-hour mean for a working shift and all drivers' exposure were more than three orders of magnitude below that limit. For comparison, the recommended mean concentration in the WHO guideline for indoor environment for B(a)P, corresponding to a mean lifetime exposure leading to an excess lifetime cancer risks of 1/100 000 increase, is 0.12 ng/m$^3$ (WHO 2010) and the ambient air quality standard (AAQS) for annual mean concentration is 1 ng/m$^3$. The measured exposures were around (both below and above) the guideline value for the indoor air, however, well below the AAQS value. Exposure to B(a)P was slightly higher among the passenger car drivers compared with truck drivers and the exposures measured with the stationary samplers were in most cases similar to the personal ones. Among the truck drivers two individuals (Truck 5 and 6) had much higher personal exposures than the stationary ones, the reason was suspected to be cigarette smoking.

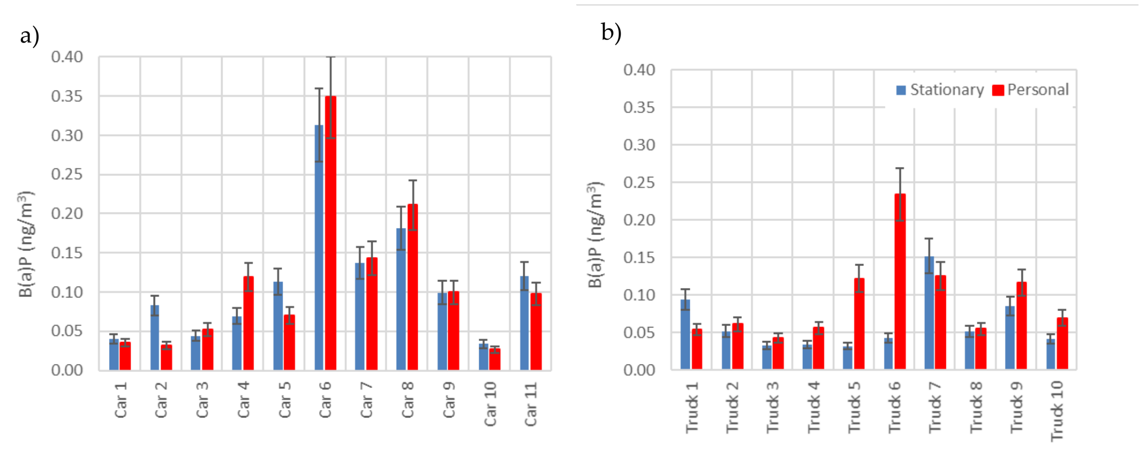


**Figure 4.22 Personal exposure to B(a)P measures with stationary and personal-carried PAH samplers during an 8-hour shift in passenger cars (a) and trucks (b). Error bars show uncertainty of the laboratory analyses.**

Figure 4.23 shows full-shift exposures to 1,3-butadiene and benzene. In cars 1-4 1,3-butadien and benzene could not be detected due to a problem with the flow rate of the pumps. The OEL for 1,3-butadiene and benzene is 1.0 mg/m$^3$ and 1.5 mg/m$^3$, respectively. The drivers' shift exposures were between 0.05 and 6 µg/m$^3$ for 1,3-butadiene and between 0.1 and 1.7 µg/m$^3$ for benzene, i.e. several orders of magnitude lower than the respective OEL. The WHO indoor air guideline values for 1,3-butadiene and benzene are 0.2-1 µg/m$^3$ (AMM recommendation) and 1.7 µg/ m$^3$, respectively (WHO 2010) and the AAQS for annual mean concentrations exists only for benzene and is 5 µg/m$^3$. The differences between the personal and stationary samples show large variability among the drivers, in cars 7 and 8 the personal exposure to both discussed VOCs substantially exceeds the stationary exposure, in some other cases the stationary exposure is higher than the personal one, as e.g. benzene exposure in Trucks 1, 7 and 9.

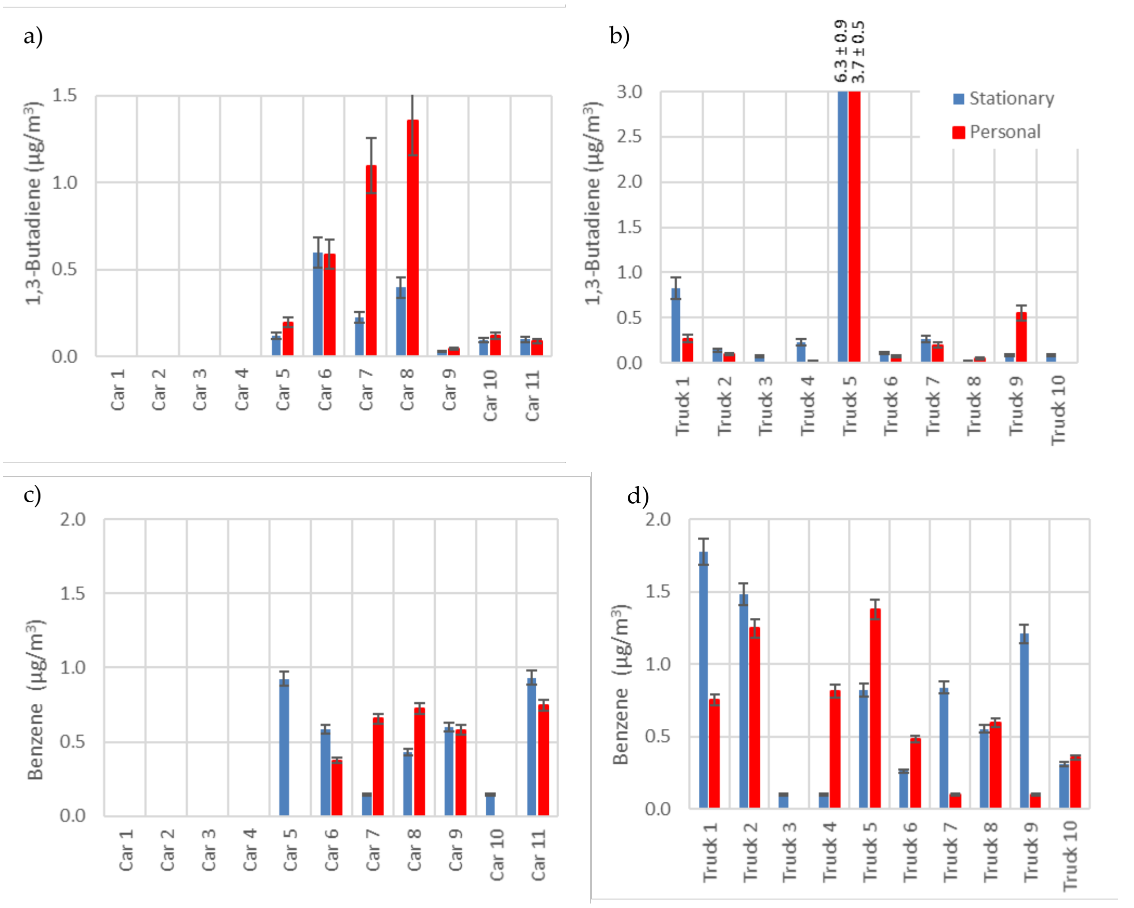


**Figure 4.23: Personal exposure to 1,3-butadiene (a, b) measured with stationary and personal-carried VOC samplers during an 8-hour shift in passenger cars (a) and trucks (b) and for benzene (c, d) measured during an 8-hour shift in passenger cars (c) and trucks (d). Error bars show uncertainty of the laboratory analysis.**

# 5. Innovations

Several different innovations have been addresses mostly focusing on developing a foundation for innovative solutions with the aim of improving air quality in vehicles. As a basis for the work, data from the project were used, along with literature data regarding air quality in vehicles, both from a health perspective and a measurement perspective. Two main goals were established within the work package i.e., developing a basis for requirement setting and developing principles for technical solutions in terms of improved air quality.

Within the project, different innovations have been studied and some have led to solutions in passenger cars. The main goal was to develop a robust requirement setting that can be used by the manufacturer for different substances, i.e. particles and gases such as VOC when new developments of the ventilation system, such as air filters or advanced air cleaning, are tested. The measurements in the project showed that the air in Gothenburg, where the majority of the measurements have been carried out, is relatively clean and that the concentrations of air pollutants vary a lot. Hence, the first challenge was to find a method that could be repeatable and useful for requirement setting. The method is based on the instrumental set-up used in the project. The requirement that has been implemented is for particles, where the Grimm Mini-WRAS is used for verification. After analyzing the particle concentration results, it was decided to split the requirement into different parts, one for absolute concentrations within the car cabin, along with one looking at the efficiency of the air cleaning system. Since $PM_{2.5}$ target levels currently have several international standards, i.e. WHO, EU EPA, this measure is used for the absolute concentration. The reason for not using $PM_{10}$ as a measure for the requirement is that existing filters in cars already remove larger particles in a satisfactory manner, i.e. particles larger than 2.5µm are seldom measured inside a running vehicle or they are the product of internal emissions, see section 4.1.3. Following the discussions in the literature stating that smaller particles are more dangerous than larger to human health (WHO 2010), it is important to take these into consideration in the requirement as well. The smaller particles have less impact on the mass concentration of $PM_{2.5}$, so another measure is needed to incorporate these sizes into the requirement. This was made by using the efficiency of the channel sizes from 10nm to 2.5 µm. The requirement developed for particles inside the cabin will be implemented in upcoming car projects at Volvo Cars.

Also studied was different solutions to improve both chemical and experienced air quality in terms of technical solutions. Along with technical solutions the project was also looking at new measurement techniques for monitoring the cabin air quality. Various technical solutions, such as ionization and different types of particle sensors were tested and evaluated. There are a variety of different sensors and they have different performance and quality. The sensors were tested in both cars while driving and in the laboratory with generated particles. A few sensors were found that could meet the technical requirements of the automotive industry.

Further, different ways to show the ambient air quality to the customers were investigated and the efforts will continue to find the best solution for visualization. Ways to use existing measurement data from air quality measurements in Sweden and in China for visualization was evaluated.

In order to further develop the area of air quality in vehicles with respect to nucleation particles, the Airmodus A11 nCNC-system measuring particles in the range 1-4 nm was tested. It was possible to use the instrument in a running car, as well as standing still. The results are in line with what we have seen with the instruments in the study. However, as the 1-4 nm particles correspond to the molecular clusters of nucleating particles, the time series of concentrations measures by this instrument reveal formation of new particles, presumably from organic substances released from the interior materials. This has to be further investigated and may be a whole project in itself.

As addressed earlier, it is not always possible to run a full-scale measurement campaign. Therefore, it was tested in China to run a scaled down campaign. Volvo Cars conducted continuous measurements of particulate matter (PM) with Grimm MINI-WRAS instruments while IVL carried out discrete pumped sampling for analyses of PM mass with a gravimetric method, PAH and of non-methane hydrocarbon compounds (NMHCs) (Moldanová et al. (2017). The aim of the measurement campaign was to test the measurement methodology developed during the project under Chinese conditions, both in terms of the measurement technique and logistics, and to investigate the indoor - outdoor ratios of concentrations of PM, PAH and NMHCs and ventilation system removal efficiencies in the air pollution situation of a Chinese city. The measurements gave consistent results demonstrating that this methodology can be successfully applied for testing of ventilation systems under real driving conditions. (Moldanová et al., 2017)

There was a broad goal within the project to investigate innovation for improved air quality, along with a few more specific goals for requirement setting and technical solutions. During the course of the project several new ideas have been tested as described above. In the end a requirement has been developed and implemented, both ionization for improved efficiency and sensors have been realised and are now part of the Volvo Cars offer. Only part of the technical solutions where tested within the project, the ideas have mainly been developed within Volvo Cars. However, the AQIFOR project have been a good base for discussion and development of testing for technical solutions. Along with that we have taken the opportunity to invite other, such as Airmodus to run their instruments in the cars. This work and has given us the opportunity to test working with smaller particles and see how this can be used in further developments and projects.

AB Volvo will use the development of measurement methodology and background material as a base to develop internal standards for requirements on reduction efficiencies of air cleaning components for particles and other air pollutants on ventilation system level coupled to this methodology. Recommendations were summarised in a separate report for AB Volvo.

A further development of the test methodology was done to study the reduction efficiency of dust and quartz for trucks employed in dusty environments. In order to cover the relevant particle sizes corresponding to the regulated PM metric, which is respirable dust and quartz content in respirable dust, pumped cumulative sampling was performed in a truck driving in a gravel pit. The sampling equipment and analyses were provided by AMM University of Örebro, and the method was complemented with parallel measurements with in-line Grimm aerosol spectrometer which also provides measurement of respirable dust. Even if this method is not accepted as testing method for the working environment certification, the online measurements provide valuable information on conditions and frequency of high-concentration episodes in the vehicle and their contribution to the full-shift mean exposure. The test duration was relatively short and more experimental work is needed for development of a robust and reliable methodology. The results of this campaign are presented in internal report by Rydström and Lindskog (2017).

# 6. Discussion, conclusions and future outlook

The air pollutants of greatest concern with respect to negative health impact from chronical exposure is $PM_{2.5}$, $PM_{10}$, ozone, $NO_2$, B(a)P and 1,3-butadiene (WHO, 2006; WHO 2018). There is increasing scientific evidence that particles in the nanoparticle size range, rather in terms of number concentration than mass concentration, and some PM compounds are more harmful than the general $PM_{2.5}$. The compound often discussed is elemental or black carbon corresponding to soot particles (WHO 2006, WHO 1013, WHO 2018 and references there in). Since road vehicles are important sources of all the primary pollutants on the list above, roads and streets, where the road vehicles move around, are concentration hotspots for these air pollutants. Ozone is a secondary air pollutant formed from emissions of NOx and VOC in photochemical reactions, however, in the first step the nitrogen monoxide (NO), which is the main NOx species in the vehicle exhaust, reacts with ozone, causing its depletion close to the emission source. Ozone concentrations in the direct vicinity of streets and roads are therefore generally low and the traffic environment is hence not considered to be coupled to increased ozone exposure. In AQIFOR we have investigated in detail system reduction efficiencies, as well as concentration levels in vehicle cabins, for size-resolved PM including short study for soot, $NO_2$, PAH including B(a)P and VOC species including 1,3-butadiene. Ozone was not measured in AQIFOR, however, tests conducted prior to the project have shown generally low ozone concentrations inside the vehicle (Winnes et al., 2014)

The project has developed a methodology framework for investigation of air quality parameters in the vehicle cabins. It has systematically been applied in a series of experiments performed with passenger cars and trucks under a variety of driving, meteorological and ambient air pollution conditions. The experimental results improved the understanding of the processes and parameters relating the air quality in the vehicle cabins to outdoor air quality, ventilation, the ventilation system reduction efficiency for the air pollutants, and interior emissions.

In agreement with previous studies, this project concluded that the most important parameters influencing the air quality in the vehicles were the cabin air filter and the air pollution levels outside the vehicle. The AQIFOR measurements showed statistically significant and repeatable differences between the different cabin air filters tested in terms of the system reduction efficiency for air pollutants targeted by the filters. It was also found that experiments employing filters with low removal efficiency or no filters showed much higher variability in the system reduction efficiency compared to experiments with more effective filters. A series of tests were performed to investigate the impact of the ventilation rate on the system reduction efficiency. Repeatable differences were found for the individual filters, with decreasing reduction efficiency for increasing ventilation rates. An additional parameter with important impact on the system reduction efficiency was the air recirculation. This parameter was not investigated in the same manner as the ventilation rate, it was however, tested in a series of shorter experiments at high air-pollution levels.

The experimental dataset was analysed considering the influence of outside parameters such as temperature and outdoor air pollutant concentrations. No impact of outside temperature on the system reduction efficiency was found. Also, the correlation to the outdoor concentrations was rather weak, and no correlation was seen at moderate and high levels of air pollutants. However, at low concentration levels the reduction efficiency decreased, most likely caused by the fact that at these levels the indoor sources of gases and particles, and the resuspension of particles from the ventilation system and cabin surfaces, became more important than the outdoor source.

The experimental driving loop was divided into three sections, city driving, highway driving and standing in the tunnel, where both the expected driving pattern and air pollution levels differed. The tunnel section represented system reduction efficiency tests at high outdoor air pollution levels common e.g. in many cities in Asia. Regarding the system reduction efficiency, the differences between the different driving cases did not show statistically significant and repeatable differences. However, the tunnel section mostly showed the highest reduction efficiencies and lowest variability in the results. These results are well in line with our current understanding of how the above discussed parameters influence the system reduction efficiency and with the fact that neither the driving pattern, nor the air pollution levels de facto differed systematically between the city and highway driving cases, while the tunnel measurements were associated with more stable conditions. Influence of the parameters related to the car operation have not been investigated up until now. Doing so will further improve the understanding of the processes involved and help with the interpretation of the data.

The different air pollutants were affected by the ventilation system in different ways. All investigated filters removed particles and by comparing the tests performed without cabin air filter to the ones with large differences can be seen: The no-filter tests showed no statistically significant removal efficiency for particles when the entire driving cycle was considered, while the mean system reduction efficiencies for $PM_{2.5}$ found in experiments with cabin air filter inserted spanned between 40 and above 80%. The no-filter experiments and experiments with less efficient filters showed also higher variability of the system reduction efficiencies compared to the experiments with the more efficient filters.

Particles in the size-range from 10 to 600 nm were found to be more difficult to remove by the filters than the coarser particles and the more efficient filters were also more efficient in this range. The small particles were found to contribute significantly to $PM_{2.5}$ in the cabin. It was found that the instruments used in this project (Grimm Mini-WRAS and TSI NanoScan & OPS) have low sensitivities in the size range 100 – 300 nm which is at the upper particle-diameter range for the nano-particle part of the instruments and below the lower particle-diameter range for the aerosol spectrometer part of these instruments. More attention needs to be payed to the instrument performance in this size range in terms of calibration with high-range SMPS instruments for PM in the traffic environment and to the development of specific calibration routines for the particle count data.

The experiments also provided valuable information on the removal efficiency for other pollutants. $NO_2$ was measured with 1 min resolution and the experiments showed statistically significant and repeatable differences for experiments employing different filters. The behaviour was similar to that of the PM.

The pumped sampling was initially burdened with experimental difficulties comprising of too high temperature at the sampling place for the outdoor samples, in case of $PM_{2.5}$ samples also with too short sampling time resulting in samples below the detection limit of the laboratory analysis. After resolving these problems repeatable results for the system removal efficiencies were found for particle-bound PAH species and for some hydrocarbon groups, especially BTEX. The highly volatile VOC and PAH species showed no reduction and the data variability was high, showing no difference from the no-filter experiments. The pumped sampling does not give the same level of detail as the on-line measurements and especially for samples taken during several driving cycles conditions when ambient concentration levels or driving pattern may change. The combination of both methods therefore gives an advantage when interpreting the low-time-resolution samples. With help of temporal patterns in the online data the trends in system reduction efficiencies in the pumped samples can often be explained as we can expect covariation of some pollutants, e.g. PM in the nanoparticle- and accumulation-size ranges and B(a)P. Advantage of the pumped samples is the possibility of a detailed analysis for organic compounds, chemical composition of PM etc.

Each driving test was accompanied with a series of recirculation tests performed in the Lundby tunnel. These tests show consistent and repeatable results which can be interpreted as a reduction rate for the filter or of the ventilation system. The results of the recirculation experiments are consistent with the results of the driving experiments and could be further used for prediction of air quality in vehicle cabins with a simple model. However, this methodology has not been finalised within the project.

The project also included a couple of campaigns investigating exposure of professional drivers to in-vehicle pollutants. These campaigns followed exposure to PAH and VOC during entire shift of these drivers. The concentrations to which the drivers were exposed to were rather low, far below the occupational limits.

The measurement results have been collected in the AQIFOR database. This database is an important complement to the measurement methodology developed in the project. It provides baseline data for testing of new innovative air cleaning components along with the possibility to further improve our understanding of the influence of different parameters and processes through further analysis of the data already available along with identifying gaps where data is missing. The measurement methodology together with the understanding of the influence of different parameters developed in the AQIFOR project, enables robust testing of air cleaning efficiency of vehicle ventilation system in real-life driving. This is already taking place at Volvo Cars. The knowledge of impact of physical parameters and ventilation-operation parameters on the reduction efficiency and the concentration levels in vehicles is beyond the state-of-the-art development of this project.

Outlook for future research in the area includes improvements of particle measurements in the size range 100 – 300 nm with the easy-to operate particle instruments as well as deeper studies on reduction efficiencies for different compounds of PM, especially BC. Improvement of our understanding of behavior and levels of PM compounds in the cabin air, e.g. of soot and volatile PM would help both the technological development regarding the filters and knowledge on health impacts of air pollution in the vehicle cabin and potential for its reduction. The measurements of VOC could be modified to TVOC specified by UBA (2019) with specific emphasis on benzene. Further, measurements of formaldehyde, compound with specified target levels and often found in the traffic environment, would be beneficial. To better understand the concentration levels, especially when the air pollution level in the cabin originating from outside is low, further research in interior sources, i.e. emissions and resuspension from interior and ventilation system, secondary formation in atmospheric reactions, leakage from fuel/exhaust system as well as emissions from the passengers, is important. The recirculation tests conducted and evaluated in the project show potential for development of a relatively simple model for prediction of air pollution levels in the cabin.

At Volvo Cars this area has expanded over the time of the project, today two PhD students are working with air quality, one focusing on air quality system development in connection to energy efficiency and one with air quality, mostly endocrine disrupting chemicals, and risk assessments. Along with that continuous work to improve the air quality for the customer is ongoing. Volvo Cars now have two technical experts working with incoming air quality and health aspects related to in cabin exposure.

# 7. References

## Internal and open reports generated in or related to the project

Borre, R., Nilsson Rabbaa, M. (2015) Comparing Volvo Cars Particle Measuring Devices With IVL's. Internal report VCC.

Langer, S. (2015) ”Vägar till förbättrad luftkvalitet i fordon”, Literature study on existing knowledge. Internal IVL report.

Moldanová, J., Salberg, H., Langer, S., Potter, A., Rehngren, E., Weidolf, S. (2017) Report on measurement campaign on car cabin air quality, 6-9 January 2017, Shanghai. IVL report U5883 (internal report).

Rydström, A.-M., Lindskog, N. (2017) Dust in a truck cabin. A study in a gravel pit. IVL internal report

Winnes, H., Peterson, K., Moldanova, J., Fridell E. (2014) Particle measurements in a car. Final report For Volvo Cars Corporation. IVL report U4855 (internal report).

## General references

Abi-Esber L, El-Fadel M. (**2008**). In-vehicle CO ingression: Validation through field measurements and mass balance simulations. Science of the Total Environment 394, 75-89.

Abi-Esber L, El-Fadel M. (**2013**). Indoor to outdoor air quality associations with self-pollution implications inside passenger car cabins. Atmospheric Environment 81, 450-463.

Adams HS, Nieuwenhuijsen MJ, Colvile RN, McMullen MAS, Khandelwal P. (**2001a**). Fine particle ($PM_{2.5}$) personal exposure levels in transport microenvironments, London, UK. Science of the Total Environment 279, 29-44.

Adams HS, Nieuwenhuijsen MJ, Colvile RN. (**2001b**). Determinants of fine particle ($PM_{2.5}$) personal exposure levels in transport microenvironments, London, UK. Atmospheric Environment 35, 4557-4566.

AFS 2018:01. Hygieniska gränsvärden – Occupational Exposure Limits. Arbetsmiljöverket - Swedish Work Environment Authority, Stockholm.

ASHRAE 62-2001: Ventilation for acceptable indoor air quality. American Society for Heating, Refrigerating and Air-conditioning Engineers. Atlanta, GA 30329.

Bakhtiari R, Hadei M, Hopke PK, Shahsavani A, Rastkari N, Kermani M, Yarahmadi M, Ghaderpoori A. (**2018**). Investigation of in-cabin volatile organic compounds (VOCs) in taxis; influence of vehicle's age, model, fuel, and refueling. Environmental Pollution 237, 348-355.

Bekö G, Uldahl Kjeldsen B, Olsen Y, Schipperijn J, Wierzbicka A, Karottki DG, Toftum J, Loft S, Clausen G. (**2015**). Contribution of various microenvironments to the daily personal exposure to ultrafine particles: Personal monitoring coupled with GPS tracking. Atmospheric Environment 110, 122-129.

BBR 21: Rules for building BSF 2014:3. Boverket - the Swedish National Board of Housing, Building and Planning. **2014** (in Swedish).

Bigazzi AY, Figliozzi MA. (**2012**). Impacts of freeway traffic conditions on in-vehicle exposure to ultrafine particulate matter. Atmospheric Environment 60, 495-503.

Bohlin P, Jones KC, Strandberg B. (**2010**) Field evaluation of polyurethane foam passive air samplers to assess airborne PAHs in occupational environments. Environmental Science and Technology 44, 749-754.

Boogard H, Borgman F, Kamminga J, Hoek G. (**2009**). Exposure to ultrafine and fine particles and noise during cycling and driving in 11 Dutch cities. Atmospheric Environment 43, 4234-4242.

Briggs DJ, de Hoogh K, Morris C, Gulliver J. (**2008**). Effects of travel mode on exposures to particulate air pollution. Environment International 34, 12-22.

Chan LY, Chan CY, Qin Y. (**1999**). The effect of commuting microenvironment on commuter exposures to vehicular emission in Hong Kong. Atmospheric Environment 33, 1777-1787.

Chan AT, Chung MW. (**2003**). Indoor–outdoor air quality relationships in vehicle: effect of driving environment and ventilation modes. Atmospheric Environment 37, 3795–3808.

Chen X, Feng L, Luo H, Cheng H. (**2014**) Analyses on influencing factors of airborne VOCS pollution in taxi cabins. Environmental Science and Pollution Research 21, 12868–12882.

Chuang H-C, Lin L-Y, Hsu Y-W, Ma C-M, Chuang K-J. (**2013**). In-car particles and cardiovascular health: An air conditioning-based intervention study. Science of the Total Environment 452-453, 309-313.

FoHMFS 2014:18. Folkhälsomyndighetens allmänna råd om ventilation - Public Health Agency of Sweden, general advice on ventilation. Folkhälsomyndigheten - Public Health Agency of Sweden, Stockholm.

Fruin SA, Hudda N, Sioutas C, Delfino RJ. (**2011**). Predictive model for vehicle Air Exchange Rates based on a large, representative sample. Environmental Science and Technology 45, 3569-3575.

Geiss O, Tirendi S, Barrero-Moreno J, Kotzias D. (**2009**) Investigation of volatile organic compounds and phthalates present in the cabin air of used private cars. Environment International 35, 1188–1195.

Gladyszewska-Fiedoruk K. (**2011**). Concentrations of carbon dioxide in a car. Transportation Research Part D 16, 166–171.

Gulliver J, Briggs DJ. (**2004**). Personal exposure to particulate air pollution in transport microenvironments. Atmospheric Environment 38, 1-8.

Harik G, El-Fadel M, Shihadeh A, Alameddine I, Hatzopoulou M. (**2017**). Is in-cabin exposure to carbon monoxide and fine particulate matter amplified by the vehicle's self-pollution potential? Quantifying the rate of exhaust intrusion. Transportation Research Part D 54 (2017) 225–238.

Houston D, Wu J, Yang D, Jaimes G. (**2013**). Particle-bound polycyclic aromatic hydrocarbon concentrations in transportation microenvironments. Atmospheric Environment 71, 148–157.

Hudda N, Kostenidou E, Sioutas C, Delfino RJ, Fruin SA. (**2011**). Vehicle and driving characteristics that influence in-cabin particle number concentrations. Environmental Science and Technology 45, 8691-8697.

Hudda N, Fruin SA. (**2013**). Models for predicting the ratio of particulate pollutant concentrations inside vehicles to roadways. Environmental Science and Technology 14, 11048-11055.

Hudda N, Fruin SA. (**2018**). Carbon dioxide accumulation inside vehicles: The effect of ventilation and driving conditions. Science of the Total Environment 610-611, 1448-1456.

ISO 16000-6 (2012): Indoor air - Part 6: Determination of volatile organic compounds in indoor and test chamber air by active sampling on Tenax TA® sorbent, thermal desorption and gas chromatography using MS or MS-FID.

Janssen, N.A.H., Hoek, G., Lawson-Simic, M., Fischer, P., Bree, van, L., Brink, H.M. ten, Keuken, M.P., Atkinson, R., Anderson, H.R., Brunekreef, B., Cassee, F., 2011. Black carbon as an additional indicator of the adverse health effects of airborne particles compared to $PM_{10}$ and $PM_{2.5}$. Environmental Health Perspective 119, 1691e1699.

Johansson C, Silvergren S, Norman M, Sjövall B. (**2013**). Halter av partiklar och NOx i fordon i relation till omgivningsluftens halter. Underlag för skattning av trafikantexponering. – Levels of particles and NOx in vehicles in relation to ambient levels. Basis for exposure estimatate of the passengers. SLB 1:2013 (in Swedish).

Joodatnia P, Kumar P, Robins A. (**2013**). The behaviour of traffic produced nanoparticles in a car cabin and resulting exposure rates. Atmospheric Environment 65, 40-51.

Jung HS, Grady ML, Victoroff T, Miller Al. (**2017**) Simultaneously reducing $CO_2$ and particulate exposures via fractional recirculation of vehicle cabin air. Atmospheric Environment 160, 77-88.

Kaur S, Nieuwenhuijsen MJ. (**2009**). Determinants of personal exposure to $PM_{2.5}$, ultrafine particle counts and CO in a transport microenvironment. Environmental Science and Technology 43, 4737–4743.

Klepeis NE, Nelson WC, Ott WR, Robinson JP, Tsang AM, Switzer P, Behar JV, Hern SC, Engelman WH. (**2001**). The national human activity pattern survey (NHAPS): a resource for assessing exposure to environmental pollutants. Journal of Exposure Analysis and Environmental Epidemiology 11, 231-252.

Knibbs LD, de Dear RJ, Atkinson SE. (**2009a**) Field study of air change and flow rate in six automobiles. Indoor Air 19, 303-313.

Knibbs LD, de Dear RJ, Morawska L, Mengersen KL. (**2009b**). On-road ultrafine particle concentration in the M5 East road tunnel, Sydney, Australia. Atmospheric Environment 43, 3510-3519.

Knibbs LD, de Dear RJ. (**2010**). Exposure to ultrafine particles and $PM_{2.5}$ in four Sydney transport modes. Atmospheric Environment 44, 3224-3227.

Knibbs L., De Dear R, Morawska L. (**2010**). Effect of cabin ventilation rate on ultrafine particle exposure inside automobiles. Environmental Science and Technology 44, 3546-3551.

Lee ES, Zhu Y. (**2014**). Application of a high-efficiency cabin air filter for simultaneous mitigation of ultrafine particle and carbon dioxide exposures inside passenger vehicles. Environmental Science and Technology 48, 2328-2335.

Lee ES, Stenstrom MK, Zhu Y. (**2015**). Ultrafine particle infiltration into passenger vehicles. Part I: Experimental evidence. Transportation Research Part D 38, 156-165.

Lewné M, Nise G, Lind M-L, Gustavsson P. (**2006**). Exposure to particles and nitrogen dioxide among taxi, bus and lorry drivers. International Archives of Occupational and Environmental Health 79, 220-226.

Lewné M, Plato N, Gustavsson P. (**2007**). Exposure to particles, elemental carbon and nitrogen dioxide in workers exposed to motor exhaust. The Annals of Occupational Hygiene 51, 693-701.

Mayer A, Burtscher H, Loretz S, Kasper M, Czerwinski J. (**2018**). High air pollution in vehicle cabins due to traffic nanoparticle emission exposure and a solution for in-use vehicles. Materials Science and Engineering 421, 032018. Doi: 10.1088/1757-899X/421/3/032018.

Ott W, Klepeis N, Switzer P. (**2008**). Air change rates of motor vehicles and in-vehicle pollutant concentration from secondhand smoke. Journal of Exposure science and Epidemiology 18, 312-325.

Pasaoglu, G., D. Fiorello, A. Martino, G. Scarcella, A. Alemanno, A. Zubaryeva, and C. Thiel (2012) “Driving and parking patterns of european car drivers-a mobility survey,” Luxembourg: European Commission Joint Research Centre.

Ragettli MS, Corradi E, Braun-Fahrländer C, Schindler C, de Nazelle A, Jerrett M, Ducret-Stich RE, Künzli N, Phuleria HC. (**2013**). Commuter exposure to ultrafine particles in different urban locations, transportation modes and routes. Atmospheric Environment 77, 376-384.

Riediker M, Williams R, Devlin R, Griggs T., Bromberg P. (**2003**). Exposure to particulate matter, volatile organic compounds, and other air pollutants inside patrol cars. Environmental Science and Technology 37, 2084-2093.

Salthammer T, Mentese S, Marutzky R. (**2010**). Formaldehyde in the indoor environment. Chemical Reviews 110, 2536–2572.

Silvergren S, Sjövall B, Johansson C. (**2013**). Koldioxidhalter relaterade till fordon – Carbon dioxide levels related to vehicles. SLB 7:2013 (in Swedish).

Tartakovsky L, Baibikov V, Czerwinski J, Gutman M, Kasper M, Popescu D., Veinblat M, Zvirin Y. (**2013**). In-vehicle particle air pollution and its mitigation. Atmospheric Environment 64, 320-328.

UBA (2019). German Committee on Indoor Guide Values. https://www.umweltbundesamt.de/en/topics/health/commissions-working-groups/german-committee-on-indoor-guide-values#textpart-1. Approached 28 May 2019.

Weichenthal S, Van Ryswyk K, Kulka R, Sun L, Wallace L, Joseph L. (**2015**). In-vehicle exposures to particulate air pollution in Canadian metropolitan areas: The urban transportation exposure study. Environmental Science and Technology 49, 597-605.

WHO (**2006**). Air quality guidelines; Global Update 2005; Particulate matter, ozone, nitrogen dioxide and sulfur dioxide. WHO Regional Office for Europe

WHO (**2010**). World Health Organization. Selected pollutants. WHO indoor air quality guidelines. Copenhagen: WHO Regional Office for Europe; 2010. p. 1-454.

WHO **(2012)**. Health effects of black carbon. WHO Regional Office for Europe.

WHO **(2013)**. Health risks of air pollution in Europe – HRAPIE project. WHO Regional Office for Europe.

WHO (**2016**). Ambient air pollution: A global assessment of exposure and burden of disease. World Health Organization. http://www.who.int/iris/handle/10665/250141, 121 p.

WHO **(2018)**. WHO housing and health guidelines. World Health Organization. https://apps.who.int/iris/handle/10665/276001, 149 p.

Xiong J, Yang T, Tan J, Li L, Ge Y. (**2019**) Characterization of VOC emission from materials in vehicular environment at varied temperatures: Correlation development and validation. PLoS ONE 10(10): e0140081. doi: 10.1371/journal.pone.0140081.

Xu B, Wu Y, Gong Y, Wu S, Wu X, Zhu S, Liu T. (**2016**). Investigation of volatile organic compounds exposure inside vehicle cabins in China. Atmospheric Pollution Research 7,215-220.

Xu B, Chen X, Xiong J. (2018) Air quality inside motor vehicles' cabins: A review. Indoor and Built Environment 27, 452-465.

Yoshida T, Matsunaga I. (**2006**). A case study on identification of airborne organic compounds and time courses of their concentrations in the cabin of a new car for private use. Environment International 32, 58 – 79.

Yoshida T, Matsunaga I, Tomioka K, Kumagai S. (**2006**). Interior air pollution in automotive cabins by volatile organic compounds diffusing from interior materials: I. Survey of 101 types of Japanese domestically produced cars for private use. Indoor and Built Environment 15, 425-444.

You KW, Ge YS, Hu B, Ning ZW, Zhao ST, Zhang YN, Xie P. (**2007**). Measurement of in-vehicle volatile organic compounds under static conditions. Journal of Environmental Sciences 19, 1208-1213.

Yu N, Shu S, Lin Y, She J, Ip HSS, Qiu X, Zhu Y. (**2017**) High efficiency cabin air filter in vehicles reduces drivers' roadway particulate matter exposures and associated lipid peroxidation. PLoS ONE 12(11), e0188498.

Zagury E, Le Moulec Y, Momas I. (**2000**). Exposure of Paris taxi drivers to automobile air pollutans within their vehicles. Occupational and Environmental Medicine 57, 406-410.

Zhang GS, Li TT, Luo M, Liu JF, Liu ZR, Bai YH. (**2008**) Air pollution in the microenvironment of parked new cars. Buildings and Environment 43, 315–319.

Zhu Y, Hinds WC, Kim S, Shen S, Sioutas C. (**2002a**). Study of ultrafine particles near a major highway with heavy-duty diesel traffic. Atmospheric Environment 36, 4323–4335.

Zhu Y, Hinds WC, Kim S, Sioutas C. (**2002b**). Concentration and Size Distribution of Ultrafine Particles Near a Major Highway. Journal of Air and Waste Management Association 52, 1032-1042.

Zhu Y, Eiguren-Fernandez A, Hinds WC, Miduel AH. (**2007**). In-cabin commuter exposure to ultrafine particles on Los Angeles freeways. Environmental Science and Technology 41, 2138-2145.

# APPENDIX A: Further information on the AQIFOR measurement campaigns and results

Table A1: Overview of AQIFOR measurement campaigns and events

| Campaign/test name | Vehicle type | Period | Aim |
|---|---|---|---|
| **Core campaign PC, winter** | Passenger car | 7-15 Dec 2015 | Full-size AQIFOR campaign |
| **Core campaign PC, summer** | Passenger car | 22-31 Aug 2016 | Full-size AQIFOR campaign |
| **Core campaign T, winter** | Truck | 20-26 Jan 2016 | Full-size AQIFOR campaign |
| **Core campaign T, summer 1** | Truck | 20-27 Jun 2017 | Full-size AQIFOR campaign |
| **Core campaign T, summer 2** | Truck | 11-21 Jun 2018 | Full-size AQIFOR campaign |
| **Demonstration campaign T, summer** | Truck | 30 Apr- 6 Maj 2019 | Reduced AQIFOR campaign |
| **Personal exposure PC** | Passenger car | November/ December 2016 | Personal exposure drivers: VOC, PAH |
| **Personal exposure T** | Truck | Mars 2017 | Personal exposure drivers: VOC, PAH |
| **Instrument comparison lab+PC** | Passenger car | Jun-Jul 2015 | Comparison on-line PM instruments laboratory and driving test |
| **Instrument comparison ambient 1** | | 2 Feb 2017 | Comparison online particle instruments, pumped PM samples and station Femman data (TEOM) |
| **Instrument comparison ambient 2** | | 25-26 Apr 2018 | Comparison online particle instruments and station Gårda data (TEOM) |
| **Method test PC Pumped PM** | Passenger car | 3-5 May 2017 | Test of methodology for pumped $PM_{2.5}$ samples |
| **Sensor tests PC** | Passenger car | | |
| **Short campaign PC Kina 1** | Passenger car | 20 Mar - 1 Apr 2015 | Test of methodology passive samplers NOx, $SO_2$, VOC, PAH, aldehydes + PM online (WRAS) |
| **Short campaign PC Kina 2** | Passenger car | 15-17 Oct 2015 | Test of methodology pumped PAH and VOC + online particles (WRAS) |
| **Short campaign PC Kina 3** | Passenger car | January 2018 | Test of methodology pumped PM + PAH (high volume), VOC + online particles (WRAS) |
| **Dust campaign T** | Truck | 16–17 Jun 2017 | Test of methodology respirable dust + quarts (pumped) + online particles (Grimm) |

Table A2. Example of campaign protocols

## a) Campaign day protocol

| | | |
|---|---|---|
| Date | | |
| Personnel | Start-up | |
| | AM | |
| | PM | |
| | Shutdown | |

| Instrument | ID | Comment |
|---|---|---|
| Online particles inside | | |
| Online particles outside | | |
| NOx inside | | |
| NOx outside | | |
| Woehler inside | | |
| Woehler outside | | |

## b) Campaign day log

| | |
|---|---|
| Date | |
| Written by: | |
| Page # | |

Driving mode:

| | |
|---|---|
| H | Highway, trunk road, from Ullevimotet to Linnéplatsen |
| C | City - från Linnéplatsen to Ullevimotet |
| LT | Lundbytunnel |
| GNT | Gnistängstunnel |
| TT | Tingstadstunnel |
| S | Special case - standing, door opened |

| Time | Filter type | Experiment | Driving mode | Note |
|---|---|---|---|---|
| | | | | |
| | | | | |
| | | | | |
| | | | | |
| | | | | |
| | | | | |
| | | | | |

### c) Protocol for pumped samples

| | | | | |
|---|---|---|---|---|
| **Protocol for Sampling point** | | | | **AQIFOR** |
| Name | Göteborg, Truck, spring 2019 | | | |
| Date | 2019-…....-……. | | | |
| Experiment | 1. | | 2. | |
| **PAH: XAD tube+PM kasette** | **2 l/min (max 1m$^3$) 2 loops** | | | |
| Experiment | 1+2 | | | |
| | **Indoor** | | **Outdoor** | |
| Pump number | | | | |
| XAD tube number | | | | |
| PM kasette number | | | | |
| Time start | | | | |
| Time stop | | | | |
| Minutes sampled | | | | |
| Volume sampled | | | | |
| Notes | | | | |
| | | | | |
| | | | | |
| | | | | |
| | | | | |
| **VOC tubes** | **~0.15 l/min, 10 l (1h)** | | | |
| Experiment | 1. | | | |
| | **Indoor** | | **Outdoor** | |
| Pump number/flux | | | | |
| Tube number | | | | |
| Time start | | | | |
| Time stop | | | | |
| Minutes sampled | | | | |
| Volume sampled | | | | |
| Notes | | | | |
| | | | | |
| | | | | |

Figure A1. Inlet for the outside air samples a) for the online particle instrument in the passenger car, b) for the pumped samples in the passenger car (the front of the car in a and b is to the right), c) for all sampling of the outside air in the truck.

# Overview of pumped samples

**Table A3. Overview of number of driving loops covered by the individual pumped samples in each series and number of analyses on pumped samples taken during the AQIFOR core campaigns**

| Campaign | Pumped 1 | | | | Pumped 2 | | Pumped 3 | |
|---|---|---|---|---|---|---|---|---|
| | *Driving loops/ sample* | $PM_{2.5}$ mass | PAH | BC | *Driving loops/ sample* | PAH | *Driving loops/ sample* | VOC |
| Core PC, winter | *2* | 6 | 6 | 6 | *2* | 6 | *1* | 6 |
| Core T, winter | *2* | 4 | 4 | 4 | *2* | 4 | *1* | 4 |
| Core PC, summer | *2* | 8+3$^{*}$ | 8 | 8 | *2* | 8 | *1* | 8+2$^{**}$ |
| Core T, summer 1 | *4* | 5+2$^{\dagger}$ | 5 | - | *2* | 6 | *1* | 10 |
| Core T, summer 2 | *4* | 8 | 8 | - | *2* | 10 | *1* | 11 |
| Demonstration T, summer | | - | - | - | *2* | 7+3$^{\dagger\dagger}$ | *1* | 7+1$^{\dagger\dagger}$ |

$^{*}$ additional 3 samples for analyses for elemental content by ED XRF

$^{**}$ additional samples for car interior emissions of VOC

$^{\dagger}$ additional $PM_{10}$ samples from gravel pit measurements

$^{\dagger\dagger}$ 1 double samples for determination of analytical uncertainties

# Description of the sampling and analyses of $PM_{2.5}$ and its chemical compounds

The sampling line for PM mass is shown in Figure A2. The sampled air first enters the IVL PM sampler, a $PM_{2.5}$ impactor, holding a thin PFTE filter. The PM sampler is coupled to a vacuum pump which is preceded by a needle valve allowing adjustment of the sample flow. From the pump the sample passes through a gas meter to measure the volume of the sampled air. The PM sampler is designed for a sample flow rate of 17 l/min, which ensures a correct size separation of the $PM_{2.5}$. The sampled volume was c.a. 2 $m^3$ in all campaigns except of the last one (Core campaign T, summer 2), where ~4 $m^3$ samples were taken. The PM mass collected in the 2 $m^3$ samples was often close to the laboratory 25 µg 'Limit of quantification', bringing too high uncertainties to the measurements.

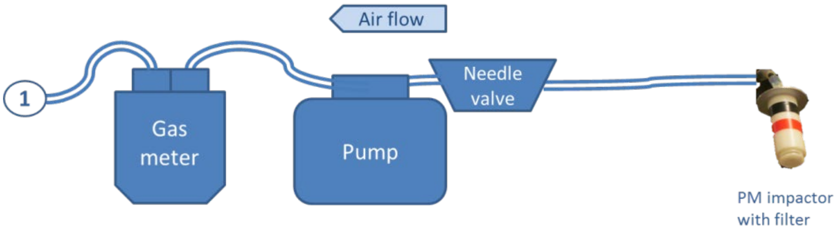


**Figure A2: Sampling line for $PM_{2.5}$**

The TF1000 filter collects the sampled PM and was later analysed with a gravimetric method for PM mass. The same filter was also analysed for PM-bound PAHs with GCMS (AMM, Bo Strandberg) and for light absorbing black carbon (BC) and UV-light absorbing PM (UVPM, corresponds roughly to organic PM), both with optical method (AMM, Peter Molnar). During the summer campaign with passenger car, additional $PM_{2.5}$ samples were collected on PTFE TEFLO filters which were analysed gravimetrically and for elemental content with ED XRF method. Table A3 above shows the full sampling and analytical matrix for the pumped samples.

# Description of sampling and analyses of polyaromatic hydrocarbons (PAH)

The sampling line for PAH included a filter holder followed by a glass column with XAD absorbent for sampling of PM-bound and gas-phase PAHs. This PAH sampler was coupled with a PTFE tubing to sampling pumps with sampling flow ~2 L/min. Different types of sampling pumps were used in this line (Gilian, Aircheck or GilAir), they were all battery-driven. The sampled volume was either logged by the pumps or was derived from the sampling time and the calibrated flow rate. The analysis of PAH followed the procedure described in Bohlin et al. (2010).

Prior to extraction, the PAH samples were spiked with 100 µL of internal standard (IS) containing deuterated US EPA 16 PAH (1 ng/µL). All samples were analyzed for PAHs by gas chromatography-mass spectrometry (GC/MS) on an Agilent 6890 GC coupled to an Agilent 5973 inert mass-selective detector. Injections were carried out in splitless mode, with a helium flow rate of 1.0 ml min-1, a front inlet temperature of 270°C and a DB5-MS capillary column (60 m x 0.25 mm x 0.25 µm). MS operated in positive electron ionization impact mode using selected ion monitoring (EI-SIM). A total of 32 PAH compounds were quantified including the 16 US EPA PAHs.

Table A4: List of quantified PAH; groups indicated with x, TEQ > 0 shown in the last column.

| 32 PAH analysed | 16 US EPA | Particulate | EPA 7 carcinogenic | EC 6 reported | EC 4 reported | B(a)P toxic equivalents (TEQ) |
|---|---|---|---|---|---|---|
| Naphthalene | X | | | | | |
| 2-Methylnaphthalene | | | | | | |
| 1-Methylnaphthalene | | | | | | |
| Biphenyl | | | | | | |
| 2,3-Dimethylnaphthalene | | | | | | |
| Acenaphthylene | X | | | | | |
| Acenaphthene | X | | | | | |
| 2,3,5-Trimethylnaphthalene | | | | | | |
| Fluorene | X | | | | | |
| 1-Methylfluorene | | | | | | |
| Phenanthrene | X | | | | | 0.0005 |
| Anthracene | X | | | | | 0.0005 |
| 2-Methylphenanthrene | | | | | | |
| 3-Methylphenanthrene | | | | | | |
| 1-Methylphenanthrene | | | | | | |
| 1-Methylanthracene | | | | | | |
| 2-Phenylnaphthalene | | | | | | |
| Fluoranthene | X | | | x | | 0.05 |
| Pyrene | X | | | | | 0.001 |
| 1-Methylfluoranthene | | | | | | |
| 1-Methylpyrene | | | | | | |
| Retene | | | | | | |
| Benzo(a)anthracene | X | x | x | | | 0.005 |
| Chrysene | X | x | x | | | 0.03 |
| 2-Methylchrysene | | x | | | | |
| Benzo(b)fluoranthene | X | x | x | x | x | 0.1 |
| Benzo(k)fluoranthene | X | x | x | x | x | 0.05 |
| Benzo(a)pyrene | X | x | x | x | x | 1 |
| Perylene | | x | | | | |
| Indeno(1,2,3-c,d)pyrene | X | x | x | x | x | 0.1 |
| Dibenzo(a,h)anthracene | X | x | x | | | 1.1 |
| Benzo(g,h,i)perylene | X | x | | x | | 0.02 |

# Description of sampling and analyses of Volatile Organic Compounds (VOC)

Sampling of NMHCs was carried out on absorbent tubes which were connected with tubing to the sampling pumps, either Scantec, Gilian GilAir Plus Personal Air Sampling Pump, or GilAir pumps. The sampled volume was 8-18 L, using a sampling air flow of 150-200 mL/min. The tubes contained adsorbents suitable to quantitatively collect $C_4$ – $C_{10}$ hydrocarbons including BTEX compounds and 1,3-butadiene.

The samples were analysed by gas chromatography with a flame ionization detector (GC-FID) or gas chromatography with a mass selective detector (GC–MS) as described further. The adsorbent tubes were thermally desorbed (Unity2 and Ultra, Markes) and the samples were analysed by gas chromatography (GC, Thermo). During the desorption stage the adsorbent tubes were heated to 250$^{o}$C (Tenax TA) or 360$^{o}$C (Carbopack X) during 10 minutes under a flow of helium. The desorbed components were refocused on a cold trap packed with Tenax TA/Carbosieve SIII cooled to –30 $^{o}$C. The trap was then heated rapidly to 300 °C in order to inject the retained analytes into the GC as a highly concentrated band of vapors. The GC was equipped with two analysis columns that were controlled by a Dean switch. The most volatile analytes (very volatile organic compounds) like butanes, butenes and 1,3-butadiene were separated on a PLOT-column (TracePLOT TG-BOND, Thermo) and detected by flame ionization detector (FID). Quantification was based on comparison of the peak area to the known peak areas of various concentrations of the standard analytes in a certified ozone precursor reference mixture (NPL). Volatile hydrocarbons (volatile organic compounds) like n-hexane to nonane were separated on a non-polar capillary column (TraceGold TG-1MS, Thermo) and detected by a mass spectrometer (ISQ LT, Thermo). Quantification by MS was based on comparison of the peak area of specific ion masses of each component. In addition to the certified ozone precursor reference mixture (NPL), a certified TO-15 reference mixture (65 components, Restek) was also used to calibrate the MS.

The limit of detection (LOD) was defined as 3 times the blank samples standard deviation of field- and lab-blanks.

**Table A5: Absorbent tubes and analytical methods used for sampling and analyses of NMHC in the different measurement campaigns.**

| Campaign | Pumped VOC samples | | |
|---|---|---|---|
| | Sampler | Analytical method | No. samples |
| Core PC, winter | Carbopack X | GC FID/FID | 6 |
| Core T, winter | Carbopack X | GC FID/FID | 4 |
| Core PC, summer | Carbopack X | GC FID+GC MS | 8+2 |
| Core T, summer 1 | Tenax/ Carbosieve SIII | GC FID+GC MS | 10 |
| Core T, summer 2 | Tenax/carbopack X | GC MS | 10 |
| Demonstration T, summer | Tenax/carbopack X | GC MS | 8 |

**Table A6**: **List of quantified VOC; groups indicated with x.**

| Compounds | VVOC | VOC | BTEX |
|---|---|---|---|
| Isobutane | x | | |
| n-Butane | x | | |
| trans-2-Butene | x | | |
| 1-Butene | x | | |
| Isobutene | x | | |
| cis-2-Butene | x | | |
| 1,3-Butadiene | x | | |
| Isopentane | x | | |
| 1-Pentene | x | | |
| n-Pentane | x | | |
| Isoprene | x | | |
| trans-2-Pentene | x | | |
| cis-2-Pentene | x | | |
| 2-Methylpentane | x | | |
| 3-Methylpentane | x | | |
| n-Hexane | | x | |
| Benzene | | x | x |
| Cyclohexane | | x | |
| Isooctane | | x | |
| n-Heptane | | x | |
| Toluene | | x | x |
| n-Octane | | x | |
| Ethylbenzene | | x | x |
| m+p-Xylene | | x | x |
| o-Xylene | | x | x |
| n-Nonane | | x | |
| Styrene | | x | |
| n-Butyl acetate | | x | |
| 1,3,5-TMB | | x | |
| 1,2,4-TMB + Decane | | x | |
| 1,2,3-TMB | | x | |

# Reduced measurement campaigns and instrument tests – supplemental information

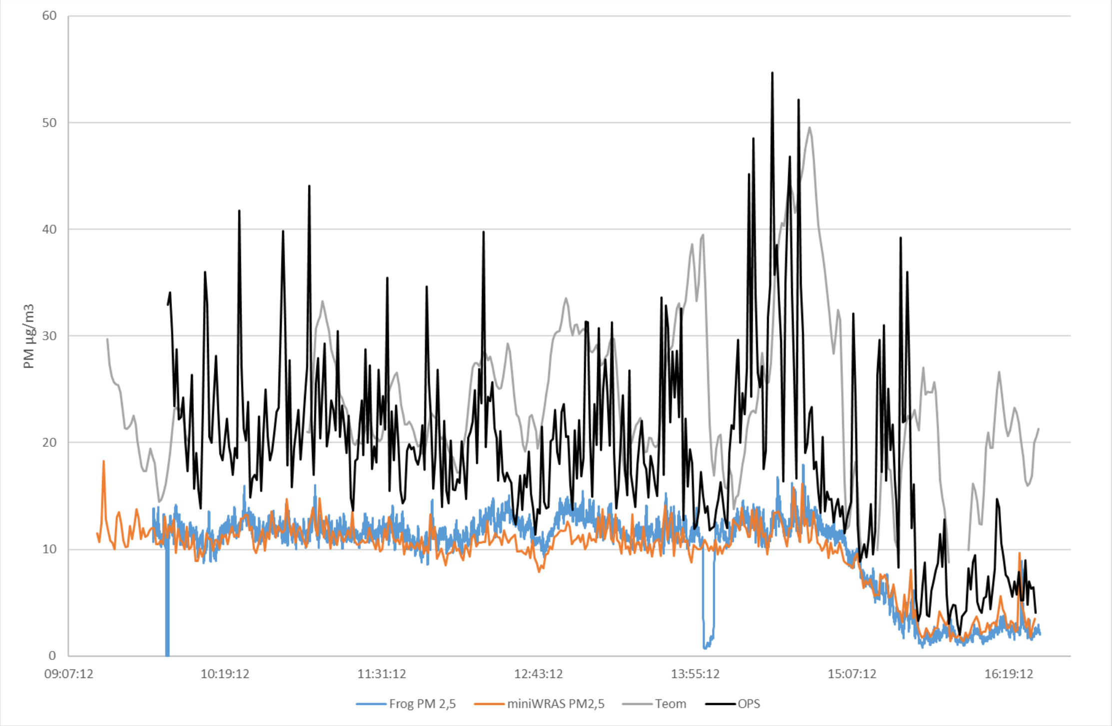


**Figure A3. Comparison $PM_{2.5}$ measured by Mini-WRAS, OPS, aerosol spectrometer Fidas Frog and TEOM at urban station Gårda in May 2018.**

# Evaluating the Results – supplemental information

## The Efficiency-of-Means

The *efficiency-of-means* is calculated by first calculating the respective arithmetic mean of the indoor and outdoor concentration:

$$\overline{PM2.5} = \frac{1}{N} \cdot \sum_{i=1}^{N} PM2.5_i = \frac{PM2.5_1 + PM2.5_2 + \ldots + PM2.5_N}{N} \qquad \text{A1}$$

where:

$N$: number of measurement points in that measurement session

The average system reduction efficiency is then calculated by mean calculating the system reduction efficiency from the mean indoor and outdoor concentrations:

$$\phi = \frac{\overline{PM2.5_{out}} - \overline{PM2.5_{in}}}{\overline{PM2.5_{out}}} \tag{A2}$$

## Mean-of-Instant-Efficiencies

The *mean-of-instant-efficiencies* is instead calculated by first calculating the system reduction efficiency at each measurement point and then calculating the mean system reduction efficiency as arithmetic mean of these immediate efficiencies:

$$\phi = \frac{1}{N} \cdot \sum_{i=0}^{N} \frac{PM2.5_{out_m} - PM2.5_{in_i}}{PM2.5_{out_i}} \tag{A3}$$

## Reasons for Using the Efficiency-of-Means

Generally speaking *the efficiency-of-means* better predicts the average reduction during a longer time period, while the *mean-of-instant-efficiencies* would theoretically better explain the instantaneous reduction during for example shorter time periods of higher concentration. One purpose of measuring PM concentrations is to evaluate the potential health impact of these concentrations. Since the health could be affected both of low average concentration during longer time periods and high concentration during shorter time periods, also the immediate reduction efficiencies are of relevance. Nevertheless, for the purpose of evaluating reduction in a vehicle´s cabin, the *efficiency-of-means* approach has mainly been used. There are mainly four reasons for this simplification:

1. There is a time lag between indoor and outdoor concentration in the cabin, as the air transport between the outside and the car interior is not immediate. To calculate the true short-time efficiencies, dynamic concentration development in the interior forced by the outside concentration and air exchange rate is needed. This systematic error will be smaller using the *efficiency-of-means* approach since the time-periods covered by the means used for calculation of efficiencies are much longer than the air exchange ratio of the car interior.
2. Sometimes there is a small time-lag between the instrument measuring inside and the instrument measuring outside since they started measuring at different times, this uncertainty becomes insignificant when using the mean concentrations.
3. In the measurement campaign, pumped samples has also been taken. These results are calculated as the *efficiency-of-means*, since the particulate matter was first collected and weighted for a certain time-period after that the average reduction was calculated.
4. Several instruments used in the measurement campaigns internally calculate the averages of several measurements points before displaying/printing their measurement. This implies that the *efficiency-of-means* is calculated first anyway.

# Standard Error and the Error Propagation Method

The results for an experiment are shown in the figures with a standard error for average indoor or outdoor concentration during a measurement session. The standard deviation for the average concentration was calculated with the following equation:

$$\sigma = \sqrt{\left(\frac{1}{N-1} \cdot \sum_{i=0}^{N} (PM_i - \overline{PM}\,)^2\right)} \qquad \text{A4}$$

When the *average system reduction efficiency* was evaluated, the difference between the indoor and outdoor concentration was calculated. One way to assess the uncertainty for this reduction was with help of the so-called error propagation method.

$$\sigma_\phi = \frac{\overline{PM_{in}}}{\overline{PM_{out}}} \cdot \sqrt{\left(\frac{\sigma_{in}}{\overline{PM_{in}}}\right)^2 + \left(\frac{\sigma_{out}}{\overline{PM_{out}}}\right)^2 - \frac{\sigma_{in,out}}{2 \cdot \overline{PM_{in}} \cdot \overline{PM_{out}}}} \qquad \text{A5}$$

where:

$\sigma_{in,out}$: is the covariance between the indoor and outdoor measurements.

Since the indoor concentration is correlated to the outdoor concentration the covariance between the two measurements must be taken into account. The covariance was calculated with the following equations:

$$\sigma_{in,out} = \sum_{i=1}^{N} \left(PM10_{in_i} - \overline{PM10_{in_i}}\right) \cdot \left(PM10_{out_i} - \overline{PM10_{out_i}}\right) \qquad \text{A6}$$

One issue with the error propagation implemented to the dataset was that it doesn’t take the time lag into account. The time lag decreases the correlation between the indoor and outdoor data points, leading to higher $\sigma_\phi$ than if this lag would be accounted for. Due to the complexity of the problem described above, we do not correct for this issue.

# Uncertainty of measurements of the pumped samples of PM2.5, PAH and VOC

The uncertainty in the measurements of pumped $PM_{2.5}$, PAH and VOC were assessed for the individual concentrations and for the system reduction efficiency. For the individual concentrations the uncertainty was associated with the laboratory analyses.

The $PM_{2.5}$ collected on the filters were associated with the uncertainty of the minimum mass that could be safely weighed: the limit of detection (LOD) of 10 µg, limit of quantification (LOQ) of 25 µg and measurement uncertainty of 20 µg.

The 32 PAH individual compounds were associated with a measurement uncertainty (standard deviation) of 15%. In the figures showing the reduction efficiency for PAH section 4.4., the error bars on the outdoor PAH concentrations on the x-axis are the standard deviation of 15% of the measured values.

The standard deviations for the individual VOC compounds were expressed as a limit of repeatability *r* from replicate samples (R%-diagram in the laboratory quality control system). The value of $r$ was calculated as the absolute value of the difference of the duplicate samples divided by the average value of the two measurements:

$$r = \frac{\mathrm{x1-x2}}{\mathrm{AVR(x1,x2)}} \qquad \text{A7}$$

where $x_1$ and $x_2$ were the measured concentration of the duplicate sample.

The standard deviation $\sigma(K)$ for the reduction efficiency was primarily calculated for the indoor-to-outdoor (I/O) ratios of the measurements:

$$\sigma(K) = \mathrm{SQRT(r1 + r2)} \qquad \text{A8}$$

The $\sigma(K)$ values were calculated for the individual VOC compounds where r1 = r2. The I/O ratios were transformed to the system reduction efficiencies $\phi = 1 - \mathrm{I/O}$. We transferred the standard deviations calculated for the I/O ratios to the reduction efficiencies.

In the sections 4.4 PAH and 4.5 VOC, the results are presented for some individual compounds but mainly the compounds are grouped according special specifications described in the Methods section. The standard deviations of the sums of concentrations in the individual groups were then calculated as follows:

For the groups of PAH compounds the plain 15% standard deviation was used and the corresponding standard deviation for the reduction efficiencies for the PAH-compound groups was calculated.

For the VOC-groups VVOC, VOC and BTEX, to obtain the standard deviation for concentration of the group, $r$, concentrations of the individual compounds in the group were multiplied by their individual standard deviations and the sum of these products was divided by the sum of concentrations of the individual compounds in the group. This procedure was applied to all individual indoor and outdoor samples. As $r$ varies with absolute VOC-group concentration in the sample, also r1 and r2 for the groups in equation A8 differ for each sample. The average $\sigma$(K) were, however, quite similar: 0.076 ± 0.010, 0.085 ± 0.026 and 0.037 ± 0.004 for the VVOC, VOC and BTEX groups. The sample specific values were applied for the calculation of standard deviations of the system reduction efficiencies for each indoor/outdoor pair and not the average values.

# Results from the measurement campaigns – supplemental information

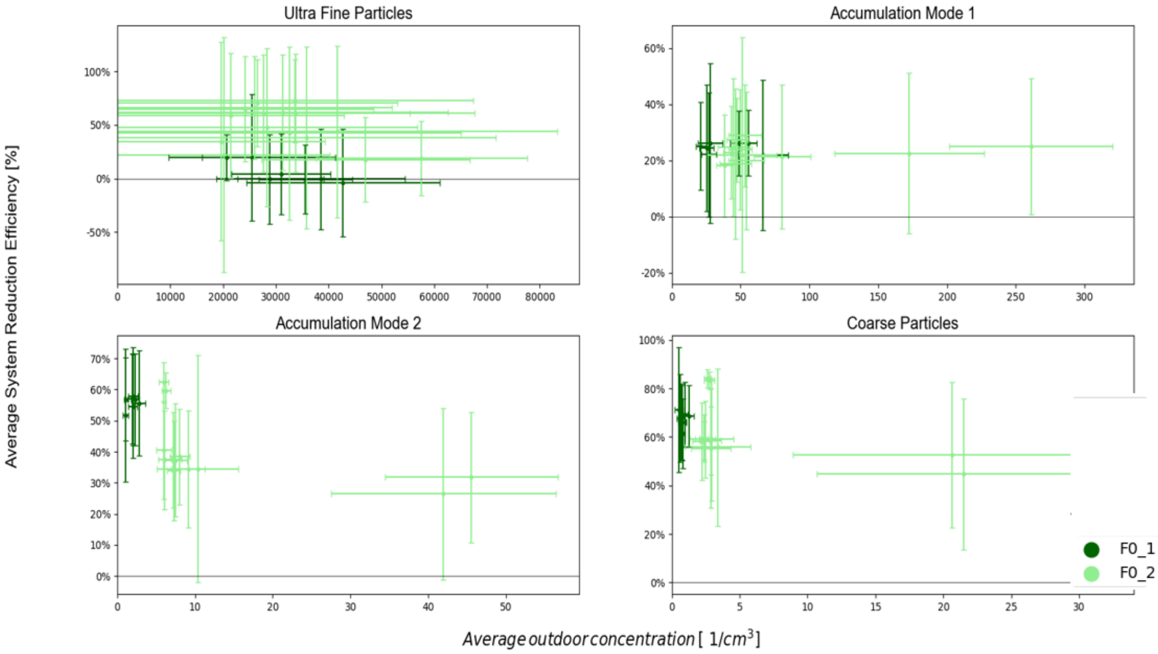


**Figure A4: Mean system reduction efficiencies in the no-filter experiments plotted for particle number concentrations in four different sizes ranges. The experiments are conducted when standing still in a tunnel, at medium ventilation rate. The error bars represent one standard deviation of the outdoor concentration (x-axis) and one standard deviation $\sigma_\phi$ for the mean system reduction efficiencies (y-axes). Each point represents one experiment.**

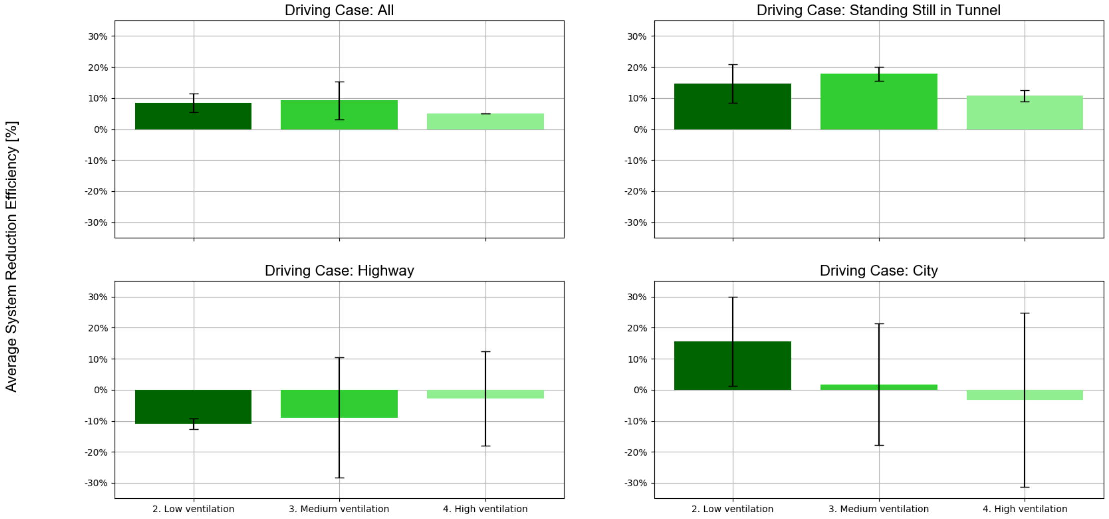


**Figure A5: $PM_{2.5}$ reduction in the no-filter experiments with passenger car (filter F0_1) for different fixed ventilation rates. The error bars represent one standard deviation $\sigma_\phi$ for the mean system reduction efficiencies.**

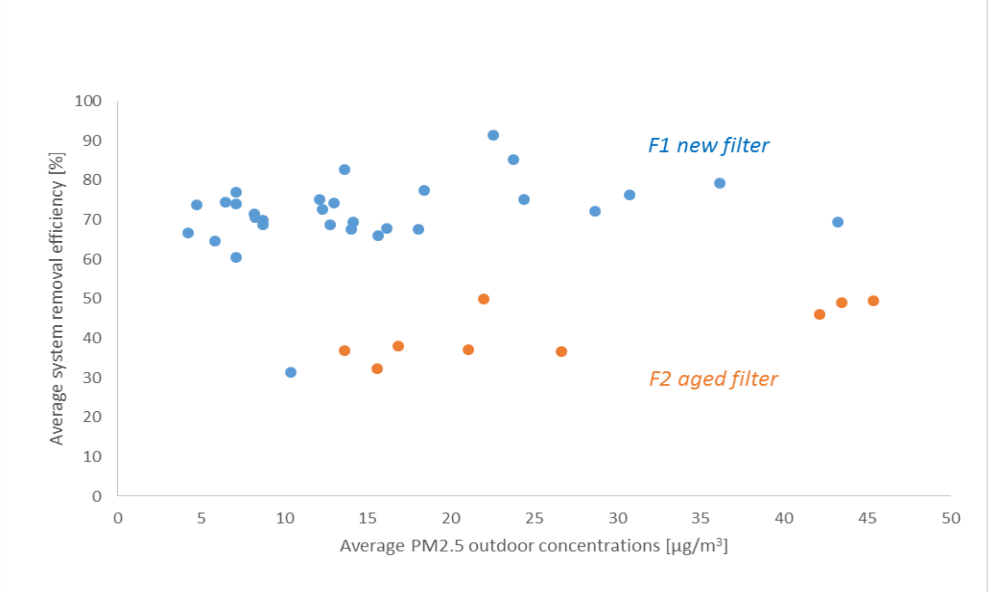


**Figure A6: Overview of the average system reduction efficiency for a new filter (F1) and an aged filter (F2). All types of driving cases (city, highway, tunnel) using the middle ventilation rate are included. Each point represents one experiment.**

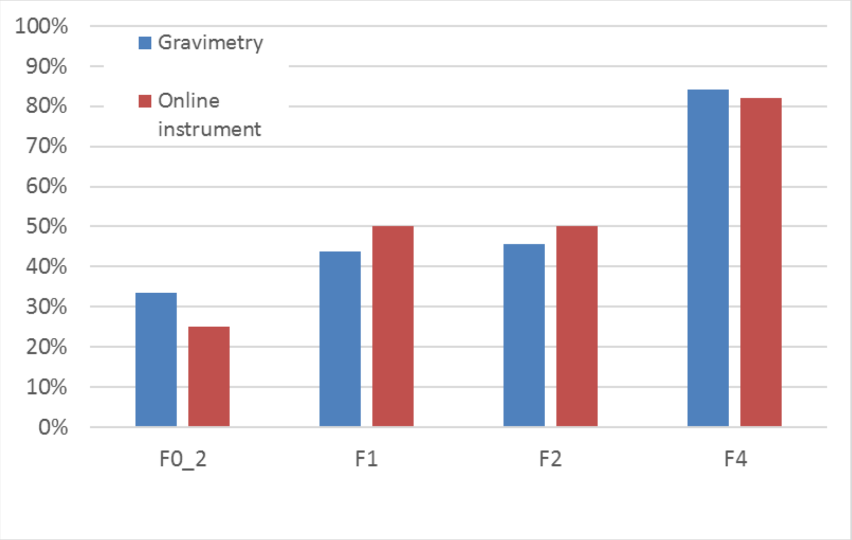


**Figure A7: Average system reduction efficiency for an $PM_{2.5}$ for no-filter experiment and filters F1, F2 and F4, comparison of results from gravimetric analysis of pumped filter samples and of the online instrument data. The number of pumped samples were: F0_2: 2, F1: 1, F2: 2, F4: 2. 2 samples were discarded due to problems during the sampling.**

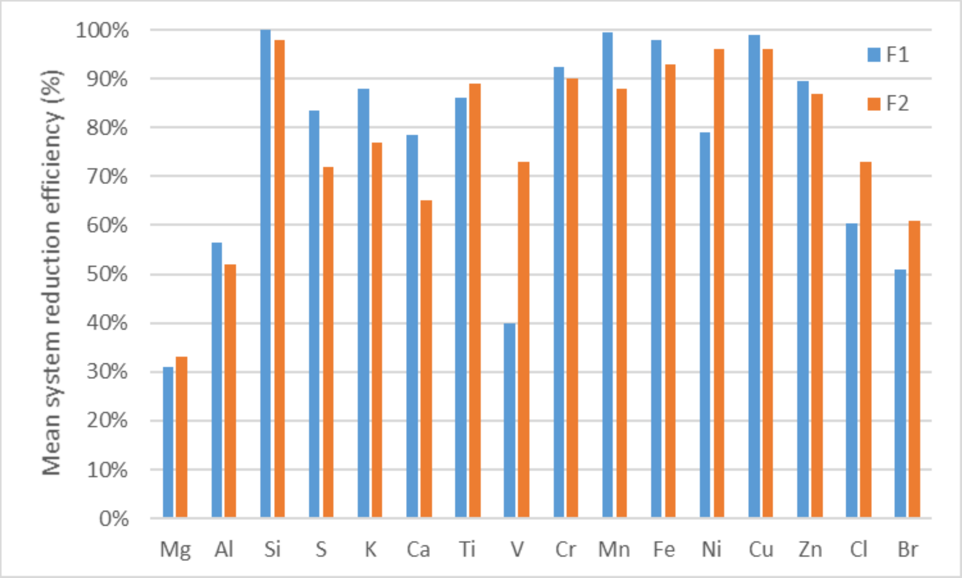


Figure A8: Average system reduction efficiency for elements in $PM_{2.5}$ collected on pumped filter samples for filters F1, F2 and F4.

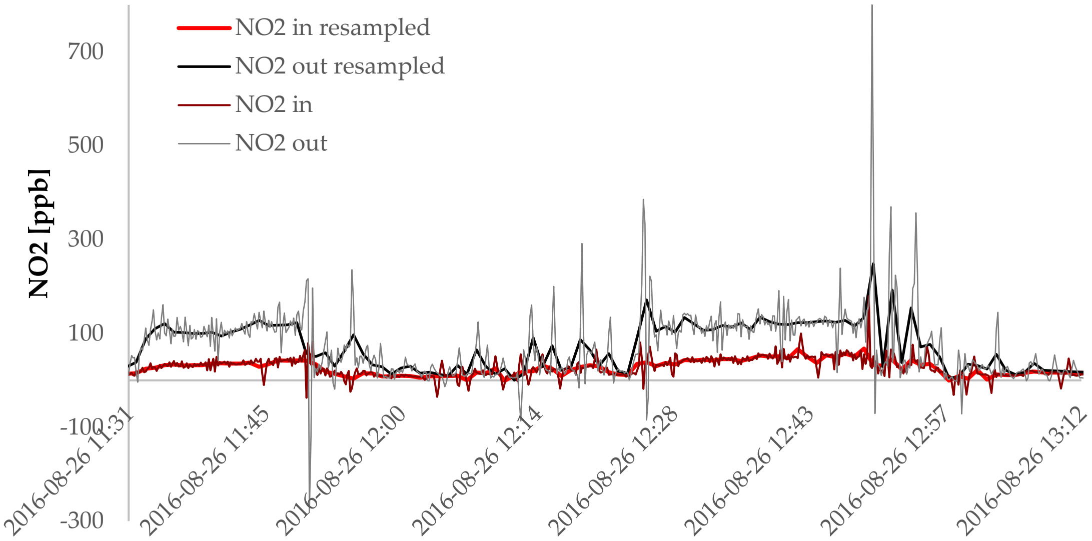


Figure A9: Example of smoothing of the $NO_2$ measurement data from 1s to 1 min means.

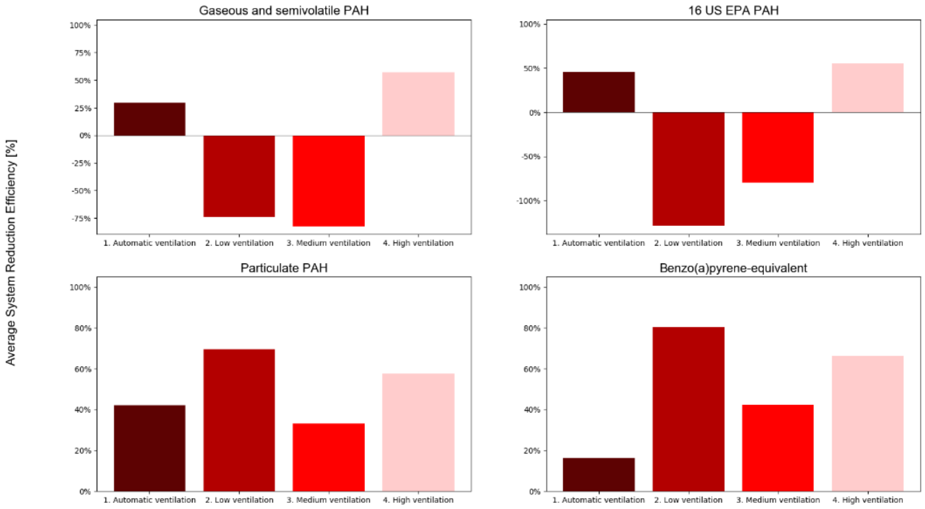


**Figure A10: Summary of reduction efficiencies for gaseous and semi-volatile PAH, 16 US EPA PAH, particulate PAH and B(a)P toxic equivalents as a function of ventilation rate; filter F1.**

# APPENDIX B: Target levels of air pollutants

**Table B1: Target levels of regulated air pollutants.**

| Compound | Recommended guideline indoors | Occupational Health Limit | AAQS* | Averaging period | Reference |
|---|---|---|---|---|---|
| **$CO_2$** | 1 000 ppm | | | actual concentrations | FoHMFS 2014:18 |
| | | 5 000 ppm | | 8 hour working day | AFS 2018:01 |
| | | | no limit value | --- | --- |
| **CO** | 9 ppm | | | 8 hours | WHO 2010 |
| | | 35 ppm | | 8 hour working day | AFS 2018:01 |
| | | | 9 ppm | actual concentrations | AAQS |
| **NO** | No guideline | 3 mg/m$^3$ | no limit value | 8 hour working day | AFS 2018:01 |
| **$NO_2$** | 40 µg/m$^3$ | | 40 µg/m$^3$ | 1 year | WHO 2010; AAQS |
| | | 960 µg/m$^3$ | | 8 hour working day | AFS 2018:01 |
| | | | 60 µg/m$^3$ | 24 hours | AAQS |
| | | | 90 µg/m$^3$ | 1 hour | AAQS |
| **$SO_2$** | 20 µg/m$^3$ | | | 24 hours | WHO 2005 |
| | | 5 mg/m$^3$ | | 8 hour working day | AFS 2018:01 |
| | | | 100 µg/m$^3$ | 24 hours | AAQS |
| | | | 200 µg/m$^3$ | 1 hour | AAQS |
| **$O_3$** | 100 µg/m$^3$ | | | 8 hours | WHO 2005 |
| | | 200 µg/m$^3$ | | 8 hour working day | AFS 2018:01 |
| | | | 120 µg/m$^3$ | 8 hours | AAQS |
| **$PM_{2.5}$** | 25 µg/m$^3$ | | | 24 hours | WHO 2005 |
| | 10 µg/m$^3$ | no limit value | 25 µg/m$^3$ | 1 year | WHO 2005; AAQS |
| **$PM_{10}$** | 50 µg/m$^3$ | | | 24 hours | WHO 2005 |
| | 20 µg/m$^3$ | no limit value | 40 µg/m$^3$ | 1 year | WHO 2005; AAQS |
| | | | 50 µg/m$^3$ | 24 hours | AAQS |
| **TVOC** | 300 µg/m$^3$ | no limit value | no limit value | --- | UBA |
| **Benzene** | 1,7 µg/m$^3$ | | | Health-based, whole life | WHO, 2010 |
| | | 1 500 µg/m$^3$ | | 8 hour working day | AFS 2018:01 |
| | | | 5 µg/m$^3$ | 1 year | AAQS |
| **Benzo(a)pyrene** | 0,12 ng/m$^3$ | | | Health-based, whole life | WHO, 2010 |
| | | 2 µg/m$^3$ | | 8 hour working day | AFS 2018:01 |
| | | | 1,0 ng/m$^3$ | 1 year | AAQS |
| **Formaldehyde** | 100 µg/m$^3$ | | | 30 minutes | WHO 2010 |

| | | | | | |
|---|---|---|---|---|---|
| | | 370 µg/m$^3$ | no limit value | 8 hour working day | AFS 2018:01 |
| **Naphthalene** | 10 µg/m$^3$ | | | 1 year | WHO 2010 |
| | | 50 mg/m$^3$ | no limit value | 8 hour working day | AFS 2018:01 |
| **1,3-butadiene** | 0,2 – 1,0 µg/m$^3$ | | | | AMM recommendation |
| | | 1 mg/m$^3$ | no limit value | 8 hour working day | AFS 2018:01 |

*AAQS Ambient Air Quality Standards – Miljökvalitetsnormer för utomhusluft: https://www.naturvardsverket.se/mknluft

### TableB2: Selected air pollutants with WHO recommended guideline value.

| **Pollutant** | **Guideline value** | **Comment** |
|---|---|---|
| CO | 10 mg/m$^3$ = 9 ppm | Actual concentration |
| $NO_2$ | 40 µg/m$^3$ | Annual average |
| $PM_{10}$ | 20 µg/m$^3$ | Annual average |
| $PM_{2.5}$ | 10 µg/m$^3$ | Annual average |
| Benzene | 1.7 µg/m$^3$ | Excess lifetime risk $10^{-5}$ |
| Benzo(a)pyrene | 0.12 ng/m$^3$ | Excess lifetime risk $10^{-5}$ |
| Formaldehyde | 100 µg/m$^3$ | 30-minute average |

### TableB3: Target levels of the VOC determined in the AQIFOR based on Occupational Exposure Limits. The pollutants with an OEL-value are in bold letters.

| **Compound** | OEL **mg/m$^3$** | Indoor value **µg/m$^3$** |
|---|---|---|
| Isobutane | | |
| n-Butane | | |
| trans-2-Butene | | |
| 1-Butene | | |
| Isobutene | | |
| cis-2-Butene | | |
| **1,3-Butadiene** | 1 | 1 |
| **Isopentane** | 1 800 | 1 800 |
| 1-Pentene | | |
| **n-Pentane** | 1 800 | 1 800 |
| Isoprene | | |
| trans-2-Pentene | | |
| cis-2-Pentene | | |
| **2-Methylpentane** | 700 | 700 |
| **3-Methylpentane** | 700 | 700 |
| **n-Hexane** | 72 | 72 |
| **Benzene** | 1.5 | 1.5 |
| **Cyclohexane** | 700 | 700 |
| **Isooctane** | 900 | 900 |
| **n-Heptane** | 800 | 800 |
| **Toluene** | 192 | 192 |
| **n-Octane** | 900 | 900 |
| **Ethylbenzene** | 220 | 220 |
| **Xylenes (o-, m-, p-)** | 221 | 221 |
| **n-Nonane** | 800 | 800 |
| **Styrene** | 43 | 43 |
| n-Butyl acetate | | |
| **Trimethylbenzenes** | 100 | 100 |
| **Formaldehyde** | 0.37 | 370 |

# APPENDIX C: Database Description

This Appendix describe the structure of the tables and some queries in the AQIFOR-database. The database is a Microsoft Access Database called: *AQIFOR.accdb*. The database can be categorised in four different table types and queries. The queries are used to connect all data and make simple calculations.

1. Measurement data
2. Campaign characteristics
3. Connection tables
4. Result tables
5. Queries

## Data Categorization General

All tables have some type of identification fields to make it possible to categories all data. The identification field are described in Table C1 below.

**Table C1: Identifiers**

| Identifier | Description |
|---|---|
| Campaign | The campaign name is used to link the data to the correct campaign. The campaign names currently in the database is: *winter2015*, *winter2016*, *summer2016*, *summer2017*, *summer2018* and *demo2019*. |
| Experiment name | The experiment name is a key to understand what type of experiment that has been conducted. The form of the experiment name is: *filter code* (FX) + *type of experiment* (OSA, REC or customer) + *ventilation type* (VL, VM, VH) + *serial number* (XX), for example *F1OSAVM01*. The experiment name in combination with the campaign name will create a unique code for each experiment, such as: *winter2015F1OSAVM01*. |
| Driving cycle | Each experiment has been categorized into different driving cycles, with the following coding:<br>1. Recirculation experiment conducted in a tunnel.<br>2. City driving<br>3. Highway driving<br>4. Standing still in a tunnel<br>The experiment name in combination with the campaign name and the code for the driving cycle will create a unique code for each driving case, such as: *winter2015F1OSAVM014*. |

# Measurement Data Tables

These tables contain measurement data from the online instruments, the pumped samples, car parameters and the weather data, a description of each table can be found in Table C2. This table type always starts with the same prefix: *1*. This number therefore implies that these tables contain some type of measurements data.

All indoor and outdoor measurement have been put next to each other in each table. If only one instrument has been measuring, the columns of the other instruments is blank. The indoor- and outdoor measurements are indicated by using the suffix "*_in*" and "*_out*" in the column name.

Apart from the identifiers explained in Table C2, the measurement tables also included the time of the indoor- and the outdoor measurement, respectively (format: *hh:mm:ss*). A third time column is also included containing the average time of the outdoor and the indoor measurement, since there is a small-time difference in some cases.

**Table C2: Measurement data from the campaigns**

| Table number | Table name | Description |
|---|---|---|
| 1_1_1 | Nanoscan | Particle number concentration. Data from two NanoScan instruments, one measuring indoor and one measuring outdoor. There are data from 13 channels, ranging from 10 nm to 421 nm. |
| 1_1_1_1 | Nanoscan_channels | Contains information on each channel's lower/upper bound and midpoint diameter. |
| 1_1_1_2 | Nanoscan_PM_concentration | The calculated particle mass concentration for each Nanoscan channel. The table is based on *1_1_1_2 Nanoscan_PM_concentration.* |
| 1_1_2 | OPS | Particle number concentration. Data from two OPS instruments, one measuring indoor and one measuring outdoor. There are data from 17 channels, ranging from 0.3 nm to 10 µm. |
| 1_1_2_1 | OPS_channels | Contains information on each channel's lower/upper bound and midpoint diameter. |
| 1_1_2_2 | OPS_constants | Flow rate and sample time |
| 1_1_2_3 | OPS_concentration | The calculated particle number concentration for each OPS channel. The table is based on *1_1_2 OPS.* |
| 1_1_2_4 | OPS_PM_concentration | The calculated particle mass concentration for each OPS channel. The table is based on *1_1_2_3 OPS_concentration.* |
| 1_3_1 | WRAS_count | Particle number concentration. Data from two Mini-WRAS instruments, one measuring indoor and one measuring outdoor. There are data |

| | | |
|---|---|---|
| | | from 41 channels, ranging from 10 nm to 35 µm. |
| 1_3_2 | WRAS_PM | Particle mass concentration. Data from two Mini-WRAS instruments, one measuring indoor and one measuring outdoor. $PM_{10}$, $PM_{2.5}$, $PM_1$, Inhalable, Thoracic, Alveolic |
| 1_4 | NOX | NO, $NO_2$ and NOx concentrations |
| 1_5 | Wohler | Temperature, RH and $CO_2$. Data from two Woehler instruments, one measuring indoor and one measuring outdoor. |
| 1_6 | Black Carbon | Light absorbing black carbon. Data from Magee Aethalometer measuring indoor. Only employed in recirculation experiments |
| 1_7_1 | Grimm PM | Particle mass concentration. Data from two GRIMM aerosol spectrometers 1.108, one measuring indoor and one measuring outdoor. $PM_{10}$, $PM_{2.5}$, $PM_1$, Inhalable, Thoracic, Alveolic |
| 1_7_2 | Grimm all sizes | Particle number concentration. Data from two GRIMM aerosol spectrometers 1.108, one measuring indoor and one measuring outdoor. There are data from 15 channels, ranging from 0.35 to 22.5 µm |

# Campaign characteristics

These tables contain description about the campaigns, such as ventilation speed and filter types. These tables always start with the prefix 2. This number therefore implies that these tables contains some type of campaign characteristics.

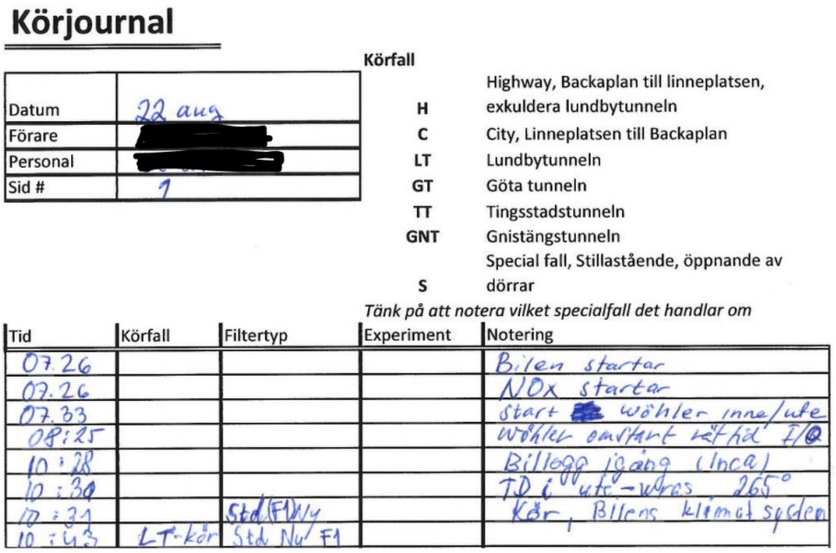
**Table C3: Campaign characteristics**

| Table number | Table name | Description of table |
|---|---|---|
| 2_1 | Digital_logbook | Is further described in the section 0. |
| 2_2 | Experiment_description | The experiment description table is based on the table logbook. Each row in this table represent one unique experiment. This table is derived from the digital logbook. The only columns that are not directly derived from the logbook is the two columns: *Failed_experiment* and *Failed_NOx_experiment*. These two columns are true or false values. A true value in the *Failed_experiment* column implies that the particle measurement results should be excluded, and a true value in the *Failed_NOx_experiment* |

| | | |
|---|---|---|
| | | implies that the NOx results should be excluded. The database includes 247 unique experiment. |
| 2_3 | Filter_description | The table describes the filters used in the different campaigns. Seven different filter types have been tried out in the campaigns and no filter have been used in two different vehicles. |
| 2_4 | Driving_cycle_description | A description of the driving cycles. |
| 2_5 | Driving_Case_ID | Each row in this table represent the unique driving cases for each unique experiment. Each row in this table therefore represent one unique experiment and driving cases. This table is derived from the digital logbook. The database includes 526 unique driving cases. |
| 2_6 | Ventilation_description | Ventilation speeds and coding used at the different campaigns. |

# The Digital Logbook

The digital version of all logbooks is called *2_1 Digital_logbook* in the database. This is perhaps the most important table in the database, since all other tables are connected, constructed or based on this table. A paper version of the driving journal was filled out during each campaign day according to the template in Figure C1. The logbook is then digitized according to the format in Table C4.



**Körjournal**

| | |
|---|---|
| Datum | 22 aug |
| Förare | [redacted] |
| Personal | [redacted] |
| Sid # | 1 |

**Körfall**

| | |
|---|---|
| H | Highway, Backaplan till linneplatsen, exkuldera lundbytunneln |
| C | City, Linneplatsen till Backaplan |
| LT | Lundbytunneln |
| GT | Göta tunneln |
| TT | Tingsstadstunneln |
| GNT | Gnistängstunneln |
| S | Special fall, Stillastående, öppnande av dörrar |

*Tänk på att notera vilket specialfall det handlar om*

| Tid | Körfall | Filtertyp | Experiment | Notering |
|---|---|---|---|---|
| 07.26 | | | | Bilen startar |
| 07.26 | | | | NOx startar |
| 07.33 | | | | start [illegible] wöhler inne/ute |
| 08:25 | | | | wöhler omstart rätt hål I/Q |
| 10:28 | | | | Billogg igång (Inca) |
| 10:30 | | | | TD i utc-wres 265° |
| 10:31 | | Std(F1)Ny | | Kör, Bilens klimat system |
| 10:43 | LT-kör | Std Ny F1 | | |

Figure C1: Example of the driving journal

**Table C4: Metadata description table.**

| Column heading | Description |
|---|---|
| Vechicle_type | Truck, Passenger car, etc. |
| Campaign_name | Campaign code name |
| Date | The date of the campaign day on the format: *YYYY-MM-DD* |
| Time_start | The time at which an experiment, driving case or note starts or is valid for. The time has the format hh:mm. |
| Case_no | Case code name |
| Experiment_name | Experiment code name |
| Pump_VOC | VOC sample field code. The experiment code is often similar to the experiment name. |
| Pump_PAH | PAH sample field code. The experiment code is often similar to the experiment name. |
| Pump_PM | PM sample field code. The experiment code is often similar to the experiment name. |
| Case_no_detailed | A more detailed numbering for the driving cases also including two other tunnels in Gothenburg. This field has only been filled out for some campaigns and is not used in results in this report. |
| Window/door | 1=open 0=closed |
| Note | Comments that were made in the driving journal |

An improvement of the methodology would be to have the logbook in a digital format directly or create a digital logbook based on the GPS-data.

# Result Tables

There are many result tables in the database, five for each instrument, the prefix for the result tables are 3. The suffix of the results tables is based on how the results are categorized, according to the list below:

1. Experiment
2. Case number
3. PM sample code
4. PAH sample code
5. VOC sample code

These tables are not so important since they are pivot-tables of the measurement tables. They are used to create result overview table *8_1 Result Comparison*, which is used to evaluate missing data and experiments for all instruments.

# Connection Tables

The connection tables are the link between the digital logbook and the measurement data tables. These tables are separated from the rest of the tables in the database by the prefix 7. The principle behind the connection table is that the ID-field is the same as in as in the measurement data tables, but all other fields are from the digital logbook. It is created by a query that goes through all rows in the measurement data and then compare it to the digital logbook according to the following equation:

$$Start\_time_{Digital\ logbook} \leq Time_{measurment\ data} < End\_time_{Digital\ logbook}$$

This table is then mainly used to update the columns: *Experiment_name* and the *Case_no* on the measurement data tables.

# Queries

The queries in the AQIFOR-database are used to connect/create/update tables and to make simple calculations. There are many queries in the database, and there numbering is based on the tables they update or create. The queries have four different functions:

1. Connect tables
2. Update tables
3. Create new tables
4. Calculations

The function of the queries that creates the connection tables are described in section 0 of the database. The tables are created by running all queries that starts with 7, such as *7_1 Create Nanoscan_connection*. The measurement data is updated with the information in the digital logbook by running all update-queries that starts 1, for example *1_1 Update Nanoscan.*

The NanoScan and the OPS measurement data are only given as number concentration (#/cm$^3$) and as number of particles (#) per channel, respectively. The mass of the particles from each channel has therefore been calculated in the database. The exported data from the OPS-instrument is given as count of particles per channel, the number concentration is therefore calculated with the following equation:

$$conc_{channel} = \frac{Count_{channel}}{(time_{sample} - time_{deadtime}) \cdot flowrate_{sample}} \ [\#particle/cm^3]$$

The flowrate for the OPS is 16.67 cm$^3$/s and the sampling time is set to 10s. The mass concentration for each channel has been calculated by assuming spherical particles and constant particle density of 1 gram/cm$^3$. With help of these two assumptions the particle mass for each channel on the OPS and the Nanoscan has been calculated according to the following equation:

$$mass_{conc} = density_{particle} \cdot \left(\frac{diameter_{particle}}{1000}\right)^3 \cdot conc_{channel} \cdot \frac{\pi}{6} \ [g/cm^3]$$

The tables containing the particle mass concentration for the OPS and Nanoscan are called 1*_1_1_2 Nanoscan_PM_concentration* and *1_1_2_4 OPS_PM_concentration*.

IVL Swedish Environmental Research Institute Ltd.
P.O. Box 210 60 // S-100 31 Stockholm // Sweden
Phone +46-(0)10-7886500 // www.ivl.se