Particles in road and railroad tunnel air

Sources, properties and abatement measures

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Abstract

High levels of air pollution are a common problem in both road and railroad tunnels. Sources and emission processes however differ significantly, as reflected by aerosols physical and chemical properties. As particle concentrations and properties affect exposure of and health effects for people on platforms and in vehicles, effective ways to reduce emissions and exposure are important. This study aims to improve the knowledge of the differences between PM$_{10}$ in the rail and road tunnel environments, their sources and the possibilities to address problems with high particulate levels. Measurement campaigns were carried out at Arlanda Central, a railroad tunnel station below Arlanda airport and in Söderleden road tunnel, a road tunnel in central Stockholm.

The results show large differences in concentration levels, size distributions and in composition of the particles. The railroad tunnel aerosol consisted of coarse particles with high iron content, while the properties of the coarse particles in the road tunnel were strongly influenced by whether the road surface was wet or dry. In wet conditions, concentrations were relatively low and iron and sulfur dominating elements, while silicon, potassium, calcium and iron from suspension and road wear dominated during dry conditions. The content of elemental carbon, most likely from the pantograph, were unexpectedly high in the railroad tunnel. An older type of train with a large proportion of mechanical brakes were suggested to be responsible to the main particle emissions in the railway tunnel. The report concludes with a discussion and proposals for action against particle sources in the various underground environments.

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Höga halter av luftförroreningar är ett vanligt förekommande problem i tunnlar för såväl väg som järnväg. Källor och emissionsprocesser skiljer sig dock åt väsentligt, vilket avspeglas i aerosolernas fysikaliska och kemiska egenskaper. Då partiklarnas halter och egenskaper är viktiga för exponering och hälsoeffekter för människor på perronger och i fordon, är effektiva sätt att minska emissionerna och exponeringen av vikt. Föreliggande studie syftar till att förbättra kunskapen om skillnaderna mellan inandningsbara partiklar (PM$_{10}$) i väg och järnvägsmiljö, partiklarnas källor och möjligheterna att åtgärda problem med höga partikelhalter. Mätkampanjer genomfördes på Arlanda Central, en järnvägsstation under Arlanda flygplats och i Söderledstunneln, en vägtunnel i centrala Stockholm.

Resultaten visar på stora skillnader i såväl halter och storleksfördelningar som i partiklarnas sammanföstning. I järnvägstunneln utgjordes aerosolen av grova partiklar med högt järninhåll, medan egenskaperna hos de grova partiklarna i vägtunneln påverkades starkt av om vägbanan var våt eller torr. Vid våt vägbana var halterna förhållandevis låga och järn och svavel viktiga element, medan kisel, kalium, kalcium och järn från suspension och vägslitage dominerade vid torr vägbana. Halten av elementärt kol, sannolikt från strömavtagare, var oväsent hög i järnvägstunneln. En äldre tågtyp med stor andel mekaniska bromsar bedömdes orsaka huvuddelen av partikelemissionerna i järnvägstunneln. Rapporten avslutas med en diskussion om och förslag till åtgärder mot partikelkällor i de olika tunnelmiljöerna.

**Titel:** Partiklar i väg- och tågtunnelluft. Källor egenskaper och åtgärdsmöjligheter.

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Preface

This project was initiated as an application to BVFF, an industry programme for research, development and innovations in road and railway construction and maintenance, in 2012. Field campaigns were conducted in first half of 2013. Part of the work was conducted as part of Saeed Abbasi’s Ph.D. studies. The authors would like to thank Håkan Wilhelmsson and Stig Englund for managing the traffic measurement system at Arlanda, Anna Kryhl at the Swedish Transport Authority for supplying railroad traffic data and finally Göran Lidén at ACES, Stockholm University, for valuable comments on the manuscript.

Linköping, in November, 2016

*Mats Gustafsson*

*Project leader*
Quality review

External peer review was performed on 30 July 2016 by Göran Lidén, Department of Environmental Science and Analytical Chemistry (ACES), Stockholm University. Mats Gustafsson has made alterations to the final manuscript of the report. The research director Mikael Johannesson examined and approved the report for publication on 21 November 2016. The conclusions and recommendations expressed are the authors’ and do not necessarily reflect VTI’s opinion as an authority.

Kvalitetsgranskning

Summary

Particles in road and railroad tunnel air. Sources, properties and abatement measures

by Mats Gustafsson (VTI), Saeed Abbasi (KTH), Göran Blomqvist (VTI), Yingying Cha (KTH)
Anders Gudmundsson (Lund University), Sara Janhäll (VTI), Christer Johansson (SLB-analys and Stockholm University), Michael Norman (SLB-analys) and Ulf Olofsson (KTH)

High levels of air pollution are a common problem in both road and railroad tunnels. However, sources and emission processes differ significantly, as reflected by the physical and chemical properties of the two aerosols. As particle concentrations and properties affect exposure of and health effects for people on platforms and in vehicles, effective ways to reduce emissions and exposure are important. This study aims to improve the knowledge of the differences between PM$_{10}$ in the rail and road tunnel environments, their sources and the possibilities to address problems with high particulate levels.

Measurement campaigns were carried out at Arlanda Central, a railroad tunnel station below Arlanda airport and in Söderleden road tunnel, a road tunnel in central Stockholm. Measurements included particle concentrations, size distributions, size resolved element content, NO$_x$, and organic and elemental carbon. Traffic and meteorology were measured and/or collected from existing databases. In Söderleden road tunnel, the campaign (non-intentionally) included both a period that was mainly wet and one that was dry. This gave the opportunity to study the differences in the importance of suspension to the contribution to particle levels.

The results show that the rail tunnel environment was characterized by relatively high concentration peaks of coarse particles and low levels of NO$_x$ and NO$_2$. Some trains were linked to emissions of ultrafine particles. The composition of the airborne particles is dominated by iron, with smaller contributions from copper, zinc and other metals. The road tunnel is characterized by high levels of ultra-fine particles, NO$_x$ and NO$_2$ and, in dry condition, also high levels of coarse particles. As the traffic is more intense than in the rail tunnel, particle levels are more constantly high during busy traffic. In humid conditions the coarse particles were dominated by iron whereas particles below about 1 micron were dominated by sulfur. In dry conditions, increases in the typical mineral elements silicon, potassium, calcium and iron were substantial. Chlorine represents a significant percentage in both wet and dry conditions, which suggests a contribution from road salt. The iron is suggested to originate from brake wear in wet conditions, and from both brake wear and road wear in dry conditions. By comparing the data with train passages and information on train types, it was found that most of the high particulate levels recorded at Arlanda C are correlated to older trains with locomotives of type Rc with their wagons. These mainly have mechanical brakes and are also braked during longer time and distance before stopping at the station. The content of elemental carbon in the air of the railroad environment was unexpectedly high, about half of the content of the road tunnel, despite lack of combustion sources. This is considered to be due to wear of graphite from the train pantograph. The main focus of action against high particle levels in railroad tunnels has been on ways to prevent exposure by separating trains from the platform or to vent contaminated air, while studies on the opportunities to prevent emissions are fewer. This study demonstrates the potential to reduce particulate emissions by identifying the types of trains and train individuals and their characteristics, technical systems that causes particle emissions, maintenance status and also how they are driven. In road tunnels abatement measures against coarse particles are linked to measures that reduce studded tire wear of road surface or reduce the suspension of deposited dust. This can be done by reducing the use of studded tires, improved pavements, effective dust control and cleaning, in addition to reducing traffic and speed. The ultrafine particles present in high concentrations originate from vehicle exhaust and may (except through further regulation of vehicle emissions) only be addressed through traffic measures such as reduced traffic and lower proportion of heavy traffic.
Sammanfattning

Partiklar i väg- och järnvägstunnlar. Källor, egenskaper och åtgärder.

av Mats Gustafsson (VTI), Saeed Abbasi (KTH), Göran Blomqvist (VTI), Yingying Cha (KTH), Anders Gudmundsson (Lunds universitet), Sara Janhäll (VTI), Christer Johansson (SLB-analys och Stockholms universitet), Michael Norman (SLB-analys) och Ulf Olofsson (KTH)


Mätningar genomfördes på Arlanda Central, en järnvägsstation under Arlanda flygplats och i Söderledstunneln, en vägtunnel i centrala Stockholm. Mätningar gjordes av partikelhalter, storleksfördelningar, storleksuppdelat elementinnehåll, NOx och organiskt och elementärt kol. Trafik och meteorologi mättes och/eller inhämtades från befintliga databaser. I Söderledstunneln inföll mätningarna under en period som var i huvudsak fuktig och en som var i huvudsak torr, vilket gav möjlighet att studera betydelsen av bidraget från suspension till partikelhalterna.

Resultaten visar att järnvägstunnelmiljön präglas av relativt höga koncentrationstoppar av grova partiklar och låga halter NOx och NO2. Vissa tåg kan kopplas till emissioner av ultrafina partiklar. Partiklarna domineras innehållsmässigt av järn, med mindre bidrag av koppar, zink och andra metaller. Vägtunneln präglas av höga halter ultrafina partiklar, NOx och NO2 och, under torra förhållanden, även höga halter grova partiklar. Då trafiken är mer intensiv än i järnvägstunneln är halterna mer konstant höga under trafikerad tid. Under fuktiga förhållanden domineras de grova partiklarna av järn, medan partiklar under cirka 1 μm domineras av svavel. Under torra förhållanden ökar de minaraltypiska elementen kisel, kalium, kalcium och järn kraftigt. Klor utgör en betydande andel under både fuktiga och torra förhållanden vilken tyder på ett bidrag från vägsalt. Järnet bedöms harröra från bromsar under fuktiga förhållanden och från både bromsar och vägslitage under torra förhållanden.

Genom att jämföra data med tågpassager och tågtyper, konstaterades att huvuddelen av de höga partikelhalter som registreras på Arlanda C kan kopplas till äldre tåg av typen RC. Dessa är främst mekaniskt bromsade och bromsas även under längre tid och sträcker innan de stannar vid stationen. Halten elementärt kol i luften i järnvägsmiljön var oväntat hög, ungefär hälften av halten i vägtunneln, kvot avsaknad av förbränningskällor. Detta bedöms bero på slitage av grafit från tågens strömvagnagare. Huvudsakligt fokus på åtgärder mot höga partikelhalterna i järnvägstunneln har legat på sätt att förhindra exponering genom att skilja av tågen från Perrongen eller att ventilerar ut förorenad luft, medan studier kring åtgärder att förhindra själva emissionen är få. Föreliggande studie visar på potential att identifiera tågtyper och enskilda tåg och deras egenskaper, tekniska system som medför partikelemissioner, underhållsstatus och övrigt hur de framförs. I vägtunnelar är åtgärdsmöjligheterna mot grova partiklar kopplade till åtgärder som minskar dubbdäcksutslaget av vägtytan eller minskar suspensionen av bildat damm. Detta kan ske genom minskad dubbdäckanständighet, förbättrade beläggningar, effektiv dammbindning och städning, förutom genom trafikåtgärder som minskad trafik och hastighet. De ultrafina partiklar (<100 nm) som förekommer i höga koncentrationer harröra från fordonsavgaser och kan (förutom genom fortsatta regleringar av utsläpp från fordon) endast åtgärdas genom trafikåtgärder som minskad trafikmängd och lägre andel tung trafik.
1. Background

Air pollution in road tunnels is a well-known problem. Exhaust gases and combustion related particles reach far higher concentrations than in street or road environments. A very large number of studies have analysed the air quality in road traffic tunnels, e.g. in Stockholm (Kristensson et al., 1999), Gothenburg (Sternbeck et al., 2002), Switzerland (Weingartner et al., 1997), California (Kirschstetter et al., 1999), Austria and the UK (Imhof et al., 2006). Typically, both the morphological, chemical and physical properties of the aerosol differ substantially between rail and traffic road tunnels. Incomplete combustion of vehicle fuels lead to high concentrations of ultrafine particles in road tunnels, which is normally not present in rail tunnels. Super-micron particles generated during mechanical wear processes are present in both environments, but with very different chemical composition due to the different processes and materials involved in the generation of these particles. Also, the gaseous air pollutant mix differ substantially. While NO and NO₂ are normally high in road tunnels during traffic hours, these gases are at background level in railroad tunnels, except when trafficked by maintenance diesel powered vehicles (Han et al., 2015).

Even though not trafficked by combustion vehicles, high levels of particulate air pollution have been recognized in several subway stations, e.g. Stockholm, London, New York city, Tokyo, Helsinki, Mexico City, Taipei, Prag, Budapest, Seoul and Rome (Aarnio et al., 2005; Birenzvige et al., 2003; Johansson and Johansson, 2003; Ripanucci et al., 2006; Seaton et al., 2005. Nieuwenhuijsen et al. 2007, Kim et al. 2008, Raut et al.,2009; more references are given in Abbasi et al. (2013) and Järvholm et al., 2013). The particle mass concentrations (PM₁₀ and PM₂.₅) are generally several times higher than in the most polluted street environments and range from tens to several hundreds of micrograms per cubic meter, depending on traffic amounts and ventilation situation.

Norman and Johansson (2005) and later Midander et al. (2012) established that the number concentration of particles was about five to ten times lower in the subway than in a densely-trafficked street environment (Hornsgatan, Stockholm). Tokarek et al. (2002) concluded that 99% of the number of particles were smaller than 2.5 µm and that 50–60% were smaller than 0.35 µm. This is not contradicting the fact that the mass concentration, as mentioned above, is higher in railroad tunnel than street environments due to the mass’ cubical dependency of particle radius. The influence of vehicle exhaust emissions on air quality in the subway will of course vary depending on the ventilation etc.

Most of the PM₁₀ from emissions in railroad environments consist of different oxidized forms of iron. In a Stockholm subway station, around 60% of the PM₁₀ was found to be iron or iron oxides, both magnetite (Fe₃O₄) and hematite (Fe₂O₃) (Johansson, 2005). Many other trace metals and metalloids have been found in elevated concentrations in subway environments, e.g. chromium, nickel, arsenic, calcium, barium, copper, antimony and aluminium (e.g. Querol et al., 2012). Also, organic material has been identified.

Based on a source receptor analysis of chemically speciated filter samples from Stockholm subway and measured composition of rails, brakes etc., Johansson (2005) found that the largest contribution (70%) to PM₁₀ was due to wear of wheels and/or rail. The rest was mainly brake wear material.

Due to the high concentrations of PM₁₀ in subway environments, there is a concern that high concentrations might be found also in regular railroad environments, where many travellers and train and rail personnel are exposed.

There are much fewer particle measurements in ground level railroad environments compared to subways (Järvholm et al., 2013). Gehrig et al. (2007) measured the PM₁₀ concentration close to the tracks of the railroad station in Zurich, Switzerland. They found a small but significant contribution to long-term average PM₁₀ concentrations (about 1 µg m⁻³) mainly consisting of iron, but also with contributions from copper, manganese and chromium from the railroad. Bukowiecki et al. (2007) showed that the iron was mainly (72%) in the coarse mode PM₁₀-2.₅. Gustafsson et al. (2012) measured
PM$_{10}$ concentrations on platforms of 4 ground level stations in Sweden and concluded that none of the environments risked to exceed the EU limit values.

Concentrations and characteristics of inhalable particles from different sources are of great interest since these particles have been shown to cause health effects in the population. The health effects of particles generated in railroad environments are not well known, but some studies have been performed. A comprehensive review has recently been made by Järvholm et al. (2013).

Järvholm et al. (2013) compared the health risks of exposures in subway and railroad tunnels with risks due to exposures in road traffic tunnels and discussed possible metrics to be used to regulate air quality in railroad tunnels. Based on the different air pollution mixtures in the different tunnel environments as well as health studies (including toxicological studies, experimental studies on humans and animal studies), they conclude that health risks are likely larger for exposures in road traffic compared to railroad tunnels, but also that rather few health studies are available.

Some results concerning railroad particles reviewed by Järvholm et al. (2013) and by Gustafsson (2009) are referred in the following. Chillrud et al. (2005) have compiled the health effects of iron, manganese and chrome, which are among the most enriched metals in subway environments. Iron is suspected to have negative health effects because of its capacity to form free radicals, which has been connected to diseases such as Parkinsonism, Alzheimer’s and multiple sclerosis. Manganese poisoning is known to cause Parkinsonism, while chromium is a well-known carcinogen. There is no epidemiological evidence, however, that relatively low exposure to these metals in subway air is related to these diseases.

A cell study using PM$_{10}$ particles from a Stockholm subway station showed that the iron rich particles were 8 times more genotoxic and 4 times more likely to cause oxidative stress in lung cells than PM$_{10}$ from an urban street environment (Karlsson et al., 2006; Karlsson et al., 2005). The higher genotoxicity is most likely caused by highly reactive surfaces giving rise to oxidative stress (Karlsson et al., 2008). On the other hand, a simultaneous cell study focusing on inflammation potential concluded that PM$_{10}$ particles from the subway were less inflammatory for human macrophages than city street PM$_{10}$ (Lindbom et al., 2006, Gustafsson et al., 2008).

In 2008 an epidemiological study concluded that lung cancer incidence was not increased among subway drivers in Stockholm, Sweden, claiming this result gives some evidence against the hypothesis that subway particles would be more potent in inducing cancer than other particles in ambient air (Gustavsson et al., 2008).

Klepczynska Nyström et al. (2010) exposed healthy volunteers to a subway environment and a control environment (office) and found no significant differences in lung function or in inflammatory response. Significant effects were however found in blood where fibrogene and regulatory T-cells increased. This is the very first published controlled human exposure study of subway particles.

For road tunnel environments, there are some human experimental studies made in Stockholm, but otherwise it seems that there are not as many studies as for subway tunnels (see Järvholm et al., 2013). Svartengren et al (2000) showed that exposure to air pollution in road tunnels may significantly enhance asthmatic reactions to subsequently inhaled allergens. Larsson et al. (2007) showed that a 2-hour exposure of healthy subjects to traffic pollution in the same road tunnel resulted in increased inflammatory response in lower airways, but no increase in blood coagulation factors, cellular infiltration or effects on lung function were seen.

The aim of this study was to measure, characterize and investigate the differences in particle properties in a road and a railroad tunnel and to identify their main sources to be able to suggest relevant and effective measures to reduce high particle concentrations.
2. Methodology

2.1. Tunnel sites

2.1.1. Arlanda C – railroad tunnel

The subterranean station Arlanda Central (C) is situated north of Stockholm below Arlanda airport. The platform is 400 m long with one track on either side with traffic in opposite directions. The tunnel is approximately 5 km long and the station is trafficked by mixed long distance, regional and commuter trains passing the airport. The tunnels are self-ventilated. The only active ventilation is smoke evacuation fans only activated if fire occurs.

![Arlanda tunnel and its entrances (upper) and Arlanda C platform with placement of the measurements.](image)

*Figure 1. Arlanda tunnel and its entrances (upper) and Arlanda C platform with placement of the measurements.*

The traffic in the tunnel is a mix of electrical trains, from long distance to commuter trains of different age and construction (Figure 2).
Apart from these electric trains, different types of maintenance vehicles traffic the tunnel, mainly in night time, when there is no regular train operations ongoing. These vehicles can be either electrically or diesel driven.

The train traffic at Arlanda C was measured by a photocell equipment designed at VTI registering each train arrival approximately 10 meters from the Arlanda C train station platform at each track (track 1= northbound, track 2 = southbound). The two rays were placed perpendicular to the rail, hence the speed could be calculated using equation 1 and 2:

\[
\text{Front Speed} = \frac{2}{T_{\text{ray2dis}} - T_{\text{ray1dis}}} \quad \text{(equation 1)}
\]

\[
\text{Aft Speed} = \frac{2}{T_{\text{ray2res}} - T_{\text{ray1res}}} \quad \text{(equation 2)}
\]

where \( T_{\text{ray1dis}} \) is the timing at when the first ray is disrupted, \( T_{\text{ray2dis}} \) is the timing at when the second ray is disrupted, \( T_{\text{ray1res}} \) is the timing at when the first ray is resumed and \( T_{\text{ray2res}} \) is the timing at when the second ray is resumed.

Since both the train front speed (by disrupting the rays) and train aft speed (by resuming the rays) could be calculated, the train length (equation 3) and speed retardation (equation 6) could be calculated using equation 4 and 5 for calculating the mean timing of the front and aft speed, i.e. when the train front and aft are exactly in the mid-point between ray 1 and ray 2.

---

**Figure 2.** Train types trafficking Arlanda C. All photos from järnväg.net.
Train length = \( \frac{\text{Front speed} + \text{Aft speed}}{2} \times (T_{\text{aft speed}} - T_{\text{front speed}}) \)  
(equation 3)

\[ T_{\text{front speed}} = \left( T_{\text{ray1dis}} + \frac{(T_{\text{ray2dis}} - T_{\text{ray1dis}})}{2} \right) \]  
(equation 4)

\[ T_{\text{aft speed}} = \left( T_{\text{ray1res}} + \frac{(T_{\text{ray2res}} - T_{\text{ray1res}})}{2} \right) \]  
(equation 5)

Retardation = \( \frac{\text{Aft speed} - \text{Front speed}}{T_{\text{aft speed}} - T_{\text{front speed}}} \)  
(equation 6)

The photocells (brand: IR transmitter IFM Efector 200 OA5101, IR receiver IFM Efector 200 OA5102) were fixed two meter apart on aluminium rods held by two tripods at approximately 130 cm above the rails (Figure 3). The timing of ray disruption and resuming was logged by a TA89-logging equipment (VTI notat T147).

During the measurement period 2013-01-28–2013-02-11, a total of 885 northbound and 959 southbound train arrivals were registered.

Train definitions used in the data analyses are: Length 20–400 m, acceleration/retardation between 2 and -2 m s\(^{-2}\), front speed above 2 m/s.

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*Figure 3. Train traffic counting equipment in the tunnel. Measurement rays across the rail is indicated by red lines in the photo.*
2.1.2. Söderleden road tunnel

Söderleden road tunnel in central Stockholm is 1.5 km long and is unidirectional in two separated bores, each bore with traffic distributed over two lanes. In this study measurements were made in the southbound tunnel bore, ca 1060 meters from the north entrance. The tunnel bore has a slight downward slope from the entrance to the sampling point. The tunnel is ventilated through two ventilation towers (see Figure 5).

![Figure 4. Söderleden road tunnel. Arrows indicate main tunnel entrances.](image)

![Figure 5. Ventilation situation in Söderleden road tunnel. The idea with the ventilation slot is that northbound air should return in the southbound tunnel tube and be ventilated in the tower at Skansbrogatan. This is to avoid polluted air from exiting through the northern tunnel opening.](image)

Unfortunately, there are no traffic counting inside the tunnel, but traffic is recorded by the congestion charge portal on a bridge (Johanneshovsbron), which is directly connected to the tunnel.
2.2. Measurements

Measurements were performed between 2013-02-12 and 2013-03-06. An overview of instruments used and parameters measured is given in Table 1, followed by more detailed descriptions in sub-chapters.

*Table 1. Instruments used during the campaign.*

<table>
<thead>
<tr>
<th>Instrument</th>
<th>Parameter</th>
<th>Arlanda C railroad tunnel</th>
<th>Söderleden road tunnel</th>
<th>Mobile measurements on X60 Commuter train</th>
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<td>R&amp;P TEOM</td>
<td>PM10</td>
<td>X</td>
<td>X</td>
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<td>AC31M Thermo Electronics</td>
<td>NO, NO₂, NOₓ</td>
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<td>Particle size distribution (optical)</td>
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<tr>
<td>TSI SMPS 3080</td>
<td>Particle size distribution (mobility)</td>
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<tr>
<td>TSI SMPS Nanoparticle Sizer 3910</td>
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<td>ELPI+</td>
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<td>Leckel SEQ 47/50</td>
<td>Sampling on filters for analyses of EC/OC</td>
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2.2.1. PM$_{10}$

Monitoring of PM$_{10}$ was performed using tapered element oscillating microbalance (TEOM) instruments (Thermo Fischer Inc., USA, model 1400a). The inlet at Arlanda were placed 2 m above the platform. Due to installation difficulties, a vertical inlet could not be used in the road tunnel. There were probably some losses in the inlet tubing to the TEOM, but these were not estimated further. The PM$_{10}$ levels in the road tunnel are therefore most likely underestimated. The inlet in Söderleden road tunnel was placed 2.5 m above the road. The TEOM was logged with one minute time resolution.

2.2.2. Size distribution measurements and particle sampling

Particle size distributions were measured using TSI Aerodynamic Particle Sizer (TSI APS 3321) and Scanning Mobility Particle Sizer (TSI SMPS 3080). These instruments measure, in tandem, the particle number distribution in the size range 7 nm – 18 µm. SMPS measures over the range 14.6-661.2 nm, and the measurement results are presented as particle number distribution, while the coarser particles are measured with APS over the range 0.523-14.6 µm and are presented as mass distribution. The reason for this is that submicron particles are best represented by number since they have very low mass in relation to the coarse particle fraction. The time resolution for APS was 20 s, while the SMPS was set to 90 s. In the conversion from number to mass, a particle density of 5,000 kg m$^{-3}$ is used for particles > 0.5 µm in the railroad tunnel (density of iron oxide) and for smaller particles a particle density of 1,000 kg m$^{-3}$. In the road tunnel the APS used a density for 2,800 kg/m$^{-3}$, which is a representative density for rock. For APS the Stokes correction was used which corrects for APS's overestimate of the particle size when the particle density is much higher than 1,000 kg m$^{-3}$. Total particle number were measured using a TSI CPC 3022. It covers the size range from 7 nm and upwards. It was measured with 1-minutes time resolution.

Complementing, shorter measurements of size distributions and size fractionated sampling of particles were made using Dekati ELPI+ and TSI fast mobility particle sizer spectrometer (FMPS 3091). The ELPI+ is an electrical low pressure impactor Dekati model ELPI+ referred to hereinafter as ELPI. This device composed to 15 stages and measured real time PM$_{10}$, PM$_{2.5}$ and particle size distribution (PSD) of airborne particles in 14 channels from 6 nm to 10 µm in aerodynamic diameter. The data acquisition can be set to either 1 or 10 Hz according to the operator's desire. Also, it enables collection of samples on 25-mm-diameter polycarbonate filters in 14 size fractions to do further chemical investigation. The instrument was set to a sampling frequency of 10 Hz and a particle density of 1000 kg m$^{-3}$ based on results from laboratory studies on airborne brake wear particles (Ulf Olofsson, oral communication, 2016). In order to eliminate any bouncing effects, all filters were coated by DS-515 grease spray before mounting in ELPI stages. During each coating process, we kept one coated filter as laboratory blank filter to compare the blank filter with the field filters. The sampling airflow was 10 lpm in ELPI.

The (FMPS), TSI model 3091 is referred to hereinafter as FMPS. The FMPS measured PSD of submicron particles from 5.6 nm to 560 nm in diameter in 32 channels every second. Its sampling flow rate was 10 lpm and the particle density was set to 1000 kg m$^{-3}$. 
2.2.3. NO\textsubscript{x}

NO\textsubscript{x} including both NO and NO\textsubscript{2}, were measured using the chemiluminescence analyzer AC31M from Thermo Electronics. The instrument measures NO and NO\textsubscript{x} while the NO\textsubscript{2} concentrations are calculated automatically from the measured values. The inlet was placed 2 m above the platform at Arlanda and 2.5 m above the road in Söderleden road tunnel. The NO\textsubscript{x} values were logged with a 1-minute time resolution.

2.2.4. EC/OC

Aerosol samples are collected on quartz-fibre filters (Munktell T 293, \(\varnothing\) 47 mm) by a Leckel SEQ 47/50 sampler with a standard EU PM10-inlet. The filters are pre-fired at 800 °C for 10 hours before sampling to drive of all organic compounds.

Organic (OC) and elemental carbon (EC) is analysed by the Thermo-optical Transmission method (TOT) developed by Birch and Cary (1996). The TOT method employs stepwise heating of a filter sample, first in a non-oxidizing helium atmosphere where carbon is volatilized or decomposed. An optical correction for any pyrolytic carbon (PC) formed during the analysis is done by continuously monitoring the transmittance through the sample with a laser beam (670 nm). The initial transmittance decreases when any PC is created in the sample as the sample becomes darker during heating. The non-oxidizing helium atmosphere is then replaced with an oxygen-containing atmosphere, the sample will start to combust and at some time the level of transmittance will be back to the initial level. This is termed the OCEC split point and all carbon measured before this is assigned as organic carbon (OC) and the carbon measured after this is assigned as elemental carbon (EC).
For the analysis, the Sunset Laboratory Lab OC-EC Aerosol Analyzer is used. The stepwise heating is made according to the EUSAAR 2 protocol (Cavalli et al., 2010).

We have noticed that the filters are coloured (not white as they should) after the final oxidation of all EC. This has been noted also for samples from the subway in Stockholm and is likely due to iron oxides (Midlander et al., 2012). According to Sunset Laboratories this should not usually cause problems, as long as there are not heavy deposits because the absorbance of the red laser light is usually small compared to the absorbance of any EC present and because the absorbance almost always remains constant during the analysis. We have not made any correction for this.

2.2.5. Element analysis

Particle samples collected using the ELPI+ were analysed using PIXE (particle induced x-ray emission), a method of very high sensitivity, for trace element analysis (Johansson et al, 1995). Using PIXE, it is possible to determine up to 35 elements simultaneously in quantities of around one nanogram or lower for elements with the atomic number (Z) higher than 12. Identification of the detected peaks in the x-ray spectrum is performed automatically by the identification and adaptation program GUPIX. A description of the analytical set-up and calibration of this is found in Shariff et al (2002).

At Arlanda C the analysed sample is from 2013-01-29 between 09:14 – 16:19. From Söderleden road tunnel the samples are from 2013-02-16 16:30 to 2013-02-17 16:20 (wet period) and 2013-03-06 14:40 to 2013-03-07 14:20 (dry period).

2.2.6. Data from and measurements made on X60 commuter train

An X60 train (train 6076 door 34) used as a commuter train between Uppsala C and Älvsjö C, passing Arlanda C, was instrumented with particle counters inside the train cabin and also under the train, near the bogie.

Figure 7. Route and time schedule of the measured commuter train, in Stockholm.
Two portable optical particle counters and one portable SMPS were used. The portable SMPS instrument was a SMPS Nanoparticle Sizer 3910 measuring particles from 10 nm to 420 nm with 1 minute time resolution. The optical counters used were two TSI Optical Particle Sizers 3330 with a size range of 0.3 – 10 μm in 16 channels, with 1 second time resolution. Copper tubes were used as pipes to pump the air to the particle counters in the train cabin. Particles were assumed to be spherical with a density of 1000 kg/m³. The Nanoscan and one optical counter were used to sample particles below the train and the other optical particle counter was used to sample the air inside the train cabin, see Figure 8.

The same technique has previous been used for on-board studies of cars and trains in Wahlström et al., (2010) and Abbasi et al. (2012). Tests were done during two days passing Arlanda C in both directions.

*Figure 8. Two sampling points, outside sampling point under the train between two bogies (left a and b), inside sampling point above the seat (right c), attached to particle counters in train cabin.*
3. Results

3.1. Traffic characteristics

3.1.1. Arlanda C

The mean number of trains registered per hour during the measurements is shown in Figure 9. Traffic peaks are found in the morning and in late afternoon and a minimum during late night hours.

![Figure 9. Mean number of trains per hour during the measurements at Arlanda C.](image)

Using the national system LUPP for monitoring punctuality and disturbances train traffic in co-operation with Trafikverket (Anna Kryhl) it was possible to identify each train registered by the traffic measurements system. In Figure 10 the speed histograms for the registered train fronts and rears are shown. The northbound traffic has a slightly higher speed and a faster retardation than the southbound traffic when passing the traffic detection system. The reasons to this might be manifold. E.g., the configuration of the railroad approaching the platform might differ and affect the driving in the two directions. The placing of the traffic detection system in relation to the normal braking procedure might also cause an apparent difference.

In Figure 11, the calculated train lengths are shown. Obvious groupings can be seen that are similar in both southbound and northbound direction. Most trains are between 80 and 240 m long. From the frequency and lengths, it can be concluded that the high peak at around 105 m are single set commuter trains and the peak at 210 m double set commuter trains.
Figure 10. Speed histograms showing the front and rear speed of trains in south- and northbound directions.

Figure 11. Distribution of train lengths in south- and northbound directions.
3.1.2. Söderleden road tunnel

Figure 12 shows, as an example, the total number of vehicles recorded by the congestion charge portal divided into northbound and southbound traffic for the first week of the measurements. Between this portal and the measurement point inside the tunnel there is only one exit ca 420 meters from the main exit of vehicles from the road tunnel. How many vehicles that take this exit is not known, but likely 10% or less of the number passing the portal on the Johanneshovsbron. The northbound traffic peaks during the morning rush hour while the southbound peaks during the afternoon. During the weekend, no northbound morning rush hour peak is noticed, but traffic peaks at about noon. The southbound traffic peaks at about the same time as during Monday – Friday, but with lower traffic amounts.

![Figure 12. Total traffic north and south passing the congestion portals at Johanneshovsbron during part of the measuring period (15 minutes time resolution).](image)

Total daily mean traffic during the measurement period was around 53,000 vehicles per day during weekdays. However, it was also a variation with the traffic increasing during the week from Monday to Friday as seen in Figure 13. The total traffic northbound during weekday was about 27,500 vehicles per day compared to 25,500 southbound as presented in Figure 13. The average total traffic during weekends were 40,000 vehicles per day and the northbound traffic peaks earlier in the day than the southbound. The difference between the directions were smaller (around 1,000) during weekends compared to weekdays.
Different vehicle classes were measured during weekday daytime. The average diurnal variation for the traffic southbound is presented in Figure 14. Between 06:00 and 19:00 passed 76.2% of the total daily traffic. For the daytime traffic, 71.9% of the traffic was personal cars, 23.2% was light duty trucks, 3.7% was heavy duty trucks and 1.3% was busses. Both personal cars and light duty vehicles show a peak in the afternoon while the heavy-duty trucks show a higher frequency during daytime and buses have maxima in the morning and late afternoon.
Figure 14. Mean daytime diurnal variation in number of vehicles for different classes when passing the congestion portals at Johanneshovsbron. Almost the same number of vehicles pass by the measurement station inside the tunnel.

Unfortunately, there are no traffic speed measurements available in the tunnel but the signed speed was 70 km/h.

The number of vehicles using studded tyres are counted manually at different streets and roads in Stockholm. No counting was made in the road tunnel but on Nynäsvägen which is the same road as Söderleden road tunnel, some 2 km south. Around 300 cars were counted on the 22 February and the studded tyres share on personal cars were 53% (Brydolf et al., 2013).

3.2. Particle and NO\textsubscript{x} concentrations (TEOM, NO\textsubscript{x}, CPC)

3.2.1. Temporal patterns in the Arlanda C railroad tunnel and relation to traffic and meteorology

Figure 15 shows NO and NO\textsubscript{x} concentrations as well as PM\textsubscript{10} (TEOM) and particle number concentration (CPC). During the first week of measurements, the high NO concentrations showed that exhaust was occasionally present in the tunnel, probably due to planned maintenance activities using diesel engines. These occasions were identified by high nitrogen oxide concentrations, when the ratio of particle number to nitrogen oxides was similar to published data on road traffic exhaust, i.e. around 1.5-2.4*10\textsuperscript{14} particles per gram of NO\textsubscript{x} (Janhäll and Hallquist, 2005). During the rest of the campaign, the nitrogen oxides were present in low concentrations and the ratio varied.

In the railroad tunnel the concentrations of nitrogen oxides are low compared to outdoor urban air, while the PM\textsubscript{10} concentrations are extremely high and particle number concentrations are mainly in level with outdoor urban background air (SLB, 2014).
Figure 15. NO and NO\textsubscript{x}, particle number and PM\textsubscript{10} concentrations at Arlanda C.
Figure 16. PM$_{10}$ relations to NO$_x$ and particle number and particle number relation to NO$_x$ at Arlanda C. Data has been classified based on some criteria as discussed in the text.

When all data is included in the scatterplots the correlations between pollutants are low at Arlanda C. Figure 16 shows scatterplots of particle concentration, both as mass and number, together with NO$_x$ concentration. The data has been divided into two main categories and an “odd fraction” of data, defined by a time period of high NO and low NO$_2$ concentrations. Occasions with particle number concentrations above 15 000 / cm$^3$ and NO$_x$ concentrations of above 1 µg/m$^3$ was separated as being connected to influence of combustion particles. The PM$_{10}$ concentrations are below about 75 µg/m$^3$, but increasing linearly with increasing particle number (lower right in Figure 16). Occasions with lower particle number but PM$_{10}$ ranging from 0 to almost 500 µg/m$^3$ are considered as being connected to coarser wear particles. Plotting NO$_x$ to PM$_{10}$ (lower left), shows that the data considered as combustion-related show a similar trend as number concentration to PM$_{10}$, but also reveals that some of the data considered as dominated by wear (PM$_{10}$ above 100 µg/m$^3$), also are connected to higher NO$_x$ concentrations (“odd fraction”). NO/NO$_2$-data show that these data from the 31$^{st}$ of January have high NO concentrations, practically no NO$_2$ and low particle number, why vehicle exhaust does not seem to be a likely source. A possible source to NO could be electrostatic discharges from the trains’ conductors. This is a process common during lightning in the atmosphere (e.g., Bond et al., 2001), but it is unknown whether the dissipation energy formed is enough for this process to occur in the railroad tunnel.
Most of the vehicle exhaust was present during night-time and connected to maintenance vehicles, but for 31st of January, the source for NO with concentrations above 20 µg/m³ is unknown. The PM₁₀ concentrations are shown in green in Figure 17 for data without exhaust present and in yellow when concentrations of NOₓ were above 4 µg/m³. The relation to specific trains and vehicles are further investigated in chapter 3.3.1.

![Figure 17. PM₁₀ concentration at Arlanda C with the occasions with NOₓ above 4 µg/m³ marked as yellow.](image)

The diurnal variation of particle concentrations is shown in Figure 18, averaged over each hour of a day, where the blue line is traffic data from the last week of the campaign when no diesel exhaust emissions were detected. Particle concentrations reach minimum during early mornings just before morning traffic starts, while maximum concentration, averaged to around 200 µg/m³ or 7000 /cm³, are generally related to morning rush hours. After the morning peak the particle number decrease rapidly again only affected by the evening rush hour, while PM₁₀ is high most of the day with rather similar concentration in the morning and evening peak.

![Figure 18. Particle concentration averaged over a full day, one week without exhaust for the particle concentrations and the full campaign for the train passages.](image)
Equally many train passes in the morning as in the evening peak hour, but the concentrations are much higher during the morning for particle number, while particle mass peaks are more similar. Both concentration peaks are delayed compared to the traffic peaks, even though the time difference is not possible to see, as even a small delay might move the particle concentration peak to be included in the next hour average.

The particles within the tunnel are probably related both to wear of rails and wheels etc., and to resuspension of the dust present in the tunnel. As wet grounds are known to reduce particle resuspension, occasions of precipitation are shown together with PM\textsubscript{10} concentrations in Figure 19. The precipitation data is corrected for snow where the data is purple. During rain events, the data is blue. During 6-8 of February, the PM\textsubscript{10} concentrations seem to be reduced by the precipitation recorded while dryer periods have higher concentrations of PM\textsubscript{10}. This calls for a study of the impact of outdoor meteorological conditions like temperature and humidity, on the concentrations inside the tunnel. It seems that the outdoor conditions affect suspension of PM inside the tunnel. If all concentrations are averaged over either the non-precipitation or the precipitation events, the concentration is 20\% lower during precipitation events.

![Figure 19. PM\textsubscript{10} concentration and precipitation from the Arlanda meteorological station. PM\textsubscript{10} during diesel contamination is given in yellow and precipitation corrected for snow is purple.](image-url)
3.2.2. Temporal patterns in the Söderleden road tunnel and relation to traffic and meteorology

Nitrogen oxides are displayed together with particle number and mass concentrations in Figure 20, showing the high concentrations of nitrogen oxide in the road tunnel as compared to the previously described train tunnel. The nitrogen oxides have lower daytime concentrations during weekends, as five consecutive peaks are followed by two lower peaks due to lower weekend traffic. In addition, particle number is on average 20–30 times higher compared to the concentrations in the train tunnel. The high concentrations are present during a larger part of time. The PM$_{10}$ concentrations are about three times as high as in the railroad tunnel, and very low in the first part of the campaign.

The low PM$_{10}$ concentrations in the first part of the campaign coincides with high humidity outside the tunnel indicative of wet road surface conditions suppressing particle suspension due to traffic turbulence in the tunnel, see Figure 21. Rain events in the area were recorded. The Söderleden road tunnel data are analysed together with meteorological data from Torkel Knutssonsgatan (www.slb.se), situated nearby the northern entrance of the tunnel.

![Figure 20. NO and NO$_2$, number concentration and PM$_{10}$ in Söderleden road tunnel (15-minute mean values). PM$_{10}$ data was lost after 6th of March due to a tunnel washing occasion that flooded the instrument inlets.](image)

VTI rapport 917A
Figure 21. Traffic counts from bridge south of Söderleden road tunnel, particle concentrations and rain events causing humid conditions, defined as situations with vapour pressure deficit below 0.03.

The effect of humidity is visible as the first part of the campaign had wet conditions that later dried up, as shown in Figure 21. Events of vapour pressure deficit\(^1\) below 0.03 are marked as purple and the effect on PM\(_{10}\) concentrations is shown. The effect of using relative humidity instead of VPD is not large, but as the focus here is on possible condensation of water, VPD is a better defined physical property. Particle number, on the other hand, follows the traffic in the tunnel relatively well. For pollution ratio analyses the five wet days during the campaign were compared to five dry days keeping the weekday constant to minimize the effect of weekend traffic on the result. The ratio dry/wet for PM\(_{10}\) was about six, while ratios for NO and particle number were close to one. The dry/wet ratio for NO\(_2\) was slightly higher than one, which might relate to nitrate chemistry, but that is merely a speculation.

The correlation between traffic and air pollutants is around 0.7–0.9 for all situations apart from PM\(_{10}\) during dry conditions, where the correlation is only 0.3. This shows that during dry conditions PM\(_{10}\) has a more complex relationship to traffic, likely to include fleet composition characteristics, such as ratio of heavy duty vehicles and time delay due to resuspension and deposition processes, whereas particle number and NO\(_x\) are controlled by exhaust emissions.

\[ \text{VPD calculated from relative humidity and temperature according to } \frac{1-RH}{100} \times 0.6108 \times e^{\frac{17.27+RH}{237.3}} \]

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\(^1\) VPD calculated from relative humidity and temperature according to \( \frac{1-RH}{100} \times 0.6108 \times e^{\frac{17.27+RH}{237.3}} \)
Correlation between pollutants divided into wet and dry period are shown in Figure 22. A population of data during the wet period seems to belong to the dry period and vice versa, which shows that the actual dry and wet conditions are more detailed and separated in time than the simple division in periods made here.

In Figure 23, the variation of PM$_{10}$ during the wet and the dry periods in connection to meteorology at rooftop height is shown. Meteorological parameters affect PM$_{10}$ concentration, but the very scattered data in the plots indicate that the effects are likely to be subordinate to the occurrence of dry road surfaces when it comes to PM$_{10}$ concentrations.

Figure 22. PM$_{10}$ relations to NO$_x$, and particle number and particle number relation to NO$_x$ during the wet (-2013-03-21) and the dry (2013-03-21-) periods.
Figure 23. PM$_{10}$-concentrations during the wet and the dry periods and the relations to meteorological parameters. Meteorology taken from rooftop measurements at Torkel Knutssongatan, ca 1 km from the northern tunnel opening.
Figure 24. Concentrations of NO and NO\textsubscript{2} together with traffic counts in the southerly direction.

Figure 24 shows the nitrogen oxides together with traffic counts for south direction. NO clearly follows the traffic weekly variation with low weekend concentrations, while the weekend effect on NO\textsubscript{2} is visible but smaller. The daily cycle differs between traffic and NO, as the peak is high for NO during morning and for traffic during afternoon, shown also in the diurnal variation (Figure 25). This could be due to influence from the other tunnel bore that has the traffic peak during the morning.

Figure 25. Diurnal variation in southbound traffic, NO, and NO\textsubscript{x} during workdays, as an average between the dry and the wet periods (thus referred to as “tot”).
Figure 26 shows the traffic counts averaged over each hour of the workday divided into wet and dry period, and weekend, respectively. The traffic during the two weeks of dry and wet workdays is almost identical, while the weekends lack the morning peak.

The particle number concentration has one continuous peak between 07:00 and 17:00 for wet and dry conditions but ends slightly earlier during dry conditions (Figure 27, upper). The peak is higher for wet conditions even though traffic counts does not differ. During weekends, the peak is even smaller than for dry conditions and peaks slightly later in the day. PM$_{10}$ during dry conditions peaks in the morning but falls off slightly again at lunch time, and another peak is formed during the early evening, still later than the traffic afternoon peak (Figure 27, lower). PM$_{10}$ concentrations are similar for wet workday and weekend conditions, even though the night-time concentrations are higher during weekends.
The particle number concentration has one continuous peak between 07:00 and 17:00, while the PM\textsubscript{10} peaks in the morning but falls off slightly again at lunch time. This might also be related to wetter conditions during afternoon. Varying meteorological conditions during this short period should have some impact on the results. Wet conditions increase particle number as compared to dry conditions, i.e. a smaller effect than on PM\textsubscript{10} but in the opposite direction.

### 3.3. Particle size distributions (APS, SMPS, ELPI)

Mean size distributions for the wet and the dry period in Söderleden road tunnel and the whole period for Arlanda C is shown in Figure 28. The mass size distribution for dry period in the road tunnel has a peak at about 4 µm while the distribution at wet conditions is bimodal with peaks at 3 and 0.7 µm. The Arlanda C distribution peaks at 2–3 µm.

At Arlanda C the mean number concentration is generally very low. An example of a short particle number peak can be seen in Figure 28, with a primary peak at 30 nm and a tendency for a secondary peak at 50 nm.
The particle size distributions of the Söderleden road tunnel in Figure 28 are averaged over the same time of day, but split into wet and dry periods, respectively, and show much higher concentrations of 3–4 µm particles during dry conditions, and slightly higher sub-micrometre mass concentrations. The distributions during both periods have a primary number peak at 20–30 nm and a secondary peak at about 100 nm. The wet period has lower concentration of most particle sizes, but also a slightly lower particle size in the peak (25–30 nm). The lower concentrations during wet conditions might be due to the fact that the data is from a Saturday as compared to the dry data sampled on a Monday.

Variations in size distributions over 24 h during wet and dry periods in Söderleden road tunnel can be seen in the examples presented in Figure 29 and Figure 30. Observe the differences in mass size distribution between dry and wet periods. The bimodal distribution of the coarse particles is obvious during the wet period, but the two modes do not necessarily coincide in time, pointing at different sources or processes affecting the concentrations. The wet period day showed in Figure 29 is a Saturday, with rather high traffic flows during the early hours (likely to be taxis, buses etc.) and a late morning rush hour starting at around 9. A rather distinct lunch hour minimum (13–15) can be seen in both the coarse fraction of the mass distribution and number distributions, while the finer fraction in the mass distribution does not display a similar minimum.

The number concentration distribution does not display obvious differences in wet and dry conditions while the mass size distribution has much higher concentrations and is coarser during dry conditions. The concentration is high also during night time, even though traffic is much lower. The concentration peaks also seem more isolated than during daytime, interpreted as suspension by single vehicles, likely to be heavy duty vehicles. For comparison, size distribution time series for a day at the railroad station Arlanda C is shown in Figure 30. The low traffic intensity compared to the road tunnel is reflected in the concentration peaks and the much lower number concentration stands out from the road tunnel data. A shift to a coarser background number distribution is seen at about 5 P.M. Number peaks associated with train passages does not seem to be affected by this shift, why it is likely to be related to an outdoor shift in air mass properties penetrating into the tunnel.

Figure 28 Mean mass and number particle size distributions during wet and dry period in the Söderleden road tunnel and at Arlanda C.
Figure 29. Time series of particle mass and number size distribution during a day with **wet road surface** in Söderleden road tunnel.

Figure 30. Time series of particle mass and number size distribution during a day with **dry road surface** in Söderleden road tunnel.
Figure 31. Time series of particle mass and number size distribution Friday 1st of February 2013 at Arlanda C.

In Figure 32, the total particle mass concentration as a function of geometric mean particle size is shown for the two tunnel sites. The Söderleden road tunnel has two distinct data clusters, representing the initial wet period, with low PM$_{10}$ concentrations and the final longer period with high PM$_{10}$ concentrations. During the wet period the concentration is relatively low and the particle size range is wide from about 1.1–4 µm, with the highest concentrations when particle mean size is around 2–3 µm. During the night between the 19th and 20th February between 19 PM to 6 AM, concentration and mean particle size successively grow followed by a high variation in concentrations but within a smaller particle size span between 3–4 µm, indicating dominating contribution from road dust suspension. At Arlanda C the size/concentration plot resembles the road tunnel plot during the wet period.
3.3.1. Detailed studies of train type (individual) effects at Arlanda

From data on time series of particle size distributions, trains related to peaks in mass size distributions of PM$_{10}$ and number size distributions of ultrafine particles have been identified on individual level using the LUPP-system and traffic data collected during the measurements. The amount of data is voluminous and it has not been possible to analyse the whole dataset. However, in Figure 34 and Figure 35, two days of size distribution time series from APS and SMPS instruments are shown. On this scale, pinpointing the arrival of RC-trains (older trains with locomotive and coaches) seem to
initiate mass peaks of the coarse fraction detected by the APS and, in some cases, ultrafine number peaks detected by of the SMPS. It can also be seen that the arrival of the specific train set RC6 1419 three times during the day in Figure 34 results in similar mass peaks. RC6 1344 generates an obvious ultrafine peak, while RC6N 1388 and the “Long night train” only results in small ultrafine particle concentrations and “RC 1419” no ultrafine peaks at its three passages of the station. In Figure 35 the high ultrafine peak from a diesel driven maintenance vehicle is seen at 2 AM and exposes a slightly finer size distribution than the ultrafines from the electric trains during daytime. The shifts in peak size among mass size distributions suggest size differences between particles emitted from different trains. The only RC-train that passes twice (RC6 1342) generates similar mass size peaks, but the ultrafine particle generation is unclear. At 9 AM the train RC6P 1336 generates a comparably long-lived ultrafine peak. As can be seen in the two figures, there are some peaks that do not relate to RC-train passages.

How about other train types than RC? In opposite to the irregular arrival pattern of the regional trains, the X60 commuter trains arrive almost simultaneously from both directions every 30 minutes from 8 AM to 11 PM. The lack of periodically recurring mass- or number concentration peaks that coincide with the periodical commuter traffic indicates that these train sets are not major particle sources (Figure 34 - Figure 36). Also, the X50/X55 and the X40 train sets seldom coincide with particle peaks, but are harder to discern from the RC arrivals. Exceptions occur, like the most obvious ultrafine particle peak in Figure 35 which seems to be related X40 (or even X60) trains. One should note that there is a difference in braking time for the RC locomotive train sets compared to the other train types operating at Arlanda C. The braking time is longer and has more variability since it is also dependent on the number of wagons. Furthermore, the wagons can either have disc or block brakes. Also in terms of electrical braking the RC locomotive train sets differs since they are without electrical braking in contrast to the commuter trains operating at Arlanda C.
Figure 34. Mass-weighted particle aerodynamic size distributions from APS (right) and number-weighted particle mobility equivalent size distributions from SMPS (left) on 2013-02-03-04 from 1 AM to 1 AM with arrivals of RC-trains at arrows. Numbers relate to train individuals.
Figure 35. Size distributions from APS (left) and SMPS (right) on 2013-02-05-06 from 1 AM to 1 AM with arrivals of RC-trains at arrows. Numbers relate to train individuals.
Figure 36. Size distributions from APS (left) and SMPS (right) on 2013-02-01. Numbers relate to train individuals.
The daily overviews seem to rather clearly suggest that most particle generation is connected to older type RC-trains. But if arrivals of all train types are plotted in shorter time series, the picture gets more complex. In Figure 37 – Figure 42, 2-hour time frames of size distributions are shown from the APS and the FMPS instruments used during this day starting at 11 AM and ending at 11 PM. Due to the rather high frequency of trains and the unknown effect of particle concentration time lag in relation to the detected arrival of each set, concentration peaks are not always synchronized with the detected arrival. In Figure 37, the RC-train U1351 is well synchronized with both a mass and a number size distribution peak, while the possible contribution from subsequent train arrivals seem to be overshadowed by the slowly decreasing peak from U1351.

The fast FMPS seems to register two or even three distinctively different size fractions in the ultrafine number distributions. One a between 10-20 nm, one at 40-110 nm and one at 100-300 nm. In Figure 37 the air is almost free from ultrafines until 11:32, when something emits two peaks at 10-20 and 40-110 nm. No train was detected at the platform. Another peak is detected before two X60 trains actually arrives and the undulation goes on until U1351 generates a distinct increase in the 10-20 nm fraction while the 40-110 nm fraction seems unaffected. The emission of 10-20 nm particles during these 30 minutes lingers on and decreases slowly until about 5 PM (see Figure 38 and Figure 39). The peaks in the 100-300 nm interval are very short-lived (minute) and do not seem to correlate to any train arrivals at the platform. Generally, peaks not related to detected arrivals might be a result of contaminated air pulses from other parts of the tunnel, moved around due to piston effects and natural ventilation, complex air circulation due to trains coming in from both directions simultaneously or might be originating in the parallel tunnel for the Arlanda Express train, which has some connections to the studied tunnel.

In Figure 38 the RC-trains 1411 followed by U1351 results in a higher mass concentration but their arrivals are not exactly synchronized with concentration increase. The high increase in mass concentration at 14:50 seems not simultaneous with any detected train arrival. In Figure 39 there are no distinct concentration changes that seem correlated to train arrivals, while in Figure 40, the RC-train 1422 generates an obvious shift to higher mass size distributions. The arrivals of RC-train and a X50 (or X55) train overlap at 11:12 and results in a wide ultrafine particle peak (10–100 nm). In Figure 41 the X60-trains 6055 and 6064 seem to correlate with the mass concentration peak at 19:10. Generally, it could be hypothesized that sudden coarse particle mass concentration changes should be related to a short distance from the emitting source and slower changes to polluted and more diluted air parcels moving along the tunnel. A train making the main speed reduction using mechanical brakes in the beginning of the tunnel will emit a cloud of brake dust, which can arrive at the platform later than the actual train set. This could be an explanation e.g. to the high peak starting at 14:50 in Figure 38.
Figure 37. Size distributions from APS (upper) and FMPS (lower) on 2013-01-31 from 11 AM to 1 PM with type specific train arrivals.

Figure 38. Size distributions from APS (upper) and FMPS (lower) on 2013-01-31 from 1 AM to 3 PM with type specific train arrivals.
Figure 39. Size distributions from APS (upper) and FMPS (lower) on 2013-01-31 from 3 PM to 5 PM with type specific train arrivals.

Figure 40. Size distributions from APS (upper) and FMPS (lower) on 2013-01-31 from 5 PM to 7 PM with type specific train arrivals.
Figure 41. Size distributions from APS (upper) and FMPS (lower) on 2013-01-31 from 7 PM to 9 PM with type specific train arrivals. Data lacking for FMPS between 19-20.

Figure 42. Size distributions from APS (upper) and FMPS (lower) on 2013-01-31 from 9 PM to 11 PM with type specific train arrivals.
The mean concentrations of elemental carbon (EC) and organic carbon (OC) are substantially higher in the road tunnel compared to the railroad tunnel. EC levels are a factor two higher and OC levels a factor of seven higher in the road tunnel (Table 2). Most of the total carbon (EC+OC) is EC in the railroad tunnel, 70%. For the road tunnel, most is OC, with only 43% being EC (Figure 43).

In the road tunnel, both EC and OC show a strong diurnal variation with lowest values during the night (Figure 43). Both EC and OC is directly related to traffic exhaust and thus correlate strongly with particle number and NO\textsubscript{x}. The correlation between EC and NO\textsubscript{x} is slightly higher (r=0.94) than between OC and NO\textsubscript{x} (r=0.89), likely reflecting the different emissions of EC, OC and NO\textsubscript{x}, depending on variations in vehicle type, fuel and possibly also driving conditions. The mean ratio of EC to NO\textsubscript{x} is 12 mgEC/gNO\textsubscript{x} in the road tunnel. For OC the ratio is 17 gOC/gNO\textsubscript{x} (Table 2).

Obviously, these values mainly reflect the ratio in the exhaust from the vehicles. The OC/EC ratio increase during night and this could be due to changing emissions with relatively more OC than EC due to less heavy duty diesel vehicles during night time. It could also partly be due to lower temperature at night which increase condensation of semi volatile organics in exhaust.

Table 2. Mean concentrations of EC, OC, NO\textsubscript{x}, NO\textsubscript{2}, total particle number and PM\textsubscript{10} during periods with simultaneous measurements of these parameters in the Arlanda railroad station and in the Söderleden road tunnel 2013.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Arlanda railroad tunnel</th>
<th>Söderleden road tunnel</th>
<th>Ratio Söderleden/Arlanda</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Jan 30 - Feb 10 (only 6 AM to 9 PM)</td>
<td>March 7 6 AM – March 8 9 AM</td>
<td></td>
</tr>
<tr>
<td>EC (µg/m3)</td>
<td>9.90</td>
<td>23.3</td>
<td>2.35</td>
</tr>
<tr>
<td>OC (µg/m3)</td>
<td>4.18</td>
<td>29.7</td>
<td>7.10</td>
</tr>
<tr>
<td>EC/(EC+OC) g/g</td>
<td>0.70</td>
<td>0.43</td>
<td></td>
</tr>
<tr>
<td>NO\textsubscript{x} (µg/m3)</td>
<td>1.79</td>
<td>2268</td>
<td>1265</td>
</tr>
<tr>
<td>NO\textsubscript{2} (µg/m3)</td>
<td>0.57</td>
<td>486</td>
<td>858</td>
</tr>
<tr>
<td>Particle number &gt;7 nm (cm\textsuperscript{-3})</td>
<td>3408</td>
<td>76 646</td>
<td>22</td>
</tr>
<tr>
<td>PM\textsubscript{10} (µg/m3)</td>
<td>129.7</td>
<td></td>
<td></td>
</tr>
<tr>
<td>EC/ NO\textsubscript{x} g/g</td>
<td>5.9</td>
<td>0.012</td>
<td></td>
</tr>
<tr>
<td>OC/ NO\textsubscript{x} g/g</td>
<td>2.5</td>
<td>0.017</td>
<td></td>
</tr>
</tbody>
</table>

As seen from Table 2, the mean EC level in the railroad tunnel is quite high, 9.9 µg/m\textsuperscript{3}, almost half of the concentration measured in the road tunnel. As the EC and OC measurements were made on filters collected during 15 hours every day (06:00 to 09:00), we cannot relate the concentrations to train movements. But it is clear that there is a substantial non-combustion source of EC; it is not coming from diesel fuelled trains. Likely this EC is due to the wear of graphite during the sliding contact between the pantograph collectors and the overhead contact lines that distribute the electric power to the trains. Some EC could also come from brake wear if they contain e.g. sintered iron-graphite alloys. A small fraction of the EC can also enter via ventilation air from outside the tunnel, but this must be
small as indicated by the low NO$_x$ concentrations (EC is normally much lower than NO$_x$ in ambient air).

The origin of OC is less obvious. Some of it can be from the ambient air via the ventilation. Other possible sources are lubrication oils used to minimize abrasive wear of contact lines or rails, re-suspended organic particulate material that had been deposited after running diesel trains, organic debris from commuter’s clothes etc. or other sources of organic debris.

Figure 43. Concentration of OC and EC within the Söderledstunnel, 3hr samples during 27hr (upper), and the part of total carbon that consists of elemental carbon (lower).
3.5. Size segregated element composition

In the road tunnel, there is a large difference in PM$_{10}$ composition between the periods with dry and wet road surfaces (Figure 44 and Figure 45). For both periods iron is abundant. Iron is uni-modal at around 2 µm during wet conditions and during dry conditions a second coarser mode appears at around 5-6 µm, where also other mineral related elements peak. The concentrations of iron are three-folded during dry conditions. Mineral related elements, e.g. silicon, calcium and potassium are all only a small fraction in wet conditions compared to in dry conditions. The mineral elements peaks are at 4-5 µm, which makes the Fe peak bimodal during dry conditions. Sulphur has a peak around 500 nm during wet conditions, while the peak during dry conditions is trimodal, with a lower peak at about 500 nm but also a higher bimodal peak coinciding with mineral related elements like Si, Ca, K and Ti at around 5-6 µm. The peak at 500 nm is almost ten folded during wet conditions. This indicates that the smaller sulphur containing particles at 500 nm deposit on larger dust during dry conditions and/or that S-sources like tyre and bitumen wear are stronger during dry conditions.

Similar to iron, copper is unimodal at wet conditions peaking at 2 µm. At dry conditions a slightly coarser mode is added in the same size range as mineral dominated elements. The relatively high peak at wet conditions (only about doubled at dry conditions) indicates that a relatively large proportion of copper at a size of around 2 µm is emitted directly to the air from brake wear and not emitted from the road surface.

As for copper, zinc and manganese peaks at 2 µm, with an additional coarser peak that correlates to peaks of crustal elements related to road wear during dry condition in the road tunnel. Zinc is the most abundant metal in tires, with up to 15 times higher concentration compared with the concentration in brake pads used for cars (Apeagyei, Bank, & Spengler, 2011). Brake wear and bitumen are likely sources of manganese.

Since the measurements were made during winter, it could be argued that chloride originates in road salt. However, the size distributions follow the pattern of the brake wear related elements, with a single peak at about 2 µm during wet conditions and a bi-modal peak at dry conditions, which renders this conclusion uncertain. A possible explanation for the bi-model appearance during dry conditions is that Cl might be mixed or connected to mineral dust from road pavements, as Si also have a bi-modal shape. The amounts of Cl are an order higher during dry than wet road conditions.

For the railroad tunnel the large importance of iron in the particles is clear (Figure 46). The concentration of iron is of the same magnitude as in the road tunnel. Iron peaks at approximately the same particle size as the lower peak (2-3 µm) in the road tunnel indicating a common source such as brake systems, in which iron contents reported as much as around 60 wt. % (Kennedy & Gadd, 2003; Wahrlström, Olander, & Olofsson, 2012). The wider peak and its tendency of bimodality indicate additional iron source(s) for PM$_{10}$ in the railroad environment, for example, the mechanical wear processes of wheel-to-rail contacts, which can generate ultrafine and fine particles at high temperature and coarse particles at low temperatures (Sundh & Olofsson, 2011).

Pantograph (current collector) and catenary system (in which copper wire is used to supply electricity) is a contributing origin of copper in railroad traffic, together with braking material with copper accounting for 7–16 wt. % (Kukutschová et al., 2011; Wahrlström et al., 2012). In the railroad tunnel, there is a tendency to a finer copper peak below the above speculated brake wear related peak that could be related to pantograph and catenary system. On the other hand, it coincides well with the modes of the iron distribution and could also be related to wheel-to-rail wear. In Table 3, it is clear that the average copper content in railroad particles was quadruple of the concentration in road particles. And the ratio of Cu/Fe released in railroad was higher than in the latter environment, implying additional contributors to copper emission.

As in the road tunnel, Zn peaks at about 2 µm, indicating a common source for Cu and Zn in both environments. However, in Table 3, it can be seen that the zinc component produced in Arlanda...
railroad was almost an order of magnitude higher than in Söderleden road tunnel, suggesting a higher emission of zinc (compared to copper) in the railroad environment.

Manganese and chromium coincides with the coarser mode of iron in the railroad tunnel, indicating lower contribution from brake wear and a higher from wheel and rail steel. Mn concentration generated from trains (from Table 3) was much higher than from road vehicles, indicating a higher wear rate than in the road tunnel. In terms of the Mn/Fe ratio, in the road tunnel, the Mn/Fe ratio (0.0096) was very close to 0.01, which is very similar to the ratio in manganese steel used in brake materials (Moreno et al., 2015). However, it is of key interest that all the Mn/Fe ratios monitored in Arlanda railroad tunnel were below 0.01 (the maximum number is 0.0089), with just 0.0059 in average ratio value. This indicates that except origins from Mn steel which was often used to manufacture wheels and rails (Mn/Fe >0.008, both for wheels and rails), there should be a large contribution of iron derived from other sources, such as brake pads (the Mn/Fe ratio was 0.0034 in brake pads used for X20).

Table 3. Diameter range and concentration statistics from the railroad and road sites.

<table>
<thead>
<tr>
<th>Place</th>
<th>Name</th>
<th>Fe</th>
<th>Cu</th>
<th>Zn</th>
<th>Mn</th>
<th>Mn/Fe (ratio)</th>
<th>Cu/Fe (ratio)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Railroad</td>
<td>Arlanda C</td>
<td>dg (µm)</td>
<td>0.01-12.25</td>
<td>0.04-12.25</td>
<td>0.135-12.25</td>
<td>0.51-12.25</td>
<td></td>
</tr>
<tr>
<td>c (ng/m³)</td>
<td>(min-max)</td>
<td>1.4-8655.8</td>
<td>3.4-674.9</td>
<td>0.8-348.7</td>
<td>6.6-60.2</td>
<td>0-0.0089</td>
<td>0 – 1.185</td>
</tr>
<tr>
<td>Average value</td>
<td></td>
<td>2095.7</td>
<td>231.4</td>
<td>121.1</td>
<td>30.84</td>
<td>0.0059</td>
<td>0.152</td>
</tr>
<tr>
<td>Road tunnel</td>
<td>Söderleden</td>
<td>dg (µm)</td>
<td>0.04-12.25</td>
<td>0.21-12.25</td>
<td>0.8-12.25</td>
<td>0.3-5.5</td>
<td></td>
</tr>
<tr>
<td>c (ng/m³)</td>
<td>(min-max)</td>
<td>1.5-4237.4</td>
<td>1.8-180</td>
<td>0.3-50.7</td>
<td>1.9-23.8</td>
<td>0 – 0.01</td>
<td>0 – 0.074</td>
</tr>
<tr>
<td>Average value</td>
<td></td>
<td>1093.2</td>
<td>54.75</td>
<td>19.93</td>
<td>11.16</td>
<td>0.0096</td>
<td>0.037</td>
</tr>
<tr>
<td>Ratios</td>
<td>C/Söderleden</td>
<td>CArlanda C</td>
<td>0.52</td>
<td>0.24</td>
<td>0.16</td>
<td>0.36</td>
<td></td>
</tr>
</tbody>
</table>
Figure 44. Size segregated element concentration in PM$_{10}$ in Söderleden road tunnel during the period with wet road surface. Main elements in upper diagram and elements with lower concentrations in lower diagram.
Figure 45. Size segregated element concentration in PM$_{10}$ in Söderleden road tunnel during the period with dry road surface. Main elements in upper diagram and elements with lower concentrations in lower diagram.
Figure 46. Size segregated element concentration in PM$_{10}$ in Arlanda C railroad tunnel. Main elements in upper diagram and elements with lower concentrations in the two lower diagrams.
Figure 47. Size resolved relative concentration of elements in PM$_{10}$ from Söderleden road tunnel during wet (upper) and dry (lower) conditions.
Enrichment factors were calculated using Ti as reference element and upper continental crust (UCC) data from Wedepohl (1995) (Figure 49 and Figure 50). The Söderleden road tunnel data show that the enrichment factors for S, Cr, Mn, Fe, Ni, Cu, Zn, Br, Zr, Mo and W are obviously higher during wet conditions, indicating a relation to direct vehicle emissions. At dry conditions, a higher contribution from road wear and resuspended road dust reduces the enrichment factors as crustal components contribute to the elemental composition. Rb, Sr and Sn only appear during dry conditions. Factors for Rb and Sr are close to unity and can be regarded to be of crustal origin, while Sn is enriched 40-fold and is likely to be present in road dust, but originating from other sources than vehicles or road wear.

The lack of Br, Rb, Sr, Zr, Mo and Sn in the Arlanda C enrichment factors also supports a crustal origin of the samples, since the road wear source is lacking in the railroad tunnel. Otherwise, the pattern of enrichment of Si, S, Cl, Cr, Mn, Fe, Ni, Cu, Zn and W at Arlanda C is very similar to that of the road tunnel. The enrichment of Cr, Mn, Fe, Ni, Cu and Zn are several tens- or even 100-fold higher than for Söderleden road tunnel during dry conditions, revealing strong metallic sources in the railroad system. All these elements are common in wheels, rails and brake systems. Tungsten has a widespread use in technical and electrical components (Koutsospyros et al., 2006), e.g. common in welded materials, why the enrichment at Arlanda C could originate in e.g. wear of welds in tracks.

Figure 48. Size resolved relative concentration of elements in \( PM_{10} \) from Arlanda C.
Figure 49. Enrichment factors based on composition of upper continental crust (UCC) Wedepohl, (1995).

Figure 50. Comparison of enrichment factors during dry and wet conditions in Söderleden road tunnel.
3.6. Results from on-board measurements from a X60 train

Example of results from on-board measurements on an X60 commuter train operating from Älvsjö C to Uppsala C is presented in Figure 51 – Figure 54. Before starting the journey, the train had a stop at the maintenance stop at Älvsjö. When the train started, it had no passengers except two people maintaining the test instrument and had been standing still for 40 minutes. At the stop at Uppsala C all passengers left the train except the two persons maintaining the instruments. On its way, it passes the two underground stations: Stockholm Södra and Arlanda C. Note that the underground station Arlanda C is the same platform as stationary platform measurements. The results show an increase in the total number of airborne particles when stopping in the two underground stations as measured by the OPS both in the cabin and under the train (upper plots in Figure 52 and Figure 53). The levels are higher outdoor under the train and also more transient. Several peaks were measured outside the train that for a short time interval was higher than when stopping at the two underground stations.

![Graphs showing PM levels under and inside the train](image)

Figure 51. PM$_{2.5}$ and PM$_{10}$ levels for the overall forward trip (P1) and in tunnel air measured under and inside the train (a) PM$_{2.5}$ under train (b) PM$_{10}$ under train (c) PM$_{2.5}$ inside train, and (d) PM$_{10}$ inside train. The box comprises values between the upper 25% and lower 25%, the line in the box represents the median value and the square inside the box shows the mean value. The whiskers show 1.5 times the inter-quartile range upper and lower the central box.
In addition, PM$_{10}$ and PM$_{2.5}$ were calculated from the OPS data using a unit density and an assumption of spherical particles. The results are presented in Figure 51. The results show an increase in the PM$_{10}$ level when stopping in the two underground stations as measured by the OPS in the cabin as well as under the train. The levels are higher outdoor under the train and also more transient. Several peaks were measured outside the train that for a short time interval was higher than when stopping at the two underground stations.

![Graph showing particle number concentration](image)

Figure 52. (a) inside train particle number concentration on 29th of Jan., measured by OPS, T-A: Arlanda C tunnel, T-C: short tunnel, T-D: Södra station tunnel (b) outdoor (under the train) particle number concentration on 29th of Jan, measured by OPS.
More detailed study of the particle concentration variation (Figure 53, lower) indicates that the braking before each stop induces a particle pulse outside the train, which decreases as the train stops. The concentration inside the train increases after the train have stopped, likely caused by the outdoor air entering the cabin as the doors are opened, and then decreases slowly until the next stop. Comparing the particle size distribution measurement results from the Nanoscan presented in Figure 54, one can identify that there are more ultrafine particles at the Stockholm Södra platform compared to the platform at Arlanda C. The concentration measured by the Nanoscan was lower in general after Arlanda C going in the direction of Uppsala.
Figure 54. Particle size distribution measured outside the compartment with a Nanoscan SMPS (range 10 nm to 420 nm). X60 train operating Älvsjö C to Uppsala C. T-A: Arlanda C tunnel, T-C: short tunnel, T-D: Södra station tunnel.
4. Synthesis and discussion

4.1. Identification of the main PM sources

4.1.1. Railroad tunnel

A comparison between the PM composition found in the railroad tunnel with material composition data, as representative as possible to the Arlanda tunnel traffic, has been compiled from literature and analyses made at KTH.

Railroad vehicle brakes consist of a variety of brake discs and pads with different elemental compositions. There are three main groups of brake pads; metallic, composite and sintered. Examples of analyses of these three types can be seen in Table 4. The compositions are very different. While Becorit is mainly consist of Mg, S and Fe, the sintered type is dominated by Cu, while the composite type is mainly C. How these brake pads are distributed among trains trafficking Arlanda C has not been further investigated.

Wheel material consists of iron (Fe) to over 96%. Carbon, chromium, nickel, copper and silica are the main trace elements (Table 5). The R8t material is common in wheels for commuter trains, while W.T. is common in wheels for locomotive driven train sets.

The chemical composition of copper alloy (CuAg 0.1) wires in Arlanda is 99.8% Cu, 0.0005% Bi, 0.04% O and 0.12% Ag. (source: Anders Lindgren, Swedish Transport Administration and in accordance to SS-EN 50149).

The results of this study confirm that the PM$_{10}$ fraction in railroad tunnels are characterized by a very high metal content and dominated by iron. This implies that wear of metallic components in the system is the main particle source. Possible sources include brake components, wheels, rails, pantograph and electric current lines.

Size segregated element composition suggests that brake wear dust peaks at 2 µm and is mainly composed of Fe, Cu and Zn. In the railroad tunnel Fe coinciding with Mn and Cr peaks, is suggested to also be related to wear of wheels and rail. The relatively high amounts of EC in the railroad tunnel is suggested to originate in pantograph collectors, but composite brakes may also be an important source.
Table 4. Brake pad element compositions. %TS = % total solid content.

<table>
<thead>
<tr>
<th>Element</th>
<th>Brake material (mg/kg)</th>
<th>Brake material (%TS)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Becorit</td>
<td>Sintered</td>
</tr>
<tr>
<td>Ag</td>
<td>0.696</td>
<td>7.96</td>
</tr>
<tr>
<td>Al</td>
<td>5480</td>
<td>210</td>
</tr>
<tr>
<td>As</td>
<td>57.1</td>
<td>2.51</td>
</tr>
<tr>
<td>B</td>
<td>21</td>
<td>9.35</td>
</tr>
<tr>
<td>Ba</td>
<td>1270</td>
<td>1.54</td>
</tr>
<tr>
<td>Be</td>
<td>&lt;0.5</td>
<td>&lt;0.3</td>
</tr>
<tr>
<td>Bi</td>
<td>1.76</td>
<td>0.926</td>
</tr>
<tr>
<td>Ca</td>
<td>7190</td>
<td>481</td>
</tr>
<tr>
<td>Cd</td>
<td>0.515</td>
<td>&lt;0.01</td>
</tr>
<tr>
<td>Co</td>
<td>31.3</td>
<td>9.84</td>
</tr>
<tr>
<td>Cr</td>
<td>36.2</td>
<td>1450</td>
</tr>
<tr>
<td>Cu</td>
<td>767</td>
<td>617000</td>
</tr>
<tr>
<td>Fe</td>
<td>41800</td>
<td>14900</td>
</tr>
<tr>
<td>K</td>
<td>4170</td>
<td>103</td>
</tr>
<tr>
<td>Li</td>
<td>2.07</td>
<td>0.34</td>
</tr>
<tr>
<td>Mg</td>
<td>64600</td>
<td>225</td>
</tr>
<tr>
<td>Mn</td>
<td>142</td>
<td>103</td>
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<td>Mo</td>
<td>8.08</td>
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<td>Na</td>
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<td>Ni</td>
<td>27.3</td>
<td>152</td>
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<tr>
<td>Pb</td>
<td>88.7</td>
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</tr>
<tr>
<td>Rb</td>
<td>40.7</td>
<td>0.405</td>
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<tr>
<td>Sb</td>
<td>41.3</td>
<td>1.21</td>
</tr>
<tr>
<td>Se</td>
<td>10.6</td>
<td>0.0896</td>
</tr>
<tr>
<td>Si</td>
<td>9830</td>
<td>1080</td>
</tr>
<tr>
<td>Sr</td>
<td>147</td>
<td>1</td>
</tr>
<tr>
<td>Th</td>
<td>2.58</td>
<td>&lt;0.02</td>
</tr>
<tr>
<td>Ti</td>
<td>490</td>
<td>25</td>
</tr>
<tr>
<td>Tl</td>
<td>0.523</td>
<td>&lt;0.05</td>
</tr>
<tr>
<td>U</td>
<td>1.86</td>
<td>0.0149</td>
</tr>
<tr>
<td>V</td>
<td>13.2</td>
<td>14</td>
</tr>
<tr>
<td>Zn</td>
<td>461</td>
<td>775</td>
</tr>
<tr>
<td>P</td>
<td>661</td>
<td>268</td>
</tr>
<tr>
<td>S</td>
<td>45900</td>
<td>43.8</td>
</tr>
<tr>
<td>Zr</td>
<td>508</td>
<td>0.282</td>
</tr>
<tr>
<td>Hg</td>
<td>0.342</td>
<td>&lt;0.03</td>
</tr>
<tr>
<td>C</td>
<td>12.4</td>
<td>93.1</td>
</tr>
<tr>
<td>N</td>
<td>&lt;0.10</td>
<td>1.06</td>
</tr>
<tr>
<td>H</td>
<td>&lt;0.10</td>
<td>&lt;0.10</td>
</tr>
</tbody>
</table>
An important approach of the current work is to investigate whether certain train types or train individuals contribute more to particulate pollution than others. If a train type is more frequently connected to high particle concentrations, a common brake system or wheel configuration might be considered an important source. Individual trains connected to high concentrations indicate an individual brake system or wheel configuration or a maintenance issue.

Many train arrivals practically coincide, which makes it difficult to identify individual trains as specific particle sources. Nevertheless, looking at a certain 24-hour period (Feb 3rd, 2013) with 118 trains passing the station and studying the occurrence of train arrivals in relation to high \( \text{PM}_{10} \) concentration peaks, reveals that 14 out of 18 RC train arrivals (78 %) coincide with peaks, while 1/15 (6.7 %), 2/11 (18 %) and 14/67 (21 %) coincide with X55, X40, and X60 train arrivals, respectively. These figures support the hypothesis that emission of \( \text{PM}_{10} \) in the tunnel most likely is related to RC train arrivals. Even though not investigated closer in this study, a possible explanation is...
that the RC trains have longer braking time and also more individual behavior since braking time depends on the number of wagons that are mechanical braked. Also in terms of electrical braking the RC locomotives are different since the locomotive itself is electric braked and the wagons are mechanically braked. The longer braking time may also make the particle emission periods from the RC trains longer and therefore also easier to detect depending on the sampling time of the instrument. However, not all RC trains are connected to high PM$_{10}$ concentrations, and occasionally some other train types are connected to particle concentration peaks, indicating that there are differences in the systems emitting the particles and/or how systems are used during the deceleration before finally stopping at the station. Differences could be associated to materials, state of system maintenance or different driving patterns.

4.1.2. Road tunnel

Particle sources in the road tunnel are both tailpipe particles and wear particles from pavement, tyres and brakes. Some contribution from winter maintenance salting might also be present. Since there are no possibilities to sample particles from individual vehicles during the measurements, the exhaust and non-exhaust particle composition is taken from literature.

Tailpipe particles originate in combustion and mainly consist of organic and elemental carbon (Pio et al., 2013). Pant and Harrison (2013) summarized trace elements for brake wear, tyre wear and re-suspension used in 22 studies between 2004 and 2012. The variation of tracers is large (Table 6), but copper is the most abundant tracer for brake wear, Zn for tyres and Al for resuspension (even though few studies have included this source). From road simulator studies, it is clear that PM$_{10}$ from SMA (stone mastic asphalt) pavements used in Sweden is dominated by Si, Ca, K and Fe in different proportions depending on rock type (Gustafsson and Johansson, 2012).

The data from the road tunnel were divided into two classes, from the wet road surface period and from the dry road surface period. A wet surface reduces the contribution of re-suspension and direct emissions from the interaction between tyres and road. This fact enables us to conclude that the elements detected in PM$_{10}$ during the wet period originates in sources without contact with the road surface, such as exhaust pipes and brakes. The distinct, coinciding, uni-modal Fe and Cu peaks at 2 µm is therefore suggested to be related to brake wear. This source is of course also present at dry conditions, but then the mode at 2 µm is accompanied by a coarser mode originating in suspension of road dust and/or direct emission from tyre/road surface interaction. Strong Si, Ca and K peaks in the coarser mode supports this conclusion. Studded tyre use makes the road surface source wear and dust suspension source strong during winter (e.g. Gustafsson et al., 2009a). The high Cl concentrations are suggested to originate in road salt.
Table 6. Range of elemental composition in car brake linings and car brake dust according to Thorpe and Harrison (2008).

<table>
<thead>
<tr>
<th>Metal</th>
<th>Car brake linings (mg/kg, unless indicated otherwise)</th>
<th>Car brake dust (mg/kg, unless indicated otherwise)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Al</td>
<td>3765</td>
<td>330–2500</td>
</tr>
<tr>
<td>As</td>
<td>&lt;2–18</td>
<td>&lt;2–11</td>
</tr>
<tr>
<td>Ba</td>
<td>2638</td>
<td>5900–74,400</td>
</tr>
<tr>
<td>Ca</td>
<td>14,300</td>
<td>920–8600</td>
</tr>
<tr>
<td>Cd</td>
<td>&lt;1–41.4</td>
<td>&lt;0.06–2.6</td>
</tr>
<tr>
<td>Co</td>
<td>6.4–45.8</td>
<td>12–42.4</td>
</tr>
<tr>
<td>Cr</td>
<td>&lt;10–411</td>
<td>135–1320</td>
</tr>
<tr>
<td>Cu</td>
<td>11–234,000</td>
<td>70–39,400</td>
</tr>
<tr>
<td>Fe (%)</td>
<td>1.2–63.7</td>
<td>1.1–53.7</td>
</tr>
<tr>
<td>K</td>
<td>857</td>
<td>190–5100</td>
</tr>
<tr>
<td>Li</td>
<td>55.6</td>
<td>Not reported</td>
</tr>
<tr>
<td>Mg</td>
<td>6140</td>
<td>83,000</td>
</tr>
<tr>
<td>Mn</td>
<td>181–3220</td>
<td>620–5640</td>
</tr>
<tr>
<td>Mo</td>
<td>0.4–215</td>
<td>5–740</td>
</tr>
<tr>
<td>Na</td>
<td>15,400</td>
<td>80</td>
</tr>
<tr>
<td>Ni</td>
<td>3.6–660</td>
<td>80–730</td>
</tr>
<tr>
<td>Pb</td>
<td>1.3–119,000</td>
<td>4–1290</td>
</tr>
<tr>
<td>Sb</td>
<td>0.07–201</td>
<td>4–16,900</td>
</tr>
<tr>
<td>Se</td>
<td>&lt;1–15</td>
<td>4.5–115</td>
</tr>
<tr>
<td>Sr</td>
<td>81.4</td>
<td>300–990</td>
</tr>
<tr>
<td>Zn</td>
<td>25–188,000</td>
<td>120–27,300</td>
</tr>
</tbody>
</table>

4.2. Abatement of identified sources

4.2.1. Railroad tunnel

Since RC trains are most commonly connected to high PM$_{10}$ peaks, reducing the amount of RC trains in the tunnel is likely to reduce particle concentrations. A preliminary conclusion is also that brake wear debris is dominating the contribution to airborne particles over wheel and rail wear, why adaption of type, composition and state of brake systems is an important abatement tool. Since the RC trains are an important source, a first focus should be on the brake systems of this train type. Further investigations on the differences in type and state of Rc braking systems might enable emission reductions. Also, driving behaviour might be important for brake wear emissions. How and when braking is performed in relation to the tunnel and the platform is likely to be important for the particle concentrations on the platform, but to our knowledge no studies have, so far, investigated this matter.
Laboratory studies show that there are small differences in coarse particle ($\text{PM}_{10,2.5}$) emissions between cast iron, organic and sintered brake pads. However, for fine and ultrafine particles, cast iron pads have the highest emissions followed by organic and sintered pads (Abbasi et al., 2012).

Elemental carbon (EC) is suggested to originate in pantographs and composite brake linings. EC has a mean concentration of about 10 $\mu$g/m$^3$ during the sampling, corresponding to around 8 % of the total $\text{PM}_{10}$ concentration.

Since this study focus on concentrations, size distributions and elemental composition of aerosols in tunnels and the relation to traffic and vehicle characteristics, no new information on how high particle concentrations can be abated by i.e. ventilation or screen doors has been investigated. A short overview of recent studies on these matter is given below.

Platform screen doors are used as a measure against high PM concentration in the platform areas of subways and a number of studies, mainly from Korea, have evaluated the effects. Kim et al. (2012) reported a significant 16 % decrease in $\text{PM}_{10}$ after installing screen doors. $\text{PM}_{2.5}$ also decrease, but not significantly (12 %). Similar improvements were found by Han et al. (2012) and Son et al. (2013). A major draw-back, published by Son et al. (2014b), is that PM levels inside subway trains have increased significantly with almost 30 % after installation of platform screen doors and on some subway lines up to over 100 %. Suggested measures to improve the situation were to synchronize ventilation with the daily variation in traffic flow either regulated by the time schedule or by sensors and artificial intelligence. Also, the importance of locating natural ventilation ducts in areas with low air pollution is stressed (Son et al., 2014b). Another suggestion, based on the fact that a high share of subway tunnel dust is iron and a large fraction of that magnetite, is to use magnetic filters. Son et al. (2014a) could demonstrate a removal efficiency of 56 % for $\text{PM}_{10}$, 46 % for $\text{PM}_{2.5}$ and 38 % for $\text{PM}_{1}$.

Few studies have focused on the influence of ventilation on PM concentration in railroad tunnels. A recent publication from Barcelona, Spain, has studied different ventilation modes in tunnel and platform in subway stations with different geometries and age (Martins et al., 2015). Ventilation is normally strong during daytime and low during night time for climate reasons. During warm periods, the ventilation is stronger resulting in lower PM concentrations. Impulsion of outdoor air during night time resulted in lower PM concentrations than extraction of indoor air. Piston effect alone was not effective for reducing particle concentrations and resulted in 29 % higher concentrations of $\text{PM}_{2.5}$ during the test week. In a newer subway system with platform screen doors it could be concluded that the ventilation of the tunnel was important for the air quality while the ventilation of the platform was not. Air conditioning inside trains reduced the PM concentrations with 15 % compared to platforms.

Rail oilers has been installed in sharply curved sectors in the Seoul subway system and tunnels are cleaned both with high-pressure water and magnetic dust collectors (Park et al., 2014).

### 4.2.2. Road tunnel

During dry conditions, road dust is a main source to $\text{PM}_{10}$ in Söderleden road tunnel during the campaign. High concentration of road dust in $\text{PM}_{10}$ can be abated through reducing the production of dust or through reducing the emission. The production is reduced through reducing traffic volume, speed, use of studded tyres and through improving road pavement wear resistance (Gustafsson et al., 2009a; Gustafsson et al., 2009b; Gustafsson et al., 2014; Gustafsson and Johansson, 2012).

Studded tyres are currently prohibited on Hornsgatan, Fleminggatan and Kungsgatan in Stockholm, and this has resulted in lower studded tyre shares on all roads in and around Stockholm. The reduction of their use is considered one of the main reasons for $\text{PM}_{10}$ concentration reductions in Norwegian cities since 2007 (Oslo Kommune, Bymiljöetaten, 2016). Evaluation of the effect on $\text{PM}_{10}$ concentration at Hornsgatan has been made using the Nortrip model (Denby et al. 2013). The impact of reduced studded tyre share and traffic volume results in reduced road wear of 72%. Combining this reduction with other emissions and meteorological factors results in an observed and modelled $\text{PM}_{10}$
net concentration reduction of around 50% averaged over the last four years. Around 40% of this 50% decrease is due to studded tyre reductions.

Another way of reducing the wear source to PM$_{10}$ is improving road pavements. Research generally confirms that reduced total wear of asphalt pavements also results in lower PM$_{10}$ emissions (Gustafsson and Johansson, 2012). Stone mastic asphalt pavements with coarse and wear resistant rocks is generally chosen to reduce pavement wear and PM$_{10}$ emission.

Both formation and emission of particles (direct and suspended) from the tyre and road interface are dependent on vehicle speed. A reduction in speed signage from 80 to 60 km/h on a major arterial road outside Oslo, Norway, resulted in an observed change of average speed from 75 to 65 km/h, and a PM$_{10}$ overall reduction of 36%. This was partly attributed also to a slight reduction in traffic volume (4%) and a reduction in maximum share of vehicles with studded tyres with 3%. Using the Nortrip model the observed decrease in PM$_{10}$ concentration could be reproduced and the model showed that about half of the reduction was due to change in speed. Also in Sweden, speed reductions have been used as a PM$_{10}$ mitigation tool. Analyses show an overall reduction in PM$_{10}$ emission factor of around 68 mg/vkm per 10 km/h reduced speed under conditions with high (60–70%) studded tyre use on a highway in Stockholm (Johansson et al., 2009).

Suspension of dust from the road surface can be reduced through e.g. dust binding with hygroscopic solutions keeping the road surface moist and effective sweeping techniques. For dust binding of paved roads, hygroscopic salts, such as CMA (calcium magnesium acetate), MgCl$_2$ (magnesium chloride) or CaCl$_2$ (calcium chloride) are used. The method is an effective but short-lived measure to abate suspension of road dust. The effect is highest when the road dust contribution is high and the reduction in the daily PM$_{10}$ concentration has been shown to be between 15% and 60%. The large variation is mainly depending on the actual contribution from suspended road dust to the total PM$_{10}$ concentration, but also depends on street type and traffic volume, type of dust binding solution and the amount, timing and frequency of the dust binding events.

Street cleaning would seem to be an effective method to reduce road dust, but the efficiency of today’s street sweepers to pick up and contain particles as small as PM$_{10}$ is generally low, which has been shown in several studies (Amato et al., 2010). Even though it is often difficult to see a direct effect on PM$_{10}$ from sweeping it is often hypothesized that reducing the load of coarser fractions will reduce precursors for PM$_{10}$ and is beneficial for air quality over time. Some modern sweeper techniques have been shown to reduce PM$_{10}$ also on short term (from day to day). For example, dry vacuum cleaning with very high vacuum in Stockholm was shown to reduce the local contribution to PM$_{10}$ by approximately 20% (Gustafsson et al., 2011) and in Finland a “street scrubber” using high pressure flushing, brushes and vacuum was shown to be effective when the dust load was high (Männikkö et al., 2014). During wet conditions, PM$_{10}$ is low and seems to be dominated by brake wear and exhaust particles. In laboratory studies, it has been shown that non-asbestos organic (NAO) and sintered brake pads emit lower amounts of particles than cast iron pads (Wahlström, 2011). The potential to reduce particle emissions from brakes depend on the current use of different types and which incentives to switch to lower emitting pads that are available and viable. Also, the need for braking in the tunnel is depending on the traffic flow and the occurrence of congestions. The traffic data available for this project has not been sufficient for any analyses of effects of reduced braking, but likely this will have only a very small effect on total PM$_{10}$ concentrations since brake wear constitute a very small part of PM$_{10}$ (<10%).

The high number concentrations in the road tunnel during both dry and wet conditions is due to exhaust particles, and highly influenced by diesel vehicles. From a health point of view, these particles have been shown to enter inside cars in tunnels to a much higher degree than the road dust particles (Johansson et al., 2013), why mitigation measures against ultrafine particles should have high priority in road tunnels where the exposure is mainly taking place inside cars. The Stockholm environmental zone, regulating heavy vehicle EURO-classes admission into the central parts of the city already
includes the Söderleden road tunnel. The increased number of diesel cars might increase the problem with ultrafine particles, but a decreasing trend of particle number concentrations in Stockholm seem to support a positive effect of modern diesel cars’ particle filters. For NOₓ, there is a tendency for slightly increasing concentrations (Omstedt et al., 2010).

The concentration of air pollutants in a tunnel depend on the rate of ventilation and type of ventilation. For the Söderleden road tunnel there is a longitudinal ventilation with jet fans in the roof. The opening in the tunnel wall to avoid tunnel air from exiting in the northern end lead to increased concentrations in the southbound tunnel bore. It might be questioned if this solution is optimal.

Possibly the operation of the jet fans could be optimised if PM₁₀ concentrations is the main target since then the wetness of the road is an important controlling parameter (less need to ventilate when roads are wet). But for exhaust related pollutants traffic intensity and congestion is probably most important for controlling ventilation. Another aspect that might be worth considering is the leaching of polluted tunnel air from the southbound tunnel exit into the northbound tunnel entrance. This can be prevented by a wall between the exit and entrance.

4.3. Future work

The rail road tunnel results indicate that at Arlanda C certain train types are more important contributors to PM₁₀ than others and it is suggested that type and state of brake systems are of high interest for abatement measures. As a consequence, continued research should focus on detailed examination of brake types and maintenance state, but also on driving behaviour and how or if it differs between train types when arriving at a tunnel station.

To attribute certain train sets to particle peaks was often not straight forward, due to time lags in train arrivals and concentration peaks. Detailed measurements of air movements in the tunnel combined with particle measurements with high time resolution should provide better accuracy in connecting train sets and particle concentrations.

The use of road tunnels has increased in Stockholm and is increasing even more in the future with the 18-km long tunnel as part of Förbifart Stockholm, a planned 18 km long road tunnel bypassing Stockholm. More time spent in road tunnels will reduce the beneficial effects on exposure (and health) of future decreasing vehicle emissions. In the case of the bypass it has been estimated that the tunnel exposure may result in 20.6 (95 % CI: 14.1–25.6) premature deaths annually (Orru et al., 2015). A number similar to the reduced number of deaths among the general population associated with reduced exposures in central city areas (outside the tunnel). The implications on the total car commuter or professional driver exposure has not been assessed, but since concentrations in the tunnels are much higher than in the ambient air it may lead to increased exposure at least for some groups of the population. On the other hand, vehicle ventilation filters are being improved, reducing penetration of particles into the cabins.

For road tunnels, the main options to reduce exposures are likely related to pavement types, effective cleaning and dust binding, speed regulations and other traffic regulations and air ventilation. So far different ways of removing pollutants from the air have not been very cost effective.
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The Swedish National Road and Transport Research Institute (VTI), is an independent and internationally prominent research institute in the transport sector. Its principal task is to conduct research and development related to infrastructure, traffic and transport. The institute holds the quality management systems certificate ISO 9001 and the environmental management systems certificate ISO 14001. Some of its test methods are also certified by Swedac. VTI has about 200 employees and is located in Linköping (head office), Stockholm, Gothenburg, Borlänge and Lund.