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Modelled atmospheric concentration of tyre wear in an urban environment

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ABSTRACT

Microplastics are emitted to the environment from many sources, and transported via water, soil and air. The airborne component is less well studied and measurements often focus on particles larger than 10 μm, while smaller particles are worse from a health perspective and likely more abundant. Measurement studies do often not include tyre wear, which is a large source of microplastics in urban areas. To improve the understanding of the microplastics concentration in urban areas, simulations of emissions and concentrations of atmospheric tyre wear particles (TWP) over Stockholm are performed. The results show that TWP are ubiquitous in urban areas, with highest concentrations along busy highways and in poorly ventilated street canyons. Yearly average roof level concentrations of TWP-PM_{10} range from 0.2 μg m^{-3} in areas with moderate traffic to 1.2 μg m^{-3} close to busy highways. Average yearly concentrations at three selected street canyons range from 0.7 to 1.1 μg m^{-3}. The modelled concentrations correspond to 4–6% of the total measured PM_{10} concentration. TWP emissions are expected to increase with increasing traffic in future, and possibly also with the increase in vehicle weight with electric vehicles and SUV:s.

1. Introduction

Microplastics (plastic particles <5 mm diameter) are present everywhere in the environment, in both water, soil and air. Many studies have focused on the occurrence and transport of microplastics in water (Zhang et al., 2020), but the airborne component has received less attention.

Microplastics are emitted to the environment from several different sources. One of the larger sources of microplastics is road traffic constituting, for example, around half of the total primary emissions of microplastics in Norway according to an estimation by Sundt et al. (2014). Microplastics from road traffic are mainly tyre wear, but also to a smaller degree wear of road markings and asphalt which contains polymer modified bitumen. Tyre wear is likely comprising more than 90% of the total microplastic emissions from traffic (Andersson-Sköld et al., 2020; Magnusson et al., 2016) and therefore, this study's main focus is tyre wear.

Tyre wear emissions arise due to the friction between tyres and the road surface, causing the tyre tread to be worn down. Tyres lose 10–20% of their mass during their lifetime (Andersson-Sköld et al., 2020). Some studies also suggest that particles may be emitted by volatilization during high temperatures and subsequent condensation (e.g. Dahl et al., 2006). The microplastic, i.e. rubber content, in tyres is estimated to vary between around 35–50% (Magnusson et al., 2016; Sommer et al., 2018).

There are only a few reports of measurements of microplastic concentration in air or surface deposition to date. Published microplastic concentration range from 0 to 175 particles m^{-3} in urban air (Abbasi et al., 2019; Asrin and Dipareza, 2019; Gaston et al., 2020; Liu et al., 2019) and deposition from 29 to 925 particles m^{-2} day^{-1} (Wright et al., 2020; Zhang et al., 2020). However, most of these measurement studies had a minimum size detection limit larger than 10 μm and not all of them included tyre wear in their definition of microplastics, meaning that they are likely to be underestimating the real values. Panko et al. (2013) observed concentrations of 0.004–1.34 μg m^{-3} of tyre wear particles in PM_{10} in different parts of the world and at different distances from roads. Rausch et al. (2022) observed mean tyre wear concentrations of 0.28 μg m^{-3} in the size range PM_1-PM_{10} in urban background and 2.24 μg m^{-3} at a kerbside location. Järlskog et al. (2022b) observed a tyre wear concentration of 1.4–2.2 μg m^{-3} in the size range 5–80 μm at a rural site next to a motorway. Thus, not many estimates of the PM_{10} concentration of microplastics are available, and the reported data are typically point measurements with limited time coverage.

Magnusson et al. (2020) measured microplastic deposition in different locations spread out over Sweden. Microplastics from tyre wear

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were mainly found in larger cities, and there in higher concentration than any other type of microplastic particles. Sommer et al. (2018) showed that TWP (tyre wear particles) comprise 34% of the investigated airborne particles close to busy roads in Germany and Rausch et al. (2022) showed that TWP comprise at least 11% of the PM_{10} concentration at a busy road in Switzerland. However, Rausch et al. (2022) mention that the $PM_{2.5}$ fraction may be underestimated due to limitations in the measurement method. Amato et al. (2013) showed that TWP comprised 16–41% of the road dust in urban areas.

Airborne microplastics, from tyre wear as well as from other sources, are likely harmful to human health (Prata, 2018). For example, workers in textile, flock and VC or PVC factories exposed to high concentrations of microplastic particles have shown respiratory symptoms (Prata, 2018). Due to the large surface area of microplastic particles, potentially harmful microorganisms and chemicals, e.g., PAH:s (polycyclic aromatic hydrocarbons) from vehicle exhaust, are easily adsorbed to the surface of microplastic particles (Prata et al., 2020). Microplastic particles from tyre wear also contain additives, some of which are hazardous for the environment and human health (Prata et al., 2020). Few studies are available on health effects of TWP and they show varying results (Kreider et al., 2020; Fussel et al., 2022).

For emission estimates and modelling studies, accurate tyre wear rates are needed. Literature estimates vary widely due to different assessment methods. According to a compilation by Polukarova et al. (2023) from a number of studies between the years 1974–2021, the values range from 4 to 400 mg/vkm for passenger cars. Looking only at studies from the last 20 years which are based on measurements, the values range from 45 to 214 mg/vkm (Table 1). One study instead used receptor modelling to estimate the TWP-PM_{10} in air and found a value of 2.2 mg/vkm (Table 1), while a previous compilation of TWP-PM_{10} emission factors in literature varied from 2.4 to 13 mg/vkm (Panko et al., 2018). Fewer measurements exist for heavy vehicles compared to passenger vehicles. Tyre wear rates for heavy vehicles are also expected to cover a wide range because of the large variations in weight, vehicle load, number of tyres etc.

Tyre wear rates are affected by several factors, such as tyre properties (tyre type, inflation pressure, alignment, brand, width etc.), road surface properties, vehicle properties, driving style, speed and meteorological conditions (Andersson-Sköld et al., 2020). Variation in these factors and in the measurement setup can explain the high variability in tyre wear rate estimates.

Modelling studies of airborne microplastics are few and limited to trajectory modelling in order to determine sources or possible end locations of microplastics (Zhang et al., 2020). To increase the knowledge of microplastic concentration in air caused by urban emissions, we will use model simulations to estimate the atmospheric concentration of tyre wear particles over a whole city. This is to our knowledge the first study estimating microplastic concentration in urban air and will be important to achieve an increased understanding the health effects. The simulations cover Stockholm, Sweden, an urban area where tyre wear is expected to be a major source of microplastics. In the simulations, the most recent tyre wear rates are used, calculated for the Swedish vehicle fleet.

2. Method

Model simulations of tyre wear emissions and atmospheric concentrations of TWP over Stockholm, Sweden, covering one year are performed. Stockholm is the capital of Sweden, having approximately 1.5 million inhabitants, where road traffic is the dominant local emission source for most atmospheric pollutants (Segersson et al., 2017). In the city center, buildings are commonly flanking the roads, promoting bad ventilation and elevated air pollution levels.

For simulations covering the urban area of Stockholm (Fig. 1), tyre wear emissions are calculated using the multi-road version of the NORTRIP emission model. The results are fed into a gaussian dispersion model to calculate the background concentrations over a domain covering the urban area. The results represent the concentrations at 2 m above ground level, or above roof level where there are buildings. These background levels are complemented with simulations with the NORTRIP-OSPM model at street level on three streets (see Fig. 1) in the city center.

2.1. NORTRIP emission model set-up

The NORTRIP (NOn-exhaust Road TRaffic Induced Particle) model (Denby et al., 2013a; 2013b) calculates the road dust load and emissions of non-exhaust particles. The emission sources include road, tyre and brake wear, sanding and salting. Processes describing the emission of particles, retention on the road surface, abrasion, crushing and removal by cleaning and snow ploughing depending on meteorological conditions and vehicle traffic are included. In this study we are only interested in tyre wear, and therefore the road and brake wear are set to zero and no sanding or salting is included. Snow ploughing by a standard rule was activated. The tyre wear is a function of the number of vehicles, vehicle speed, vehicle type (heavy/light) and tyre type (summer/friction/-studded) and the wetness and depth of snow/ice on the road surface. The tyre wear is not dependent on pavement characteristics or the driving style. Above a certain threshold of snow/ice height, 0.06 mm water equivalent, no road or tyre wear occurs.

The required meteorological input data for the NORTRIP model are taken from measurements in the Stockholm area. Precipitation type was not measured, so instead it is assumed that the precipitation is in the

Table 1
Examples of tyre wear rates and TWP-PM_{10} emission factors reported in literature. The unit (mg/vkm) shall be interpreted as mg TWP emitted per km driven by each respective car (mg per vehicle-km).

Tyre wear rate (mg/vkm)	TWP-PM_{10} (mg/vkm)	Type of vehicle	Type of measurement	Reference
4–400 average 112	–	Passenger car	Compilation of 48 studies from the years 1974–2021 based on both estimations and measurements	Polukarova et al., (2023)
55–214	–	Passenger car	Measurement of weight loss after driving in road simulator	Grigoratos et al., (2018)
45-57, average 51.1	–	Passenger car	Approximation of weight loss during tyre lifetime	Lee et al., (2020)
224	–	Lightweight truck	“	“
779	–	Bus	“	“
949	–	Heavyweight truck	“	“
Average 97, but more representative average was 74	–	Passenger car	Weighing of tyres from five cars	Luhana et al., (2004)
–	2.4–13	Passenger car	Compilation of 14 studies 1995–2013 based on estimations, modelling and measurements	Panko et al. (2018)
–	2.2 mg/vkm	Vehicle mix at street	Receptor model based on PM_{10} measurements at a street with known traffic amounts	Sjödin et al., (2010)

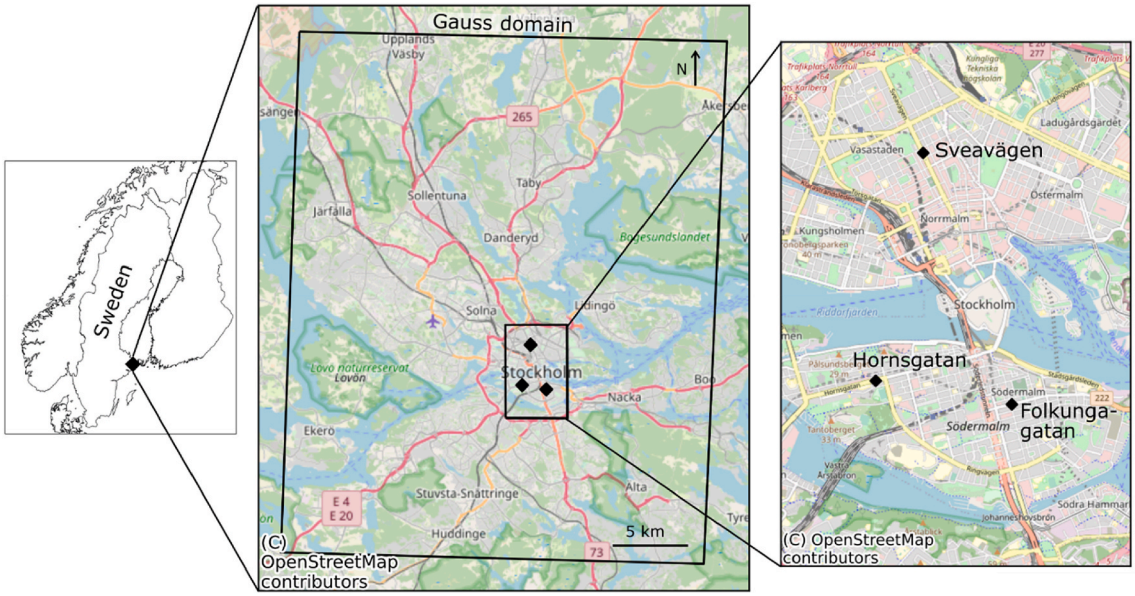

Fig. 1. Map showing the location of Stockholm in Sweden, the Stockholm city with the Gauss model domain marked by a black rectangle and Hornsgatan, Folkungagatan and Sveavägen marked by diamonds and a zoom in on the city center with the three streets marked.

form of snow during temperatures below zero. Road surface wetness is calculated through a mass-balance approach where precipitation, condensation, evaporation, drainage, splash and spray, refreezing and road maintenance activities are considered following standard rules in the NORTRIP model (Denby et al., 2013a).

The tyre wear rates are calculated based on the methods described in Polukarova et al. (2023), which aim to improve the tyre wear estimation for the Swedish vehicle fleet. For light vehicles (passenger cars and light duty vehicles) the wear rates are based on measurements of tyre weight loss and thereafter scaled to different road types (rural, motorway and urban). Wear rates for rural roads are lowest. Motorway wear rates are 24% higher and urban wear factors 57% higher than for rural roads. In this study, the urban wear rates are used for all roads in the simulation. Tyre wear rates for passenger cars (PC) and light duty vehicles (LDV) are combined to a single wear rate assuming that the share of PC and LDV is 88% and 12% respectively, based on driven mileage statistics from Sweden (Trafikanalys, 2020).

For heavy vehicles (vehicles with total weight of more than 3.5 tonnes), Polukarova et al. (2023) used tyre wear rates based on literature combined with statistics of the Swedish vehicle fleet (weight classes and number of tyres) and detailed mileage statistics from a previous study. The same method is used here but using only the urban wear rates weighted together using the urban mileage statistics. The resulting tyre wear rates for light and heavy vehicles are presented in Table 2. In the used standard settings of the NORTRIP model, 10% of the tyre wear is released as PM_{10} and 1% as $PM_{2.5}$. The model is run from October 1, 2018–December 31, 2019. The first three months are used to allow buildup of the road dust and only data from the year 2019 are shown in the results.

Only light vehicles (passenger cars and light duty vehicles) are assumed to change tyres during the winter season, while heavy duty vehicles are assumed to use summer tyres all year round. During the period when winter tyres are required (1 December – 31 March), the share of studded tyres for light vehicles in the urban area simulation is 40% in most of the inner-city streets and 50–60% in the suburbs (see Fig. S4). This is extrapolated from targeted measurements during the winter season (see for example Brydolf and Elmgren (2020)). Based on observations, the share of winter tyres is set to increase linearly with time starting two months before 1 December and to decrease linearly during one month after 31 March. Light vehicles that do not have studded tyres are assumed to have studless winter tyres during the winter and the intermediate periods in autumn and spring.

Table 2
Tyre wear rates used in the NORTRIP model.

Tyre wear rates	Summer tyres (mg/vkm^{-1})	Studless winter tyres (mg/vkm^{-1})	Studded tyres (mg/vkm^{-1})
Light vehicles	105	205	139
Heavy vehicles	1029	1029	1029

2.2. NORTRIP-gauss model setup

For the urban area simulations, the NORTRIP multi-road version is used with traffic data from the East Sweden Air Quality Management Association emission data base (Säll, 2021), which is administered by SLB Analys. The emission data base contains information of the average diurnal, weekly and seasonal traffic variation, the share of light and heavy vehicles, speed and tyre type based on measurements and estimations. Figs. S1–S4 shows information on the traffic data in the emission data base. The amount of heavy traffic is 6–8% on most of the smaller urban roads and 10–20% on some of the transit routes. Mean daily traffic varies from a few thousand on the smaller streets to 50 000–150 000 on the motorways through the city.

The emissions calculated by the NORTRIP model are fed into the Airviro Gauss (Apertum, 2021a) dispersion model, which is a gaussian dispersion model, where particles are treated as an inert tracer following the wind. Particles with sizes up to 10 μm are considered. The meteorological data needed as input are taken from measurements in Stockholm (Högdalen) and the wind field is thereafter calculated with the Airviro wind model (Apertum, 2021b). The calculations are performed in a grid with 100 m × 100 m grid cells over an area covering 28 km × 34 km, including the city and most of its suburbs. Fig. 1 shows the model area and the selected streets in the city center. The model is run with 1-hourly time resolution covering the year 2019.

2.3. NORTRIP-OSPM model setup

Three streets in central Stockholm are selected for more detailed

calculations with the single-road version of NORTRIP and OSPM. The streets are chosen because there is a relatively high amount of traffic with homogenous buildings on both sides and because PM_{10} measurements are available. The OSPM (Operational Street Pollution Model (Berkowicz, 2000),) is a dispersion model implemented in NORTRIP. It combines a plume model for the direct traffic emissions with a box model for the recirculation within the street canyon. Particles are treated as an inert tracer and sizes up to 10 μm are considered.

Table 3 shows the characteristics of the selected streets, and photos are shown in Fig. S5. For Hornsgatan there is available traffic measurements to use as input data. For the other sites, the yearly mean traffic and share of heavy vehicles are taken from the Stockholm city open database (https://dataportalen.stockholm.se/dataportalen/). The shape of the diurnal variation (different for weekdays, Saturdays and Sundays) is taken from the Hornsgatan measurements. The street geometries and building heights are approximated from Google Earth. Studded tyre share on all three streets is based on previous measurements (Brydolf and Elmgren, 2020). The share of studless winter tyres is estimated from inspection of parked cars in Stockholm (Trafikverket, 2019).

3. Results

3.1. Yearly mean emissions and concentrations

The calculated yearly total TWP-PM_{10} emissions are shown in Fig. 2 and the corresponding atmospheric concentrations in Fig. 3. The tyre wear emissions are clearly highest along the E20/E4 motorway, running through the city from southwest to northwest, with more than 100 000 vehicles per day. The emissions are also high along other transit routes through the city. The total tyre wear emissions (all particle sizes) in the modelled area amount to 960 tonnes per year. This is approximately 9% of the estimated total emissions in Sweden (Polukarova et al., 2023). The TWP-PM_{10} fraction, assumed to be 10% of the total tyre wear emission, amounts to 96 tonnes. As a comparison, the exhaust emissions of PM_{10} from road traffic in the model domain is 51 tonnes the same year (Engardt, 2021). The exhaust and tyre wear emissions are thus of similar magnitude, but TWP tend to be larger and thereby fewer in number.

The yearly mean PM_{10} concentration from tyre wear vary from 0 to 1.17 μg m^{-3} over the modelled area (Fig. 3). The highest concentrations are found along high-emitting roads. Not surprisingly, the average concentrations are highest along the E20/E4 motorway. The average concentrations along this road are many times higher than in the rest of the city. The concentration in the city center and part of the suburbs a few hundred meters away from the highways is relatively uniform with values around 0.2 μg m^{-3}. As expected, the street canyon simulations with NORTRIP-OSPM show much higher values than the roof level concentration. The modelled concentrations correspond to 4–6% of the total measured PM_{10} concentration at four of the city's measurement stations (Lilla Essingen, Hornsgatan, Folkungagatan and Sveavägen). At the roof-level site, Torkel Knutssonsgatan in central Stockholm, the Gaussian model calculates annual mean concentrations of total PM_{10} from all sources on the order of 9 μg m^{-3} during 2019. According to the model 0.2 μg m^{-3} is from tyre wear, 1.2 μg m^{-3} is from road and break wear, 0.1 μg m^{-3} is from tail-pipe emissions. The remaining 7.3 μg m^{-3} is attributed to long-range transport of PM_{10} from sources outside the

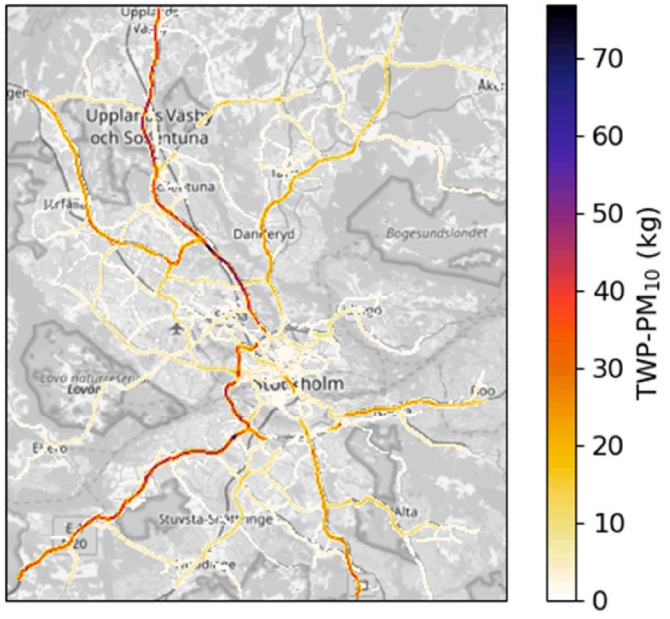

Fig. 2. Yearly total emissions of PM_{10} from tyre wear calculated in a grid of 100 × 100 m with the NORTRIP model.

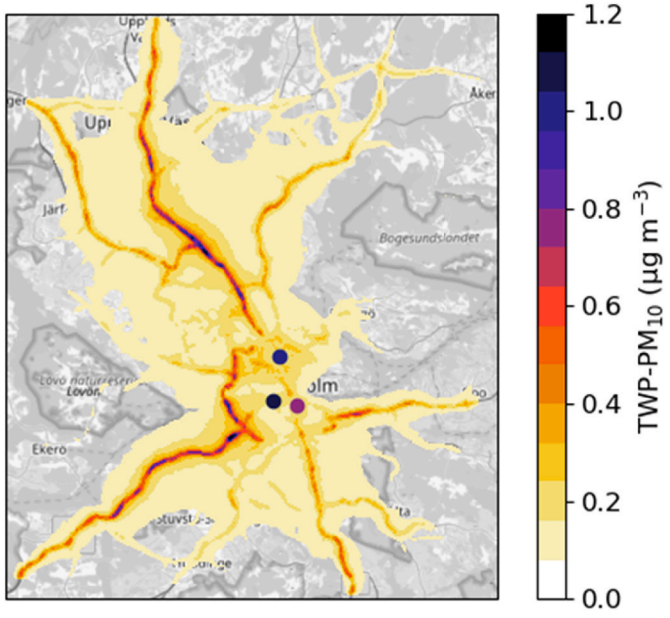

Fig. 3. Yearly mean TWP-PM_{10} concentration calculated with the NORTRIP-Gauss model on a 100 m × 100 m grid (contours) and the NORTRIP-OSPM model (points).

Table 3
NORTRIP model settings for the street level simulations. The coordinates give the location of the middle of the chosen stretch.

Street	Hornsgatan	Sveavägen	Folkungagatan
Coordinates	59.3172 N 18.0488 E	59.3410 N 18.0584 E	59.3148 N 18.0784 E
Building height (m)	24	28	South side: 21 North side: 27
Street canyon width (m)	23	32	23
Yearly average daily traffic (nr of vehicles/day)	21 314	22 000	14 000
Share of heavy vehicles (%)	8	8	12
Share of studded/ friction/summer tyres for light vehicles during winter (%)	24/75/1	33/66/1	33/66/1
Signed speed limit (km h^{-1})	30. Average speed from measurements: 41	50	30

modelling domain (Engardt, 2021).

The results agree reasonably well with the three existing studies which measured atmospheric mass concentrations of TWP, which were in the range 0.004–2.24 μg m^{-3} (Järlskog et al., 2022a; Panko et al., 2013; Rausch et al., 2022). Since the spatial resolution of the simulation is 100 m × 100 m, the concentration will be smoothed. Just next to roads, higher concentrations than the modelled maximum can thus be expected.

The results are not directly comparable to measurement studies of microplastic particles, where the concentration is given as number of particles per m^3. The maximum measured concentration in literature is 175 microplastic particles m^{-3}, but most other studies have observed much lower concentrations (Abbasi et al., 2019; Asrin and Dipareza, 2019; Gaston et al., 2020; Liu et al., 2019; Zhang et al., 2020). While the NORTRIP model does not calculate particle number concentration, the mass concentrations obtained in the present work generally correspond to much higher number concentrations than the published measurements. Many of the studies focused on particles coarser than 10 μm. For example, in their measurements of TWP at an urban kerbside during one year, Rausch et al., (2022) showed that including TWP particles in the range 1–80 μm gave a concentration of 13.47 μg m^{-3} and in the range 1–10 μm gave 2.24 μg m^{-3}. While these particles have environmental effects, airborne, inhalable particles are more relevant from a human health perspective. Measurements of TWP from road dust show that the number of particles smaller than 20 μm is very abundant (Järlskog et al., 2022a).

Values shown in Fig. 3 are yearly mean values, while there can be large fluctuations in the time series data. The average TWP-PM_{10} concentration during dry days (days with no precipitation or snowfall and very little road wetness at Hornsgatan) follows a very similar geographical pattern but with a maximum value of 1.5 μg m^{-3}. 56% of the days were defined as dry during the modelled year.

3.2. Time variation in concentrations

Fig. 4 shows the monthly average and monthly maximum TWP-PM_{10} concentrations from the NORTRIP-Gauss model and Fig. 5 shows the corresponding values for the three selected streets. Clearly, average monthly concentrations are highest in April, both in urban background and in the three streets. This can be explained by the very low amounts of precipitation this month (in total 2 mm) combined with a large accumulated tyre dust load after the winter season. The lowest concentrations are found in November in the Gauss domain, Hornsgatan and Folkungagatan, likely due to many days with wet roads.

The monthly mean values at the three streets are substantially higher than the Gauss model estimate at the same locations, showing clearly the street canyon effect (Fig. 5). Monthly mean values at the three streets are often of similar magnitude as the maximum monthly value in the Gauss domain (Fig. 4), even though the traffic is lower here than in many of the most heavily frequented roads in the area of the Gauss domain. However, the Gauss simulations are performed with a resolution of 100 m × 100 m, meaning that concentrations are likely higher very close to the most highly emitting roads. Hourly maximum concentrations in the Gauss domain are below 6.3 μg m^{-3} 90% of the time, but hourly values up to 80 μg m^{-3} occur.

Time periods with zero tyre wear concentrations occur when emissions are suppressed due to wet or frozen road surfaces which often happens in winter. For example, there is a long period of no emissions from 17 January to 9 February (not shown), contributing to the relatively low average values in January and February. The highest daily mean concentrations occur in winter, in both the Gauss domain and for the three streets, due to a combination of higher wear rates for winter tyres and stable meteorological conditions, inhibiting dispersion. Fig. 6 shows the amount of tyre wear particles on the road (tyre dust load) at Hornsgatan as modelled in the NORTRIP model. The tyre dust load accumulates in winter because of wet or frozen conditions on the road and is resuspended in spring as the roads dry up. At Hornsgatan, 25% of the emissions are explained by resuspension. The resuspension is lowest in February, when the amount of tyre wear particles in the road is at its maximum.

According to the Swedish air quality regulations the yearly mean PM_{10} concentration may not exceed 40 μg m^{-3} and the daily mean PM_{10} concentration may not exceed 50 μg m^{-3} more than 35 days per year. The TWP-PM_{10} is well below these limits by itself, but contributes to the total PM_{10} concentration. During days when the particle concentration is close to the limits, every extra μg m^{-3} can contribute to an exceedance. The health effects of TWP by themselves or as a mixture with other road particles is not yet fully understood, and their contribution is

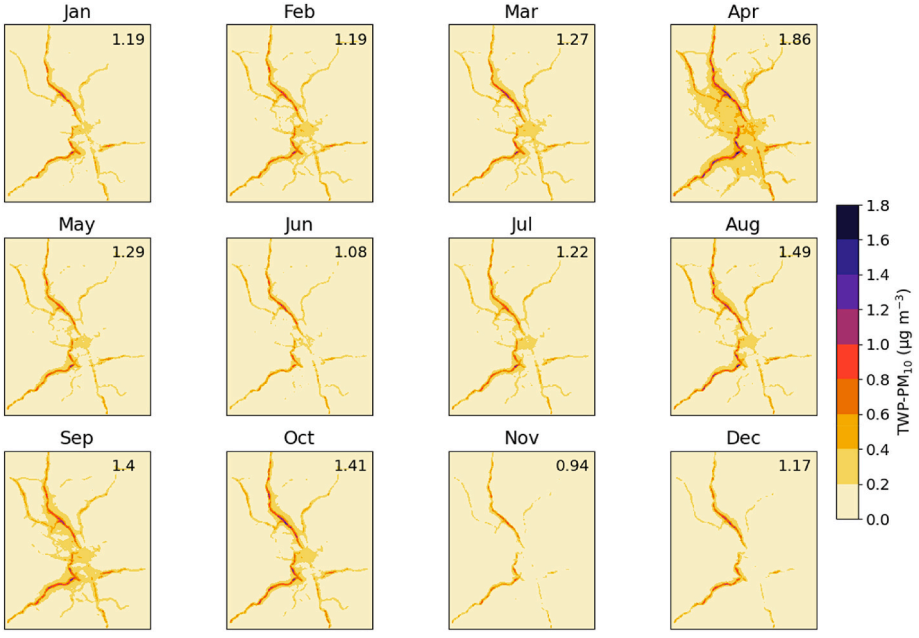

Fig. 4. Monthly mean concentration of TWP-PM_{10} calculated with the NORTRIP-Gauss model. The maximum grid cell value during each month is given in the upper right corner (in μg m^{-3}).

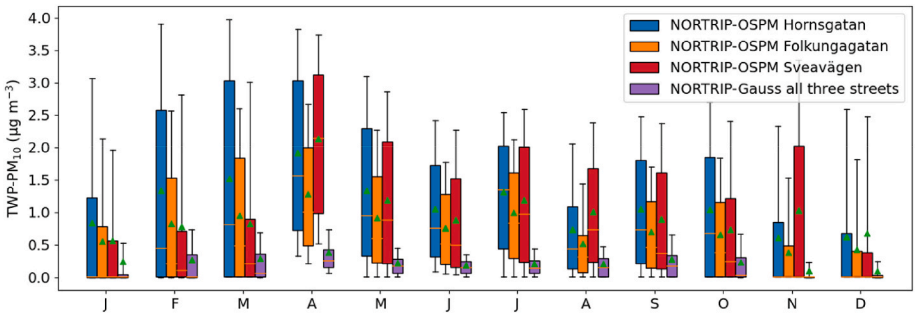

Fig. 5. Monthly statistics of TWP-PM_{10} concentrations from tyre wear at three streets in central Stockholm calculated with the OSPM model and the Gauss model. The statistics are based on the hourly average data each month. The extent of the boxes shows the interquartile range and the whiskers shows the 10-th and 90-th percentiles. The median is marked by the horizontal line and the mean by a triangle.

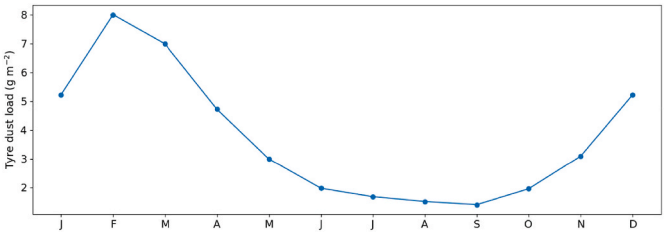

Fig. 6. Time series of road particle load from tyre wear at Hornsgatan calculated with the NORTRIP model.

important to quantify.

Fig. 7a shows the averaged diurnal variation of traffic and atmospheric concentration of TWP-PM_{10} calculated with the OSPM model. The concentrations are highest during daytime, with a peak at 16 local time, corresponding to the maximum traffic flow. The TWP-PM_{10} follows the traffic rhythm closely, indicating that the number of vehicles is an important parameter. Fig. 7b shows the averaged diurnal variation of TWP-PM_{10} during the four seasons. The atmospheric concentration is much higher in spring (MAM), than during the rest of the year, due to the increased resuspension of TWP after the winter season. Small differences in the shape of the diurnal variation exist between the seasons mainly due meteorological conditions.

3.3. Limitations and sensitivity study

There are several factors impacting the model results, which are examined by sensitivity simulations with the NORTRIP-OSPM model at Hornsgatan. Uncertainties in the model simulation arise from uncertainties in traffic data, wear rates, vehicle speed, tyre type and the year-to-year variation. The traffic data is based on measurements and extrapolated to all streets in the city. While there may be uncertainties, e.g. in the share of heavy traffic and day-to-day variations, the Airviro EDB is the best available traffic source covering all of Stockholm. The traffic speed for the multi-road version of NORTRIP covering all of Stockholm is generally based on speed limits on the respective roads and does not take overspeeding or congestion into account. In the street level simulation at Hornsgatan, actual traffic measurements are used and the average speed is 41 km h^{-1}. To test the importance of overspeeding, the maximum speed is set to 30 km h^{-1} in a sensitivity simulation, which is the maximal allowed speed.

To study the sensitivity to the wear rates, the wear rates for passenger cars are varied between 45 and 214 mg vkm^{-1}, based on literature estimates from the latest 20 years (Grigoratos et al., 2018; Lee et al., 2020; Luhana et al., 2004). For this specific case they are assumed to be the same for all tyre types. There are very little available wear rate estimates for light and heavy duty vehicles, and the heavy vehicles also differ considerably in weight and number of wheels. Therefore, the wear rate is set to vary within a range, with limits chosen at −20 to 20%. The new wear rates are thus 823–1235 mg vkm^{-1} for heavy vehicles. Light vehicles are a combination of passenger cars (45–214 mg vkm^{-1}) and light duty vehicles (108–318 mg vkm^{-1}), giving a new wear rateof 53–220 mg vkm^{-1}.

There are some factors that could affect the tyre wear rate that are not implemented in the NORTRIP model, e.g. the temperature of the road surface and the tyres and the driving style. To our knowledge, there are no available data sets on tyre and road temperature that can be used to tune a model. The driving style is partly covered by using urban wear rates, which take into account that there is on average more braking and acceleration in urban areas compared to in rural areas. It is possible to set the pavement characteristics of each road surface, but this would require data from all roads in the area, why no differentiation between different roads has been made. In the NORTRIP model it is assumed that 10% of the direct tyre wear is released as TWP-PM_{10}. The few studies that have measured this have shown varying results (Grigoratos and Martini, 2014).

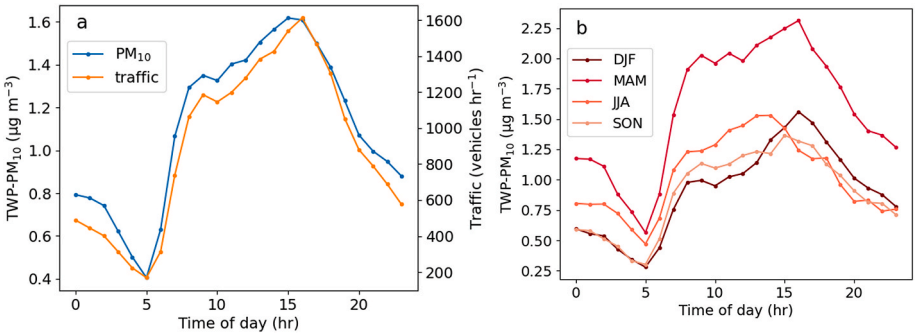

Fig. 7. a) Average diurnal variation of observed traffic and modelled TWP-PM_{10} concentration at Hornsgatan. b) Average modelled diurnal variation of TWP-PM_{10} concentration at Hornsgatan during the four seasons. DJF: December–February, MAM: March–May, JJA: June–August, SON: September–November.

The car fleet is right now undergoing changes. There is a general trend in many parts of the world of passenger cars becoming heavier, due to the popularity of SUV:s (IEA, 2019). In Sweden the total weight of the passenger car fleet has increased by 22% and the average weight by 7% over ten years (2010–2020), due mainly to an increased number of cars in the segment 1701–2000 kg (Trafikanalys, 2022). Also, the number of light duty vehicles is increasing as well as the share of electric passenger cars. While the total number of electric cars is still low in Sweden (2% in 2021), the number of vehicles sold increases fast, and will likely at some point be the dominating engine type. Electric cars are heavier than their combustion engine counterparts. According to Beddows and Harrison (2021) the average weight of electric vehicles is 21% higher than similar models of diesel and petrol cars. To study the future prospect that all passenger vehicles are electric, it is assumed that the weight of light vehicles increase by 21%, but that the heavy vehicle weight is the same. There is much work going on in electrification of heavy vehicles (Earl et al., 2018), but the weight of heavy vehicles is more dependent on the freight weight and therefore not so sensitive to vehicle weight. Since vehicle weight is not a variable in the NORTRIP model, the increase in weight has to be approximated by an increase in wear rate, and it is assumed that there is a linear relationship between vehicle weight and tyre wear emission (Timmers and Achten, 2016), giving new tyre wear rates for light vehicles of 127, 248 and 168 mg vkm^{-1} for summer, studless winter and studded tyres respectively.

The results of the sensitivity simulations are shown in Fig. 8. The uncertainty in wear rate has the largest impact on particle concentration, highlighting the necessity of improving the estimations of these. The speed decrease of 11 km h^{-1} results in a reduction of the concentration by 20%, highlighting also the importance of speed. However, the relationship between speed and emissions is not fully understood. In NORTRIP, a linear relationship is used, but other shapes or slopes could also be possible. The weight increase of passenger cars, assuming 100% electric light vehicles, increases the concentration by 13%. There are, however, many other aspects that affect the tyre wear emissions in future. The size distribution of the vehicle fleet may change, e.g. if people buy smaller electric cars to improve the range or if the share of SUV:s keep increasing. It is not only the vehicle weight that affects emissions, but the driving behaviour may also change, e.g. the possibility for faster acceleration with electric cars. On the other hand, manufacturers of tyres argue that tyres for electric cars are created to withstand wear better (Johannesson and Lithner, 2022). The improved traction control of electric vehicles may also decrease the wear during acceleration and braking (Johannesson and Lithner, 2022).

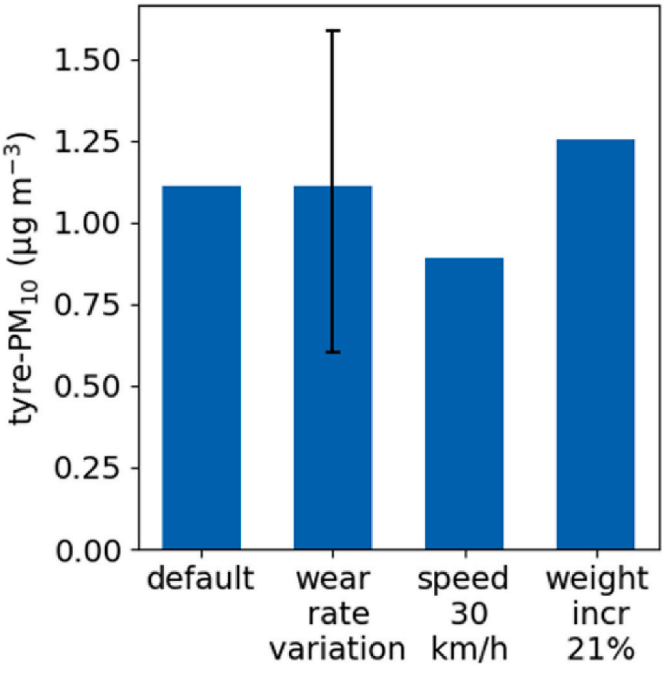

Fig. 8. Annual average TWP-PM_{10} concentration for four different scenarios at Hornsgatan. The error bar shows the maximum and minimum wear factor variation.

4. Discussion

Previous measurement studies have shown that microplastics are present in urban air, but these studies have only been snapshots in time and space and have often not included TWP. This model study shows that TWP-PM_{10} are ubiquitous in urban air, albeit in small numbers, during all seasons. Lack of concurrent real-world measurements of TWP prohibited an on-line comparison with observational data in Stockholm during the study period but our results are generally in agreement with the few published estimates of observed concentrations of TWP in the urban atmosphere. Although this study only focuses on one city, there are many other similar locations around the world. As car traffic is an important component of almost any urban area many people will be exposed to airborne TWP. Microplastics have been observed in human lungs (Jenner et al., 2022), showing that the airborne pathway is important. Microplastic particles may be harmful in themselves when inhaled, but also act as vectors for other substances (Prata et al., 2020).

As well as possibly being directly harmful to humans, the tyre wear emissions also pose a threat to the environment. As Stockholm is situated by the Baltic Sea coast and the governing wind direction is towards the sea, TWP will be transported by the winds and deposited over sea, where they can have negative effects on marine life. They can for example be ingested by fish (GESAMP, 2015) and deposited in sediments. Many cities are, like Stockholm, situated by water bodies, which gives a direct source of TWP into water. Moreover, sediments in urban areas are often stirred, e.g. during dredging to maintain sufficient water depth for shipping operations. Thus, particles in sediments may again be released into the water body.

This model study is a first approach to describe the extent of the problem with airborne microplastic pollution from tyre wear. To improve future model simulations, more knowledge on the emission rates and particle characteristics is needed. There is a large variation in the tyre wear rates in literature and especially a large uncertainty regarding the effect of speed, cornering, acceleration and braking. There is also less data on tyre wear from heavy vehicles, which is a group of vehicles with large variations in terms of size, load, number of tyres etc. In this study, urban wear rates are used throughout the domain. In future studies, wear rates could possibly be differentiated by road type or traffic situation to improve the estimations. The share of PM_{10} of tyre wear is set to 10% of total mass based on available estimations. The accuracy of this estimation will improve with increasing knowledge. Also, pavement characteristics impact the wear rate (Gustafsson et al., 2009).

More knowledge is also needed on the particle characteristics. Larger or heavier particles are likely deposited on or near the road and transported via runoff to end up in sediment and soils. Some are likely to be removed through road cleaning to end up in waste material deposits (Järlskog et al., 2020). Smaller particles in both air and runoff water tend to be transported longer distances. Only local TWP sources within the model area were included in this modelling study, excluding the long-distance transport. Due to the lack of measurements of finer TWP (<2.5 µm), there is very little knowledge on the long-range transport of these particles. Long-distance transport of microplastic particles has been reported, but focusing on larger particles (e.g. Allen et al., 2019; Bergmann et al., 2019; González-Pleiter et al., 2021) and seldom included tyre wear. While it has been found that TWP are generally encrusted with mineral particles (Rausch et al., 2022; Sommer et al., 2018), increasing their density, finer TWP emitted directly to the air may possibly be purer and thus lighter and more easily transported

longer distances than those found on streets, roads and in stormwater. To increase the understanding of this dispersion of particles, more measurements of particle properties are needed, especially focusing on the smaller particles. Measurement methods for particle identification also need to be improved, reducing the manual work and increasing the number of particles that can be analysed in situ. With more data on TWP characteristics, more details can be introduced into models.

This study has shown that TWP-PM_{10} emissions are of the same magnitude as exhaust emissions, but while the particle exhaust emissions are continuously decreasing throughout Europe (AQEG, 2019), tyre wear emissions are likely to increase with increasing traffic, and possibly with heavier vehicles as discussed above, increasing both their absolute and relative importance. Therefore, it may still be important for urban planners to consider air pollution in areas with heavy car traffic, even with a fully electrified car fleet. Areas with low ventilation or walking and cycling paths next to busy roads are environments with high risk of exposure. This is extra important to consider since there is a general densification of many urban areas today.

5. Conclusions

This study presents simulations of atmospheric TWP-PM_{10} concentration in an urban area. Roof level concentrations are calculated in a domain covering the Stockholm urban area, complemented with street level calculations in three selected street canyons. The results show that TWP are ubiquitous in urban areas, although in much higher concentrations along busy highways and in poorly ventilated street canyons. Highest daily and hourly concentrations occur in winter due to stable conditions while highest monthly concentrations occur in April due to resuspension of road dust.

The airborne microplastics pose a potential health risk and can also be transported to the surrounding landscape and to the Baltic Sea. As traffic is projected to increase in future, the TWP emissions will also increase. The possible increase in weight of the vehicle fleet due to electrification and SUV:s may add to increasing TWP emissions.

CRediT authorship contribution statement

Nina Svensson: Conceptualization, Formal analysis, Investigation, Writing – original draft, Writing – review & editing, Visualization. **Magnuz Engardt:** Methodology, Investigation, Writing – review & editing. **Mats Gustafsson:** Conceptualization, Writing – review & editing. **Yvonne Andersson-Sköld:** Conceptualization, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

The authors do not have permission to share data.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.aeaoa.2023.100225.

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