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Health and economic assessment of ultrafine particles in Stockholm: Impacts of electrification and local policies

Lina Broman^a, Magnuz Engardt^{a,b}, Sanna Silvergren^a, Hedi Kriit^{c,e}, Michael Norman^a, Christer Johansson^{a,d,*}

^a Environment and Health Administration, SLB-analys, Box 8136, 104 20 Stockholm, Sweden

^b Present address: Swedish Meteorological and Hydrological Institute, 601 76 Norrköping, Sweden

^c Department of Public Health and Clinical Medicine, Umeå University, 901 87 Umeå, Sweden

^d Department of Environmental Science, Stockholm University, 106 91 Stockholm, Sweden

^e Heidelberg Institute of Global Health, Medical Faculty and University Hospital, Heidelberg University, Heidelberg 69210, Germany

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ABSTRACT

High resolution air quality dispersion modelling was applied to assess the health risks of long-term exposure to ultrafine particles (UFP) emitted from the transport sector and to evaluate the health benefits of different local interventions in Stockholm. Health impacts, based on associations between exposure, premature mortality and various diseases, were quantified as quality adjusted life years (QALYs) and monetized as health-related cost-irrespective of local mitigation measures in the city, fleet renewal driven by vehicle emission regulations is projected to reduce UFP exposure by 86 % between 2021 and 2030, saving almost 1700 QALYs corresponding to approximately €500 M in health-related costs. The renewal of the fleet affects mainly diesel passenger cars and heavy goods diesel vehicles equipped with particle filters, which efficiently remove most exhaust particles, including UFPs. Among the local mitigation policies, the introduction of a low emission zone (LEZ) in 2030 covering 16 km² of the inner city is estimated to save around 40 QALYs. In contrast, a much smaller LEZ (0.2 km²) approved by the city council would yield minimal health benefits, saving only 0.1 QALYs. Electrification of road vehicles under the business as usual (BAU) scenario is projected to save around 70 QALYs in 2030, exceeding the gains of any local policy scenario. Comparisons using PM_{2.5} and NO₂ as predictors of health-related emissions, shows that gains in adverse health impacts of local mitigation actions may be severely underestimated if only health impacts of UFP exposure is considered.

1. Introduction

1.1. Global burden of disease attributable to air pollution

Air pollution is a major global public health threat and remains one of the primary risk factors for premature mortality. According to the Global Burden of Disease (GBD) 2019 assessment, exposure to ambient levels of fine particulate matter (PM_{2.5}) alone accounted for approximately 4.2 million premature deaths in 2019, primarily from cardiovascular and respiratory diseases (GBD, 2021). This health burden is disproportionately concentrated in urban areas, where high population densities coincide with elevated emission levels. As a result, urban residents experience significantly higher PM_{2.5} concentrations compared to rural populations (Brauer et al., 2012; Lelieveld et al., 2015). As of 2018,

55 % of the world's population lived in urban areas, a share projected to increase to 68 % by 2050 (UN DESA, 2018; UN-Habitat, 2022). The ongoing shift towards urban living exposes billions of people to higher levels of air pollution, underlining the need for targeted mitigation strategies in cities. In Sweden, the urbanization has been ongoing for a long time, resulting in the majority of the population (88 % as of 2023) living in urban areas (Statistics Sweden, 2024). The concentration of PM_{2.5} in Sweden has decreased by 68 % from 1990 to 2023, mainly due to reduced industrial emissions, increased use of district heating, more efficient woodburning boilers, and lower emissions from the transport sector (SEPA, 2025). Despite this significant reduction in PM_{2.5} levels, a health impact assessment for 2019, based on high-resolution dispersion modelling together with gridded population data and exposure-response functions, estimated that exposure to PM_{2.5} from regional

* Corresponding author at: Department of Environmental Science, Stockholm University, 106 91 Stockholm, Sweden.

E-mail address: christer.johansson@aces.su.se (C. Johansson).

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background levels and local hotspots was attributable to 4262 premature deaths annually (Wylde et al., 2023).

1.2. Urban air pollution and the role of transportation

In densely populated areas, emissions from transport, residential heating, industry and energy production accumulate, resulting in persistently high levels of air pollution. Among these sources, transport is a major contributor to PM_{2.5} concentrations, through both tailpipe and non-exhaust sources such as brake wear, tire wear, and road dust resuspension (Amato et al., 2014; Segersson et al., 2017). In Sweden, long-range transport remains the major source of PM_{2.5}, followed by emissions from residential wood combustion. In the country's two largest cities, Stockholm and Gothenburg, transport constitutes the third-largest source of population exposure to PM_{2.5} (Segersson et al., 2017). PM_{2.5} has for long served as the primary indicator of exposure in global assessments. However, several studies suggest that relying solely on PM_{2.5} as an exposure metric may underestimate the health risks associated with other particle size fractions and pollution emission sources, particularly ultrafine particles (UFPs, defined as particles less than 100 nm in diameter) which are not adequately represented by PM_{2.5} mass concentration as they only contribute to a fraction of the total particle mass (Lanzinger et al., 2016; Weichenthal et al., 2017).

1.3. Ultrafine particles (UFPs): sources and characteristics

UFP are emitted from both natural and anthropogenic primary sources, and can also be formed secondarily from gaseous precursors of natural and anthropogenic origin (Kumar et al., 2014; Rönkkö and Timonen, 2019; Ehn et al., 2014; Patel et al., 2024). While the larger particles size fractions PM_{2.5} and PM₁₀ are regulated in terms of particle mass per volume of air ($\mu\text{g m}^{-3}$), UFPs, due to their negligible mass, are typically characterized by their particle number concentration (PNC) per unit volume (particles/cm³). Previous studies have found that UFPs account for more than 80 % of the total PNC (Trechera et al., 2023). Consequently, the terms UFP and PNC are often used interchangeably, as is also done in this study. Whereas annual concentrations of PM_{2.5} tend to be uniformly distributed across cities, largely due to the importance of long-range transport, UFPs measured as total PNC, show large spatial variability because of the dominating influence of local sources within urban areas (Johansson et al., 2007; Puustinen et al., 2007; Saha et al., 2020; Clark et al., 2024). The transport sector includes significant sources of UFPs such as road traffic, ships and aviation (Kukkonen et al., 2016; Li et al., 2023; Cassee et al., 2019; Hopke et al., 2022; Viana et al., 2014; Alzahrani et al., 2024; Qin-Qin Li et al., 2023; Rivas et al., 2020). In most cities, diesel vehicle exhaust emission is the main contributing source of UFP (Li et al., 2023; Garcia-Marlès et al., 2024). Other sources are residential wood burning (Corsini et al., 2019) and secondary particle formation (Li et al., 2023; Hopke et al., 2022; Qin-Qin Li et al., 2023).

1.4. Health impacts of ultrafine particles

A growing body of evidence from both toxicology and epidemiology shows that UFPs pose significant health risks (Vallabani et al., 2023). UFPs enter the human body through the lungs. Due to their small size, they easily penetrate deep into the distal regions of the respiratory tract and deposit across the extensive alveolar surface area. A fraction of UFPs are further capable of crossing epithelial barriers and enter the bloodstream, where they may be transported to other organs (Leikauf et al., 2020; Miller et al., 2017; Terzano et al., 2010). Exposure to ambient UFPs has been linked to a wide range of health issues, including issues related to the respiratory system (Peters et al., 1997), the cardiovascular system (Schraufnagel, 2020), brain disorders (Maher et al., 2016; Garciduenas and Ayala, 2022), diabetes (Bai et al., 2018) and associations between elevated UFP concentrations and overall hospitalizations for

high-burden diseases and interactions to extreme cold (Qi et al., 2024). There is still limited toxicological and epidemiological data on the health effects of individual sources (Vallabani et al., 2023).

Although the risk of developing health issues increases with the level of exposure to UFPs, there are today no legally defined exposure limits or specific Air Quality Guidelines for UFPs. However, the World Health Organization (WHO) has provided Good Practice Statements for UFPs to guide authorities towards measures to reduce UFPs based on new research findings. In short, the guidelines recommend to (1) Quantify ambient levels of PNC in terms of number concentration from < 10 nm, (2) Integrate UFP monitoring into the existing air quality monitoring, (3) Distinguish between low and high PNC levels, specifically low (< 1000 particles/cm³, 24-hour mean) and high (> 10,000 particles/cm³, 24-hour mean or 20,000 particles/cm³, 1-hour mean), and (4) Utilize emerging science and technology to improve methods for assessing of exposure to UFP for application in epidemiological studies and UFP management. The reader is referred to WHO global air quality guidelines for further details (World Health Organization, 2021).

1.5. Health and economic assessment of UFP exposure

There are several studies regarding the effects on exposure and health of policy interventions to reduce NO₂, PM_{2.5} and PM₁₀. (e.g., Segersson et al., 2017, 2021; Johansson et al., 2009; Orru et al., 2015; Kriit et al., 2021; Burns et al., 2020; Khreis et al., 2023), but limited studies of the impact of interventions on UFP exposure and health (Khreis et al., 2023; Patel et al., 2023). We have not found any study comparing health effects associated with emissions of UFP from different types of transport and furthermore comparing with effects of PM_{2.5} and NO₂ emissions using a consistent exposure modelling framework and considering risks associated with long-term exposure of several common health outcomes and costs.

The large spatial variability of UFPs, as well as many other factors, has contributed to uncertainties in exposure quantification and to inconsistencies in assessing human health risks in epidemiological studies (Cassee et al., 2019). While for PM_{2.5} several epidemiological studies have relied on central monitoring data as representative for contrasts in population exposures between cities (Segersson et al., 2021), assessment of UFP exposure cannot rely on a single central monitoring station. To properly account for the large spatial variability in long-term exposure of UFP, high resolution modelling at scales below 1 km is required.

1.6. Knowledge gaps and objectives of the study

While there are a number of publications assessing health and economic impacts of policies targeting PM_{2.5}, PM₁₀ and NO₂, studies addressing UFPs remain scarce. Existing work has either focused on source apportionment, health impacts of single sources and short-term exposures, or local monitoring evaluation, but to date no study has systematically compared the health and economic consequences of long-term population exposure across multiple transport sources within a consistent high-resolution modelling framework. Furthermore, previous assessments rarely benchmark UFP-related impacts against those of PM_{2.5} and NO₂, limiting the ability to understand whether interventions targeting UFP deliver unique or additional benefits beyond established pollutants. The design of optimal policies remains constrained by the limited availability of research that simultaneously incorporates detailed spatial heterogeneity in UFP exposure, associated long-term health risks, and their economic valuation.

In this study, we address the knowledge gaps by firstly quantifying the source contribution of UFP long-term/annual exposure using high resolution modelling in Stockholm. Secondly, we estimate the adverse health and economic impacts attributable to annual UFP exposure. Thirdly, we assess the potential health and economic gains of different policy interventions aiming to reduce the annual UFP exposure, including low-emission zones and electrifications. The economic

valuation of health impacts is an important contribution for policy makers to be able to include and compare the effects of different interventions across sectors (e.g. transport, health etc.) using the same metric.

We make use of recently published epidemiological studies associating long-term UFP exposure with different health outcomes to quantify effects of UFP exposures on both mortality and morbidity in terms of quality-adjusted life years (QALYs) gained and their corresponding monetary values. Additionally, we compare these health impacts with those related to exposure to PM_{2.5} or NO₂. The study aims to provide policymakers with guidance on prioritizing the most effective strategies to mitigate the health risks posed by transport emissions.

2. Methodology

The health impacts and associated costs of nanoparticle emissions from transportation sources are assessed following the impact pathway approach (Silveira et al., 2016). The methodology utilizes the same models and underlying activity data as previously applied in Stockholm to evaluate the health benefits of congestion tax (Johansson et al., 2009), increased bicycle commuting (Johansson et al., 2017), and assessing the health and economic impacts of a fossil fuel-free fleet in Stockholm (Kriit et al., 2021). The modelled UFP concentrations, along with corresponding exposure estimates, represent total particle number concentrations > 4 nm. We used ArcGIS Pro and Python to conduct the analyses and to produce the figures.

2.1. Emissions

To assess effect of transport-related mitigation measures, we include the following transport source categories

- road traffic exhaust
- road traffic non-exhaust (brake wear, clutch wear, tyre and road wear)
- railway, including above-ground sections of the metro
- shipping
- aviation

In addition, emissions from residential heating and contributions from non-local sources are included to enable evaluation of model performance through comparison with measured total particle number concentrations.

Emission factors for road vehicles are based on the Handbook Emission Factors (HBEFA) version 4.2 (Notter et al., 2022). The HBEFA database specifies non-volatile particle number exhaust emission factors for various vehicle categories, speeds, driving situations, and EURO classes, with a lower cut-off at 23 nm. The Stockholm road network is represented in high spatial detail. Traffic volumes, including heavy traffic, fleet composition, and temporal variations are based on municipal data updated yearly. Vehicle fleet composition is derived from the national registry and categorized into passenger cars (petrol, diesel, ethanol and gas and electric), light commercial vehicles (petrol, diesel, electric), heavy good vehicles (petrol, diesel, electric), buses (diesel, gas, electric). Emission factors for each vehicle type are calculated as weighted averages based on vehicle kilometers travelled per Euro class. UFP emissions from tyre and road wear, brake wear and clutches are included as described by Chen et al. (2024a).

Railway emissions are based on network links (i.e. track segments) containing data on average weekday train frequency by train type, using 2022 data. Emissions from tunnels are excluded. UFP emissions from shipping are based on the Shipair 2 model (van Dongen et al., 2023), developed by the Swedish Meteorological and Hydrological Institute and integrated in the Airviro Air Quality Management system. Shipair 2 uses Automatic Identification System (AIS) data to determine the ship location and identity, which are used with ship specifications to estimate fuel consumption. PNC emission factors are based on the Horizon 2020 project SCIPPER (Grigoriadis et al., 2021). A comprehensive description

of all emission factors and activity data for all transport sources is provided in Chen et al. (2024a).

2.2. Dispersion modeling

To quantify the spatial distribution of annual average particle number (PN) concentration across Stockholm we apply the Airviro Gaussian dispersion model, complemented with the Operational Street Pollution Model (OSPM) (both models are components of the Airviro Air Quality Management system, <https://www.airviro.com/airviro/>). The dispersion models are forced by meteorological data from a meteorological mast located in southern Stockholm. The results consist of high-resolution maps of annual mean UFP concentrations in Stockholm.

Nanoparticles are treated as if they were inert, i.e., we do not consider particle dynamics or deposition processes. The primary uncertainty introduced by this assumption (for annual mean exposure estimates) is that we neglect dry deposition of particles as noted by Gidhagen et al. (2005). However, when evaluating the relative differences in exposure (e.g. with and without a specific emission category), the effect of dry deposition will be minor.

We also neglect potential changes in UFP exposure resulting from nucleation because of the change in local sources emissions associated with a particular policy intervention. Changes in UFP exposure for populations outside the model domain are not included. This includes both changes in emissions outside the domain, transported particles emitted inside the domain and secondary particles formation driven by changes in precursor emissions. Hence, the health effects of policies may be underestimated.

2.2.1. Meteorology and wind model

Meteorological data are obtained from observed wind and temperature profiles. To ensure that the calculated UFP concentrations and the effects of the various policy interventions are not biased by specific weather conditions, all simulations use a representative meteorological year. The same climatological dataset is applied in all policy scenarios. Climatological data spanning a multi-year period (1998–2019) are used as input to the diagnostic wind model developed by Danard (1977). The annual average UFP concentrations are obtained from 360 hourly mean values as part of the climatology using 15 years of hourly mean air temperature, vertical temperature gradient, standard deviation of the horizontal wind direction and of the vertical wind velocity, the solar radiation and the precipitation. Meteorological parameters are collected from a 50-meter-high mast located in Högdalen, southern Stockholm. The wind model generates a locally adjusted wind field across the study domain, accounting for local topography, surface roughness, and vertical heat transfers. The data is classified according to 60 wind directions (sectors). All events falling into a specific sector will be classified according to the atmospheric stability conditions, i.e. 6 intervals of Monin-Obukhov lengths. All data are then sorted, frequencies of all classes are estimated and the median values (of the Monin-Obukhov lengths) of each class (in this case 360 classes) are identified, including the specific date/hour when each class example occurred. The emission data base provides the emissions during the specific hour that correspond to the meteorology. The temporal variations of emissions are based on typical operating conditions or, in the case of vehicle traffic, based on historic measurements on different road types and vehicle types. Based on the meteorological conditions and emissions the concentrations can then be obtained using the Gaussian dispersion model. The average of the 360 cases represents an annual average. The consistency in calculating annual averages estimated using climatological weather versus using hourly time series calculations has been demonstrated previously by Segersson et al. (2017).

2.2.2. Grid configuration

The Airviro Gaussian model is used to calculate the horizontal distribution of air pollution concentrations 2 m above ground level. In

densely build-up areas, the calculations represent concentrations 2 m above roof level.

Emission sources from different sectors are represented using slightly different grid configurations. The model employs a variable resolution grid (quad grid), with resolution determined by the emission-strength of road and railway sources. For UFP emissions from road traffic the grid sizes range from 30×30 m to 500×500 m, with finer resolution in areas of higher emissions. For railway traffic, the grid sizes range from 10×10 m to 500×500 m. The grid configurations for road traffic and railway traffic are shown in Fig. 1 (left and top middle, and right and bottom right respectively). For shipping, aviation and residential heating emissions, a uniform grid of 100x100 meters is used.

2.2.3. Ospm model

The conditions for mixing and diluting air pollutants vary across different street canyons. Wide streets devoid of buildings can accommodate significantly higher exhaust emissions without resulting in unacceptably high concentrations, compared to narrow streets flanked by tall buildings. The layout and dimensions of the street canyon, as well as whether it is open or enclosed, play a significant role in street ventilation and concentration levels. To calculate concentrations of air pollutants near ground level or on the street in densely built-up areas, we use the street canyon model OSPM (Berkowicz, 2000) integrated into the Airviro system. This model has been previously tested (Kakosimos et al., 2010) and found to accurately model NOx concentrations in street canyons, exemplified by Hornsgatan in Stockholm in Ottosen et al. (2015). In this study, the OSPM model is employed to calculate particle number concentrations in a double-sided street canyon for the model evaluation only, not for population exposure concentrations.

2.2.4. Validation of modeled UFP concentrations by comparison to observations

The modelled UFP concentrations were evaluated against observations from fixed monitoring stations in Stockholm, covering both street-level and urban background locations. At the street canyon site Sveavägen 59 (operational from 2020), total particle number concentrations (PN4) were measured using a condensation particle counter (CPC, TSI model 3775, USA) with a 50 % cut-off diameter at 4 nm. The urban background site is located on a rooftop at Torkel Knutssongatan in central Stockholm. Measurements used for model evaluation cover the period 2020–2023 and were carried out using a Scanning Mobility Particle Sizer (SMPS) and a CPC. Model evaluation also included the street canyon site Hornsgatan 108, where PN measurements were active between 2002 and 2020. CPC instruments with lower cut-off sizes (7 nm in 2006, 4 nm in 2013) were used. In this analysis we used data from the period 2018–2020. After a measurement interruption in 2020, Hornsgatan station was replaced by the Sveavägen site. In addition, we have used data from the regional background station (RB) situated about 90

km north of Stockholm (Norunda station) operated by Stockholm University and the Swedish Environmental Research Institute. At the RB-site a Differential Mobility Particle Sizer (DMPS) with particle range (10–410 nm) with CPC is used. Together, these stations provide a very unique long term observational dataset. For street canyon geometries, the reader is referred to Krecl et al (2011).

We evaluated the modelled annual particle number concentrations (PNC) at the urban background site Torkel Knutssongatan and the street canyon sites Sveavägen and Hornsgatan. At the urban background site, the model initially underestimated concentrations by approximately 44 %, suggesting that certain sources may be missing or underestimated in the emission database (EDB). Potential contributors not fully captured may include natural emissions, stationary combustion sources, off-road machinery, or heating plants. For the street canyon sites Sveavägen and Hornsgatan, the model initially underestimates the PN concentrations with 14 % and 31 % respectively. To adjust the model, we correct for the missing sources in the modelled UB by adjusting the modelled UB by a factor 1.9. With the model adjusted using a correction factor, the discrepancy between model and observations decreased to –14 % (underestimation), +15 %, and –5% (underestimation) for Torkel Knutssongatan, Sveavägen, and Hornsgatan, respectively. The reader is referred to Chen et al. (2014a) for additional details.

It should be noted that this study focuses on the change in annual concentrations and population exposure due to the change in emissions with different policies, i.e. they relate to a baseline situation without the policy compared to a scenario with the policy. In this case, calculation errors are not due to different meteorological conditions. Furthermore, emissions other than local emissions do not influence the conclusions. The main uncertainty in this methodology lies in the estimated change in the transport emissions related to the policy.

2.3. Population exposure assessment

Population-weighted exposure was derived by overlaying annual pollutant concentrations with the residential population distribution. Residential population distributions were obtained from Statistics Sweden, with data mapped at a 100 x 100-meter grid resolution for the Stockholm region.

Population weighted concentration, C_{Pop} , is calculated as:

$$C_{Pop} = \frac{\sum C_i P_i}{\sum P_i} \quad (1)$$

where C_i and P_i are concentrations and population in each calculation grid cell in the area studied. Population weighted concentrations are calculated for each individual source as well as for all transport sources.

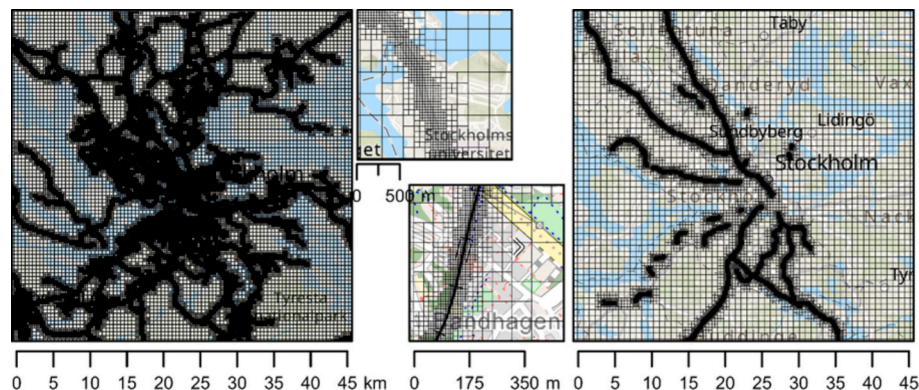


Fig. 1. The model domain and quad grid applied for the dispersion modelling of road traffic (left) and railroad traffic (right) in Stockholm. The middle figures display a zoomed in version of the quad grid (35 m x 35 m) close to major road (top) and the quad tree (10 m x 10 m) of railway (bottom).

2.4. Health impact estimation

2.4.1. Methodology

The number of premature deaths or the number of people affected by various diseases is estimated based on population-weighted exposure concentrations and relative risks of mortality and morbidity (Martenies et al., 2015):

$$Y = Y_0(1 - e^{(-\beta C_{pop})}) \quad (2)$$

where ΔY = is the change in the number of cases (e.g. mortality per 100,000 inhabitants), Y_0 is the baseline frequency, β is the dose–response function and ΔC_{pop} is the change in population-weighted concentration associated with some emission or change in emission for some abatement scenario. The dose–response function is here taken to be the natural logarithm of the hazard ratio (or relative risk, RR).

Health costs associated with mortality and morbidity are estimated according to valuations described by Kriit et al. (2021). These valuations are based on the Swedish ASEK report which presents the rationale, principle and recommended values for use in social cost-benefit analyses (CBA) by the Swedish Transport Administration (Söderqvist et al., 2019).

2.4.2. Relative risks

Relative risk (RR) estimates for various health outcomes, such as premature mortality, type 2 diabetes, myocardial infarction, and chronic obstructive pulmonary disease (COPD), were drawn from epidemiological studies. As shown in the review of Vallabani et al. (2023) there is very limited epidemiologic evidence on the health impacts of UFP exposure associated with different transport related sources, but some evidence related to total UFP exposure. An updated literature survey since Vallabani et al. (2023) was made to identify epidemiological studies associating health effects with long-term exposures. The RRs for each health outcome were assumed to be the same irrespective of source of exposure.

We have identified four epidemiological studies that have considered associations between mortality and long-term particle number exposure (Pond et al., 2022; Bouma et al., 2023; Lloyd et al., 2024; Qi et al., 2024). Bouma et al. (2023) and Lloyd et al. (2024) base the health effect estimates on high resolution exposure modelling using land-use regression in combination with measurements and other modelling. Lloyd et al. (2024) examined non-accidental and cause-specific mortality in a cohort of approximately 1.5 million adults residing in Toronto and Montreal. An increase in long-term exposure to outdoor ultrafine particles of 10,000 particles cm^{-3} was associated with an increased risk of non-accidental mortality (hazard ratio, $\text{HR}^1 = 1.073$; 95 % confidence interval (CI): 1.061, 1.085). They observed a possible confounding by particle size and suggested that previous studies have not considered this and may have underestimated or missed important health risks associated with ultrafine particles. Qi et al. (2024) analyzed mortality data from New York State Vital Records from 2013 to 2020 (883,725 deaths) and found significant associations between UFP exposure and total non-accidental mortality ($\text{RR} = 1.10$ per 834 particles cm^{-3} ; 95 % CI: 1.05, 1.17). Daily exposure from 2013 to 2020 was obtained from a chemical transport model with aerosol microphysics, controlling for multi-pollutant effects (O_3 , SO_2 and NH_3) and other factors.

The HR of Bouma et al. (2023) (1.012 per 2723 particles cm^{-3}) for all cause natural mortality was based on a national cohort (The Netherlands) of 10.8 million adults aged ≥ 30 years, followed from 2013 until 2019. Land-use regression based on mobile measurements was

used for exposure calculations. The HR remained significant in two-pollutant models. Also Pond et al. (2022) used land-use regression based on mobile measurements and fixed monitors to analyze associations between mortality and long-term UFP exposure in the USA (617,997 adults). In a single-pollutant model UFP was found to be significantly associated with all-cause mortality with a HR of 1.03 per 5005 particles cm^{-3} . The association was robust to controlling for PM_{10} , $\text{PM}_{2.5}$ and SO_2 , but not $\text{PM}_{2.5}$, which could partly be due to fact that they aggregate exposure to census tract level as discussed by Lloyd et al. (2024). Based on the studies by Pond et al. (2022), Bouma et al. (2023), Lloyd et al. (2024) and Qi et al. (2024) a pooled random effect on all-cause mortality was calculated to be 1.0058 per 1000 particles cm^{-3} (95 % CI: 1.0039, 1.0076) (Table 1).

Three studies have associated UFP with type 2 diabetes (Bai et al 2018; Sorensen et al., 2022; Yang et al., 2025). The study population of Bai et al. (2018) included all Canadian-born residents aged 30–100 years who lived in the City of Toronto, Canada, from 1996 to 2012. Sorensen et al (2022) analyzed all persons living in Denmark for the period 2005–2017; in total, 2.6 million persons aged > 35 years. Yang et al. (2025) studied the impact of long-term UFP exposure on diabetes prevalence using data from a multi-center cohort (from 2017 to 2021). The study included 93,990 participants, with a diabetes prevalence of 10.97 %. The studies found significant associations between UFP and type 2 diabetes. They report HR of 1.06 (95 % CI: 1.05, 1.08 per 9948 particles cm^{-3}) (Bai et al) and 1.052 (95 % CI: 1.042, 1.063 per 4248 particles cm^{-3}) (Sorensen et al., 2022), and Yang et al. (2025) report OR of 1.20 (95 % CI: 1.14, 1.26 per 6326 particles cm^{-3} overall IQR). Downward et al., (2018) analyzing a cohort in The Netherlands and Bai et al (2019) analyzing a cohort in Tonto (Canada) report significant associations between long term UFP exposure and myocardial infarction with HR of 1.59 (95 % CI: 1.13, 2.24 per 10,000 particles cm^{-3}) and 1.05 (95 % CI: 1.02, 1.07 per 10,029 particles cm^{-3}), respectively. Poulsen et al. (2023) report significant associations between running 5-year mean residential UFP concentrations and stroke for all persons aged > 50 years and living in Denmark in the period 2005–17 with HR 1.039 (1.026, 1.052 per 4248 particles cm^{-3}).

For the same cohort in Canada as used by Bai et al. (2019), Weichenthal et al. (2017) found significant associations between UFP

Table 1

Relative risks (RR) used for estimates of mortality and morbidity. Pooled random effects were estimated according to Neyeloff et al. (2012).

Health outcome	RR calculated per 1000 particles cm^{-3} a (95 % CI)	References
Mortality associated with long-term exposure	1.0059(1.0035, 1.0084)	Pooled random effects based on Bouma et al. (2023), Pond et al. (2022), Lloyd et al. (2024) and Qi et al. (2024)
Diabetes type 2	1.014(1.007, 1.022)	Pooled random effects based on Bai et al. (2018), Sorensen et al. (2022) and Yang et al. (2025)
Myocardial infarction	1.022 (0.98, 1.064)	Pooled random effects based on Downward et al. (2018) and Bai et al. (2019)
Stroke	1.0090(1.0061, 1.012)	Poulsen et al., 2023
Lung cancer	1.014(1.010, 1.017)	Bouma et al. (2023)
COPD	1.0058 (1.0039, 1.0086)	Weichenthal et al. (2017)
Childhood asthma	1.0027(1.0000, 1.0054)	Lavigne et al (2019)

^aOriginal reported RR_{orig} per N number of particles cm^{-3} was converted to RR_{1000} per 1000 particles cm^{-3} by $\text{RR}_{1000} = \text{RR}_{\text{orig}}^{(1000/N)}$. Health risk estimates in individual studies expressed as relative risks (RRs) or hazard ratios (HRs) were considered numerically equivalent to RRs in line with the “rare disease assumption” (Greenland and Thomas, 1982).

¹ Health risk estimates in individual studies expressed as relative risks (RRs), hazard ratios (HRs) or percentage increase in the risk were considered numerically equivalent to RRs in line with the “rare disease assumption” (Greenland and Thomas, 1982).

exposure and COPD. The HR was 1.06 (95 % CI: 1.04, 1.09 per 10,097 particles cm^{-3}).

Lavigne et al. (2019) report significant associations (1.03; 95 % CI: 1.00, 1.06 per 10,862 particles cm^{-3}) and childhood asthma (up 6 years old) when analyzing 160 641 singleton live births in 2006–2012 in Canada.

No studies have been found associating UFP exposure with the other health outcomes included in the impact assessment by Kriit et al. (2021), i.e. adult asthma, preterm birth, dementia, and restricted activity days. For lung cancer, Bouma et al. (2023) report significant associations with UFP exposure based on a national cohort in The Netherlands, but since we have not found any QALY value it is not included in our calculations.

2.4.3. Quality-adjusted life years (QALYs) and costs

We calculated QALYs lost due to the health outcome considered as described in Kriit et al. (2021). Shortly, a literature review was conducted to retrieve outcome specific QALY weights. The number of lost QALYs per outcome (Table 2) was the difference between average population QALY weight times the duration of the specific outcome. The total number of QALYs lost for outcome (o) associated with the number of cases, ΔY (from equation (2)), is calculated as:

$$QALY_o = \Delta Y \cdot B_o \cdot Q_o \cdot F_o \cdot P \quad (3)$$

where, for each outcome, o , B_o is the baseline number of cases in the population, Q_o is the number of QALYs lost per case, F_o is the fraction of the population affected and P is the total population.

The total health cost is the sum of the values of the QALYs and health direct costs. The value of a QALY is taken to be €300,000 based on a willingness-to-pay study including data collected from 800 individuals in the Swedish adult general population (Olofsson et al., 2019). The direct health costs for each morbidity outcome are taken from Kriit et al. (2021), calculated for 2024, and presented in (Table 3).

A Monte-Carlo analysis was undertaken to estimate the uncertainty in the total number of QALYs and total costs associated with the uncertainties in the RRs of the health outcomes. Assuming that the Log (RR)'s of all outcomes follows a normal distribution, a 95 % CI of QALYs and costs were calculated based on 10,000 estimates. For UFP exposure levels in the scenarios examined here the 95 % CI of total number of QALYs and costs are within around 30 % of the mean value.

2.5. Description of scenarios

Health and economic outcomes were compared across local policy scenarios, quantifying population exposures and QALYs gained relative to a baseline scenario (BAU) without the respectively policy. An overview of the five scenarios analyzed is provided in Table 4. All policy scenarios were evaluated as if they were implemented in four different years, 2030, 2040 and 2050. The scenarios were identified based on planned policies in Stockholm and input gathered from a user group

Table 2

Basic data for calculating QALYs for diseases and number of QALYs lost.

Health outcome	Years of life lost/ duration of disease	Baseline (cases per 100 000 population)	Number of QALY lost	Fraction of population affected ^a
Mortality	11	915	9.7	0.64
Myocardial infarction	1	246	0.09	0.64
Stroke	10	253	2.85	0.38
Diabetes II	9	400	1.5	0.82
COPD	19	157	5.51	0.38
Childhood asthma	17	1800	1.7	0.012

^a Based on the age distribution of the Swedish population (and health outcome/epidemiological study).

Table 3

Direct health costs (rounded numbers for 2024 from Kriit et al., 2021).

Health outcome	Direct health cost (Euro)
Myocardial infarction	26,000
Diabetes II	61,000
Stroke	61,000
COPD	14,000
Childhood asthma	18,000

Table 4

Description of policy scenarios for the city of Stockholm.

Policy scenario	Transport mode	Description
Low emissionzone (small area)	Road vehicles	Implementation of the designated environmental zone Class 3 planned for 2025. Area covered: 0.2 km ² (350 m x 600 m).
Low emissionzone (extended area)	Road vehicles	A hypothetical low emission zone applying the same regulations as those of Class 3, but extended to cover the same area as the congestion tax zone. Area covered: 16 km ² .
Electrification of ports	Maritime transport	Estimated effect of the ongoing port electrification in Stockholm, reaching 100 % compared to a scenario without electrification.
Measures to reduce traffic demand	Road vehicles	Effect of a 30 % reduction in traffic volumes by 2030 compared to the reference scenario 2030, in line with the political target set for Stockholm. The reduction applies citywide (187 km ²).
Closure of Stockholm's local airport Bromma	Aviation	Effect of closing the Bromma Airport, located in central Stockholm.

workshop as part of the EU project “Nanoparticle emissions from the transport sector” (nPets, <https://www.npets-project.eu/>, accessed January 2025). Electrification of road vehicles can also be very important for reducing UFP emissions and is included here as part of the BAU scenarios. The share of electric vehicles depends not only on actions taken by the city, but more importantly, on factors such as economic development, vehicles and fuel costs and decisions by national and international authorities. Since the importance of local actions for electrification of road vehicles made by the city is difficult to quantify it is part of the BAU scenarios.

2.5.1. Business as usual scenarios

The Business as usual (BAU) scenarios for 2030, 2040 and 2050 refer to model simulations of transport emissions considering road vehicle fleet renewal, electrification and reduced emissions from shipping, while assuming constant total vehicle kilometers travelled and unchanged emissions from aviation and railway. Meteorological data are the same in all BAU and future policy scenarios.

Known vehicle fleet composition for 2021 (“current situation”), based on national statistics of registered on-road vehicles was used to compare measurements with calculated concentrations for evaluation of the model calculations as referred to earlier (section 2.2.4). The fleet in future year scenarios is based on politically decided and planned regulations of both light and heavy vehicles, especially regulations related to CO₂ emissions as part of the Swedish climate emission reporting, and how the regulations affect registrations of new vehicles (SEPA, 2024). All data needed to calculate vehicle emissions in all scenarios have been implemented in the HBEFA (version 4.2) data base for Sweden by the Swedish Transport Administration.

Fig. 11 (in Appendix A) shows the shares of total vehicle kilometers (vkm's) by different types of vehicles in the Stockholm region during 2021 to 2050. Electrified vehicles make 13 %, 33 %, 61 % and 78 % of

the total vkm's in 2021, 2030, 2040 and 2050, respectively. It should be noted that the expected shares of electric vehicles in 2030, 2040 and 2050 are averages for Sweden as a whole; for Stockholm electric vehicle shares are likely somewhat higher than this. (The sensitivity of this for the health impact is presented in [section 3.3](#) below).

In 2021 most UFP emissions (according to HBEFA 4.2) are due to heavy goods diesel vehicles (HGV) (45 %) but only 3 % of total vehicle km's, while diesel passenger cars (PCs) contribute with 35 % of total emissions and make 76 % of total vkm's. In 2050 PC diesel is the main contributor to UFP emissions (56 %) while HGV contribute by 19 % to total emissions.

Both for gasoline and diesel PC, LCV (Fig. 12 in Appendix A) and HGV vehicles (Fig. 13 in Appendix A) there is a substantial change in the distribution of the vehicles between different Euro classes when comparing 2030, 2040 and 2050 with 2021. Already by 2030 there are very few vkm's driven not fulfilling Euro 6 limits. For HGV almost all vkm's fulfill Euro VI already by 2030.

The renewal (including electrification) of the vehicle fleet will lead to substantial reductions in emissions of particles as shown in [Table 5](#). Compared to 2021 the emissions from PCs decrease by 76 %, 93 % and 98 % 2030, 2040 and 2050 respectively. For HGVs emissions are reduced by 92 %, 97 % and 99 %, 2030, 2040 and 2050 respectively.

Emissions from shipping are also expected to be reduced. [Trosvik et al. \(2020\)](#) present four scenarios for emissions after 2015 until 2045. They include four pollutants, SO₂, CO₂, PM₁₀ and NO_x, but unfortunately not UFPs. The scenarios are i) baseline with energy efficiency improvements, ii) higher energy efficiency improvement, iii) lower transport demand, iv) lower transport demand and fuel switch ([Table 6](#)). We simply assume that UFP emissions will follow the trends of NO_x. By translating their numbers into trends starting from 2021 by linear interpolation from 2015, we obtain the values presented in [Table 6](#). We adopted the baseline scenario with emissions being reduced by 8 %, 37 % and 57 %, in 2030, 2040 and 2050 respectively, compared to the emissions in 2021.

2.5.2. Impact of electrification of road vehicle fleet

There are many factors affecting the rate of electrification of road vehicles, including the economic development, costs of vehicles and fuels and different national and international policies that may promote more sales of electric vehicles. Registered shares of electric vehicles 2021 and shares for the scenarios 2030, 2040 and 2050 based on HBEFA 4.2 are provided in [Table 7](#). For PCs the share of electric vehicles increases by approximately 3 % per year. For HGVs, LCVs and Urban buses the annual rates of electrification increase are 0.6 %, 0.7 % and 2.4 %, respectively.

Table 5

Particle number emissions (10¹⁰ vkm⁻¹) from PCs, HGVs, LCVs and buses 2021, 2030, 2040 and 2050 according to HBEFA 4.2. Percentages indicate decrease since 2021. The emissions are weighted by the vkm of each fuel (petrol, diesel, gas, ethanol), Euro class and include emissions from PHEVs.

Vehicle type	2021	2030	2040	2050
PC	138	33 (-76 %)	10 (-93 %)	3 (-98 %)
HGV	1456	121 (-92 %)	37 (-97 %)	22 (-99 %)
LCV	980	101 (-90 %)	13 (-99 %)	5 (-100 %)
Bus	1297	46 (-96 %)	7 (-99 %)	3 (-100 %)

Table 6

Percent reductions in particle number emissions from domestic ships at berth in 2030, 2040 and 2050 relative to emissions in 2021 (estimated based on [Trosvik et al., 2020](#)).

Scenario	2030	2040	2050
Baseline with energy efficiency improvements	-8%	-37 %	-57 %
High energy efficiency	-11 %	-41 %	-62 %
Lower transport demand	-20 %	-49 %	-75 %
Lower transport demand and fuel switch	-34 %	-64 %	-98 %

Table 7

Shares vehicle kilometres and electric vehicles for the different vehicle types (based on HBEFA 4.2). Shares of vehicle kilometres (and total vehicle kilometres) is the same for all years. PC = passenger cars, HGV = heavy goods vehicles, LCV = Light commercial vehicles.

Vehicle type	Share of vehicle kilometers	Share of electric vehicles			
		2021	2030	2040	2050
PC	81 %	7.2 %	41 %	75 %	94 %
HGV	3.5 %	0 %	1.3 %	6.9 %	16 %
LCV	15 %	0.5 %	3.9 %	12 %	21 %
Urban bus	1 %	3.0 %	25 %	58 %	69 %

respectively.

As can be seen in [Fig. 2](#), BEV² and PHEV³ gasoline passenger vehicles make similar vehicle kilometers up to 2030. After this year BEV vehicles km's become more and more dominating and in 2050 almost 90 % of the electric PC's vehicle kilometers are made by BEV's. Note that "PHEV only electric" means vehicle kilometers driven only by electricity (some of the vehicle kilometers driven by PHEV is by either petrol or diesel engines).

Regenerative braking will effectively reduce brake wear in electric vehicles, but the higher weight of electric cars compared to conventional diesel and gasoline cars will increase tire and road wear (the methodology for calculating wear emissions is presented in [Chen et al., 2024a](#)).

2.5.3. Scenarios with low emission zones

The impact on population exposure and health of two low emission zone scenarios were analyzed. The first scenario reflects a zone that has been approved by the Stockholm city council for implementation in the city center. The zone covers a very small area; 0.2 km² ([Fig. 3](#)). For comparison a second hypothetical scenario was included in which the LEZ is extended to a larger area of 16 km². The boundaries of this extended zone correspond to those of the existing congestion tax area, which is monitored via automatic number plate recognition cameras, as described by [Johansson et al. \(2009\)](#).

The same vehicle regulations apply in both low emission zones. Passenger cars, light trucks, and light buses must meet one of the following criteria to be permitted:

- Fully electric vehicles (plug-in hybrids are not permitted)
- Natural gas vehicles compliant with Euro 6 emission standards
- Fuel cell vehicles

For heavy trucks and heavy buses, the following vehicle categories will be permitted:

- Fully electric vehicles
- Natural gas vehicles compliant with Euro 6 emission standards
- Fuel cell vehicles

² BEV refers to a fully electric vehicle powered exclusively by a lithium-ion battery, with no need for an internal combustion engine.

³ PHEV refers to a plug-in hybrid electric vehicle equipped with an internal combustion engine paired with an electric motor.

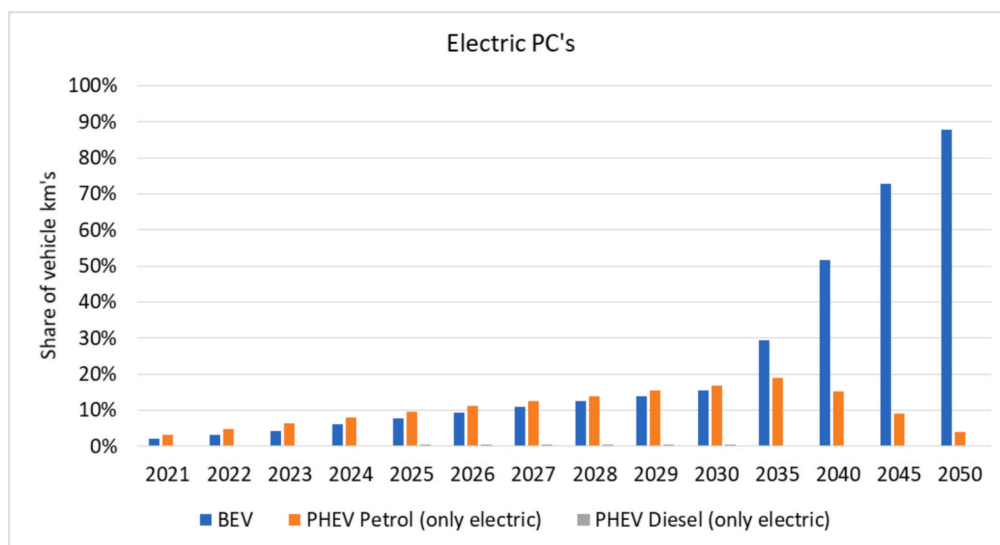


Fig. 2. Share of vehicle km's driven by electric passenger vehicles 2021–2050. For PHEV only the kilometers driven by electricity is included.

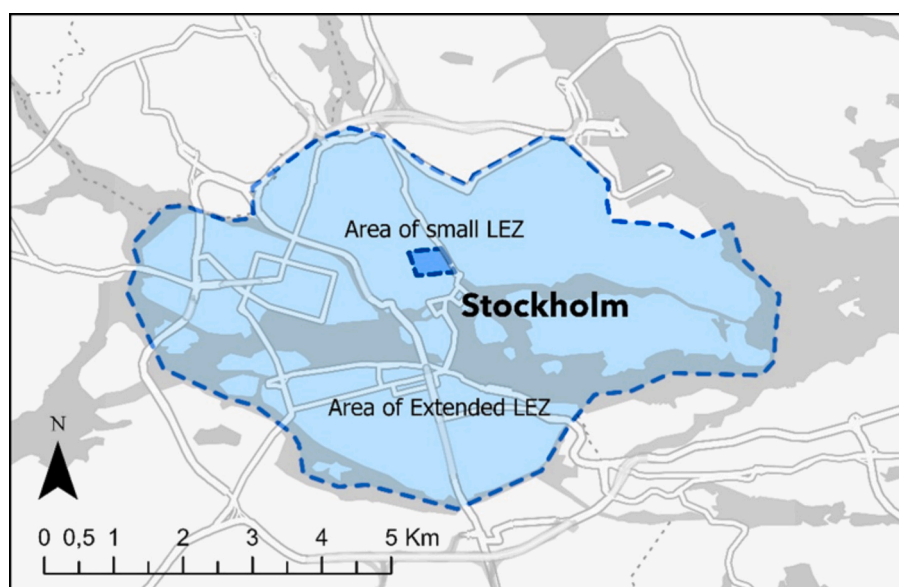


Fig. 3. Area of the low emission zones in Stockholm. The total area of the small zone is 0.2 km² and the area of the large zone is 16 km².

- Plug-in hybrid vehicles compliant with Euro 6

No general exemptions are granted for residents or workers in the area. However, certain vehicles, such as private mopeds and motorcycles, emergency vehicles, and road maintenance vehicles, will be permitted. The area includes numerous workplaces, shops, and other establishments.

2.5.4. Scenario of port electrification in Stockholm

To support the City of Stockholm's target of achieving fossil-free shipping by 2040, the Stockholm Climate Action Plan 2020–2023 stipulates that all ships operating within the city must be powered by renewable fuels and connected to electricity while docked. A significant share of the regular ferry traffic cruising in the Stockholm archipelago already has shore power capability. However, international cruise ships arriving during the summer season currently lack this infrastructure. To address this, The Ports of Stockholm are participating in the joint EU project *Coordinated Supply of Onshore Power in Baltic Seaports* (2019-EU-

TM-0125-W), in collaboration with three other Baltic ports – Copenhagen/Malmö, Aarhus, and Helsinki. The objective is to invest in shore power connections, thereby reducing maritime emissions within the port areas.

The ports of Stockholm are located on the eastern side of the city (Fig. 4, right). The ports of Värtahamnen and Stadsgården handle intensive ferry traffic of goods and passengers from Finland and the Baltics. During the summer season, international cruise ships dock at Frihamnen, as well as Stadsgården and Skeppsbron. The Energihamnen port manages coal and oil deliveries, while Södertälje hamn, situated 30 km southwest of central Stockholm, serves industrial and cargo vessels. Prevailing southwesterly winds on average transport ship emissions away from the city center, however, there is a large spatial and seasonal variation.

To assess the potential health benefits of this initiative, we constructed an onshore power scenario representing the impact of 100 % port electrification in Stockholm. This scenario estimates the reduction in pollution levels and associated health benefits if the ports in

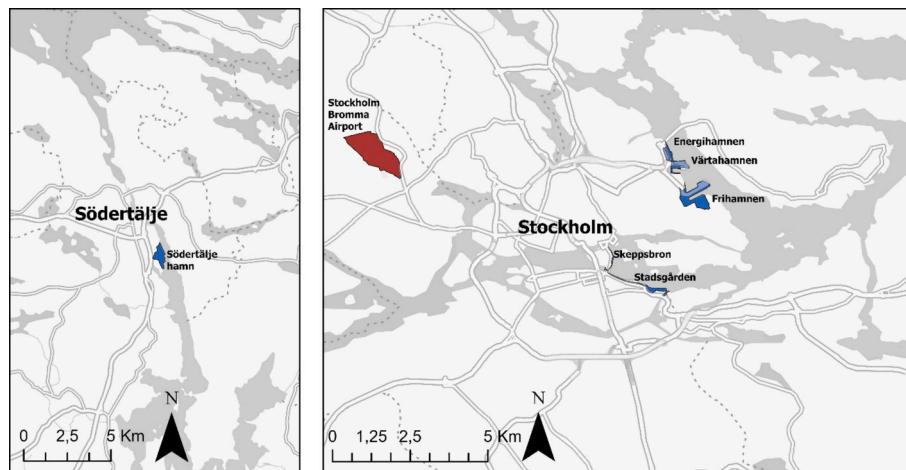


Fig. 4. Location of the ports of Stockholm (right) and the city of Södertälje 30 km SW of Stockholm (left) are indicated in blue, while the location of the local airport, Bromma, is marked in red. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Stockholm were electrified. The UFP emission data from shipping are based on the Shipair 2 model, as previously described by [Chen et al. \(2024a\)](#).

2.5.5. Scenario of a 30 % reduction in traffic

According to the City of Stockholm's environmental program for 2020–2023, adopted by the city council on May 25, 2020, the municipality aims to become entirely fossil-free as an organization by 2030, with a city-wide target of 95 % fossil-free status by 2040. To support these goals, the City of Stockholm has set an ambitious target to reduce traffic volume within the municipality by 30 percent between the baseline year of 2017 to the target year of 2030.

In this scenario, we assess the potential health benefits of achieving this traffic reduction in 2030, focusing on the associated decrease in UFP exposure. A partial reduction has already been achieved by 2022, which has been accounted for in our analysis. The scenario assumes that traffic reduction is uniformly distributed across the entire municipality, acknowledging this as a simplification. Other parameters, including speed limits, the proportion of heavy vehicles, traffic composition, and the influx of air pollutants, are assumed to remain unchanged from the baseline scenario.

2.5.6. Scenario of local airport closure

Bromma airport is located less than 10 km northwest of Stockholm's city center ([Fig. 4](#)). It is state-owned by Swedavia, and although an agreement between Swedavia and the City of Stockholm permits operations until 2038, there are ongoing discussions about the possibility of an earlier closure. An increasing number of observational studies have reported high concentrations of UFP near airports, including contributions to UFP levels detected several kilometers downwind from the airport ([Stacey et al., 2020](#); [Lopes et al. 2019](#); [Keuken et al., 2015](#)).

To assess the potential health benefits of closing Bromma airport, we constructed a scenario comparing population exposure to UFPs in the event of an airport closure against a reference scenario in which airport operations continue until 2038. UFP emissions were derived from modelled NO_x levels during landing, takeoff and taxiing, using a scaling factor based on measured UF- to- NO_x ratios around the airports in Barcelona and Milan, as described in [Chen et al. \(2024a\)](#).

The airport's most frequently used runway for departure is oriented northwest, and aircraft typically take off into the prevailing winds from the southwest. This means that while planes typically head northwest initially, their emissions can be carried towards densely populated areas around the airport by prevailing wind pattern.

3. Results

3.1. Business as usual

[Fig. 5](#) shows the spatial variations in UFP concentrations ($>4\text{nm}$) due to emissions from different transport sources in 2021. The total concentrations from all transport sectors is dominated by the emissions from road vehicles. The highest concentrations occur in central parts of Stockholm, which are most densely populated and where most workplaces and commercial activities are located. In contrast, the UFP emissions from shipping and aviation differ significantly due to their locations outside the city center.

Highest population weighted UFP concentrations are obtained for road vehicle exhaust emissions 1940 cm^{-3} of which HGVs and PCs account for 62 % and 26 %, respectively. Emissions from shipping and aviation result in 196 cm^{-3} and 2.9 cm^{-3} , respectively. The exposure due to emissions from railway is negligible ([Table 8](#)).

It should be noted that biomass burning may also represent an important local source of UFP exposure in some residential areas located outside of the central, less densely populated parts of Stockholm (not shown here). For the entire model domain, the population weighted exposure from biomass burning is $68\text{ particles cm}^{-3}$, corresponding to 3.5 % of the exposure from road vehicle exhaust emissions. Although the relative contribution from biomass burning may become more important in the future as road traffic emissions decline, the estimates for biomass emissions are associated with greater uncertainty compared to vehicle exhaust emissions.

The total population weighted exposure concentration in the BAU scenarios is reduced by 78 % in 2030 compared to 2021 ([Fig. 6](#)). This is mainly due to reduced emissions from road vehicle exhaust ([Table 8](#)), which in turn is mainly a result of the renewal of the vehicle fleet but to some extent also due to electrification of vehicles. Between 2030 and 2040 the exposure is further reduced by 55 % and between 2040 and 2050 by 41 % ([Fig. 6](#)). The emissions from ships also contribute to reduced exposure, but the relative importance for the total exposure increase from 2021 to 2050 ([Fig. 6](#)). By 2040 ship emissions will be the main source of UFP exposure in Stockholm. Also, the importance of road and tire wear increase in importance as the exhaust emissions are being reduced. In 2050 one third of the UFP exposure from road traffic is due to non-exhaust emissions, mainly due to tire and road wear.

As shown in [Table 8](#) the importance of electrification increases from 2021 to 2050. The total exposure concentrations due to transport emissions would be 3 %, 17 %, 40 % and 93 % higher in 2021, 2030, 2040 and 2050, respectively if there were no electric vehicles. This is mainly due to reducing the contribution from diesel passenger car

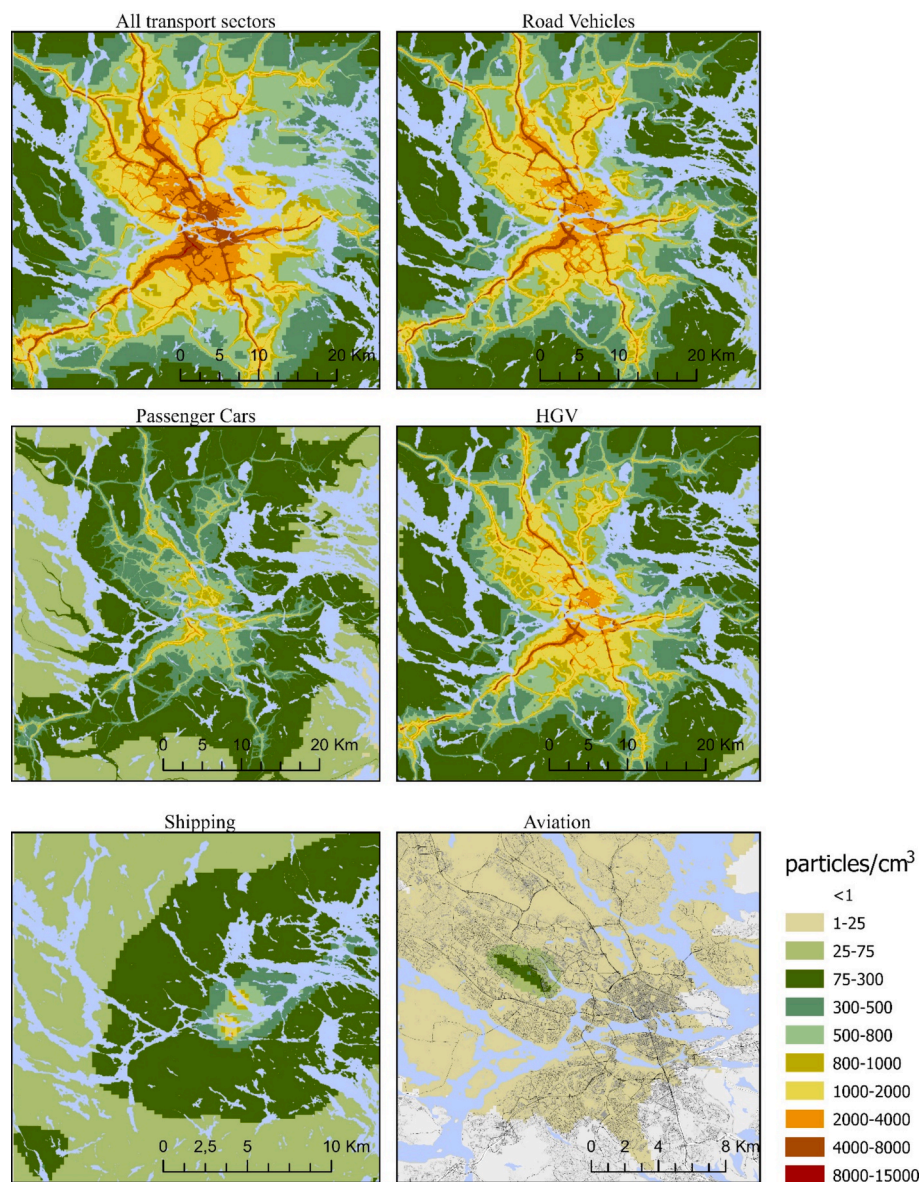


Fig. 5. Annual average UFP concentrations due to emissions 2021 from all transport sectors and separately for all road vehicles, passenger cars, heavy goods vehicles, shipping and the local airport. Spatial resolution varies depending on emissions and source (from 10 m x 10 m up to 500 m x 500 m). The road network is shown in brown and water areas in blue. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 8

Population weighted exposure concentrations (cm^{-3}) due to emissions from transport sources in Stockholm for the business-as-usual scenarios.

Exposure	2021	2030	2040	2050
Road traffic exhaust	1,940	270	70	23
Road traffic tire and roadwear	9.4	9.7	10.4	11.1
Road traffic brake	1.1	0.9	0.8	0.8
Road traffic clutch	0.0	0.0	0.0	0.0
Sea traffic	196	183	124	86
Aviation	2.9	2.9	2.9	2.9
Train	0.0	0.0	0.0	0.0
Sum	2,150	466	208	124
Sum without electrification of road vehicle fleet (i.e. only renewal of fleet)	2205	543	291	240
	(+3%)	(+17 %)	(+40 %)	(+93 %)

exhaust emissions. Note that both BEVs and PHEVs are defined as electric vehicles, which means that the emissions from electric vehicles are not zero, but depend on the share of PHEVs and the driven

kilometers using the internal combustion engine (as shown in Fig. 2).

Road, tire, brake and clutch wear make very small contributions to UFP emissions, but the contribution increase in importance for UFP exposure from 2021 to 2050 due to increased number of relatively heavy electric vehicles with higher road and tire wear emissions (on average electric vehicles are heavier than the fleet of conventional vehicles; Gustafsson et al., 2024).

3.2. Impact of policy scenarios on population weighted concentrations

Table 9 and Fig. 7 shows the impact on the population weighted concentrations if the different policy scenarios were implemented in 2030, 2040 or 2050. The total exposure due to all transport emissions is reduced by between 1 % up to 19 % depending on policy and when implemented. Electrification of ports reduces exposure by 11 %, 16 % and 19 % if implemented 2030, 2040 or 2050, respectively. Implementation of a small LEZ and closure of the airport has very small impacts on the exposure. But the extended LEZ reduces exposure by up to 9 %, which is similar to reducing traffic by 30 % (Table 9).

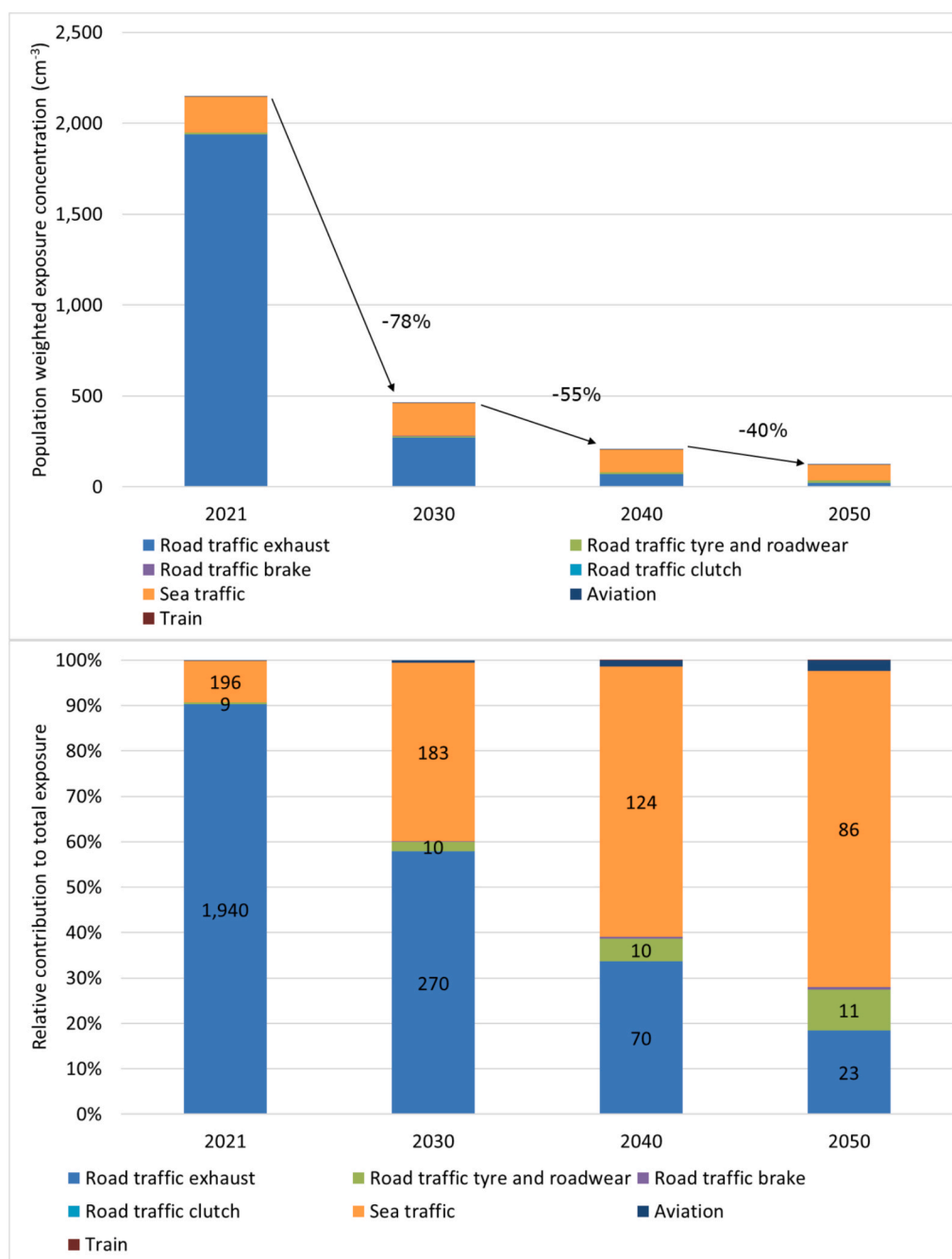


Fig. 6. Population weighted exposure concentrations due to emissions from transport sources in Stockholm (upper panel) and relative contributions from different sources (lower panel) for the BAU scenarios. The numbers on the lower panel refer to population weighted exposure (particles cm⁻³) for road traffic exhaust, non-exhaust and shipping.

Table 9

Contribution to total population exposure concentrations (cm⁻³) from all transport sources in business as usual and five local policy scenarios. Percentages refer to reduction in exposure if policy is implemented compared to BAU (i.e. without the policy). Note that BAU include both renewal of the fleet and electrification of road vehicles.

Scenario	2030	2040	2050
BAU	466	208	124
With small LEZ	465 (-0%)	208 (-0%)	124 (-0%)
With extended LEZ	423 (-9%)	199 (-4%)	120 (-3%)
With 30 % less traffic	422 (-9%)	195 (-6%)	119 (-4%)
With electrification of ports	416 (-11 %)	174 (-16 %)	101 (-19 %)
With closure of local airport	463 (-1%)	205 (-1%)	121 (-2%)

Fig. 7 shows the absolute reductions in population weighted exposure concentrations of the five local policy scenarios and anticipated road vehicle electrification. With the exception of the closure of the airport, the largest reductions in exposure are seen for 2030.

Foreseen road vehicle electrification is much more important for reducing exposure than any of the local policy scenarios. This is mainly attributed to the increase in vehicle kilometers by electric passenger cars (BEV's) in 2030 to 2050 (see Fig. 2) replacing Euro 5 and 6 diesel PC's.

3.3. Impact of policy scenarios on QALYs gained

Since local emissions from road traffic and shipping decrease with time, reductions in exposures by local policies addressing these sectors

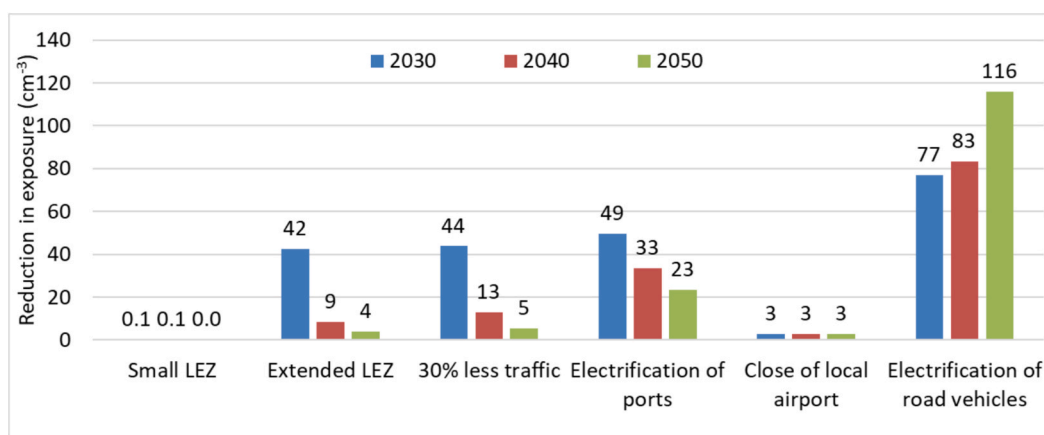


Fig. 7. Reduction in population weighted exposure to UFP compared to BAU in population weighted concentrations of the different policies when implemented in 2030, 2040 or 2050. Also shown is the reduction in exposure due to electrification of road vehicles compared with no electrification.

will also become smaller. Therefore more QALYs are gained if policies are introduced as early as possible as illustrated in Fig. 8. Electrification of ports, 30 % less traffic and an extended LEZ would save 48, 42 and 41 QALYs, respectively if implemented in 2030, whereas 32, 12 and 8 QALYs would be saved in 2040 and 23, 5, and 4 would be saved in 2050. The small LEZ saves less than 2 % of the number of QALYs saved with the extended LEZ. Closing the local airport saves only 3 QALYs.

As mentioned earlier the shares of electric vehicles in BAU refer to the average for Sweden. For Stockholm electrification is expected to proceed somewhat faster than Sweden as a whole. Fig. 9 illustrates the sensitivity of gained number of QALYs when increasing the share of electric vehicles in 2030. In BAU 2030 there are 41 %, 1.3 %, 3.9 % and 26 % electric PC, HGV, LCV and busses, respectively, and this saves 74 QALYs as shown in Fig. 8. If the shares of electric vehicles in 2030 are increased by 2 %, 5 % and 10 % the gain in number of QALYs attributable to all vehicles increase by 6, 15 and 29 QALYs, respectively (Fig. 9), i.e. from 74 QALYs to 79, 88 and 102 QALYs respectively. This corresponds to 8 %, 20 % and 39 % more QALYs, respectively, compared to the shares of electric road vehicles in BAU.

The impact on QALYs of increased electrification of a certain vehicle type depend on the relative shares of different Euro standards and the relative share of vehicle kilometers by that vehicle type. Most QALYs gained in 2030 is achieved by increasing electrification of PCs for which an increase from 41 % in BAU to 43 %, 46 % and 51 % saves 3, 8 and 15 QALYs, respectively.

In 2021 an increase in shares of electric PCs from 7.2 % in BAU (with 2 %, 5 % and 10 %) to 9.2 %, 12.2 % and 17.2 % is estimated to gain 12, 31 and 62 QALYs, respectively (Fig. 14 in Appendix A). In 2040 an

increase in shares of electric PCs from 75 % in BAU to 77 %, 80 % and 85 % saves 2, 4 and 9 extra QALYs, respectively.

3.4. Relative health risks associated with UFP, $PM_{2.5}$, and NO_2 exposure due to road traffic emissions

Mitigation policies often affect more than one pollutant and the effect on emissions, exposure and thereby health risks may differ between pollutants. While UFP serves as reliable indicators of exhaust emissions, UFP exposure does not adequately reflect health impacts of non-exhaust emissions from road traffic. In Stockholm non-exhaust emissions are already today the dominant local source of PM from road traffic (Gustafsson et al., 2024). Consequently, assessing the health impacts of mitigation actions targeting non-exhaust particles need to properly account for the health risks associated with particle mass exposures. In addition to particles, transport emissions include NO_x and thus affects exposure of NO_2 , which in itself, independently of particles, have been shown to be associated with mortality and morbidity. The objective of this section is to assess the relative impact on health risks associated with exposure to UFP, $PM_{2.5}$ and NO_2 from road traffic emissions. Maps showing the spatial distributions of annual mean UFP, $PM_{2.5}$ and NO_2 concentrations are provided in Appendix B.

The population weighted exposure due to road traffic emissions of $PM_{2.5}$ are calculated as the sum of the exhaust-PM emissions based on HBEFA 4.2 emission factors and non-exhaust emissions based on the NORTIP model (Denby et al., 2013 as described in Segerström et al., 2021). The number of QALYs for $PM_{2.5}$ is calculated based on both mortality and morbidity in the same way as for UFP. For the different

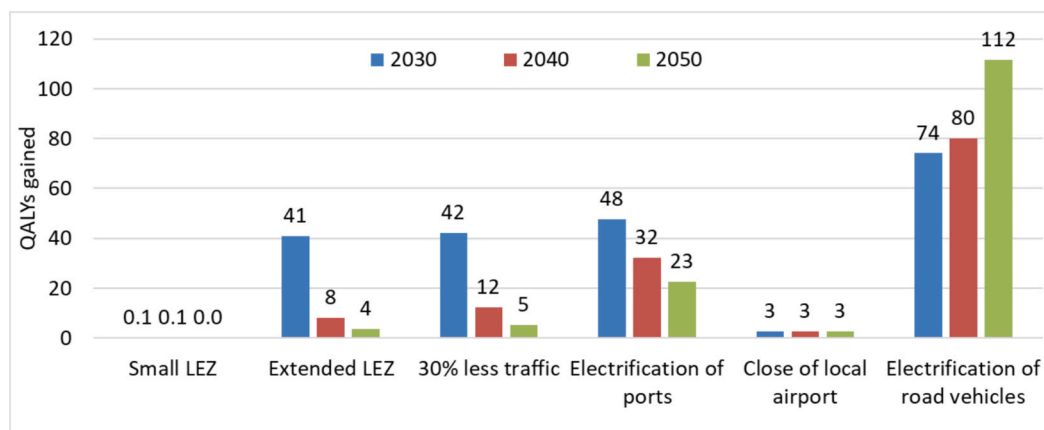


Fig. 8. Number of QALYs gained with the electrification of road vehicles and with the different local policies when implemented 2030, 2040 and 2050, respectively.

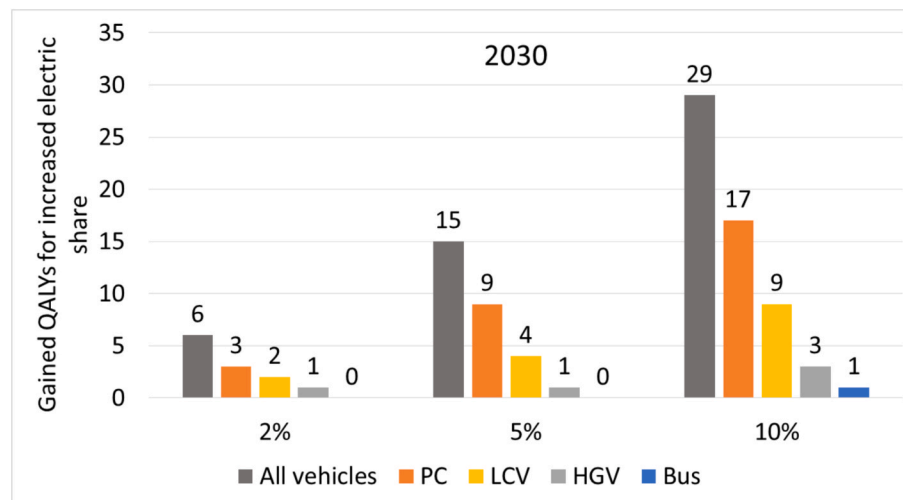


Fig. 9. Number of QALY gained in 2030 for 2%, 5%, 10% increase in electric vehicle share compared to BAU for all vehicles, LCV only, PC only, HGV only and bus only. The relative composition of non-electric vehicle types is kept the same as the baseline for each year. Electric vehicles are a mixture of BEV and PHEV as shown in Fig. 2.

morbidity outcomes the HRs are based on the recommendations for revision of the Swedish Transport Administration analysis method and values used for socio-economic calculations in the transport sector (Söderqvist et al., 2019) updated based on a literature survey by Oudin et al. (2022), see Table 10. Baselines, population data and years of life lost etcetera are the same as those used for UFP as presented earlier. For mortality we use the HR ratio reported by Turner et al. (2016), 1.26, which is higher than recommended by WHO in the HRAPIE report, 1.06 (HRAPIE, 2013), but motivated by the fact that we assess health risks of near-source exposures using high resolution modelling (see Segersson et al., 2021).

The annual mean population exposure to NO_2 is based on calculated annual mean of NO_x . An empirical relation between NO_2 and NO_x is used to obtain NO_2 concentrations⁴ (Edlund et al., 2024). The HRs for NO_2 are presented in Table 10 and are taken from a literature survey by Oudin et al. (2022). In addition to $\text{PM}_{2.5}$ and NO_2 a HR for road traffic exhaust PM exposure based Stockfelt et al. (2021) is used to compare number of premature mortalities using particle number (UFP) and particle mass ($\text{PM}_{\text{exhaust}}$) as health indicators of vehicle exhaust. Stockfelt et al. (2021) observed positive associations with natural mortality for the last five years of exposure to road traffic exhaust particles of 1.12 (95 % CI

1.03–1.21) per $1 \mu\text{g m}^{-3}$. The study is based on high-resolution dispersion modeling of PM from different sources and address registries assigning annual individual residential mean concentrations to participants in four cohorts in Gothenburg, Stockholm and Umeå (Sweden) for the years 1990–2011. QALY's cannot be calculated for $\text{PM}_{\text{exhaust}}$ due to lack of HRs for other health outcomes.

UFP and $\text{PM}_{\text{exhaust}}$ are associated with comparable numbers of premature deaths (Table 11), which is expected given that both represent estimates of road traffic emissions and are therefore not strictly additive, even though the UFP exposure include a very small fraction due to non-exhaust emissions (brake, tire, road and clutch wear). However, it should be noted that the estimated number of premature deaths is based on completely different epidemiological studies. The HR for UFP include health risks of exposure to all sources, whereas the study for $\text{PM}_{\text{exhaust}}$ include only road traffic exposure. Any deviations between health

Table 10

Hazard ratios (HR) used for quantifying health impacts of $\text{PM}_{2.5}$ and NO_2 exposure.

Health outcome, age (years)	HR NO_2	Reference	HR $\text{PM}_{2.5}$	Reference
Mortality, >30	1.047 per 10 ppb	Stieb et al. (2021)	1.26 per $10 \mu\text{g m}^{-3}$	Turner et al. (2016)
Myocardial infarction, >30	1.03 per $10 \mu\text{g m}^{-3}$	Gandini et al. (2018)	1.13 per $5 \mu\text{g m}^{-3}$	Cesaroni et al. (2014)
Stroke, >30	1.08 per $10 \mu\text{g m}^{-3}$	Wolf et al. (2021)	1.10 per $5 \mu\text{g m}^{-3}$	Wolf et al. (2021)
COPD, >50	1.07 per $10 \mu\text{g m}^{-3}$	Park et al. (2021)	1.20 per $10 \mu\text{g m}^{-3}$	Park et al. (2021)
Diabetes type II	Not enough evidence	–	1.25 per $10 \mu\text{g m}^{-3}$	He et al. (2017)
Childhood asthma, 2–18	1.05 per $4 \mu\text{g m}^{-3}$	Khreis et al. (2017)	1.03 per $1 \mu\text{g m}^{-3}$	Khreis et al. (2017)

Table 11

Annual premature deaths, years of life lost and QALY's lost associated with road traffic emissions of UFP, $\text{PM}_{2.5}$, NO_2 and $\text{PM}_{\text{exhaust}}$ according to baseline scenarios 2021, 2030, 2040 and 2050. Values in parenthesis indicate 95% CIs based on the HRs 95% CI.

	UFP	$\text{PM}_{\text{exhaust}}$	NO_2	$\text{PM}_{2.5}$
2021				
Premature deaths	144(85,202)	88(23,135)	159(75,230)	104(78,130)
Years of life lost	1394 (821,1964)	851 (219,1308)	1547 (723,2234)	1005 (753,1259)
QALY	1879	NA	1910	948
2030				
Premature deaths	21(12,29)	27(7,41)	57(27,82)	100(75,126)
Years of life lost	201 (118,284)	262(67,401)	554 (258,797)	972 (728,1218)
QALY	270	NA	681	917
2040				
Premature deaths	6(4,8)	16(4,25)	24(11,34)	100(75,125)
Years of life lost	58(34,82)	156(40,239)	233 (108,335)	967 (724,1212)
QALY	78	NA	286	912
2050				
Premature deaths	3(2,4)	13(3,20)	16(8,24)	99(75,125)
Years of life lost	25(15,35)	128(33,196)	159(74,228)	965 (723,1209)
QALY	34	NA	195	910

⁴ $\text{NO}_2 = \text{NO}_x \cdot \left(\frac{30}{36 + \text{NO}_x} + 0.173 \right)$

impacts based on UFP and PM_{exhaust} exposure, especially in later years (2040 and 2050), may also be due to uncertainties in emission factors.

For 2021 all exposure metrics show similar health impacts, as indicated by overlapping 95 % confidence intervals. For future years, 2030, 2040 and 2050, the number of premature deaths and QALY's lost due to exposure to road traffic using PM_{2.5} or NO₂ as health risk predictor is larger compared to using UFP (Fig. 10). This is likely a reflection of the fact that UFP is not a good predictor of health risks associated with non-exhaust, which constitute a large fraction of road traffic PM_{2.5} emissions 2030 and onwards. In the case of NO₂ it likely reflects the continued emissions of NO_x mainly from heavy duty vehicles for which the electrification is not progressing as fast as light duties. Emissions of UFP and PM_{exhaust} is effectively reduced by the particle filters both on light and heavy duty vehicles.

Thus, using only UFP and PM_{exhaust} in health impact assessments will underestimate impacts of future mitigations of road traffic emissions. Fig. 10 shows that the number of premature deaths are 5, 17 and 40 times higher 2030, 2040 and 2050, respectively, using PM_{2.5} instead of UFP as health impact exposure metric. Compared with NO₂ the number of deaths are 3, 4 and 7 times higher in 2030, 2040 and 2050 than if UFP is used. Both exhaust-PM, non-exhaust-PM and NO₂ need to be considered in order to assess the overall health risk associated with exposure to road traffic emissions. However, due to the correlations between pollutants health impact estimates for different pollutants cannot simply be added without using multipollutant risk estimates (Dominici et al., 2010). Recently, Chen et al. (2024b) demonstrated how two-pollutant risk models could be used to consider the combined effect of PM_{2.5} and NO₂, but so far this has not been done considering effects of UFP exposures.

3.5. Importance of different health outcomes for total QALYs

Premature mortality is the main contribution to the total number of QALYs gained for all pollutants considered here as can be seen in Table 12. Mortality associated with long-term UFP exposure contributes with 74 % to all QALYs, whereas PM_{2.5} exposure contributes with 84 %. UFP exposure seem to constitute a relatively larger risk of developing diabetes compared to PM_{2.5} exposure, 15 % compared to 7 % of all QALYs for UFP and PM_{2.5}, respectively. The relative importance of different health outcomes will be the same for different transport sectors since we apply the same relative risks for all sectors (due to lack of information). This also means that the relative importance of different health outcomes for the total number of QALYs gained is the same in all policy scenarios presented here. Currently there is very limited knowledge of the health impacts of source-specific UFPs related to different transport modes (Vallabani et al., 2023).

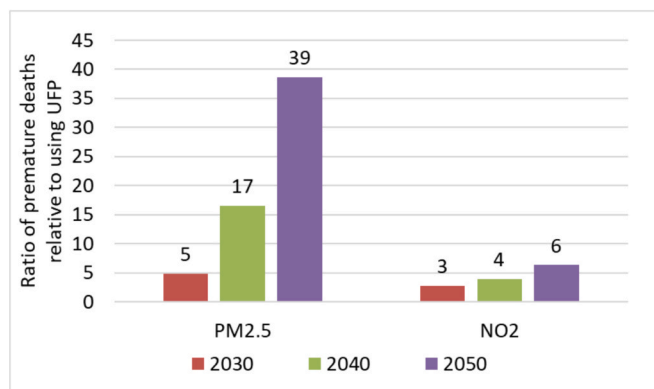


Fig. 10. Ratio to UFP of annual number of premature deaths of PM_{2.5} and NO₂.

Table 12

Relative contribution of premature mortality and different morbidities to total number of QALYs for UFP, NO₂ and PM_{2.5} in the baseline scenario (see Table 11).

Health outcome, age	UFP	NO ₂	PM _{2.5}
Mortality, >30	74 %	75 %	84 %
Myocardial infarction, >30	0.7 %	0.2 %	0.2 %
Stroke, >30 ^a	5.4 %	11 %	3.5 %
COPD, >50	4.2 %	12 %	4.0 %
Diabetes type II	15 %	— ^b	7.1 %
Childhood asthma, 2–18	0.2 %	2.2 %	0.7 %

^a For UFP > 50.

^b Not enough data.

3.6. Economic evaluation of policy actions

In 2021 the exposure of the population in Greater Stockholm (2.14 million inhabitants) due to the total emissions from all local transport (road traffic, shipping, aviation, and train) is estimated at 2150 particles cm⁻³. Due to this exposure we estimate that 2070 QALY would be lost, which correspond to a total costs of around €640 M. Road traffic, shipping, aviation, railway contribute with 91 %, 9.1 % 0.1 % and 0.0 % to the total cost, respectively (Table 13).

The reduction in road traffic emissions 2021–2050 correspond to an estimated monetary reduction from €581 M in 2021 to €10 M in 2050. Road and tire wear increases and brake wear decreases, but not until 2050 road and tire wear become a significant part (9 %) of the total road traffic costs. Brake wear decreases slightly with the electrification but the contribution to the overall health related costs is small (<1%). Health related costs associated with UFP emissions from sea traffic is estimated to be reduced from €59 M (9 %) in 2021 to €26 M (70 %) in 2050. Emissions from train and aviation are kept constant and the contributions to overall costs in Stockholm are negligible.

Road vehicle electrification saves 3 %, 17 %, 40 % and 93 % of the total estimated costs without road vehicle electrification in 2021, 2030, 2040 and 2050, respectively (Table 13).

It should be noted that these estimated health costs refer to UFP emissions only. If PM_{2.5} and/or NO₂ is used as health predictors instead of UFP health costs would be larger. For PM_{2.5} this is mainly associated with the exposure due to emissions from road, tire and brake wear as discussed earlier (section 3.4).

Table 13

Annual health related costs (million €) of UFP exposure in Stockholm due to different types of local transport emissions for 2021 and the business-as-usual scenarios 2030, 2040 and 2050. Percentage share of costs for each year is shown in parenthesis.

Source	2021	2030	2040	2050
Road traffic exhaust	578(90 %)	80(58 %)	21(34 %)	7(18 %)
Road traffic tire and road wear	2.8(0.4 %)	2.9(2.1 %)	3.1(5.0 %)	3.3(9.0 %)
Road traffic brake wear	0.3(0.0 %)	0.3(0.2 %)	0.2(0.4 %)	0.2(0.6 %)
Road traffic clutch wear	0.00 (0.00 %)	0.00 (0.00 %)	0.00 (0.00 %)	0.00 (0.01 %)
Sea traffic	59(9.1 %)	54(39 %)	37(60 %)	26(70 %)
Aviation	0.9(0.1 %)	0.9(0.6 %)	0.9(1.4 %)	0.9(2.3 %)
Railway	0.0(0.0 %)	0.0(0.0 %)	0.0(0.0 %)	0.0(0.0 %)
Sum	641	139	62	37
Total costs without electrification of road traffic (percent of total costs saved by electrification)	657(3 %)	162(17 %)	87 (40 %)	72(93 %)

4. Discussion

4.1. Fleet renewal and emission reductions

The study highlights the importance of fleet renewal, particularly the transition to Euro 6/VI vehicles with particle filters, emerged as the most impactful intervention for reducing UFP exposure. This is consistent with what has been observed in historic trends of particle number emissions and concentrations in Stockholm (Krecl et al., 2024). Stricter emission regulations, especially with diesel particulate filters (DPF) in Euro 5, 6, and VI vehicles, led to a reduction of the estimated real world emission factor by 55 % between 2013 and 2019. As demonstrated in this study, in addition to the emission regulations, the electrification of the fleet will also speed up the reduction of UFP exposure and related health and economic burden in the future.

4.2. Effectiveness and limitations of low emission zones

As shown in the comparison for Stockholm the difference in health impacts between two zones, with the same regulations, but with different sizes can be very large. In these scenarios we assume 100 % compliance and that only emissions from traffic within the borders of the zones are affected. In reality compliance depends on the size of the penalty, enforcement method and how well-informed drivers are. It is also likely that the regulations within the zone will affect emissions outside the borders, either because some drivers will take detours, making journeys longer and thereby increasing emissions or because the overall vehicle fleet is renewed as a result of the regulations within the zone and thereby reducing emissions more than if only emissions within the zone were considered. However, both these aspects are very difficult to quantify. Several studies have demonstrated the health benefits of introducing low emission zones in cities but mostly by analyzing impacts from $PM_{2.5}$, NO_2 or PM_{10} (Chamberlain et al., 2023). When using only UFP as marker, the health impacts of changes in non-exhaust emissions will not be properly accounted for as we have demonstrated (see also discussion in section 4.5 below).

4.3. Shipping

Shipping is known to be an important source of both primary and secondary air pollutants and to have significant impacts on public health (e.g. Andersson et al., 2009). Still, relatively few studies have addressed the emissions of air pollution from port emissions and what cities can do to reduce impacts (Ducruet et al., 2024). Connecting docked ships to onshore electricity instead of using auxiliary engines has been possible in several cities in the US and Europe (Gillingham and Huang, 2020; Winkel et al., 2016). The extent of the reduction of population exposure depends on the type of fuel burned from the ships, size of the port, time at berth and vicinity of the port to the city (Winkel et al., 2016). Ramacher et al. (2020) modeled the impact of ship emissions on air quality and human health in the Gothenburg area, Sweden. They simulated the impact of onshore electricity implementation for shipping in 2040 and found up to 3 % lower $PM_{2.5}$ and up to 30 % lower NO_2 , while O_3 increased of up to 3 %, indicating that important health benefits can be gained by onshore electricity.

We have not found any study making a quantitative assessment of the impacts on health of UFP emissions from docked ships. Mueller et al. (2023) reviewed health impact assessments of shipping and port-sourced air pollution on a global scale. They found 32 studies with health impact assessments of shipping emissions, however none of the studies found used UFP as marker. Karl et al. (2020) modeled the dispersion and dynamics of aerosols in the plumes of ships in the vicinity of harbors. The hourly mean UFP concentrations increased by a factor of two or more up to a distance of 3600 m, compared with the corresponding UFP concentrations in the urban background. They concluded that the ship emissions contributed to increased daily mean exposures in

residential areas located in the vicinity of the harbors.

4.4. Aviation

Our calculations indicates that health impacts associated with UFP exposures due to emissions from Stockholm Bromma airport are very small. But airports can be a very important source of UFP in cities. Several studies have reported substantially increased concentrations of UFP in the surroundings of airports (e.g. Mueller et al., 2022; Keuken et al., 2015; Zhu et al., 2011). Van Seters et al. (2024) summarized the scientific evidence on the health impact of aviation UFP emissions and discussed how fuel quality could be changed to reduce emissions. They extrapolated UFP concentrations measured around Schiphol Airport to the 32 busiest airports in Europe and estimated nearly 280,000 cases of high blood pressure, 330,000 cases of diabetes and 18,000 additional cases of dementia among populations living near these major airports. However, they emphasized that these are very crude estimate and that more epidemiological studies are needed to verify the health impacts of aircraft emissions.

4.5. Comparative health impacts using UFP, $PM_{2.5}$ and NO_2 as indicators

Most often health risks due to emissions are due to a complex mixture of several pollutants and the choice of pollutants may be critical in health impact assessments of different emission policies. Alongside with reduced exhaust vehicle emissions the non-exhaust emissions increase in importance in the future. We have shown that using only UFP and $PM_{exhaust}$ in health impact assessments will underestimate impacts of future mitigations of road traffic emissions. The number of premature deaths are 5, 17 and 39 times higher in 2030, 2040 and 2050, respectively, using $PM_{2.5}$ instead of UFP as health impact exposure metric. Compared with NO_2 the number of deaths are 3, 4 and 6 times higher in 2030, 2040 and 2050 than if UFP is used. Both exhaust- PM , non-exhaust- PM and NO_2 need to be considered in order to assess the overall health risk associated with exposure to road traffic emissions. In 2021 UFP exposure is estimated to be associated with 1879 QALYs lost, while $PM_{2.5}$ exposure is associated with 948 QALYs lost, indicating that current health impacts are underestimated if only $PM_{2.5}$ are accounted for in the health impact assessments of road traffic (Table 11). However, we estimate that already in 2030, the number of QALYs lost due to $PM_{2.5}$ exposure largely exceed QALYs lost due to UFP because of the estimated reduction in exhaust emissions. Then health impacts are underestimated by a factor of three if using only UFP exposure.

Our findings are consistent with Kriit et al. (2021), who used $PM_{2.5}$ as health indicator and demonstrated that the non-exhaust particles will dominate the contribution to the total the $PM_{2.5}$ concentrations and the related health impacts with increasing electric vehicles in the vehicle fleet. Since the major fraction of the particles in non-exhaust emissions are larger than $2.5 \mu m$ using PM_{10} might result in even larger health impacts than $PM_{2.5}$. Sommar et al. (2021) found 17 % increase in mortality associated with an increase in exposure of $10 \mu g m^{-3} PM_{10}$ based on cohorts in three Swedish cities. This is consistent with 22 % increase in mortality per $10 \mu g m^{-3} PM_{10}$ associated with long-term exposure to road wear reported by Stockfelt et al., (2021). Even though these relative risks are somewhat lower than for $PM_{2.5}$, exposure concentrations of PM_{10} are higher than $PM_{2.5}$, and may result in higher health impacts of PM_{10} .

As pointed out by Dominici et al. (2010), a multipollutant approach may be needed when assessing impacts of pollutant mixtures. Recently Chen et al. (2024b) proposed a model that include impacts of both $PM_{2.5}$ and NO_2 . But an important question is whether the health impacts of UFP exposure are considered if using only $PM_{2.5}$ and NO_2 or if there are toxic characteristics of UFP exposure are unaccounted for. No studies were found that have addressed this specific question.

4.6. Uncertainties in health risk estimates

The health risk estimates for long-term UFP exposure are still very uncertain due to the limited number of epidemiological studies. The most important uncertainty of total number of QALYs is associated with the relative risks of mortality. An important issue with quantification of long-term UFP exposure is the high micro-scale spatial variations of annual UFP concentrations in cities (Cassee et al., 2019). To capture this variation Pond et al. (2022), Bouma et al. (2023) and Lloyd et al. (2024) all used a combination of dedicated mobile monitoring campaigns and modelling to quantify exposure, even though Pond et al. (2022) aggregated the exposure to census tract level (3–4 km²). But the 95 % confidence intervals of the RRs per 1000 particles cm⁻³ of these 3 studies overlap (Pond: 1.004, 1.008; Bouma: 1.004, 1.005; Lloyd: 1.006, 1.008). In the study by Qi et al. (2024) the exposure was assessed with much coarser resolution, using a chemical transport model with 25 km grids. The CI of the RR per 1000 particles cm⁻³ differ marked from the other studies; 1.06, 1.2. It should be noted that the RR of Qi et al. (2024) had almost negligible influence on the pooled random RR used here (it contributed by 0.1 % to the pooled random RR).

There are many other factors than exposure misclassification that can contribute to different results as discussed by Cassee et al. (2019). Pollutants that may be important confounders are PM_{2.5} and NO₂. The mortality effect of UFP reported by Bouma et al. (2023) remained significant when controlling for both PM_{2.5} and NO₂. In the study by Pond et al. (2022) the effect of UFP was sensitive to the inclusion of PM_{2.5}, which is not surprising considering the relatively coarse resolution. Qi et al. (2024) did not control for confounding by PM_{2.5} or NO₂, while Lloyd et al. (2024) adjusted for the size distribution of UFP, but not PM_{2.5} or NO₂.

Statistical tests based on the I^2 index (Higgins et al., 2003) indicates that statistical heterogeneity is considerable for the four epi studies associating long-term UFP exposure with premature mortality ($I^2 = 88$ %, 95 % CI: 61 %, 99 %). The estimated 80 % prediction interval of the pooled random RR (per 1000 particles cm⁻³) is 1.001, 1.010 (whereas the 95 % confidence interval of RR is 1.003, 1.008). Corresponding I^2 for diabetes (3 studies) is 96 % (95 % CI: 84 %, 100 %) with an 80 % prediction interval of the pooled random RR (per 1000 particles cm⁻³) of 0.99, 1.038 (whereas the 95 % confidence interval of RR is 1.007, 1.022).

However, as noted by Borenstein (2023) an estimate of statistical heterogeneity based on less than five studies is “not likely to be informative”. We use the pooled random RR that account for the variance between the studies. This is considered to be a more conservative approach when heterogeneity is suspected.

We do not have the age distribution in every grid but make a correction of the total population by multiplying with the fraction older than 30 etc., depending on what is relevant for the relative risk. This means that there might be misclassifications if adults (>30, or > 50) or children (<18) are unevenly distributed in relation to the spatial variation in the concentrations. If e.g. children with asthma live mostly in areas with lower concentrations their exposure will be overestimated. Since we are interested in effects of different mitigations, it is the difference in exposure with and without mitigation that we are looking at. We believe that such errors are smaller than other errors involved in our health impact assessments.

4.7. Strengths of our study

Our study employs detailed local data on traffic and vehicle fleet enabling high resolution modeling of total nanoparticle (>4 nm) concentrations and its health impacts associated with annual mean exposures. A variety of policy scenarios are evaluated, including electrification, low-emission zones, and traffic reduction, offering a comparative perspective on potential interventions. Modeling future projections (2030, 2040 and 2050) allows for understanding long-term

trends and the potential importance of fleet renewal and electrification.

Health impact assessments are grounded in recent epidemiological studies and includes estimating both mortality and morbidity. By quantifying health benefits in terms of QALYs and associated economic costs, the study provides actionable insights for policymakers. The QALY value used in this study to value the health impacts monetarily, was shown to have limited variation in the value of a QALY across wide variation of the size of estimated QALY loss and when using different methods and samples (Olofsson et al. 2019). Using a QALY estimate to value the health impacts of air pollution allows comparison of health interventions across sectors. Non-exhaust sources (brake wear, tire wear) are included in future scenarios with a renewed fleet and increased electrification. The study includes a comparison with using PM_{2.5} and NO₂ as health risk indicators to estimate health impacts of future road traffic emissions.

4.8. Limitations of our study

Scenarios depend on assumptions about future fleet composition, electrification rates, and shipping emissions. Changes in these factors (e.g., slower adoption of electric vehicles) could significantly alter the projected outcomes. The study evaluates effects of primary UFP emissions from local sources, excluding formation of new nanoparticles as a result of nucleation associated with local emissions and their potential effects on exposure and health.

Degree of compliance and behavioral adaptation affecting traffic displacement to surrounding areas have not been considered in LEZ scenarios. Potential reductions of UFP emissions due to stricter future regulations (Euro 7) on new vehicles have not been considered. This regulation includes rules for the exhaust emissions of road vehicles, but also for tire abrasion and brake wear emissions. It introduces specific limits for braking of electric vehicles.

In this study we have not explored the effect of multiple policy mitigations. The cumulative impact of policies may lead to the synergistic effects being superior to the effects of implementing a single policy (Feng et al., 2025).

5. Conclusions

We present a comparative assessment of five policy mitigation scenarios in Stockholm: i) a small low emission zone in the city center (0.2 km²), ii) an extended low emissions zone covering a larger area in Stockholm (approximately 16 km²), iii) 100 % electrification of ports, iv) a 30 % reduction in vehicle transports and, v) closure of the local airport. The effectiveness of the scenarios is evaluated against a business-as-usual scenario without the implementation of these policies. Exposure calculations are based on high-resolution modelling with input data from detailed geographically resolved emission data.

The business-as-usual (BAU) scenarios for 2021, 2030, 2040, and 2050 demonstrate the effects on exposure and health without any local policies, but with anticipated renewal and electrification of the vehicle fleet and reduction in emissions resulting from European emission standards for new vehicles. Already by 2030 exposure due to direct vehicle exhaust UFP emissions have decreased by 90 % compared to 2021. This is estimated to correspond to 2070 QALYs gained, and to a monetary value of €640 M (excluding direct health costs). The main reason for the reduced emissions is the introduction of new diesel vehicles with particle filters, both passenger cars and heavy-duty vehicles. The most important gain in terms of health effects is reduction in premature mortality associated with long-term UFP exposure.

The electrification of ports is estimated to result in 50 QALYs gained if introduced in 2030, while the electrification of vehicle fleet is estimated to gain 74 QALYs in 2030. The most important local policies related to road traffic are the extended low-emission zone (LEZ) and measures that would reduce traffic by 30 %. They would result in more than 40 QALYs gained if introduced in 2030. Closing the local airport by

2030 would gain 3 QALYs and finally, the introduction of the small low-emission zone would save 0.1 QALYs if introduced in 2030. The effectiveness of mitigation policies targeting traffic emissions is expected to diminish over time, as most diesel vehicles will be equipped with effective particle filters. Road vehicle electrification is projected to be much more important for reducing exposure than any of the local policy scenarios. This is mainly attributed to the increase in vehicle kilometers by electric passenger cars (BEV's) in 2030 to 2050 replacing Euro 5 and 6 diesel PC's.

Since road traffic constitute the main source of UFP exposure in cities in general, policies promoting electrification, low emission zones and traffic reductions are likely to be the most efficient mitigations for reducing UFP exposures and associated health risks in densely populated cities.

Policy measures targeting both exhaust and non-exhaust UFP sources are essential for improving air quality and public health in Stockholm. Using only primary UFP as a predictor of health impacts will underestimate the impacts of future mitigations of road traffic emissions. Both non-exhaust PM and NO₂ need to be considered, especially in future years when most vehicles use particle filters that efficiently remove exhaust particles. Future policies should integrate these measures to maximize health and economic benefits.

CRediT authorship contribution statement

Lina Broman: Writing – review & editing, Validation, Methodology,

Formal analysis. **Magnuz Engardt:** Writing – review & editing, Methodology, Formal analysis, Data curation. **Sanna Silvergren:** Writing – review & editing, Formal analysis, Data curation. **Hedi Kriit:** Writing – review & editing, Validation, Methodology. **Michael Norman:** Writing – review & editing, Data curation. **Christer Johansson:** Writing – original draft, Validation, Project administration, Methodology, Funding acquisition, Formal analysis.

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.

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Appendix A

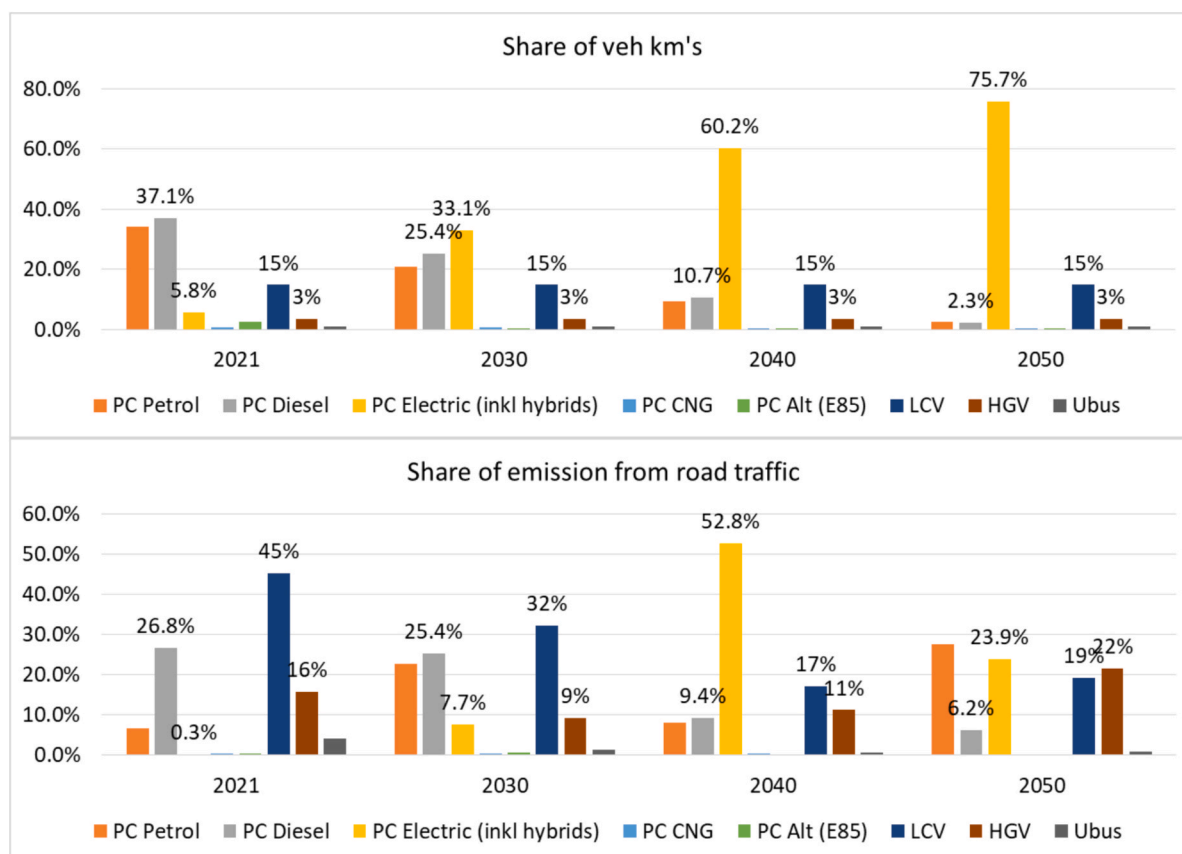


Fig. 11. Shares of total vehicle km's (upper panel) and total particle number emissions from road traffic for different types of vehicles 2021, 2030, 2040 and 2050 (according to HBEFA 4.2). CNG = Compressed natural gas, LCV = Light commercial vehicle, Ubus = urban bus, PC = passenger car, HGV = heavy goods vehicle, Alt = alternative fuel, E85 = 85 % ethanol (rest is petrol).

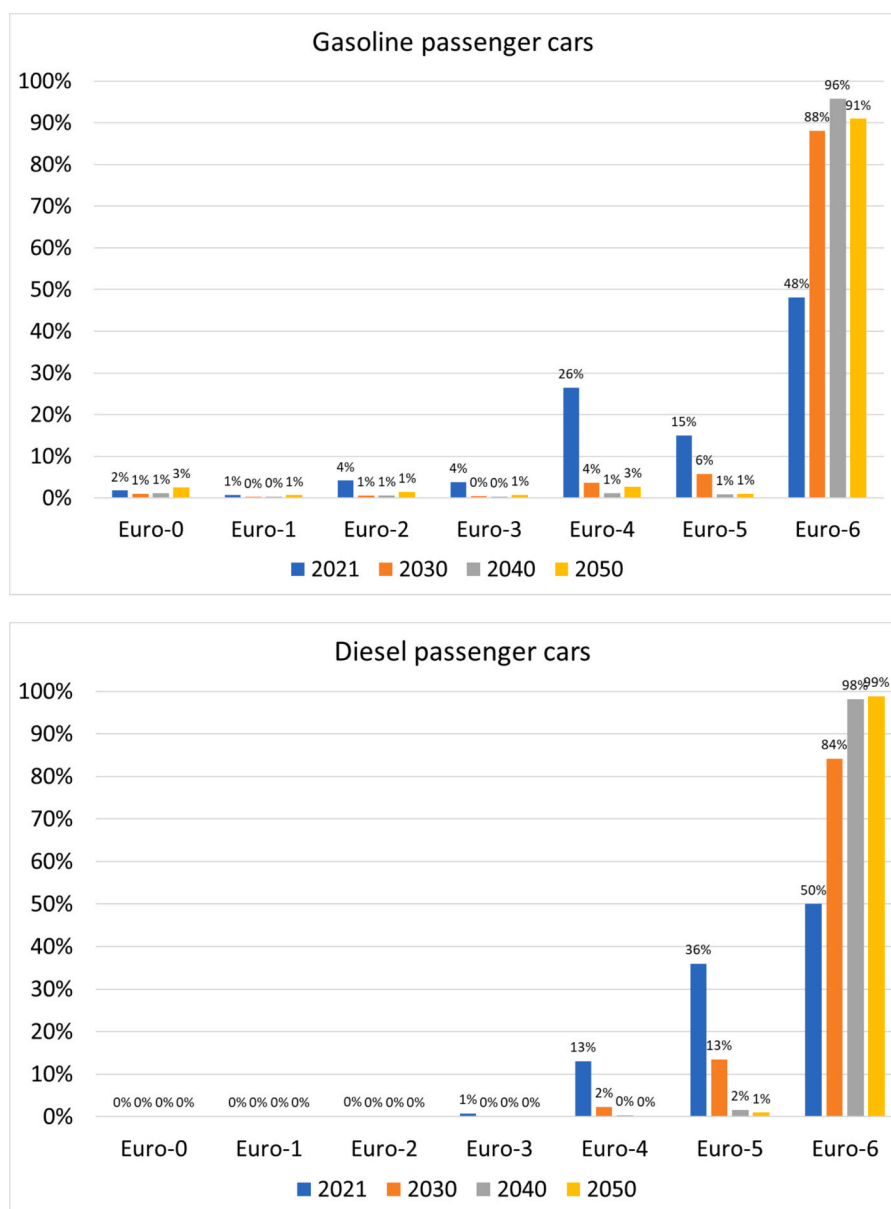


Fig. 12. Percent vehicle kilometers of different Euro classes of gasoline and diesel passenger and light commercial vehicles for the years 2021, 2030, 2040 and 2050. The percentage is per total vehicle kilometers for each fuel and vehicle type.

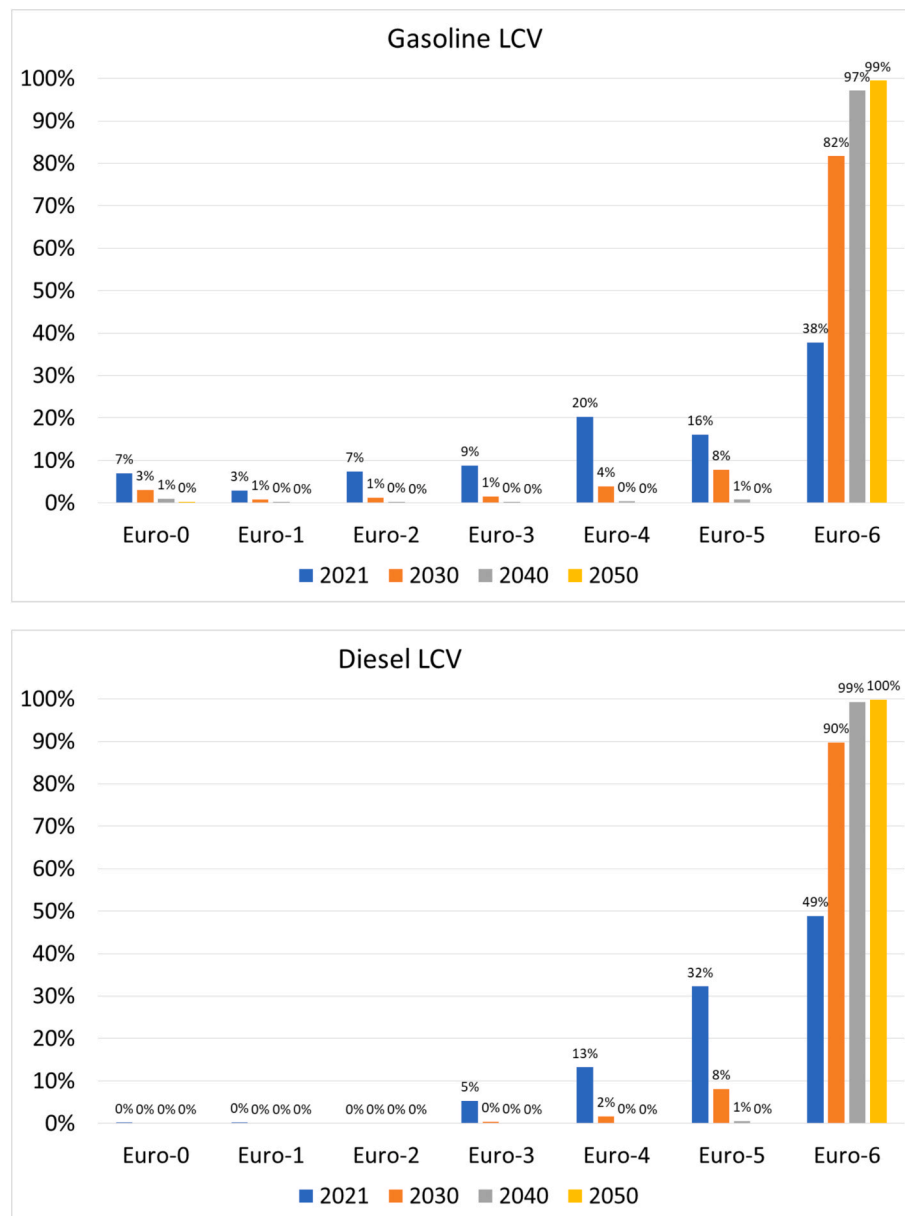


Fig. 12. (continued).

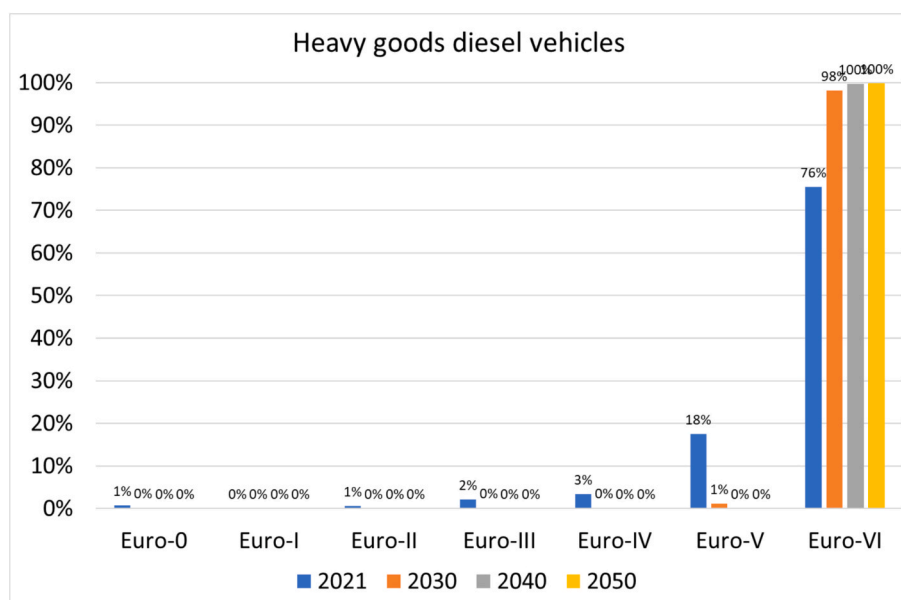


Fig. 13. Shares of vehicle kilometers of different Euro classes of heavy goods diesel vehicles for the years 2021, 2030, 2040 and 2050. The percentage is per total vehicle kilometres by heavy goods vehicles.

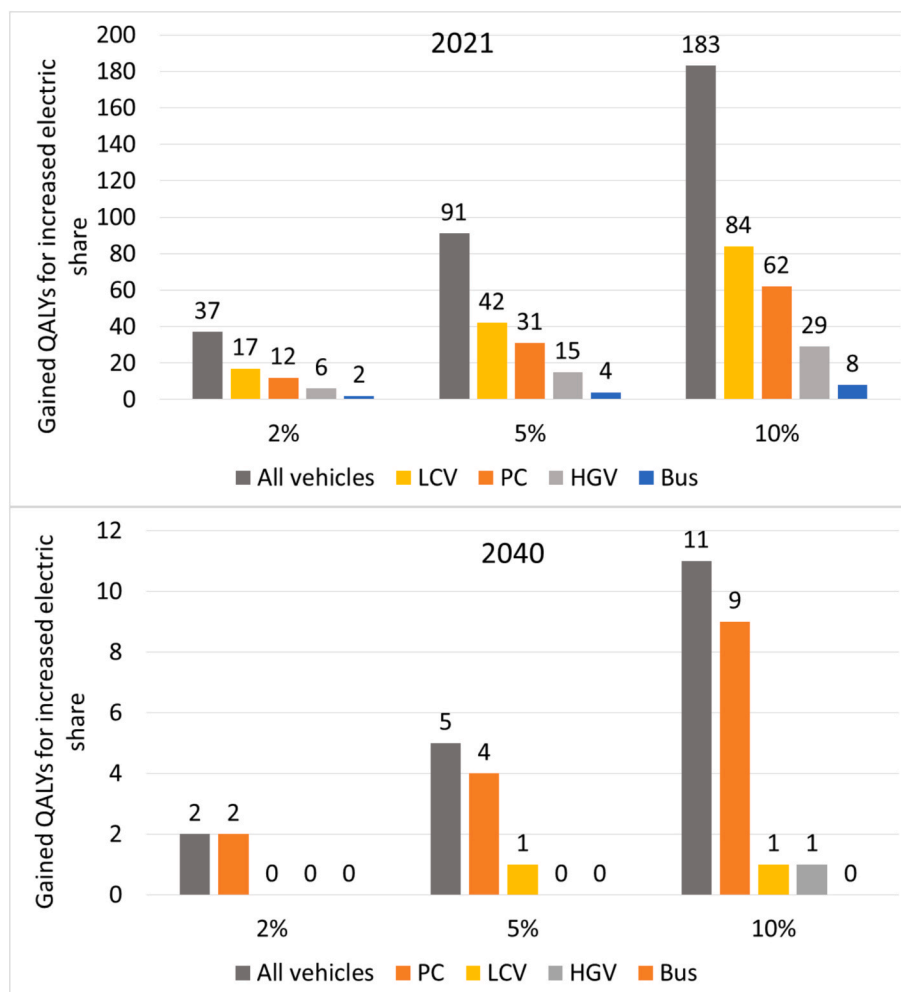


Fig. 14. Extra QALYs gained in 2021 and 2040 if shares of electric vehicles are increased by 2 %, 5 % and 10 % from the shares in 2021 ("current situation") equal to 7.2 %, 0 %, 0.5 %, 3 % and BAU 2040 equal to 75 %, 6.9 %, 11.6 %, 58 % for PCs, LCVs, HGVs and urban buses, respectively.

Appendix B

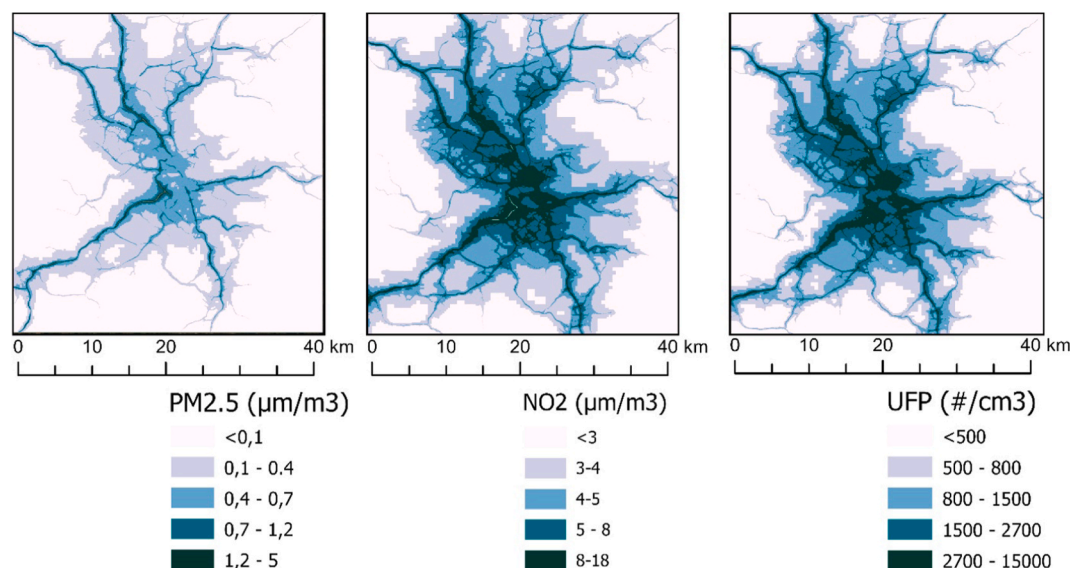


Fig. 1. Modeled annual average concentrations of PM_{2.5}, NO₂ and UFP for 2021.

Data availability

Data will be made available on request.

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