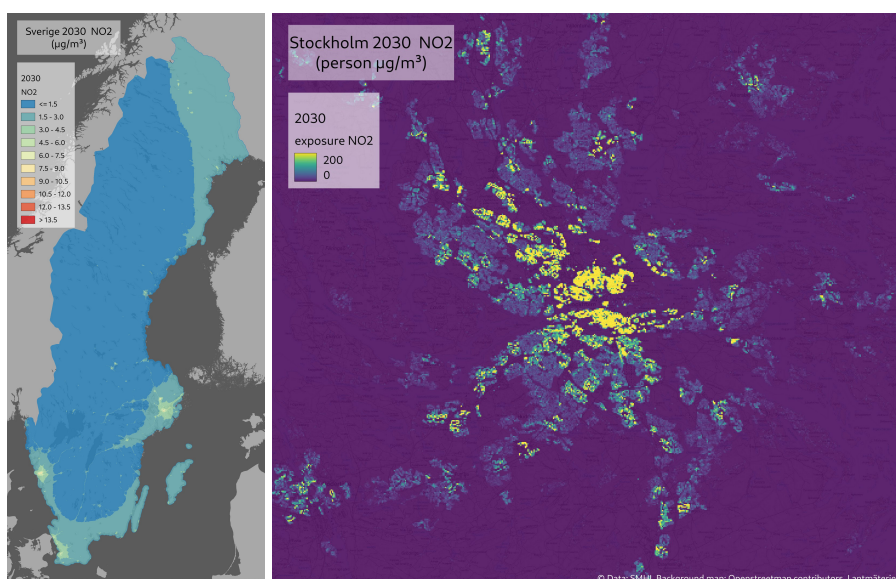


Quantification of population exposure to NO_2 , $\text{PM}_{2.5}$, PM_{10} and O_3 and estimated health impacts for 2023 and 2030

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Cover page figure:

Annual mean concentrations of NO₂ over Sweden and the population exposure to NO₂ zoomed in over Stockholm for 2030.

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Summary

A recently developed air quality modelling system is used to produce high-resolution concentrations (with street-canyon effects not included) of air pollutants (NO_2 , $\text{PM}_{2.5}$, PM_{10} and O_3) over all of Sweden. As a next step the population exposure is quantified and associated health impacts and socioeconomic costs for the years 2023 and 2030 are calculated.

The concentrations of NO_2 are predicted to decrease from 2023 to 2030 due to lower traffic exhaust emissions. Modelled exceedances of the air quality standard for the yearly mean of NO_2 are limited to the nearest vicinity of a few large roads in 2023, while there are no exceedances for 2030 (excluding street-canyon effects). The average population exposure of NO_2 decreases from $4.0 \mu\text{g}/\text{m}^3$ to $2.9 \mu\text{g}/\text{m}^3$.

PM_{10} and $\text{PM}_{2.5}$ concentrations in 2030 are predicted to remain on a similar level as in 2023, with a possible slight upward trend. The reasons for increasing concentrations are more traffic and thus higher levels of road, tyre and brake wear particles, as well as slightly higher regional background levels. Modelled exceedances of the air quality standard for the yearly mean of $\text{PM}_{2.5}$ are not expected in neither 2023 nor 2030, while PM_{10} exceedances are limited to the vicinity of larger roads in both years (again excluding street-canyon effects). The average population exposure of particles shows very little variation between 2023 and 2030 (from 8.1 to $8.2 \mu\text{g}/\text{m}^3$ for PM_{10} and 4.6 to $4.7 \mu\text{g}/\text{m}^3$ for $\text{PM}_{2.5}$).

O_3 concentrations are predicted to stay at similar levels in 2030 compared to 2023, with reductions in rural areas and locally increased concentrations close to larger roads. The average population exposure changes only slightly from $57 \mu\text{g}/\text{m}^3$ to $56 \mu\text{g}/\text{m}^3$.

The following health impact assessment is based on NO_2 , local and regional $\text{PM}_{2.5}$, respectively, and wear particles in the PM_{10} fraction, combining the high-resolution exposure modelling with concentration–response functions from recent epidemiological evidence.

For 2023, the total number of incident mortality cases is estimated at approximately 5 966 (95% CI: 3 905–8 035), with contributions from regional background $\text{PM}_{2.5}$, locally generated $\text{PM}_{2.5}$, wear particles (PM_{10}), and NO_2 . Compared with previous assessments, the methodology distinguishes more clearly between these sources, including a separate treatment of wear particles using PM_{10} -based risk estimates.

The analysis also includes a broader set of morbidity outcomes. Large numbers of cases are estimated for functional outcomes, including ~680 000 (579 000–782 000) work loss days and ~48 000 (15 700–80 400) school absence days, showing substantial impacts on productivity. Among chronic diseases, estimates include 1 083 COPD cases, 720 dementia cases, 374 diabetes cases, and 251 stroke cases annually. Additional outcomes newly included in the assessment comprise autism spectrum disorder (ASD) (1 422 cases) and allergic rhinitis (148 cases), based on emerging evidence for air pollution effects in children.

In the scenario work for 2030, in total a lower estimated mortality attributed to air pollution was estimated (5 627 cases, e.g. 6% lower), however both of the cases attributed to local wear particles and regional background $\text{PM}_{2.5}$ were higher (621 vs 554 and 3601 vs 3467 cases, respectively).

Methodologically, mortality estimation has been refined by separating $\text{PM}_{2.5}$ into regional background and locally generated components, while wear particles are assessed using PM_{10} exposure–response functions. This adjustment addresses the contribution of

coarse particles from non-exhaust traffic emissions and avoids underestimation of their health effects, although the uncertainty due to limitations in source-specific risk estimates remains.

Sammanfattning

Ett nyligen utvecklat system för luftkvalitetsmodellering används för att producera högupplösta halter (med lokala gaturumseffekter ej inräknade) av luftföroreningar (NO_2 , $\text{PM}_{2.5}$, PM_{10} och O_3) över hela Sverige. I ett nästa steg kvantifieras befolkningens exponering och de associerade hälsoeffekterna och socioekonomiska kostnaderna för åren 2023 och 2030.

Halterna av NO_2 förutspås minska från 2023 till 2030 på grund av längre trafikavgasutsläpp. Modellerade överskridanden av luftkvalitetsnormen för årsmedelvärde av NO_2 är begränsade till den omedelbara närheten till några få större vägar år 2023, medan inga överskridanden beräknas ske år 2030 (lokala gaturumseffekter ej inräknade). Den genomsnittliga befolkningsexponeringen av NO_2 minskar från $4.0 \mu\text{g}/\text{m}^3$ till $2.9 \mu\text{g}/\text{m}^3$.

Halterna av PM_{10} och $\text{PM}_{2.5}$ år 2030 förutspås ligga på en liknande nivå som 2023, med en möjlig, svagt uppåtgående trend. De ökade halterna kan förklaras av mer trafik och därmed högre nivåer av partiklar från slitage av vägar, däck och bromsar, men även av något högre regionala bakgrundshalter. Modellerade överskridanden av luftkvalitetsnormen för årsmedelvärde av $\text{PM}_{2.5}$ är inte förväntade varken 2023 eller 2030, medan överskridanden för PM_{10} är begränsade till närheten av större vägar för båda åren (återigen med lokala gaturumseffekter utelämnade). Den genomsnittliga befolkningsexponeringen av partiklar varierar mycket lite mellan 2023 och 2030 (från 8.1 till $8.2 \mu\text{g}/\text{m}^3$ för PM_{10} och 4.6 till $4.7 \mu\text{g}/\text{m}^3$ för $\text{PM}_{2.5}$).

Halterna av O_3 förutspås också ligga på liknande nivåer 2030 jämfört med 2023, med minskningar i landbygdsområden och lokala ökningarna vid större vägar. Den genomsnittliga befolkningsexponeringen förändras mycket lite, från $57 \mu\text{g}/\text{m}^3$ till $56 \mu\text{g}/\text{m}^3$.

Hälsokonsekvensbedömningen bygger på NO_2 , lokalt och regionalt $\text{PM}_{2.5}$ samt slitagepartiklar i PM_{10} -fraktionen, och kombinerar högupplöst exponeringsmodellering med koncentrations-responsfunktioner baserade på nyare epidemiologisk evidens.

För 2023 uppskattas det totala antalet incidenta dödsfall till cirka 5 966 (95 % KI: 3 905–8 035), med bidrag från regional bakgrund av $\text{PM}_{2.5}$, lokalt genererad $\text{PM}_{2.5}$, slitagepartiklar (PM_{10}) och NO_2 . Jämfört med tidigare bedömningar särskiljer metodiken dessa källor tydligare, inklusive en separat behandling av slitagepartiklar med riskestimat baserade på PM_{10} .

Analysen omfattar också ett bredare spektrum av sjuklighetsutfall. Ett stort antal fall uppskattas för funktionsrelaterade utfall, däribland cirka 680 000 (579 000–782 000) dagar med arbetsfrånvaro och cirka 48 000 (15 700–80 400) dagar med skolfrånvaro, vilket visar på betydande påverkan på produktiviteten. Bland kroniska sjukdomar omfattar skattningarna 1 083 fall av KOL, 720 fall av demens, 374 fall av diabetes och 251 fall av stroke per år. Ytterligare utfall som nyttillkommit i bedömningen omfattar autism-spektrumtillstånd (ASD) (1 422 fall) och allergisk rinit (185 fall), baserat på framväxande evidens för luftföroreningars effekter hos barn.

I scenariearbetet för 2030 uppskattas en sammantaget lägre luftföroreningsrelaterad dödlighet (5 627 fall, dvs. cirka 6 % lägre). Däremot är båda fallen hänförliga till lokala slitagepartiklar och regional bakgrund av $\text{PM}_{2.5}$ högre (621 jämfört med 554 fall respektive 3601 jämfört med 3467 fall).

Metodmässigt har skattningen av dödlighet förfinats genom att $\text{PM}_{2.5}$ delas upp i regional bakgrund och lokalt genererade komponenter, medan slitagepartiklar bedöms med hjälp av exponerings-responsfunktioner för PM_{10} . Denna justering beaktar bidraget från grova partiklar från icke-avgasrelaterade trafikemissioner och minskar risken för under-

skattning av deras hälsoeffekter, även om osäkerheten till följd av begränsningar i källspecifika riskskattningar kvarstår.

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1 Introduction

Over the last couple of years, SMHI has developed a new air quality modelling system capable of producing high-resolution concentrations over all of Sweden. An early iteration of this system was used by Alpfjord Wylde et al. (2023) to quantify population exposure and associated health impacts from NO₂, PM_{2.5} and PM₁₀. Analysis was done for the years 2019 and 2030 at a spatial resolution of 250 m x 250 m. In the following projects by Grundström et al. (2023) and Windmark et al. (2025), the method was further improved with an increased resolution of 50 m x 50 m, improved emission assumptions as well as improved model formulation including street canyon modelling for the years 2019 and 2023. The resulting data sets have been made publicly available on Luftwebb³, allowing for municipalities, governmental agencies and other professional users as well as the public to do their own analysis of the air quality anywhere in Sweden.

The Swedish Environmental Protection Agency (EPA) have recently published the results from their work on a government assignment to propose what Sweden should do to fulfil the new EU directive on ambient air quality (2024/2881). The new directive contains, among other things, stricter limit values and assessment thresholds for each pollutant as well as more concrete guidelines on what information should be included in air quality plans when the limit values are exceeded. Overall, this significantly increases the need for each country to perform air quality modelling. A proposal from the Swedish EPA is that the national modelling work by SMHI should play an important part in this, but it is still unclear exactly what will enter into Swedish law.

In this project, we have followed Alpfjord Wylde et al. (2023) and applied the above methodology to quantify population exposure and health impacts for NO₂, PM_{2.5}, PM₁₀ as well as O₃ for the years 2023 and 2030. For the year 2023, we use the same results as Windmark et al. (2025), and for 2030 we apply new emissions but keep the same meteorology as 2023. The resulting concentrations have been produced at a spatial resolution of 50 m x 50 m along with a range of other resolutions to study the effect that spatial resolution has on the results. Exposure and health impact calculations as well as a health economic assessment have then finally been made based on population data as point coordinates.

In section 2, we give a brief summary of the modelling assumptions with a focus on emissions assumptions done for scenario year 2030 as well as the exposure and health impact and economic assessment calculations. In section 3, we show results from the validation of O₃ that was introduced in this project and in section 4, we discuss the resulting concentration levels, population exposure and health impacts. Finally, in section 5, we present and discuss the results.

2 Methodology

The dispersion methodology follows a chain of models from regional down to urban scale. Relevant measures are produced in this way for NO₂, PM₁₀, PM_{2.5} and O₃ covering all of Sweden for the years 2023 and 2030. These models depend greatly on input data such as emissions, traffic, physiography, building geometries and meteorology. Exposure calculations are then done by combining the concentrations with population data, and finally the health impact assessment is carried out.

These points are all discussed in more detail in the sections below.

2.1 Emission data

For emissions over the rest of Europe, we use the gridded EMEP (European Monitoring and Evaluation Programme)⁴ data at an 0.1 ° x 0.1 ° resolution for 2023 and gridded ECLIPSE v6b⁵ emissions at

³ Luftwebb: <https://www.smhi.se/data/luftkvalitet/nationell-modellering-av-luftkvalitet>

⁴ EMEP emission database (emission year 2023): <https://www.ceip.at/webdab-emission-database>

an 0.5 ° x 0.5 ° resolution for the year 2030 (highest available resolution). This is supplemented with the daily forest fire emission data from the CAMS Global Fire Assimilation System (CAMS-GFAS). The different emission sectors are defined according to the GNFR activity categories⁶.

Over Sweden, we use the gridded SMED (Svenska MiljöEmissionsData)⁷ emissions at a higher resolution of 1 km x 1 km. To represent the year 2023, we use the then latest available (at the time of modelling) SMED emission year of 2021 for both Swedish and European sources. Further improvements were done to the geographical distribution of small-scale residential heating, industry and energy point sources as well as domestic and international shipping (described in more detail in Grundström et al., 2023 and Windmark et al., 2025). To get emissions for 2030, we scale the modified geographically distributed dataset with national total emissions obtained from SMED for 2030. The scenario has been described in a report by the Swedish EPA⁸.

At the high spatial resolution needed for this project, the emissions from the critical road transport sector were instead calculated based on road network vector data with traffic flows for 2023 from the Swedish Transport Administration (Trafikverket). The road network is then combined with vehicle fleet data for each road type and the corresponding set of emission factors in HBEFA (Handbook Emission Factors for Road Transport), a step done by the Swedish Environmental Institute (IVL). This project has used the latest available HBEFA version 4.2 (Notter et al., 2022), which was originally released in 2022. This is the same underlying methodology as the data published by SMED, but allows for significantly higher spatial resolution. For 2030, we scale the traffic from 2023 to 2030 based on the traffic forecast from Trafikverket (2022), on the same road network and use 2030 emission factors from HBEFA 4.2.

Sector-wise emissions of NO_x, PM₁₀ and PM_{2.5} from Swedish sources for 2023 and 2030 are shown in Figure 1. For particle emissions, the NORTRIP model uses traffic and meteorology to model non-exhaust emissions. These emissions are not shown in Figure 1 since the NORTRIP model is run as part of the atmospheric dispersion model and the results are not part of the emission inventory database. Thus, the total emissions from road transport are higher than depicted in the figure.

The fraction of studded tyres on road vehicles is an important parameter to determine PM emissions for Swedish conditions as the emission factors for studded tyres are significantly higher than for studless winter tyres. This data comes from manual tyre counts performed in 18 cities over six regions published by the Swedish Transport Administration. For 2023, we calculate the fractions based on the average for the years 2022-2024. As there are clear and continuous trends of the development of the fraction over the timeline of recorded data 2015 to 2024, to get values for 2030, we extrapolate the time series.

The results for both years are shown in Table 1 for the regions and in Appendix 1 (Table 11) for the individual cities. Generally, we see a gradient of increased use of studded tyres from south to north, with the lowest fraction in REG SOUTH (REG SYD) (35 %) and Malmö (13 %) to the highest fraction in REG NORTH (REG NORD) (95 %) and Luleå (97 %). The extrapolation results in relatively small adjustments between the two years for all regions but REG STOCKHOLM and the largest of the cities (Stockholm, Gothenburg, Malmö and Uppsala).

⁵ ECLIPSE emission database (emission year 2030):
<https://iiasa.ac.at/models-tools-data/global-emission-fields-of-air-pollutants-and-ghgs>

⁶ GNFR activity codes:
https://www.ceip.at/fileadmin/inhalte/ceip/1_reporting_guidelines2014/annex_i_rev18-11.xlsx

⁷ Geographically distributed emission data from SMED (submission 2023, emission year 2021):
<https://www.smhi.se/data/miljo/nationella-emissionsdatabasen>

⁸ Trender för utsläpp av luftföroreningar - Analys av statistik och scenarier inrapporterade 2023: <https://www.naturvardsverket.se/48f8a0/globalassets/amnen/luft/trendanalys-utslapp-till-luft--data-rapporterade-2023.pdf>

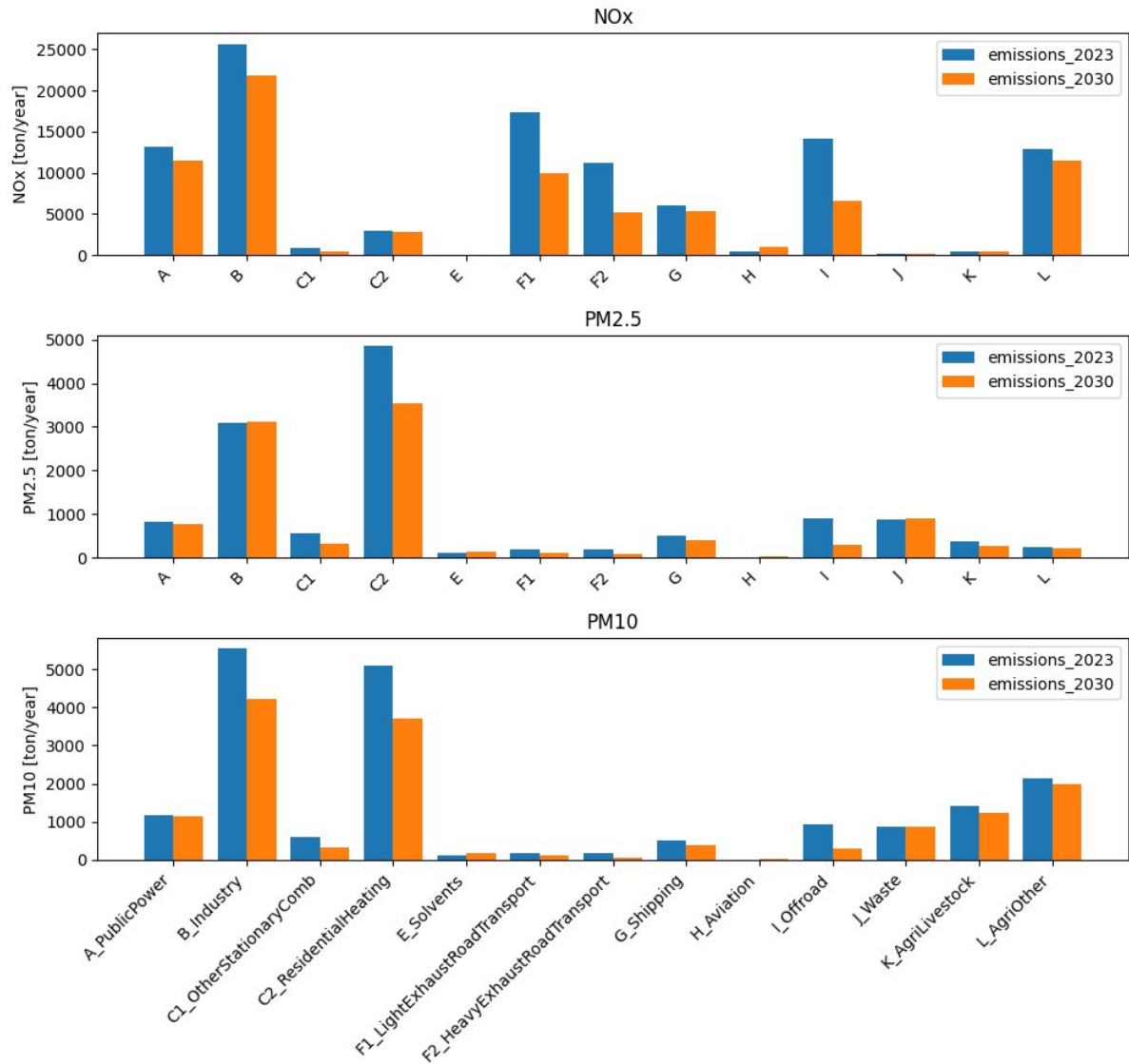


Figure 1. Comparison of 2023 emissions to 2030 emissions of NO_x , PM_{10} and $\text{PM}_{2.5}$ in Sweden. Non-exhaust particle emissions are not shown, as the results are produced through the atmospheric dispersion model and hence are not part of the emission inventory database. Thus, the total emissions from road transport are higher than depicted in the figure.

Table 1. Measured and assumed usage of studded tyres assumed for 2023 and 2030 in the larger regions as defined by the Swedish Transport Administration.

Region	2023 (%)	2030 (%)	2030/2023 (ratio)
REG SOUTH (SYD)	35.3	34.2	0.97
REG WEST (VÄST)	53.0	45.9	0.87
REG EAST (ÖST)	64.0	60.0	0.94
REG STOCKHOLM	43.7	28.5	0.65
REG MID (MITT)	89.7	91.3	1.02
REG NORTH (NORD)	95.3	95.0	1.00

2.2 Meteorological data

Meteorology is perhaps the strongest driver between year-to-year variations in air quality as the emissions tend to shift more gradually. The highest concentrations of air pollution are often created during cold and stable weather, whereas wet and windy weather tends to decrease the concentrations. Meteorology affects the dispersion, but precipitation, or more specifically road wetness, has a substantial effect on the emission of road dust handled by NORTIP.

In the regional MATCH model, 3-dimensional meteorological data for all of Europe is taken from the ECMWF weather model HRES⁹ (0.1 ° horizontal resolution). For the urban dispersion modelling, the ECMWF data is combined with hourly 2-dimensional data over Sweden from the MESAN¹⁰ model (2.5 km resolution) together with global radiation from STRÅNG¹¹.

In this work, meteorological data for year 2023 has been used to drive the modelling for both 2023 and 2030. This allows us to study the effects of the changed emissions while removing the effects of the year-to-year variations in meteorology.

2.3 Dispersion calculations

The dispersion methodology follows to a large degree the same approach as Grundström et al. (2023) that was further improved by Windmark et al. (2025). In short, the model set-up relies on regional modelling performed by the MATCH model at a resolution over Sweden of 5 km x 5 km. These results are then post-processed by a bias-correction scheme, which adjusts the modelled results to measurement data, followed by the BUDD scheme (Backtrace Upwind Diffuse Downwind; Segersson, 2021) to remove the urban contribution from the MATCH results to produce the regional background contribution only.

The urban contribution is then produced by the NG2M model (Next Generation Gaussian Model) at a resolution of up to 50 m x 50 m all over Sweden. These concentrations are modelled hour-by-hour, but due to disk usage considerations, the time series data is removed when all statistics of interest have been calculated. Throughout this modelling chain, concentrations are adjusted using regional and urban scale observations. For a more detailed description of the dispersion methodology, see Windmark et al. (2025).

2.3.1 Note on the post-processing of regional background concentrations

Due to an error in the post-processing step, incorrect MATCH results were unintentionally used for 2023. After regional bias-correction, the impact on the 2023 regional background results was negligible. However, as no actual bias-correction can be done for a future scenario due to the lack of measurement data, and the results from the bias-correction of the 2023 data were used to approximate the bias of the 2030 calculations, the final 2030 regional background concentrations turned out to be slightly overpredicted as an effect of the error in the 2023 data.

As a quality control, urban concentrations were recalculated for 2030 for a selection of highly polluted areas. A comparative analysis revealed that the yearly average concentrations, when using the correct results, was 97.2 % for NO₂, 98.8 % for PM_{2.5} and 99.1 % for PM₁₀ compared to the incorrect results. The maximum differences observed were 0.17 µg/m³ for NO₂, 0.083 µg/m³ for PM_{2.5} and 0.10 µg/m³ for PM₁₀. For O₃ there was no difference.

Despite the slight overprediction in the reported concentrations (based on the incorrect MATCH results), the study's overall conclusions regarding concentration shifts and health impacts between 2023 and 2030 remain valid when compared to the correct MATCH results. Additionally, it is important to

⁹ HRES: <https://www.ecmwf.int/en/forecasts/datasets/set-i>

¹⁰ MESAN: <https://www.smhi.se/data/sok-oppna-data-i-utforskaren/meteorologisk-analys-api>

¹¹ STRÅNG: <https://www.smhi.se/data/sok-oppna-data-i-utforskaren/yttaekande-modell-for-solstralning-strang>

note that modelling results inherently include uncertainties, particularly for future emission scenarios.

2.4 Exposure calculations

For the exposure calculations, we have used a combination of two datasets. One is a 100 m x 100 m resolution from Statistics Sweden that has been shared with SMHI, and the other is in the form of point coordinates where the detailed operations have been performed by Statistics Sweden without giving us access to the raw data.

2.5 Health impact assessment

2.5.1 Relative risk estimates

The health impact assessment (HIA) has mainly focused on PM_{2.5} because it has the strongest epidemiological and toxicological support, including concentration–response functions for mortality and morbidity. Some HIAs also include NO₂ and O₃ to capture additional health impacts and source-specific pollution patterns, but this requires careful handling of overlap, confounding, and greater uncertainty in the exposure–response estimates.

Due to the great variation in origin, composition, and physico-chemical properties across locations of fine particles, epidemiological studies differ in their ability to capture associations between specific particle types and mortality depending on the study setting, spatial resolution, and exposure metric used. For mortality the health impact of air pollution is therefore estimated separately for regional background concentrations of PM_{2.5}, local generated PM_{2.5}, locally generated wear particles in the PM₁₀ fraction, and NO₂.

Figure 2 presents the relative risk (RR) estimates used in the HIA for each included health outcome, pollutant, and underlying meta-analysis or epidemiological study. A log-linear concentration–response function was assumed within the reported exposure range. RR estimates per 10 µg/m³ increase in air pollutant concentration were extracted from the literature and are presented together with their corresponding 95% confidence intervals (CI).

The figure is structured as a forest plot, where point estimates (squares) represent the RR and horizontal lines indicate the 95% CI. A vertical reference line at RR = 1 denotes no association. Health outcomes are listed alongside the corresponding study and pollutant, and baseline frequencies (expressed as cases per 100 000 person-years) are provided to show the magnitude of disease occurrence in the population. For ASD, childhood asthma, COPD and allergic rhinitis baseline frequencies are based on prevalences. Mortality estimates are presented at the top of the figure. All other health outcomes are ordered by descending magnitude of the RR. This ordering highlights outcomes with stronger relative associations with air pollution exposure.

The inclusion of multiple studies for mortality and single-study estimates for other outcomes reflects the available epidemiological evidence.

Preterm mortality, incidence of myocardial infarction, stroke and lung cancer were estimated in persons older than 30 years of age. Diabetes was estimated in persons older than 15 years of age, COPD 50 years, and dementia 60 years. Sick days among 15–64-year-olds, and school absence days among 6–11-year-old. Asthma in children up to 18 years old.

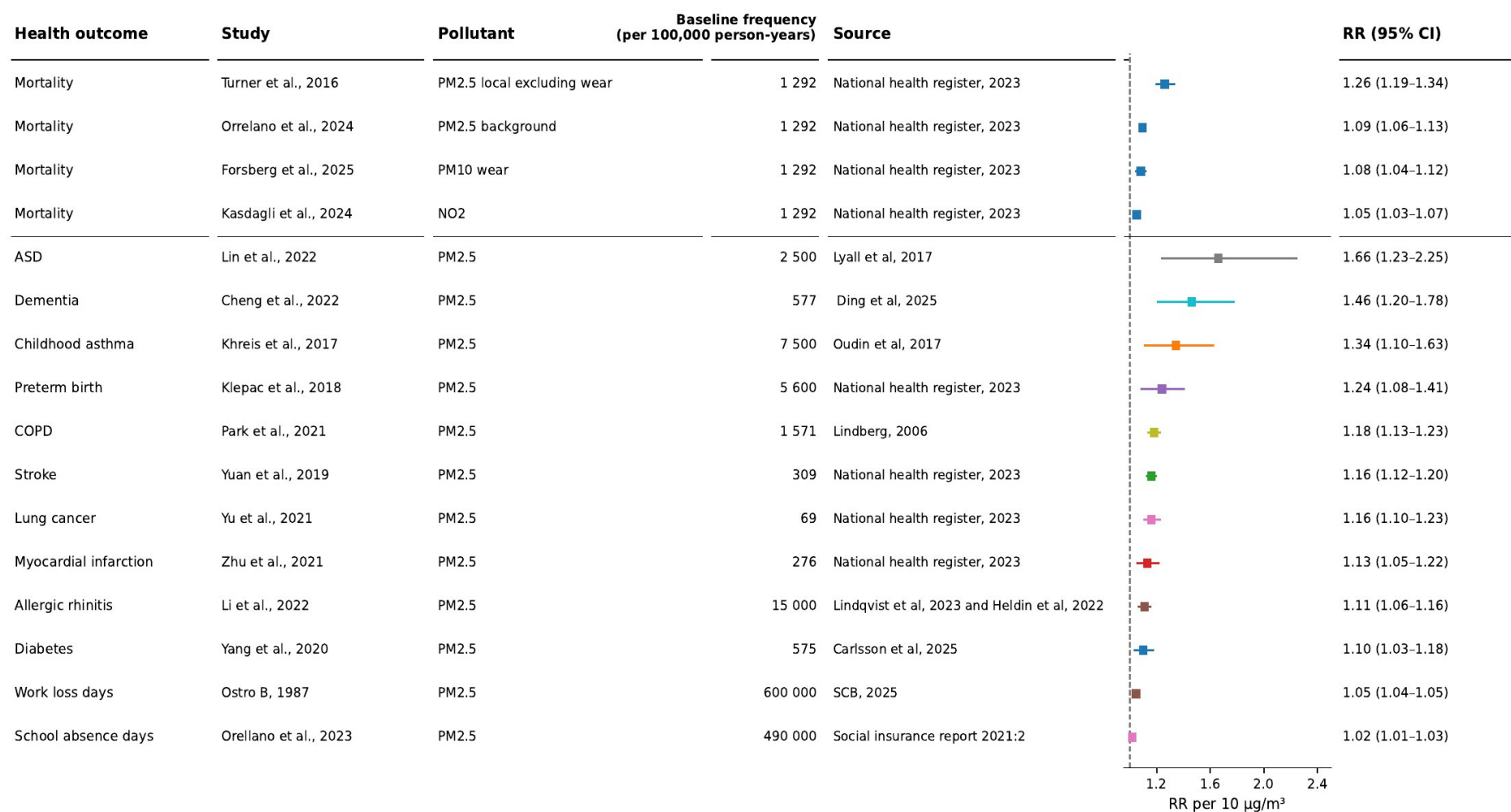


Figure 2. Relative risks (RR) for health outcomes associated with air pollution exposure. The figure presents RRs and 95% confidence intervals per 10 µg/m³ in pollutant concentration for mortality and morbidity outcomes. RRs are shown together with baseline incidence rates used in the health impact assessment. PM_{2.5} is the primary pollutant for most outcomes, while PM₁₀ is used for wear particles and NO₂ for traffic-related exposures. The forest plot displays central estimates and uncertainty intervals, with RR values >1 indicating increased risk. These concentration–response functions form the basis for the calculation of attributable cases in the health impact assessment.

2.5.2 Mortality health impact assessment

Regional background PM_{2.5}

Orellano et al. (2024) in their updated WHO-commissioned systematic review and meta-analysis of cohort studies, long-term exposure to ambient PM_{2.5} was associated with increased all-cause mortality, with a pooled relative risk of 1.095 (95% CI: 1.064–1.127) per 10 µg/m³ increase in PM_{2.5}, based on 53 effect estimates from 52 studies. The certainty of the evidence was rated as high. The authors also reported that the concentration-response relationship was generally linear or near-linear, with some studies suggesting a supra-linear shape at lower concentrations, and found no evidence of a safe threshold below which effects were absent. Although between-study heterogeneity was substantial, the direction of association was consistently positive, and sensitivity analyses excluding studies at higher risk of bias did not materially change the pooled estimate.

Local generated PM_{2.5}

Studies examining associations between local variation in long-term PM_{2.5} exposure and mortality in adult cohorts have consistently reported higher relative risks than are typically observed when all study types are considered together. Similarly elevated pooled estimates are obtained when studies conducted at relatively low PM_{2.5} concentrations are combined. This is likely because such studies, more than others, capture contrasts driven by local sources and within-city variation, rather than by differences in regional background concentrations.

For assessing the effects of local PM_{2.5} sources, particularly traffic and residential wood burning, we therefore apply a relative risk of 1.26 per 10 µg/m³ PM_{2.5}, using the 95% confidence interval reported by Turner et al. (2016).

Wear particles

Although health impact assessments generally prioritize PM_{2.5} over PM₁₀, this may lead to an underestimation of the effects of non-exhaust traffic emissions, since the coarse fraction constitutes a substantial share of exposure to wear particles. However, because current expert assessments do not support the derivation of source-specific relative risks for PM_{2.5} or PM₁₀, there is at present no generally accepted scientific basis for applying a different relative risk to traffic-related wear particles than to total PM₁₀. Based on the long-term exposure-response evidence judged most relevant for Swedish conditions, we therefore applied PM₁₀-based risk estimates to wear particles expressed as PM₁₀. The available evidence suggests that a relative risk of 1.12 per 10 µg/m³ is reasonable for all-cause mortality in adults aged 30 years and older (Forsberg et al., 2025). Because this estimate is not adjusted for NO₂, a reduction corresponding to that reported for PM_{2.5} in multi-pollutant analyses results in an adjusted relative risk of approximately 1.08 per 10 µg/m³, which was applied.

NO₂

The pooled results on long-term exposure to NO₂ and mortality from the WHO project Health Risks of Air Pollution in Europe (HRAPIE-2), published in October 2024, should be regarded as a highly influential recommendation regarding which exposure-response functions should be applied in health impact assessments (Kasdagli et al., 2024). One limitation is that the article does not provide pooled coefficients for estimates adjusted for PM_{2.5}. A possible solution is offered by the EU-funded project Urban Burden of Disease Estimation for Policy Making (Chen et al., 2024). When estimating the consequences of exposure changes that simultaneously affect both PM_{2.5} and NO_x (exhaust emissions), and when calculations for both pollutants are intended to be used and combined, the mortality effect estimate should be reduced by 22.6% for NO₂ and 32.3% for PM_{2.5}. At present, it appears appropriate to base the calculations on the pooled relative risk from HRAPIE-2 (Kasdagli et al., 2024), namely 1.05 (95% CI: 1.03–1.07) per 10 µg/m³, corresponding to a reduced relative risk of 1.038 after adjustment.

2.5.3 Morbidity

Morbidity impacts of long-term exposure to PM_{2.5} were estimated using the concentration-response functions (CRFs) recommended by Forastiere et al. (2024). That study selected CRFs from systematic reviews and meta-analyses based on causality considerations, methodological appraisal using AMSTAR2-EH, validation of meta-analytic input data, and assessment of confidence in the epidemiological evidence. For pre-term birth a meta-analysis estimate by 1.24 (1.08–1.41) per 10 µg/m³ PM_{2.5} was applied (Klepac et al., 2018). Gestational windows of exposure typically included were whole pregnancy period, 1st, 2nd, 3rd trimester, first and last gestational months.

Morbidity outcomes include outcomes with evidence category A but also include outcomes on the B+ list. Outcomes in category A provide the most robust and policy-stable estimates, while B+ outcomes represent endpoints where the evidence is sufficiently suggestive, biologically plausible, and relevant for population health, but where uncertainty remains larger. The inclusion of ASD and school absence in the B+ list is especially relevant. Even if the causal evidence is not yet as strong as for mortality, cardiovascular disease, asthma or lung cancer, these outcomes capture dimensions of health and societal burden that are important for children, families, schools and public services. Excluding these outcomes would risk underestimating the total health burden of air pollution, particularly for outcomes affecting children and long-term societal costs. Because most morbidity studies are based on within-city exposure contrasts, and because the lower end of the observed exposure range is usually above 5 µg/m³, these risk estimates have generally not been applied in health impact assessments to annual mean PM_{2.5} concentrations below the new WHO Air Quality Guideline of 5 µg/m³. This cut-off was applied also in this report.

For restricted activity days, the exposure-response function previously applied is based on older US cross-sectional studies using self-reported outcomes. This represents a comparatively weak study design, although these estimates have been widely used in earlier health impact assessments, including by the WHO in HRAPIE (Henschel & Chan, 2013).

2.5.4 Additional health outcomes in children

For allergic rhinitis, the evidence linking air pollution to allergic disease has become increasingly robust. A systematic review and meta-analysis of prenatal exposure reported pooled odds ratios of 1.36 per 10 µg/m³ PM_{2.5} for allergic rhinitis or hay fever in children (Ai et al., 2024). A broader systematic review found pooled odds ratios of 1.11 per 10 µg/m³ PM_{2.5}, mainly in children and adolescents (Li et al., 2022). Evidence also suggests that improved ambient air quality is associated with a reduced prevalence of childhood allergic disease (Chen et al., 2024). From a life-course perspective, allergic rhinitis appears to increase in prevalence from childhood into adolescence and early adulthood before declining later in life due to remission, which supports the view that it represents a substantial burden during the school-age years and beyond (Lindqvist et al., 2024; Heldin et al., 2022).

Autism spectrum disorder is another outcome for which ambient PM_{2.5} exposure appears relevant. Meta-analyses have reported elevated risks associated with exposure during pregnancy, infancy, or early life, with pooled estimates such as 1.34 per 10 µg/m³ PM_{2.5} and 1.22 per 10 µg/m³ PM_{2.5} in recent reviews (Liu et al., 2023; Amnuaylojaroen et al., 2024). A further systematic review and meta-analysis on particulate matter and neurodevelopmental disorders has also been identified as an important basis for risk assessment (Lin et al., 2022), and a broader evidence synthesis has supported the inclusion of ASD among child outcomes relevant to long-term air-pollution health assessment (Forastiere et al., 2024). Although prevalence estimates vary across studies and settings, ASD is generally considered sufficiently common for even modest air-pollution-related increases in risk to have substantial implications at population level (Lyall et al., 2017; Christensen et al., 2016; Maenner et al., 2023). This is especially important because ASD is associated with lifelong consequences for health care, education, social services, families, and labour market participation.

School absenteeism is also a meaningful and policy-relevant indicator of child health burden. A systematic review and meta-analysis of studies on restricted activity days found that school absence

increased in association with PM_{2.5} exposure, with a pooled estimate of 1.0173 per 10 µg/m³ PM_{2.5} (Orellano et al., 2023). Supporting studies have shown that particulate pollution is associated with higher absence rates and wider economic consequences, including lost school resources and parental work absence (Mendoza et al., 2020; Deng et al., 2023). School absenteeism is therefore useful not only as an indicator of children's impaired daily functioning, but also as a proxy for indirect productivity losses among caregivers.

2.5.5 Socio-economic valuation

At the valuation stage, the health costs of air pollution are assessed from a societal perspective, meaning that all relevant costs arising from an attributable health outcome are included. These costs can be grouped into three broad categories: healthcare and treatment costs, productivity losses, and losses in quality of life.

The same socio-economic cost per outcome as in Gustafsson et al. (2022) was applied, but in year 2023 value. The cost for a school absence day was assumed to be the same for a work loss day.

2.5.6 Allergic rhinitis

The costs of allergic rhinitis are based on the study by Cardell et al. (2016). That study used a questionnaire survey among adults aged 18–65 years in Sweden. The average total cost, including both direct and indirect costs, was estimated at €961.1 (2014 prices) per person-year for individuals with allergic rhinitis. Total costs were estimated to be 3.8 times higher for individuals with moderate or severe allergic rhinitis than for those with mild disease; moderate or severe allergic rhinitis accounted for 62% of cases. As much as 70% of the total costs were attributable to productivity losses. For full-time students, the direct costs of healthcare and medication were estimated at an average of €284.4 (2014 prices) per year.

To estimate treatment and productivity costs, we include only the direct healthcare and medication costs of €284.4 per year before age 18, and €961.1 per year thereafter. We assume that prevalence between ages 7 and 17 years increases from 10% to 20%, corresponding to an average prevalence of 15% among children in this age range. Summed over 10 years, this gives a cost per case of €2 844 (2014 prices). Assuming that 20% have allergic rhinitis at age 18, and that half of these cases have remitted by age 50 under a linear remission pattern, the average disease duration between ages 18 and 68 years is 37.5 years. This implies a cost per case of €961.1 per year. No treatment or productivity-loss costs are included after age 68, since prevalence and consequences are less well described in older adults (Bozek, 2017; Baptist et al., 2016), and because productivity losses constitute the dominant cost component in Cardell et al. (2016).

2.5.7 Autism spectrum disorder

Autism spectrum disorder (ASD) is associated with substantial long-term costs for the healthcare system and social services, as well as financial costs for affected individuals, their families, and society more broadly. We therefore propose using lifetime costs for ASD. Buescher et al. (2014) estimated these in the UK at £920 000 (2012 prices) per case without intellectual disability and £1.5 million (2012 prices) per case with intellectual disability. A more recent study by Zhao et al. (2024) estimated the corresponding lifetime costs in China at \$2.65 million (2020 prices) and \$4.61 million (2020 prices), respectively. Both studies included direct, indirect, and social costs. The estimates reported by Buescher et al. (2014) are considered more transferable to Sweden, since the Swedish healthcare, welfare, and education systems are more similar to those of England than to those of China. Assuming that 30% of ASD cases involve intellectual disability, the average lifetime cost is £1 094 000. Other cost studies on ASD have been limited to direct childhood costs (Järbrink, 2007; de Oliveira & Tanner, 2023) and do not account for, for example, costs related to adapted education, parental productivity losses, or productivity losses in adulthood among individuals with ASD.

To estimate quality-adjusted life years (QALYs), we propose using quality-of-life weights from Klang et al. (2022), who conducted a questionnaire study among adults diagnosed with ASD in Sweden. A total of 110 adults with ASD participated, and the reported EQ-5D index was 0.60. Since ASD prevalence increases up to adolescence, the average duration has been assumed to be 66 years.

3 Validation of the air quality modelling

Validation of the modelling results is critical to ensure high quality of the end results. As this work to a large degree is based on the work by Windmark et al. (2025), we focus the validation in this report on O_3 and refer to the previous work for validation of the other pollutants. A validation for O_3 was carried out where model results were compared to urban background measurements as well as local traffic sites. In the final results, a bias correction factor is applied to the urban concentrations, but the validation of the urban stations is performed before this step.

3.1 Data quality objectives

As the new EU directive on ambient air quality (2024/2881) will soon come into effect, we focus our validation against the modelling quality objective (MQO) (European Commission, 2025) in the new directive.

While the indicators RDE and RPE of the old directive are simple to apply, many factors are not considered with such unrefined indicators. For example, factors such as differences in the timing of events between measurement and model, and the fact that the measurement uncertainty differs depending on pollution level, cannot be evaluated with the indicators RDE and RPE. The MQO instead involves a number of statistical indicators, developed through the DELTA tool¹² software within the framework of FAIRMODE¹³. In this, a number of statistical indicators are calculated together and define a minimum level of quality to be achieved by a model for policy use. The MQO is constructed on the basis of the observation uncertainty.

How well model results compare to measurements at a station, according to the MQO, can be visualized in a Target diagram. If the station is plotted inside the coloured circle ($T < 1$), the MQO is fulfilled. The diagram also provides information about whether the model error is dominated by bias (either negative or positive) or by correlation or standard deviation. The recommended minimum number of stations for the evaluation of the MQO is 10 stations (European Commission, 2025), and MQO must be fulfilled for at least 90% of the available stations.

3.2 Validation of O_3

The validation results for O_3 are divided into validation at urban background sites and at local traffic sites.

3.2.1 Validation at urban background sites

Validation of O_3 is made against the uncorrected urban background concentrations of eleven measurement stations from Malmö Rådhuset as the southernmost station to Falun Östra Falan as the northernmost station. Similar to NO_2 , it is problematic that the entire northern half of Sweden is unrepresented by urban background measurements.

There is notably one station in Stockholm, Danderyd Mörbyskolan, that was excluded due to low data coverage. Further, we excluded another station in Stockholm, Danderyd Gymnasiet, because its location adjacent to a parking lot may introduce localised influences, not making it ideal as an urban background station.

¹² DELTA tool: https://knowledge4policy.ec.europa.eu/fairmode/delta-tool_en

¹³ FAIRMODE (Forum for Air Quality Modelling in Europe): <https://fairmode.jrc.ec.europa.eu/>

Figure 3 shows the observed and uncorrected modelled annual mean concentrations of O_3 at each measurement station. In general, the model overestimates the O_3 levels as compared to observations. The model performs really well at some sites, e.g. Djursholm Vasaskolan, Stockholm Torkel Knutssongatan, Ytterby, Rödeby, Staffanstorp and Mölndal Göteborgsvägen, where the overestimation of the modelled O_3 concentrations is less than 10 %. The largest differences are seen for Malmö Rådhuset (+26 %), Falun Östra Falan (+22 %), Helsingborg Norr (+18 %) and Lund Spyken (+17 %). The regional contribution gives the most significant contribution to the total levels, but these overestimations on the urban scale could be an indication that the ozone titration in NG2M, where NO reacts with and consumes O_3 , could need some calibration.

Figure 4 shows the Target plot of O_3 at the urban background stations. All 11 stations lie within the $MQI < 1$ circle and the MQO is fulfilled at both hourly/daily (MQI_HD) and yearly (MQI_YR) time resolution.

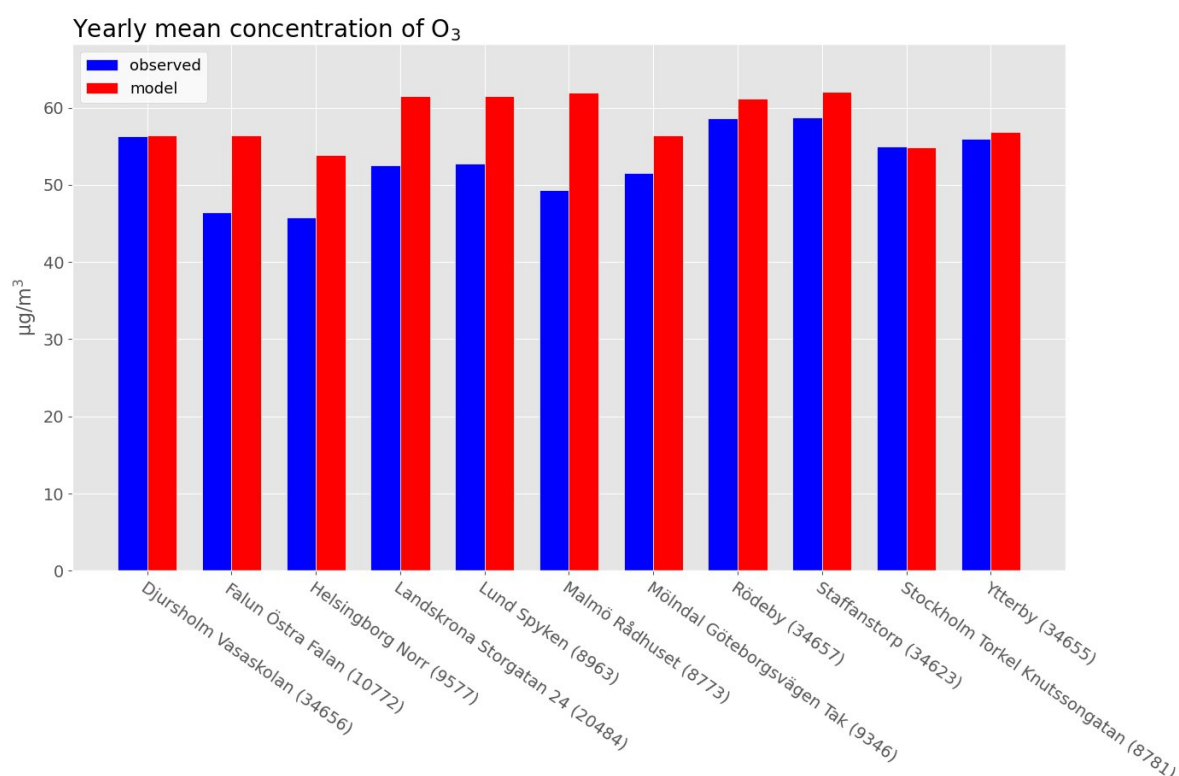


Figure 3. Observed and uncorrected modelled annual mean of O_3 at urban background sites. Modelled results include a regional and urban contribution. Measurement stations with lower data coverage than 75% have been excluded.

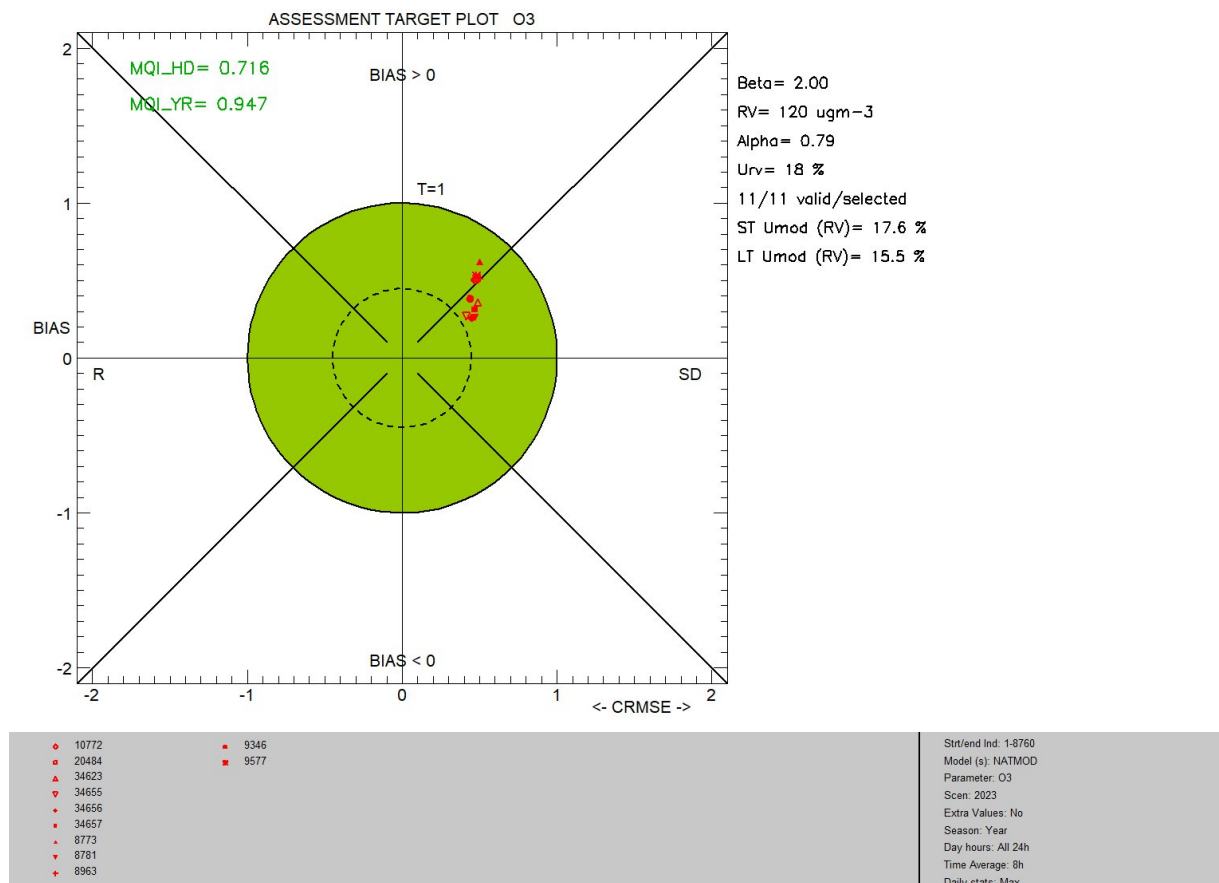


Figure 4. Target plot for O₃ at urban background sites with hourly time resolution data, without urban correction. The plot was created using DELTA Tool version 7.3 (Thunis et al., 2012).

3.2.2 Validation at local traffic sites

The validation at local traffic sites is made against the two available measurement stations: Malmö Dalaplan and Lund Trollebergsvägen. One station in Stockholm, Danderyd Vasaskolan, was excluded from this validation due to poor data coverage. As these two stations both lie in the southernmost part of Sweden, this unfortunately gives us relatively little information on how well the street canyon model performs.

Figure 5 shows the observed and modelled annual mean concentrations of O₃. The model overestimates the O₃ levels as compared to observations similarly to the urban background stations. Comparing the modelled values including urban correction (black bars) to the observations, the overestimation is roughly 10 %. We hypothesise that emissions of NO_x (nitrogen oxides) and especially NO could be underestimated by the model, as seen by the tendency towards underestimation in the validation of NO₂, leading to insufficient removal of O₃ in the street canyons.

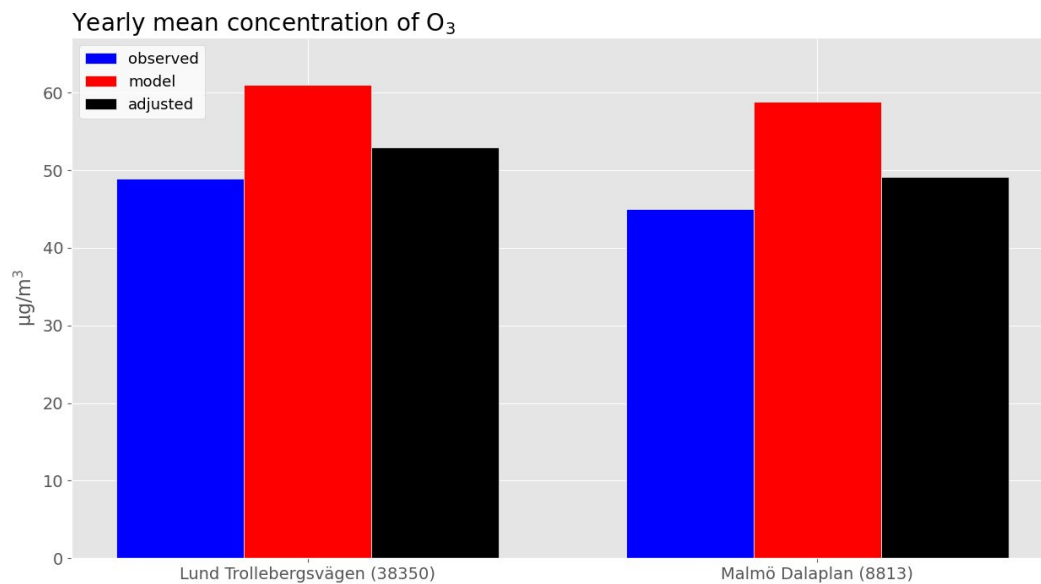


Figure 5. Observed and modelled annual mean of O_3 at local traffic sites. Modelled results include a regional and urban contribution. Red bars signify modelled values without urban correction. Black bars signify modelled values including urban correction. Measurement stations with lower data coverage than 75% have been excluded.

The low number of stations does not allow for application of MQO for statistical reasons (see section 3.1). We did however decide to show the Target plot (Figure 6), to provide insights into the potential of fulfilling MQO at local traffic sites would the stations have been more numerous. In the Target plot, both points lie within the $MQI < 1$ circle, demonstrating that the model is able to capture measured O_3 concentrations well at these two stations.

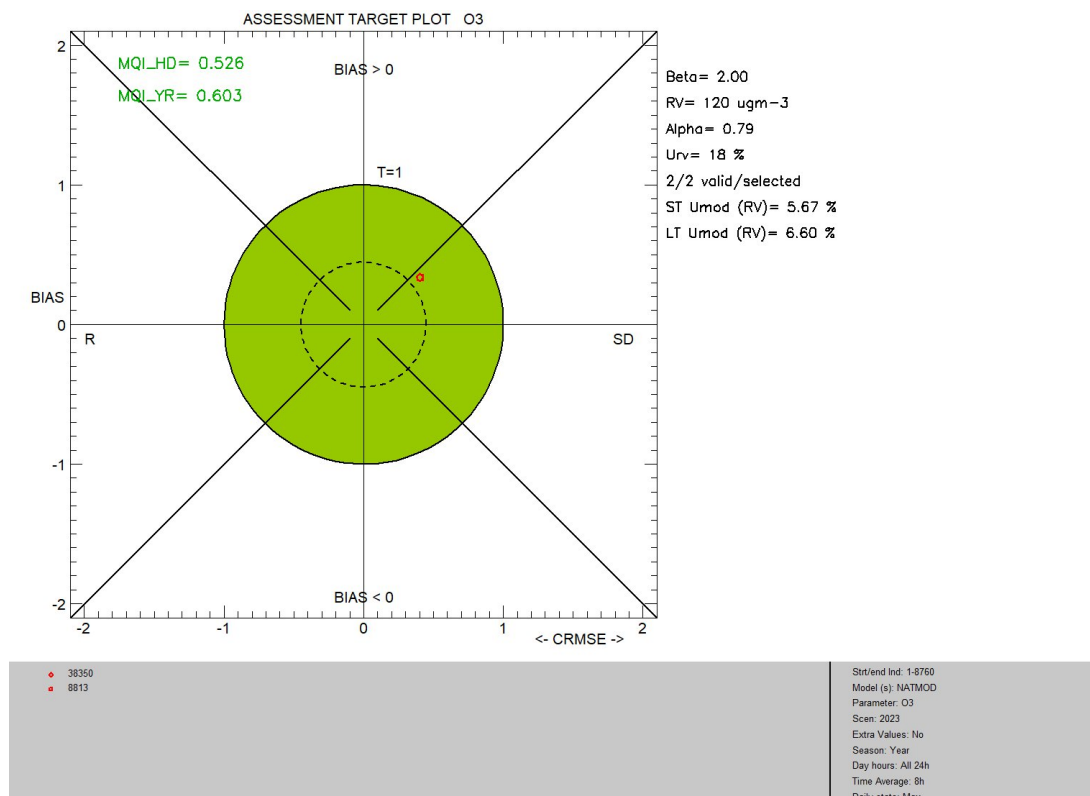


Figure 6. Target plot for O_3 at local traffic sites with hourly time resolution data, with urban correction. Note: both points are stacked on top of each other. The plot was created using DELTA Tool version 7.3 (Thunis et al., 2012).

4 Results and discussion

This section presents the results for year 2023 and 2030, with sections divided into discussions on concentrations and exposure for all pollutants (NO_2 , $\text{PM}_{2.5}$, PM_{10} and O_3) and estimated health impacts based on NO_2 , $\text{PM}_{2.5}$ and PM_{10} .

4.1 Concentration

As the health impact is calculated based only on annual mean concentrations of pollutants, we focus on the presentation of this metric rather than percentiles of daily and hourly mean concentrations.

The discussion below is divided into subsections for each pollutant, and we look at the concentrations from the regional and urban contributions both on a national level as well as on an urban level for a few selected cities; Malmö, Gothenburg, Stockholm and Umeå. We also focus the evaluation comparing to only the limit values of the air quality standards (AQS) in the new EU directive on ambient air quality (2024/2881) and the recommended levels of the WHO air quality guidelines (AQG; World Health Organization, 2021), and only do comparisons to the thresholds related to the annual mean. The AQS limit values and AQG levels are presented in Table 2.

Table 2. The concentration levels corresponding to limit values of the air quality standard in the new air quality directive (2024/2881) and the recommended levels of the WHO air quality guidelines for the annual mean of NO_2 , $\text{PM}_{2.5}$ and PM_{10} (unit: $\mu\text{g}/\text{m}^3$). There are no defined limit values nor recommended levels for the annual mean of O_3 .

Pollutant	NO_2	$\text{PM}_{2.5}$	PM_{10}	O_3
Air quality standard (annual mean) ($\mu\text{g}/\text{m}^3$)	20	10	20	Not defined
Air quality guidelines (annual mean) ($\mu\text{g}/\text{m}^3$)	10	5	15	Not defined

Note that the evaluation does not include the effect of the street canyon contribution as it is unclear how it can be implemented into health effect studies. This latter contribution can be significant for NO_2 and PM_{10} , hence exceedances compared to the AQS and AQG might occur even though they are not seen in this study.

If the street canyon contribution is included, our results for 2023 indicate a total number of 47 municipalities that exceed the AQS (annual mean or 95.1th percentile of days), with 47 municipalities exceeding the PM_{10} standards, 7 municipalities exceeding the NO_2 standards and 1 municipality exceeding the $\text{PM}_{2.5}$ standards. Looking at exceedances of the AQG, another 125 municipalities would be affected, primarily due to $\text{PM}_{2.5}$.

For 2030, this increases to 57 municipalities that are estimated to exceed one or more AQS, with 57 municipalities exceeding the PM_{10} standards, 0 municipalities exceeding the NO_2 standards and 7 municipality exceeding the $\text{PM}_{2.5}$ standards. For exceedances of the AQG, another 127 municipalities are affected, primarily due to $\text{PM}_{2.5}$.

4.1.1 Concentration of NO_2

Figure 7 shows how the annual mean concentration of NO_2 varies throughout the country for 2023 and 2030. The cities of Malmö, Gothenburg, Stockholm and Umeå are also presented at a higher resolution in Appendix 2A.

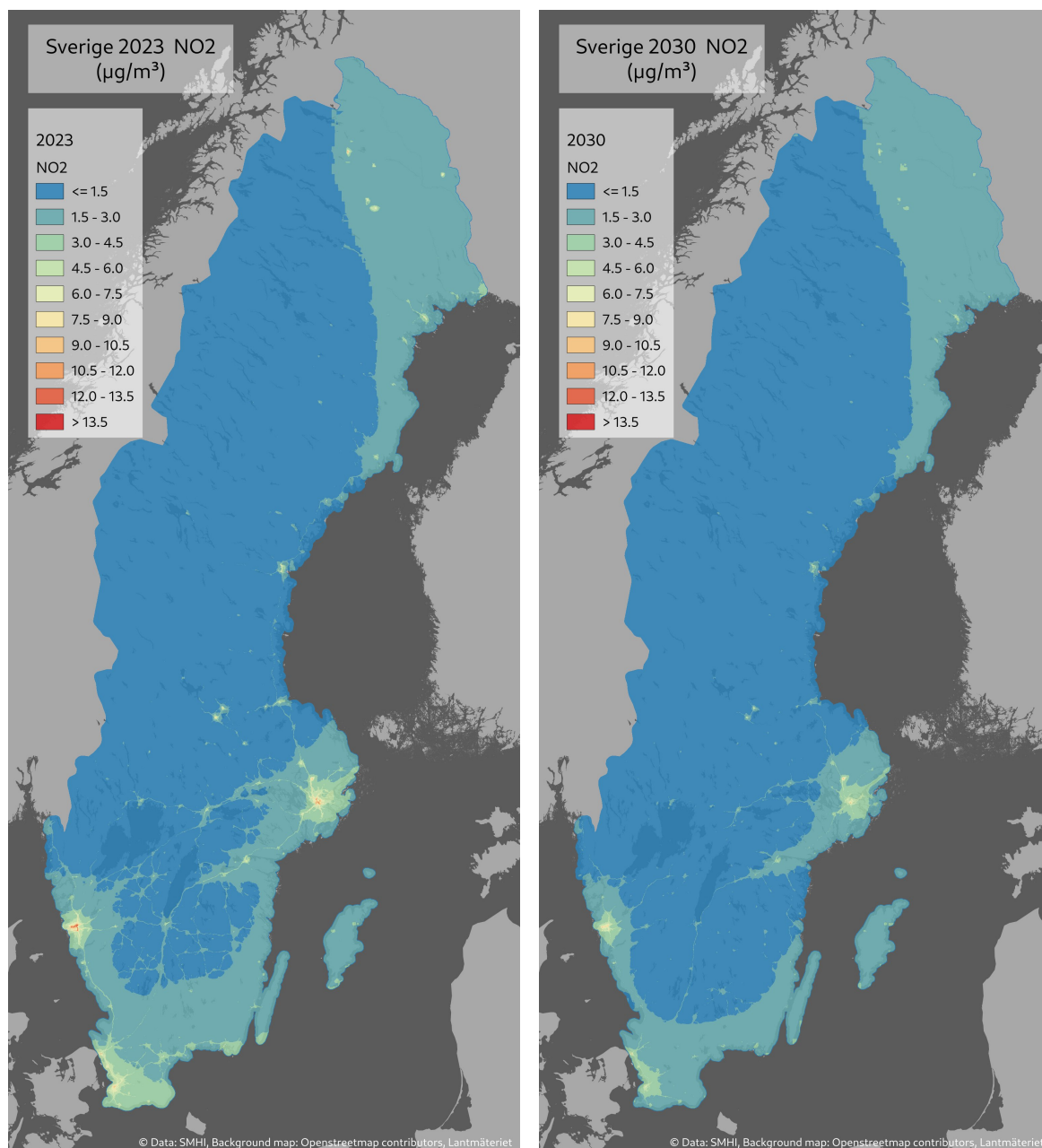


Figure 7. Concentration of NO₂ yearly mean for 2023 and 2030.

NO₂ is to a high degree a local pollutant, with the greatest contributions coming from transportation, industry and public power. The regional and urban background source contributions are illustrated for the selected cities in Figure 8 and Figure 9.¹⁴ For 2023, the regional contribution (from emissions at a distance of 10-20 km outside of the high resolution NG2M calculations) varies between 4 $\mu\text{g}/\text{m}^3$ in the south to 2 $\mu\text{g}/\text{m}^3$ in the north. The urban contribution in turn can reach 10-20 $\mu\text{g}/\text{m}^3$, with the highest contributions near the most well-trafficked roads.

A general decreasing trend is seen for NO₂. There are both regional and urban reductions from 2023 to 2030, with the largest reductions around the major roads of approximately 30-40 % and 10-20 % in rural areas. The reductions also tend to be larger in the south. Note that due to the unintentional error in the post-processing step of the regional background concentrations, described in section

¹⁴ Note that the for the urban contributions, NO_x is shown as an approximation of NO₂, since the NO_x-to-NO₂-chemistry model cannot calculate difference source contributions to urban NO₂.

2.3.1, the difference between the 2023 and 2030 results for NO₂ is slightly underpredicted (the concentrations for 2030 should be slightly lower than presented).

The electrification of the vehicle fleet as well as the improved emission reduction from internal combustion vehicles are the main contributors to the decrease of NO₂, despite increased traffic for 2030. While emission projections always carry some uncertainty, the observed decrease in this case is substantial enough to be considered reliable.

For 2023, exceedances of the new AQS of NO₂ for the yearly mean (excluding street canyon contributions) are limited to the few largest roads in or around the cities of Gothenburg, Borlänge and Stockholm. All of these exceedances are located in close vicinity to the roads (within 50-100 meters). For 2030, there are no exceedances of the NO₂ yearly mean AQS.

If this evaluation is extended to the AQG, exceedances for 2023 occur in more cities, including Malmö, Helsingborg, Jönköping, Norrköping, Falun, Sundsvall and Luleå as well as along major highways. As is the case for the AQS, exceedances are limited to areas nearest the larger roads. For 2030, there are small exceedances only in Malmö, Gothenburg, Stockholm and Luleå.

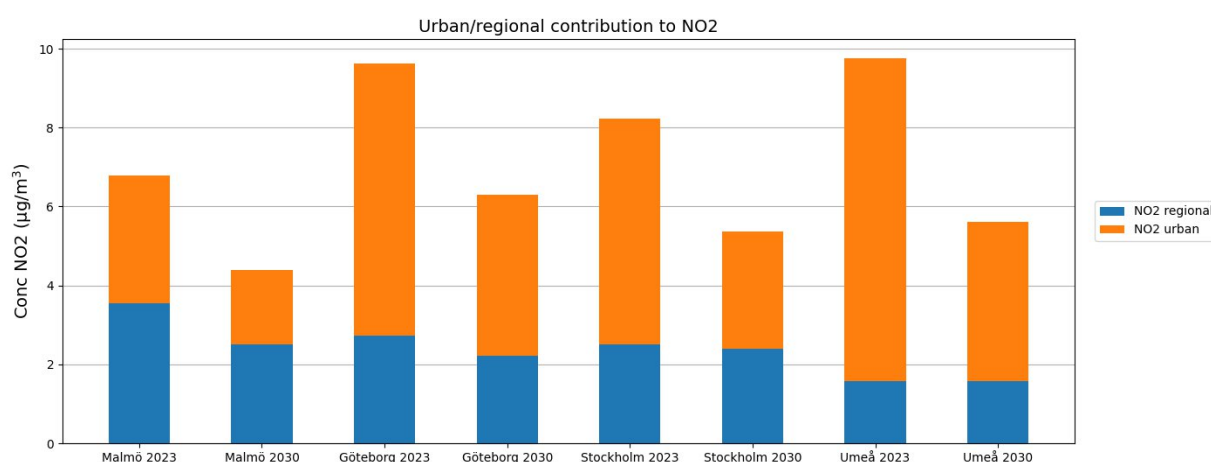


Figure 8. Regional and urban background contributions to NO₂ grid concentrations for 2023 and the 2030 scenario at urban monitoring station locations in the four cities Malmö, Gothenburg, Stockholm and Umeå in Sweden.

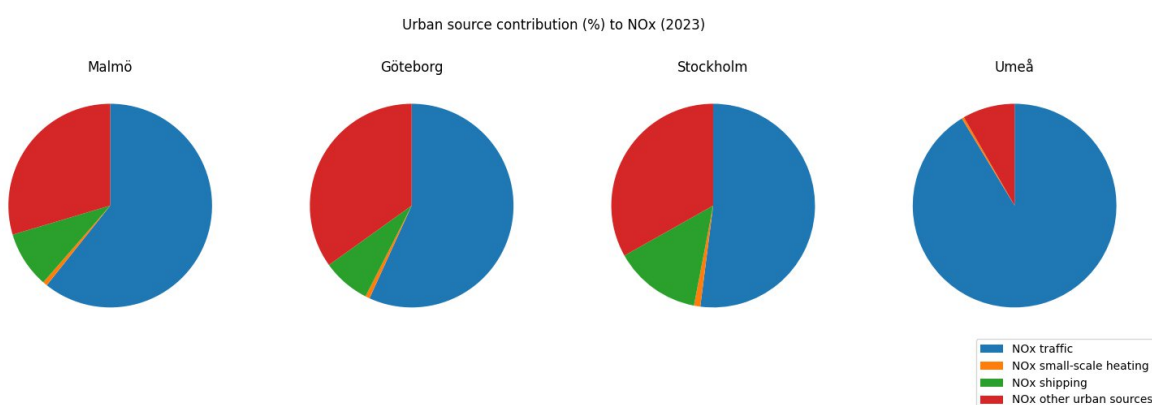


Figure 9. The urban background source contribution to NO_x (as an approximation of NO₂, see footnote 14) concentrations for 2023 are shown in pie charts for Malmö, Gothenburg, Stockholm and Umeå. Urban contributions from traffic exhaust, small-scale residential heating, shipping and other sources are presented.

4.1.2 Concentration of PM_{2.5}

Figure 10 shows how the annual mean concentration of PM_{2.5} varies throughout the country. The cities of Malmö, Gothenburg, Stockholm and Umeå are also presented at a higher resolution in Appendix 2B.

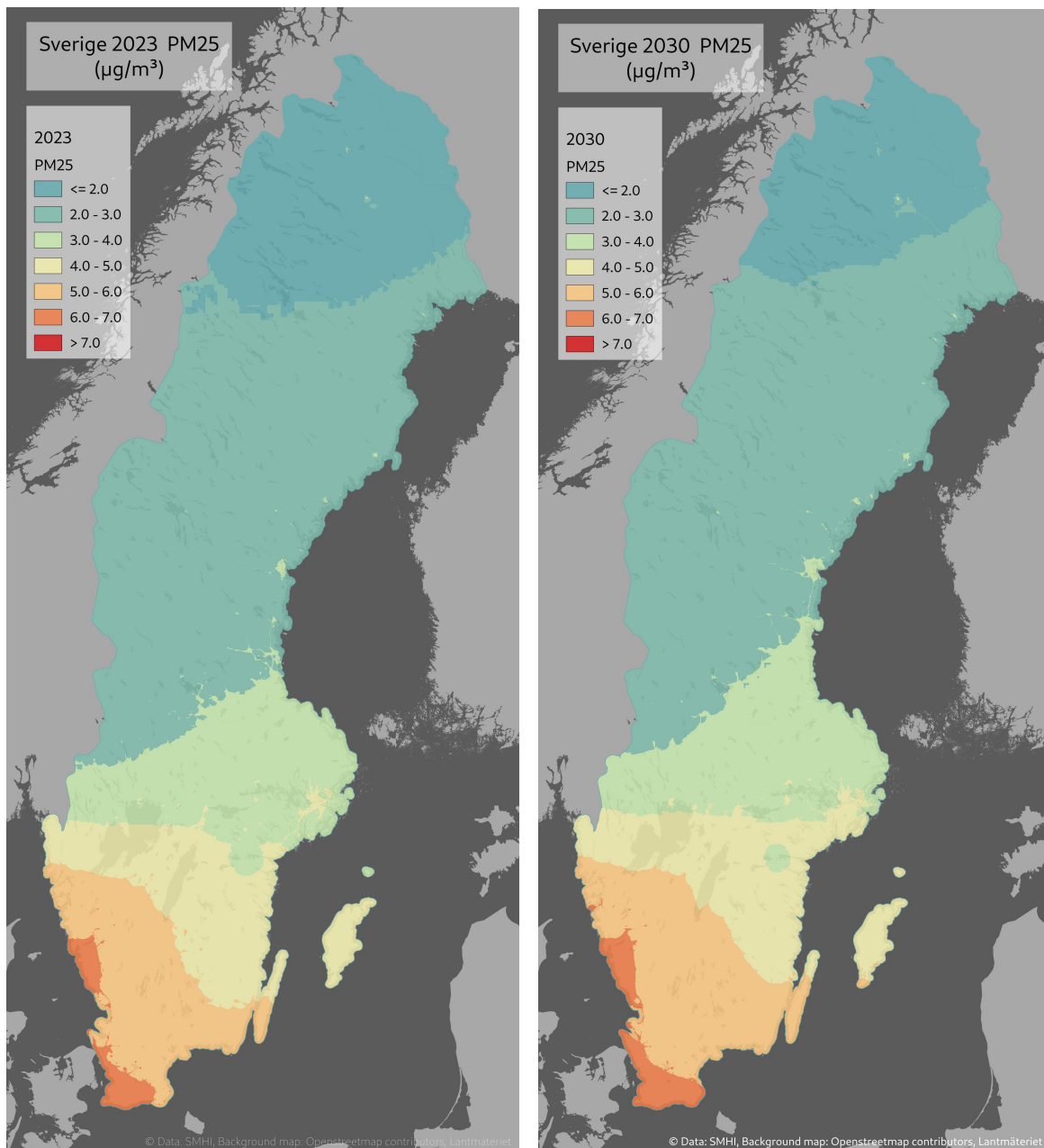


Figure 10. Concentration of PM_{2.5} yearly mean for 2023 and 2030.

Contrary to NO₂, PM_{2.5} is primarily a regional pollutant with contributions transported over large distances, with the largest urban contributions coming from wear particles and shipping. The source contributions are illustrated for the selected cities in Figure 11 and Figure 12. For 2023, the regional contribution ranges from 5-6 $\mu\text{g}/\text{m}^3$ in southern and western Sweden to 2 $\mu\text{g}/\text{m}^3$ in northern Sweden. The urban contribution in turn reaches at most 1-2 $\mu\text{g}/\text{m}^3$ in the cities.

In general, the calculations for the 2030 scenario show a small increase in PM_{2.5} concentrations compared to 2023. The sources are partly increased traffic and the resulting wear particles, partly a higher regional background contribution. However, note that due to the unintentional error in the post-processing step of the regional background concentrations, described in section 2.3.1, the dif-

ference between the 2023 and 2030 results for $PM_{2.5}$ is slightly overpredicted (the concentrations for 2030 should be slightly lower than presented). In addition to the inherent uncertainties in emission projections, the observed small increase is likely to rather reflect a stable trend in $PM_{2.5}$ concentrations from 2023 to 2030, with a possible slight upward tendency.

For both 2023 and 2030, no exceedances of the new AQS for the yearly mean (excluding street canyon contributions) are expected.

For the AQG, large parts of southern and western Sweden are expected to see exceedances due to the regional contribution. For 2023, these exceedances reach Trollhättan, Jönköping and Kalmar, with only very local exceedances around a handful of roads north of that. 2030 shows the same general pattern, with the exceedances reaching only slightly farther north beyond the cities mentioned above.

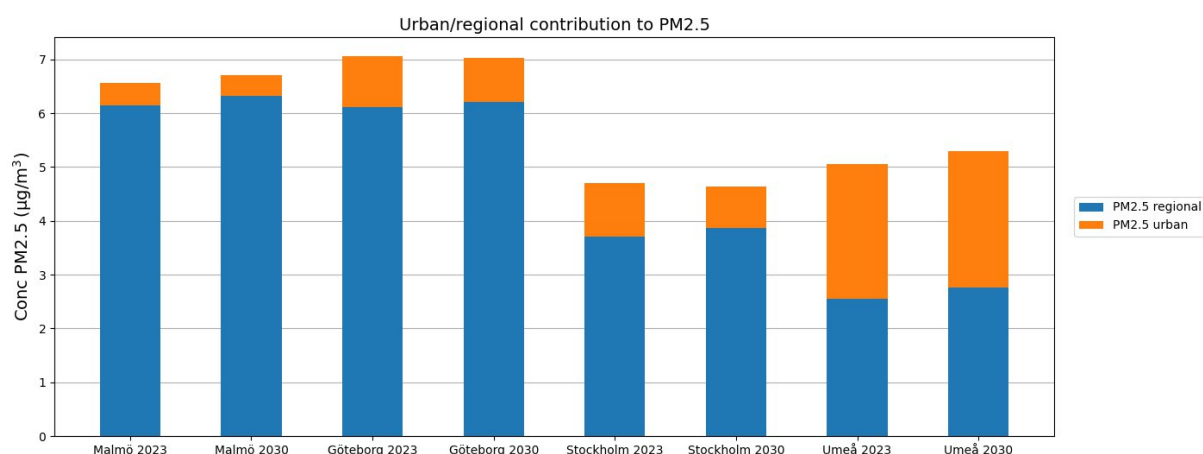


Figure 11. Regional and urban contributions to $PM_{2.5}$ grid concentrations for 2023 and the 2030 scenario at urban background monitoring station locations in the four cities Malmö, Gothenburg, Stockholm and Umeå in Sweden.

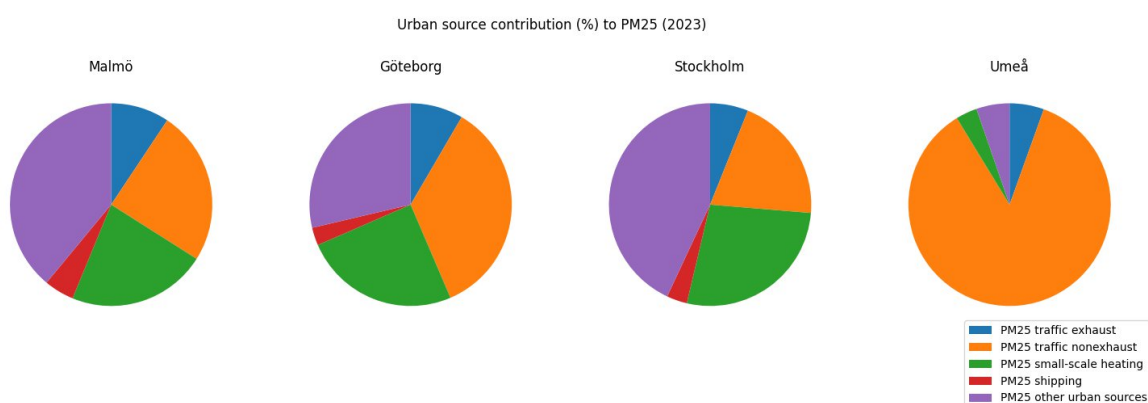


Figure 12. The urban source contribution to $PM_{2.5}$ concentrations for 2023 are shown in pie charts for Malmö, Gothenburg, Stockholm and Umeå. Urban contributions from traffic exhaust, traffic non-exhaust, small-scale residential heating, shipping and other sources are presented.

4.1.3 Concentration of PM₁₀

Figure 13 shows how the annual mean concentration of PM₁₀ varies throughout the country. The cities of Malmö, Gothenburg, Stockholm and Umeå are also presented at a higher resolution in Appendix 2C.

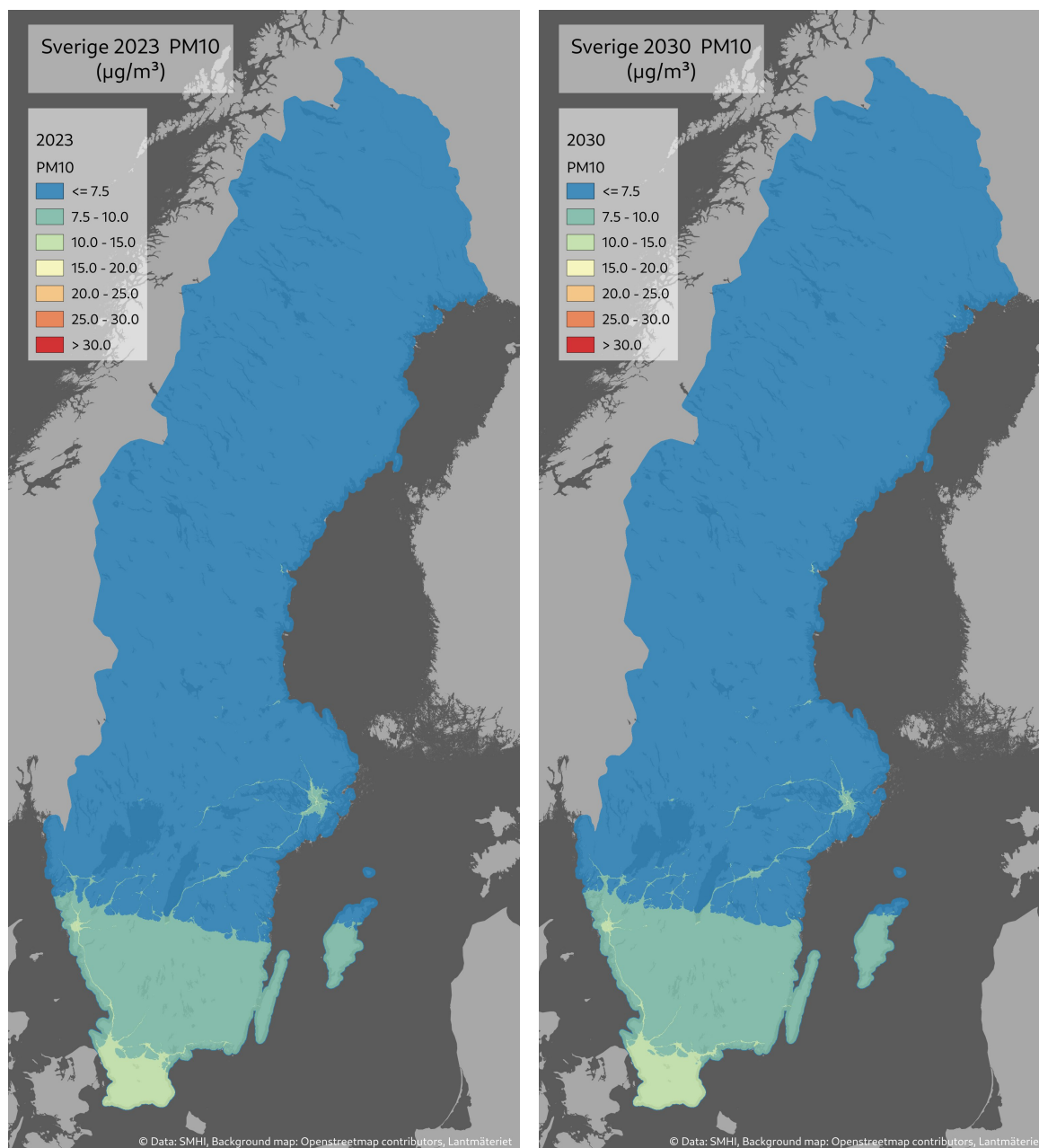


Figure 13. Concentration of PM₁₀ yearly mean for 2023 and 2030.

PM₁₀ is a varied pollutant with possibly significant contributions both from regional and urban sources, but the very highest concentrations are produced in the street canyons that are not included in this analysis. The regional and urban source contributions are illustrated for the selected cities in Figure 14 and Figure 15.

The largest sources are traffic and industry, but agriculture and public power are significant as well. In the south, the regional contribution is generally the most dominant. Farther north, however, the regional contribution decreases while the effect from road wear increases with the use of studded tyres (as seen in Table 1, for 2023, REG SOUTH (REG SYD) has a studded tyre use of 35 % while REG NORTH (REG NORD) has a use of 95 %). Comparing 2023 and 2030, a small increase in PM₁₀ concen-

trations is observed in the future scenario. However, note that due to the unintentional error in the post-processing step of the regional background concentrations, described in section 2.3.1, the difference between the 2023 and 2030 results for PM_{10} is slightly overpredicted (the concentrations for 2030 should be slightly lower than presented). As for $PM_{2.5}$, the sources are partly increased traffic and the resulting wear particles, partly higher regional background concentrations, and the small increase likely reflects a stable trend in PM_{10} concentrations from 2023 to 2030, with a possible slight upward tendency.

For 2023, exceedances of the new AQS of PM_{10} (again neglecting street canyons) for the yearly mean are, similarly to NO_2 , limited to the vicinity of the larger roads, but such exceedances occur more frequently within many cities all over the country. There are also additional cities all over the country with exceedances of only the AQG. For 2030, the number of roads and affected cities increase.

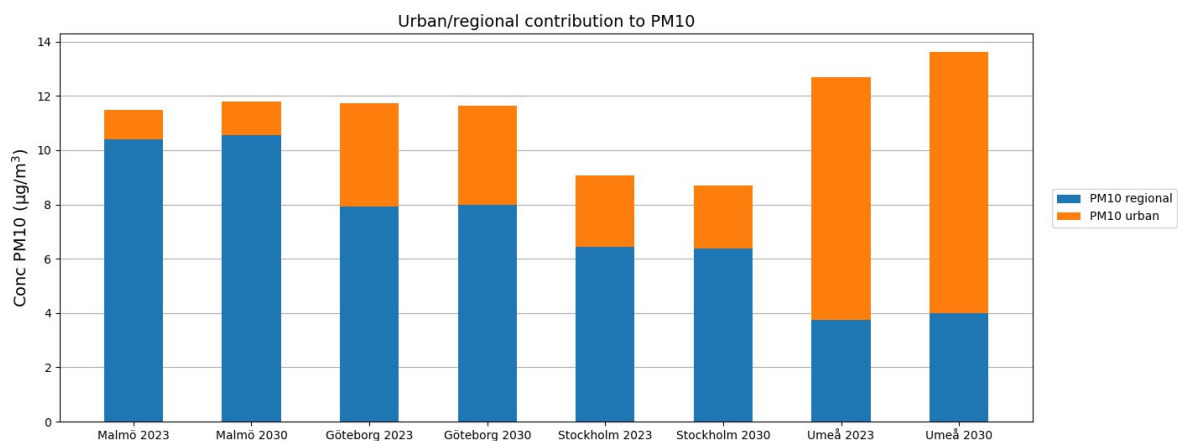


Figure 14. Regional and urban contributions to PM_{10} grid concentrations for 2023 and the 2030 scenario at urban background monitoring station locations in the four cities Malmö, Gothenburg, Stockholm and Umeå in Sweden.

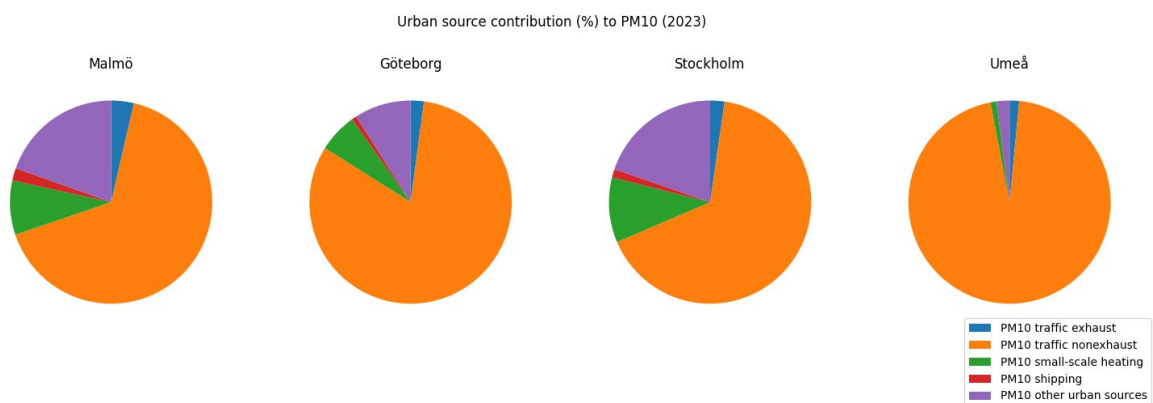


Figure 15. The urban source contribution to PM_{10} concentrations for 2023 are shown in pie charts for Malmö, Gothenburg, Stockholm and Umeå. Urban contributions from traffic exhaust, traffic non-exhaust, small-scale residential heating, shipping and other sources are presented.

4.1.4 Concentration of O₃

Figure 16 shows how the annual mean concentration of ground-level O₃ varies throughout the country. The cities of Malmö, Gothenburg, Stockholm and Umeå are also presented in Appendix 2D.

As we have limited our analysis to yearly mean concentrations, and the AQS and AQG of O₃ are related to the daily 8-hour mean, we are not able to discuss any exceedances in this section. Generally, however, an overall decrease in O₃ is observed between 2023 and 2030, especially in rural areas. In well-trafficked areas, an increase can be observed. The unintentional error in the post-processing step of the regional background concentrations, described in section 2.3.1, showed to have no effect on the concentrations of O₃ in the 2030 scenario.

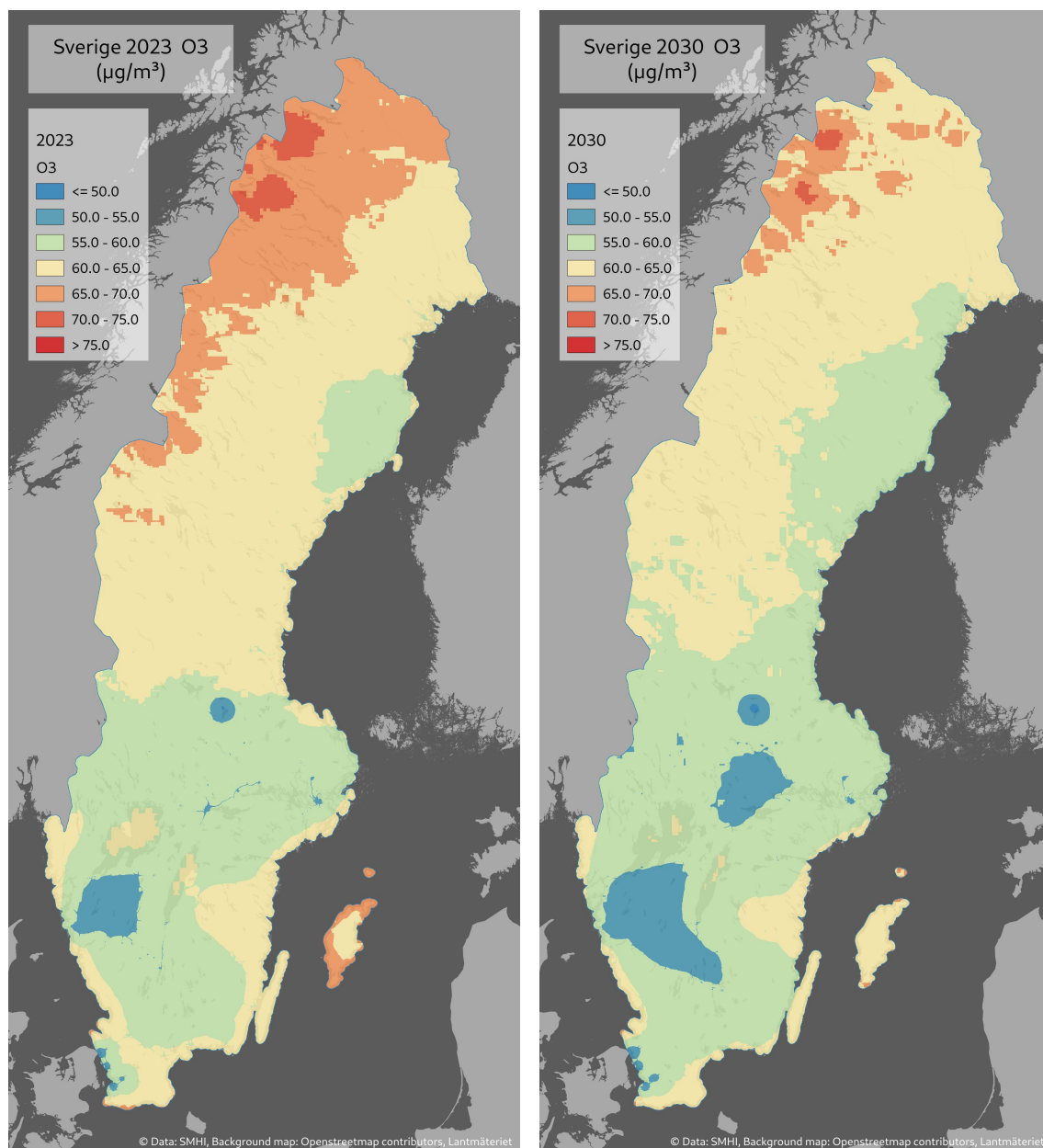


Figure 16. Concentration of O₃ yearly mean for 2023 and 2030.

The reason for a decrease in ground-level O₃ is more complex, as it is a secondary pollutant and not emitted directly, but produced through chemical interaction between VOCs (volatile organic compounds), sunlight and NO_x (nitrogen oxides). However, NO_x is involved not only in the production but also in breaking down O₃. Since O₃ can be produced in one place and thereafter transported over larger distances, areas with high NO_x emissions, such as city centres, can experience lower O₃ concentrations than areas with low NO_x emissions. It also means that the projected lower O₃ concentrations

in rural areas in 2030 can generally be said to be due to a decrease of the regional background concentrations. Close to large roads, the increase is instead due to reduced NO_x emissions, leading to less O₃ consumption.

4.2 Exposure

In this section the population exposure results of NO₂, PM_{2.5}, PM₁₀ and O₃ are presented for 2023 and 2030. To calculate the exposure, the already presented concentration results (section 4.1) are multiplied by the population density. This means that the distribution of the highest exposure is not necessarily the same as that of the highest concentrations, as the exposure results depend on the number of people who live in an area with a certain concentration of a pollutant.

Firstly, we present the exposure concentration distribution for each pollutant in 2023 and 2030 (sections 4.2.2-4.2.5). Thereafter, a comparison is done per age group (section 4.2.6) as well as to the air quality standards and WHO guidelines (section 4.2.7). In section 4.2.8 the source contributions to the exposure are presented and in section 4.2.9 a sensitivity analysis shows how the model resolution affects the results. Exposure maps for a few selected cities are further shown in Appendix 2A-D.

As described in previous sections, there has been significant improvements to the dispersion models and input data since the previous exposure study by Alpfjord Wylde et al. (2023). We therefore avoid making too many direct comparisons between the results of the two projects. One major difference is the increased resolution of the urban dispersion model, from 250 m x 250 m in the previous study to 50 m x 50 m in this. To evaluate what effect this has had on the results, we have also performed calculations of the national concentrations and subsequent exposure calculations for three other resolutions: 100 m x 100 m, 500 m x 500 m and 1000 m x 1000 m.

Further note that all figures and tables in this section are presented based on gridded population data while the health impact assessment is determined based on point coordinate population data. While this may lead to slight differences in total population exposure, the effects are negligible compared to the uncertainties in the emissions and dispersion simulation modelling.

4.2.1 Population distribution

The total population in Sweden for year 2023 was 10 531 687 inhabitants. Table 3 shows the number of inhabitants in each age span. The same population data was used for the exposure calculations for 2023 and for the 2030 scenario.

Table 3. Population distribution over different age spans for year 2023.

Population age span	Number of inhabitants
0-6	799 166
7-15	1 127 248
16-19	486 048
20-24	590 345
25-44	2 783 467
45-64	2 571 041
65-	2 174 372
Total	10 531 687

4.2.2 Exposure to NO₂

The distribution of the NO₂ exposure divided into different bins is shown in Figure 17. Comparing the distribution for 2023 and 2030, there is an overall shift from higher exposures to lower. Percentage-wise, the decrease is the largest for the bins with the highest concentrations, where the number of persons exposed to 8.5-9.0 µg/m³ has dropped significantly. The most common exposure concentration in both 2023 and 2030 is 1.5 µg/m³. As was mentioned in section 4.1.1, the decrease in NO₂ concentrations and hence overall exposure is believed to be reliable.

The average exposure of NO₂ for the total population is calculated to 4.03 µg/m³ for 2023 and 2.91 µg/m³ for 2030 (also shown in Table 4).

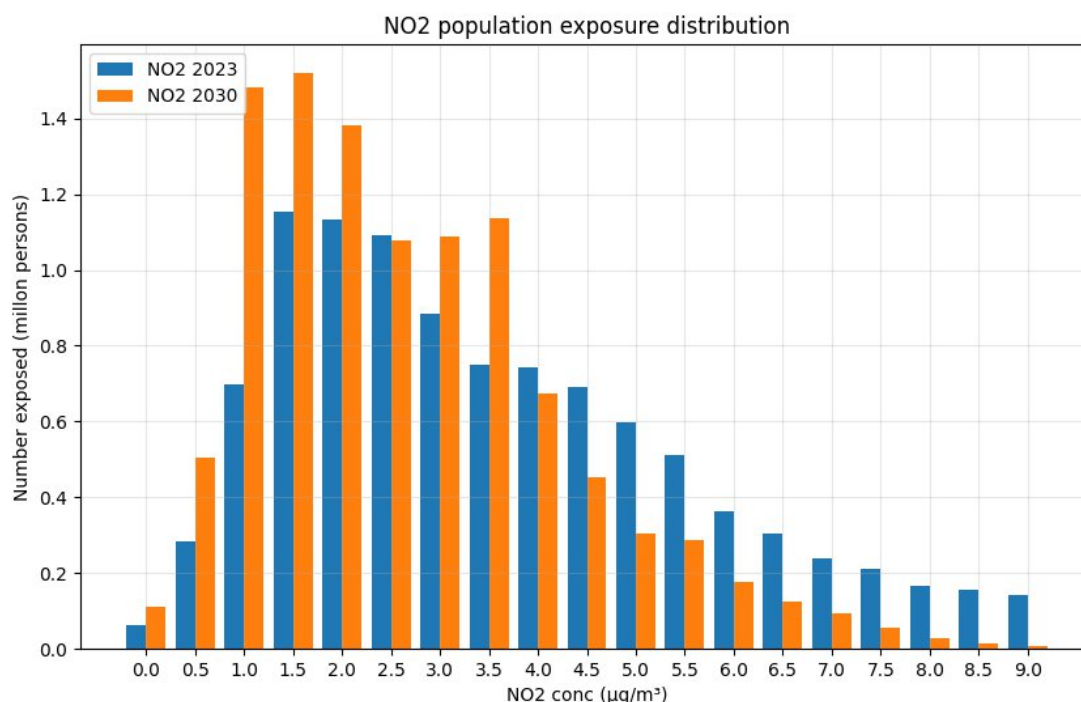


Figure 17. Distribution of population exposure of NO₂.

In Appendix 2A, Figure 29-Figure 32 show how the population exposure of the annual mean concentration of NO₂ varies throughout the cities of Malmö, Gothenburg, Stockholm and Umeå for 2023 and 2030, as compared to the annual mean concentration from section 4.1.1. Figure 33 instead shows the population exposure on a national level, however, note that this is presented at a resolution where the higher exposures are difficult to distinguish as they tend to be very focused on the city centres.

4.2.3 Exposure to PM_{2.5}

The distribution of the PM_{2.5} exposure for 2023 and 2030 divided into bins is presented in Figure 18. Between 2023 and 2030 there is a general shift from both lower exposure and higher exposure levels, towards the centre values (around 4.0 µg/m³). There is also an increase in the number of people exposed to the higher levels (around 6.5 µg/m³).

It is worth noting that the shift in exposure of PM_{2.5} between 2023 and 2030 is quite different from the distribution for NO₂ (Figure 17). This reflects the overall different trends for NO₂ and PM_{2.5}, where the exposure of NO₂ generally decreases and PM_{2.5} increases. However, as was mentioned in section 4.1.2, the increase in PM_{2.5} concentrations is less pronounced and carries more uncertainties compared to the decrease in NO₂ concentrations.

The average exposure of PM_{2.5} for the total population is calculated to 4.61 µg/m³ for 2023 and 4.71 µg/m³ for 2030 (also shown in Table 4).

In Appendix 2B, Figure 34-Figure 37 display maps of the population exposure on PM_{2.5} in Malmö, Gothenburg, Stockholm and Umeå, next to the maps of the annual mean concentration from section 4.1.2. A national map is shown in Figure 38. As is the case for the NO₂ exposure maps in the previous section it is difficult to distinguish the higher exposures on the national map, since they tend to be very focused on the city centres.

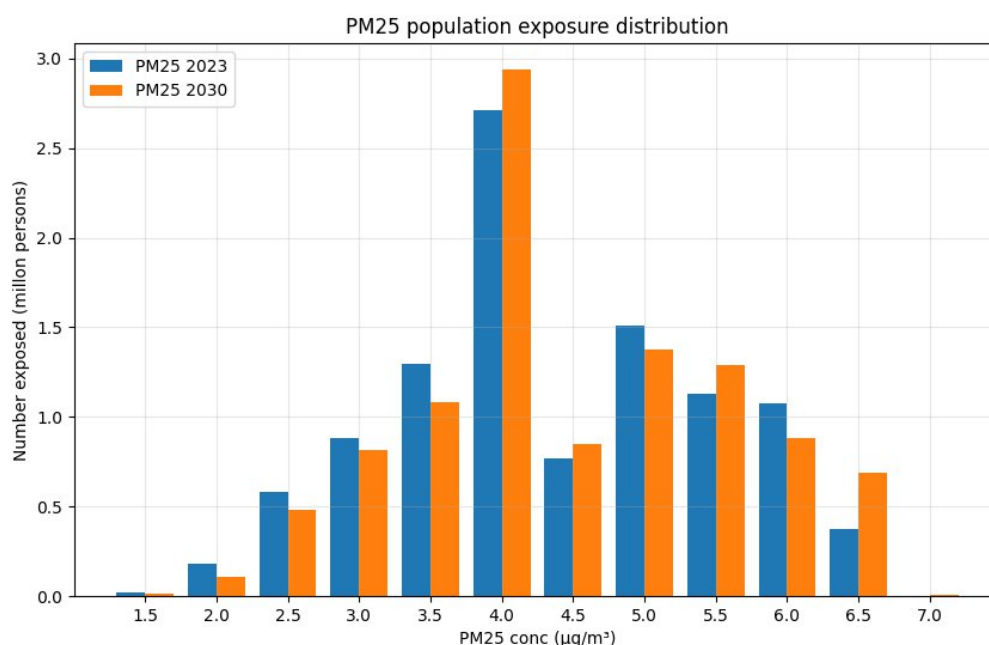


Figure 18. Distribution of population exposure of PM_{2.5}.

4.2.4 Exposure to PM₁₀

The distribution of population exposure for PM₁₀ is shown in Figure 19. The overall distribution is similar to that of PM_{2.5}, though the range is larger for PM₁₀. Between 2023 and 2030 there is a general shift from the lowest levels towards the centre values (7.5 µg/m³) and also an increase in exposure of the highest levels. As for PM_{2.5}, the overall increase in PM₁₀ exposure is small and more uncertain compared to the decreasing trend of NO₂.

The average exposure of PM₁₀ for the total population is calculated to 8.11 µg/m³ for 2023 and 8.23 µg/m³ for 2030 (also shown in Table 4).

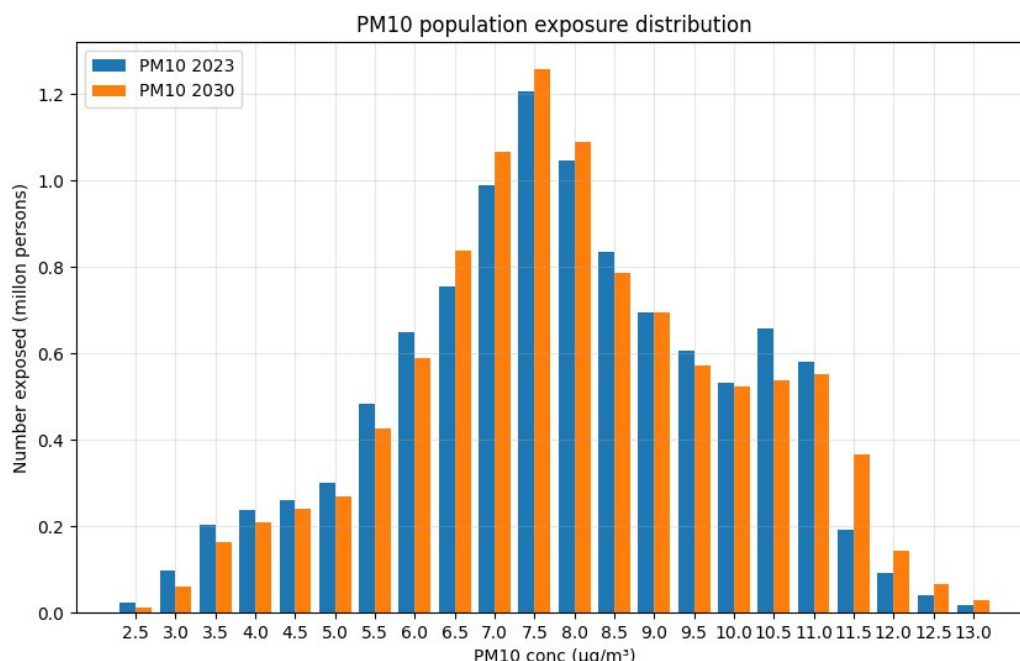


Figure 19. Distribution of population exposure of PM_{10} .

Exposure maps of PM_{10} for Malmö, Gothenburg, Stockholm and Umeå are shown in Appendix 2C, next to the annual concentration maps from section 4.1.3. No national map is included for PM_{10} as the differences are too difficult to distinguish on the national level.

4.2.5 Exposure to O_3

Figure 20 shows the distribution of exposure to O_3 . The pattern and trends between 2023 and 2030 are different compared to the other substances. Between 2023 and 2030 there is a shift from higher exposure levels towards lower, leading to a small overall decrease in population exposure. The most common exposure level in 2023 is $56 \mu\text{g}/\text{m}^3$ compared to $54 \mu\text{g}/\text{m}^3$ in 2030. However, as ground-level O_3 is a secondary pollutant, one should keep in mind that there are additional uncertainties related to the modelling of O_3 , especially when differences in the modelled concentrations are minor.

The average exposure of O_3 for the total population is calculated to $56.91 \mu\text{g}/\text{m}^3$ for 2023 and $56.18 \mu\text{g}/\text{m}^3$ for 2030 (also shown in Table 4).

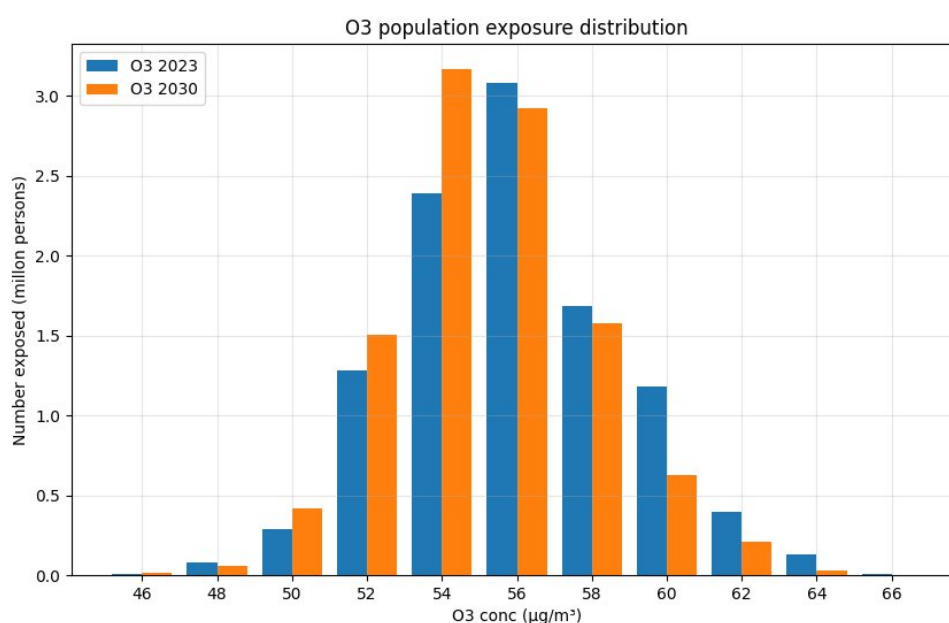


Figure 20 Distribution of population exposure of O_3 .

Exposure maps of O₃ for Malmö, Gothenburg, Stockholm and Umeå are shown in Appendix 2D, next to the maps of the annual concentration from section 4.1.4. No national map is included for O₃.

4.2.6 Exposure for different age groups

Total exposure for different age groups is shown in Table 4. While NO₂ and PM has different patterns, the differences in exposure for different age groups are fairly small. The highest exposure is for the age groups 20-24 and 25-44 years. O₃ shows a different pattern, where the highest exposure is observed for the age group 65+ years and the lowest for the age groups 20-24 and 25-44 years. This is expected since O₃ is a secondary pollutant dependent on the amount of NO_x and the regional background of O₃ (see section 4.1.4).

Table 4. Mean population weighted exposure results (µg/m³) per age group for NO₂, PM_{2.5}, PM₁₀ and O₃ for 2023 and 2030.

Age group	NO ₂ 2023	NO ₂ 2030	PM _{2.5} 2023	PM _{2.5} 2030	PM ₁₀ 2023	PM ₁₀ 2030	O ₃ 2023	O ₃ 2030
Total population	4.03	2.91	4.61	4.71	8.11	8.23	56.91	56.18
Age 0-6 years	3.98	2.88	4.62	4.72	8.12	8.23	56.89	56.15
Age 7-15 years	3.81	2.78	4.61	4.71	8.04	8.14	57.05	56.26
Age 16-19 years	3.86	2.81	4.61	4.71	8.05	8.16	57.03	56.25
Age 20-24 years	4.38	3.12	4.64	4.75	8.30	8.46	56.20	55.88
Age 25-44 years	4.42	3.14	4.63	4.73	8.30	8.42	56.54	55.92
Age 45-64 years	3.94	2.85	4.60	4.70	8.06	8.16	57.01	56.25
Age 65+ years	3.71	2.71	4.56	4.67	7.89	8.03	57.27	56.46

4.2.7 Exposure in relation to air quality standards and recommendations

Table 5 shows the percentage of the population exposed to concentrations corresponding to the limit values of the air quality standard of the current and the new air quality directive, as well as the WHO air quality guidelines.

When applying the current and new directive, ≤0.02 % of the population is exposed to the AQS (yearly mean) of all pollutants for 2023 and 2030. However, when applying the AQG, the numbers are around 1 % for NO₂, 40 % for PM_{2.5} and 0.1 % for PM₁₀ in 2023. It should be noted that these numbers only reflect exposure to the regional and urban background concentrations, and they would have been higher if the street canyon contribution had been included in the calculations. As we only compare to the yearly mean of the AQS, a comparison to the daily and hourly percentiles would likely increase the numbers even further.

Table 5. Percentage of population exposed to concentrations of NO_2 , $\text{PM}_{2.5}$, PM_{10} and O_3 corresponding to the limit values of the air quality standard in the current and new air quality directive as well as the recommended levels in the WHO air quality guidelines for 2023 and 2030. As there are no defined limit values nor recommended levels for the annual mean of O_3 , it is omitted from the table.

Limit values	NO_2 2023	NO_2 2030	$\text{PM}_{2.5}$ 2023	$\text{PM}_{2.5}$ 2030	PM_{10} 2023	PM_{10} 2030
Current AQ standard	40	40	25	25	40	40
Percentage of population exposed to current AQS (annual mean, $\mu\text{g}/\text{m}^3$)	0	0	0	0	0	0
New AQS	20	20	10	10	20	20
Percentage of population exposed to new AQS (annual mean, $\mu\text{g}/\text{m}^3$)	0.005	0	0	0	0.02	0.01
WHO AQG	10	10	5	5	15	15
Percentage of population exposed to AGQ (annual mean, $\mu\text{g}/\text{m}^3$)	1.3	0.06	39.0	40.3	0.11	0.13

4.2.8 Exposure source contribution

NO_2

The sectoral contributions to the overall NO_2 exposure in 2023 are shown in Figure 21. Exposure contributions to NO_2 , divided into a regional and an urban contribution (left) and the urban exposure contribution from different sectors (right). Note that the sector contributions (pie diagram) are based on NO_x . The figure shows the average contributions to the exposure across the whole of Sweden, although it should be noted that there might be large differences between regions.

The regional background and the urban part contribute about equally to the exposure. The largest urban contribution is the traffic sector but other urban sources (such as energy production, industries and working machinery) also play a significant role.

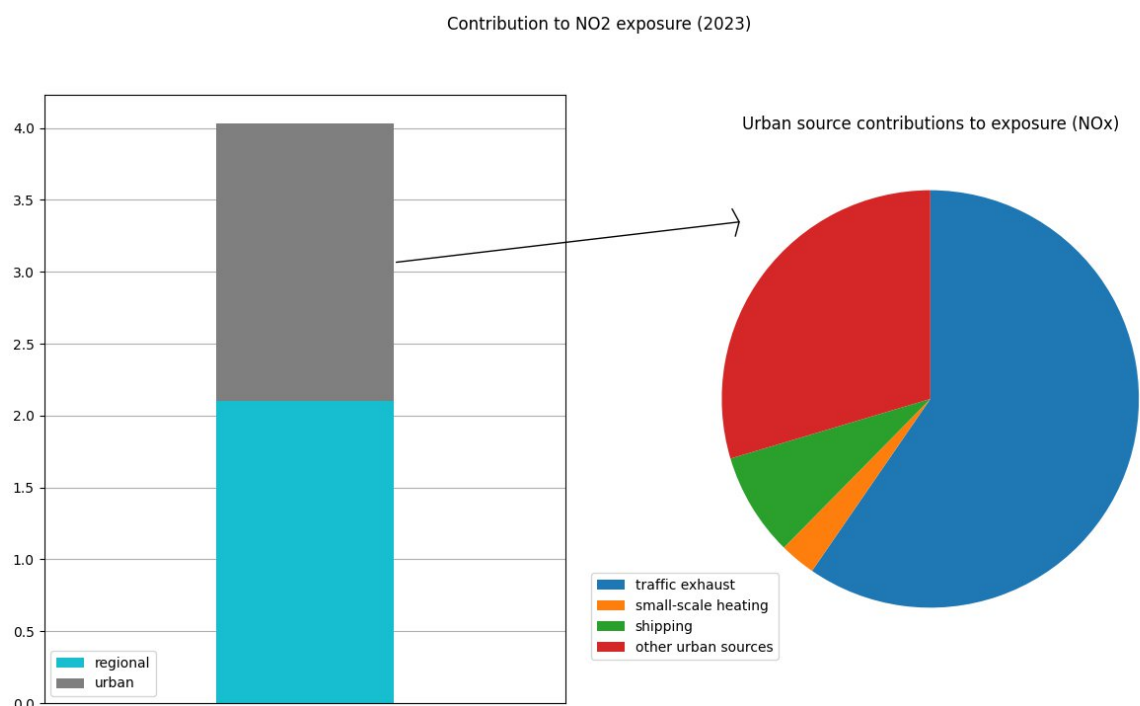


Figure 21. Exposure contributions to NO₂, divided into a regional and an urban contribution (left) and the urban exposure contribution from different sectors (right). Note that the sector contributions (pie diagram) are based on NO_x.

PM_{2.5}

The different sectoral contributions to the exposure of PM_{2.5} in 2023 are shown in Figure 22. The regional contribution is the dominating sector, compared to the urban contributions. Among the urban sectors, residential heating, non-exhaust traffic and other urban sources contribute to the exposure.

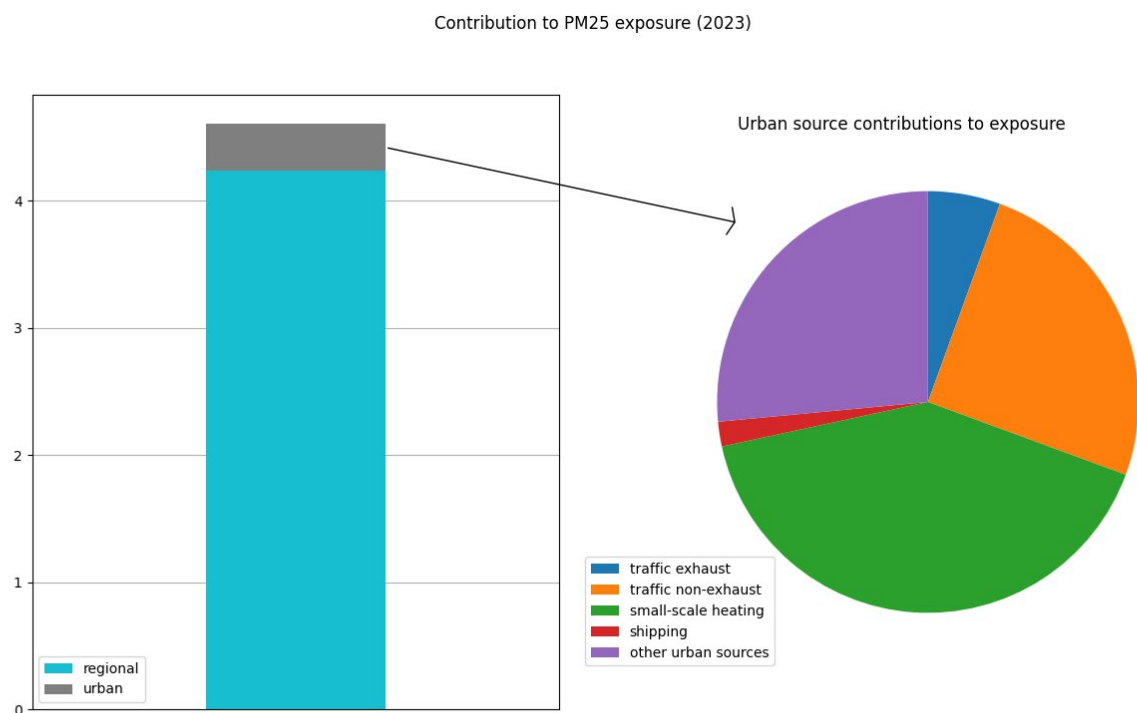


Figure 22. Exposure contributions to PM_{2.5}, divided into a regional and an urban contribution (left) and the urban exposure contribution from different sectors (right).

PM₁₀

Comparison of different sectoral contributions to the exposure in 2023 are shown in Figure 23. While the regional part is dominating, among the urban sectors the non-exhaust from traffic is the main contributor to the exposure.

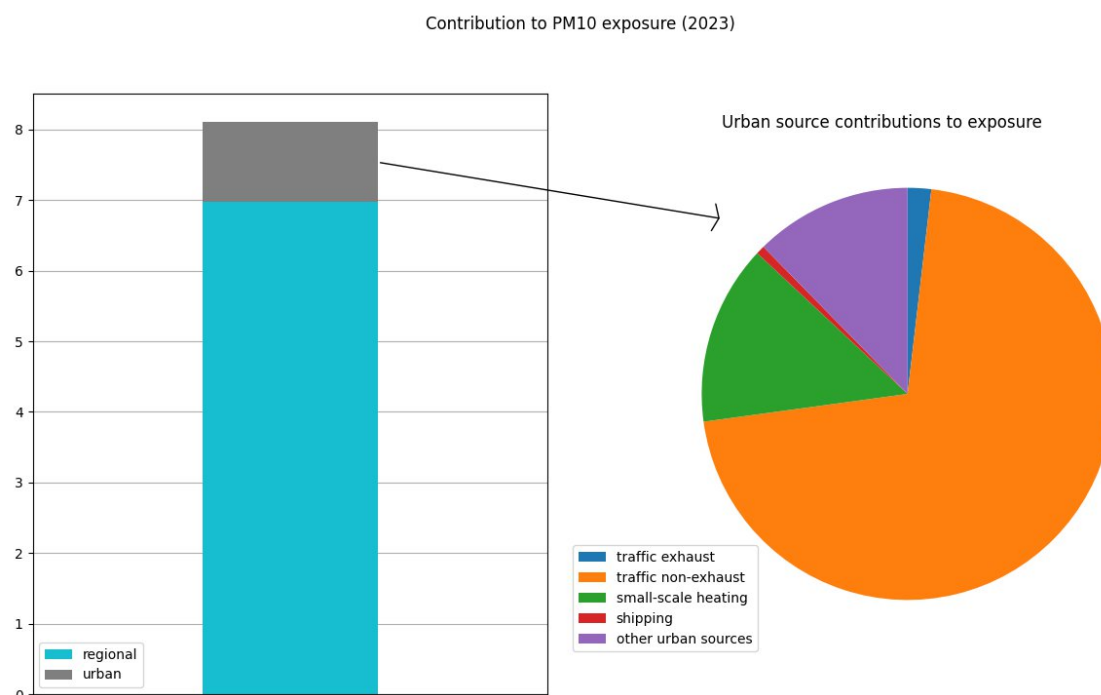


Figure 23 Exposure contributions to PM₁₀, divided into a regional and an urban contribution (left) and the urban exposure contribution from different sectors (right).

Comparison of 2023 and 2030

In addition, exposure per source category of NO₂, PM_{2.5} and PM₁₀ (and total exposure of O₃) for both 2023 and 2030 is shown in .

For NO₂, there is a decline in exposure from both urban and regional sources between 2023 and 2030, where the total exposure decreases by almost 28 %. For PM₁₀, the main contributor to the overall exposure increase is road wear from traffic and regional sources, while the other sectors remain on a similar level or experience a decrease. The pattern is similar for PM_{2.5} as for PM₁₀, although the overall urban contribution remains the same between 2023 and 2030.

As has been noted previously, it should be mentioned that since the projected change in total exposure of particles and O₃ is small (2.3 % for PM_{2.5}, 1.5 % for PM₁₀ and 1.3 % for O₃), it carries a higher level of uncertainty than the NO₂ exposure decrease.

Table 6. Mean population weighted exposure results ($\mu\text{g}/\text{m}^3$) per source category for NO_2 , $\text{PM}_{2.5}$, PM_{10} for 2023 and 2030.

Sources	NO_2 2023	NO_2 2030	$\text{PM}_{2.5}$ 2023	$\text{PM}_{2.5}$ 2030	PM_{10} 2023	PM_{10} 2030	O_3 2023	O_3 2030
Exposure contribution from urban sources	1.931	1.160	0.368	0.308	1.134	1.141	-	-
Traffic - Exhaust	-	-	0.020	0.010	0.021	0.011	-	-
Traffic - Roadwear	-	-	0.091	0.109	0.805	0.897	-	-
Residential heating	-	-	0.154	0.112	0.160	0.123	-	-
Shipping	-	-	0.007	0.007	0.008	0.008	-	-
Other	-	-	0.096	0.070	0.140	0.102	-	-
Exposure contribution from regional sources	2.099	1.747	4.237	4.402	6.971	7.085	-	-
Total exposure	4.030	2.907	4.605	4.710	8.105	8.226	56.91	56.18

4.2.9 Sensitivity analysis — effect of model resolution on exposure calculations

The results of a sensitivity analysis looking at the effect of model resolution on exposure calculations are presented in Table 7 to Table 10. The results specify the number of inhabitants exposed over different concentration thresholds for the different pollutants at different model resolutions. For all pollutants, the number of inhabitants exposed over the lowest concentration thresholds remain fairly constant, with the largest differences seen for NO_2 which varies around 9 % between the highest and lowest resolution. This is primarily due to the fact that the regional contribution is less significant for NO_2 in comparison to the other pollutants.

The results vary distinctly between the different pollutants. For NO_2 , where the local contribution is generally more dominant than for PM_{10} and $\text{PM}_{2.5}$, there is a fairly strong dependence on spatial resolution for the highest exposure levels. However, the results for PM_{10} also show that having a high spatial resolution in the dispersion modelling is important to capture the exposure to the highest levels. While these highest exposure levels may have a limited contribution to the average population exposure, they still have an impact on the health impact assessment. Furthermore, it might be important to capture the highest exposure levels as well as possible, in order to facilitate remediation efforts.

Table 7. Population exposure of NO_2 over different thresholds from varied concentration resolutions.

NO_2 resolution	# > 2.5 ug/m3	# > 5 ug/m3	# > 7.5 ug/m3	# > 10 ug/m3	# > 15 ug/m3	# > 20 ug/m3
50 m x 50 m	7 117 828	2 510 957	725 911	135 146	7 621	506
100 m x 100 m	7 085 732	2 469 357	699 343	118 778	4 406	0
250 m x 250 m	6 979 023	2 337 902	620 955	72 498	2 947	0
500 m x 500 m	6 788 141	2 136 764	586 052	50 517	0	0
1000 m x 1000 m	6 537 582	1 882 794	423 439	437	0	0

Table 8. Population exposure of $\text{PM}_{2.5}$ over different thresholds from varied concentration resolutions.

$\text{PM}_{2.5}$ resolution	# > 2 ug/m3	# > 3 ug/m3	# > 4 ug/m3	# > 5 ug/m3	# > 6 ug/m3	# > 7 ug/m3	# > 8 $\mu\text{g}/\text{m}^3$
50 m x 50 m	10 504 639	9 728 424	7 881 113	4 110 592	2 315 372	104 616	400
100 m x 100 m	10 504 379	9 726 015	7 872 425	4 099 956	2 310 752	103 612	0
250 m x 250 m	10 502 109	9 705 129	7 831 243	4 081 788	2 293 944	96 357	0
500 m x 500 m	10 497 843	9 651 960	7 701 892	4 041 430	2 224 802	80 245	0
1000 m x 1000 m	10 492 747	9 583 994	7 437 968	3 992 221	2 140 113	0	0

Table 9. Population exposure of PM_{10} over different thresholds from varied concentration resolutions.

PM_{10} resolution	# > 2.5 ug/m3	# > 5 ug/m3	# > 7.5 ug/m3	# > 10 ug/m3	# > 15 ug/m3	# > 20 ug/m3
50 m x 50 m	10 525 965	9 692 931	6 560 080	1 832 533	12 157	2 444
100 m x 100 m	10 525 923	9 686 896	6 545 652	1 810 029	8 869	1 024
250 m x 250 m	10 525 634	9 662 487	6 461 557	1 745 799	2 308	0
500 m x 500 m	10 525 050	9 621 677	6 307 929	1 628 876	0	0
1000 m x 1000 m	10 523 305	9 553 271	6 104 618	1 492 090	0	0

Table 10. Population exposure of O_3 over different thresholds from varied concentration resolutions.

O_3 resolution	# > 50 ug/m3	# > 55 ug/m3	# > 60 ug/m3	# > 65 ug/m3	# > 70 ug/m3	# > 75 ug/m3
50 m x 50 m	10 523 481	9 263 697	2 672 323	56 759	207	0
100 m x 100 m	10 526 983	9 291 679	2 685 588	57 498	207	0
250 m x 250 m	10 527 857	9 380 289	2 718 834	58 753	206	0
500 m x 500 m	10 527 477	9 488 143	2 767 373	61 342	195	0
1000 m x 1000 m	10 525 985	9 649 789	2 841 061	63 065	173	0

4.3 Estimated health impacts

4.3.1 Health impacts associated with air pollution

Figure 24 presents estimated incident cases for selected health outcomes associated with long-term exposure to air pollution in 2023. Results are shown as point estimates with 95% confidence intervals on a logarithmic scale to accommodate the wide range in effect sizes across outcomes. At the top of the figure, the combined estimate for mortality is 5 966 cases (95% CI 3 905–8 035), representing the sum of four separate mortality estimates: 3 467 cases for PM_{2.5} background, 579 for PM_{2.5} local excluding wear, 554 for PM₁₀ wear, and 1 366 for NO₂. The average years of life lost were 10.9.

In absolute terms, the largest non-fatal burden was estimated for work loss days, with 680 692 cases (95% CI 578 868–781 905), followed by sick days among school children, with 48 200 cases (95% CI 15 684–80 381). Among the disease-specific outcomes, the highest numbers were estimated for ASD (1 422, 95% CI 572–2 311), COPD (1 083, 95% CI 798–1 358), and dementia (720, 95% CI 343–1 107). These were followed by diabetes (374, 95% CI 116–652), stroke (251, 95% CI 192–309), childhood asthma (211, 95% CI 70–352), myocardial infarction (185, 95% CI 73–302), allergic rhinitis (148, 95% CI 83–211), preterm birth (104, 95% CI 34–154), and lung cancer (56, 95% CI 36–79). The ranking of outcomes reflects both the underlying incidence of disease in the population and differences in the relative risks associated with air pollution exposure.

Figure 25 presents the estimated number of incident mortality cases attributable to locally generated air pollution, disaggregated by particle emission source. For PM_{2.5} local excluding wear, the source-specific breakdown indicates that the largest contribution came from residential heating, with an estimated 321 incident deaths (95% CI 241–408), followed by other local (i.e. modelled urban emissions) sources, with 199 cases (95% CI 150–253). By contrast, the contributions from traffic exhaust and shipping were much smaller, at 42 cases (95% CI 31–53) and 13 cases (95% CI 9–16), respectively.

Between 2023 and 2030, the total number of incident deaths related to air pollution is expected to decrease slightly, from 5 966 to 5 627 cases (Figure 26), corresponding to a reduction of about 6%. However, this overall trend masks clear differences between pollutant components. The number of deaths associated with NO₂ is expected to decrease from 1 366 to 987 cases, and those associated with PM_{2.5} local excluding wear from 579 to 417 cases. At the same time, the number of deaths linked to PM_{2.5} background is expected to increase from 3 467 to 3 602 cases, and those linked to PM₁₀ wear from 554 to 621 cases. Overall, this suggests that reductions in local exhaust-related contributions are partly offset by increases in background concentrations and wear particles.

For the non-fatal outcomes, the results instead point consistently towards increases by 2030 since these are estimated in relation to total PM_{2.5}. The largest absolute increases are seen for work loss days, from 680 692 to 696 077 cases, and for sick days among school children, from 48 200 to 49 289 cases. All disease-specific PM_{2.5}-related outcomes are also expected to increase compared with 2023: ASD from 1 422 to 1 547, COPD from 1 083 to 1 181, dementia from 720 to 788, diabetes from 374 to 407, stroke from 251 to 274, childhood asthma from 211 to 230, myocardial infarction from 185 to 201, allergic rhinitis from 148 to 161, preterm birth from 104 to 114, and lung cancer from 56 to 61 cases. For most of these outcomes, this corresponds to an increase of roughly 9% between 2023 and 2030.

Taken together, the results show that the future development is not uniform. While the overall mortality burden is expected to decline slightly, mainly due to lower contributions from NO₂ and local PM_{2.5} excluding wear, several other health outcomes are projected to increase. The results therefore illustrate how changes in pollutant composition and source contributions may lead to different trends depending on the health outcome and pollutant component. Uncertainty intervals overlap between 2023 and 2030 for several outcomes, particularly those with wider confidence intervals.

4.3.2 Discussion

The present assessment incorporates two important developments in the evidence base that have implications for both the magnitude and composition of the estimated health burden from air pollution.

Firstly, new epidemiological evidence supports an association between long-term exposure to traffic-related wear particles and mortality, using PM_{10} as an exposure metric. Traditionally, health impact assessments have focused on $PM_{2.5}$, which may underestimate the contribution of non-exhaust emissions, as these are largely represented in the coarse fraction. The inclusion of wear particles as a contributor to mortality therefore represents a methodological advancement, allowing a more comprehensive assessment of traffic-related impacts. This is particularly relevant in the context of future emission scenarios, where exhaust emissions are expected to decrease due to electrification and stricter emission standards, while non-exhaust emissions (e.g. from tyre and road wear) are projected to constitute an increasing share of total traffic-related particulate matter. Consequently, even as overall particulate levels decline, the relative importance of wear particles for public health may increase.

Secondly, the inclusion of additional health outcomes in children reflects a growing body of evidence linking air pollution to early-life health effects and long-term developmental outcomes. In particular, associations with autism spectrum disorder (ASD), allergic rhinitis, and school absence have been incorporated based on recent systematic reviews and meta-analyses.

Taken together, these developments suggest that previous assessments may have underestimated both the total health burden and the relative contribution of specific sources, particularly non-exhaust traffic emissions. They also show the importance of adopting a life-course perspective in air pollution health assessments, where impacts in early life may have long-term consequences for individuals and society. At the same time, uncertainties remain, particularly regarding source-specific exposure–response relationships and potential overlap between pollutants, which should be carefully considered in interpretation and policy application.

4.3.3 Health economic assessment

In 2023, the estimated socio-economic costs attributable to long-term air pollution exposure were dominated by mortality, with a total cost of €15 430 million in 2023 (95% CI: €7 450–30 141 million 2023) (Figure 27). Within this total, the largest contribution came from $PM_{2.5}$ background, accounting for €8 968 million 2023 (95% CI: €4 515–17 161 million 2023), followed by NO_2 at €3 533 million 2023 (95% CI: €1 577–7 114 million 2023), $PM_{2.5}$ local excluding wear at €1 496 million 2023 (95% CI: €828–2 757 million 2023), and PM_{10} wear at €1 432 million 2023 (95% CI: €529–3 109 million 2023). Among the non-fatal outcomes, the highest estimated costs were observed for ASD (€2 072 million 2023), dementia (€769 million 2023), and COPD (€732 million 2023), whereas costs for the remaining outcomes were substantially lower.

By 2030, the total socio-economic cost of mortality is projected to remain at a similar level, declining only slightly to €15 299 million 2023 (95% CI: €7 353–29 941 million 2023) (Figure 28). However, this stable total contains a shift in the source-specific pattern. Costs attributable to $PM_{2.5}$ background are projected to increase to €9 315 million 2023, and those attributable to PM_{10} wear to €1 606 million 2023, whereas the corresponding costs for NO_2 and $PM_{2.5}$ local excluding wear are projected to decrease to €3 298 million 2023 and €1 079 million 2023, respectively. This indicates that declining local exhaust-related contributions are partly offset by increasing costs associated with background $PM_{2.5}$ and wear particles.

For the non-fatal outcomes, the socio-economic costs are projected to increase by 2030. The largest absolute increases are estimated for ASD, from €2 072 million to €2 254 million 2023, dementia, from €769 million to €842 million 2023, and COPD, from €732 million to €797 million 2023. More moderate increases are projected for stroke (from €182 million to €198 million 2023), work loss days (from €190 million to €194 million 2023), diabetes (from €104 million to €113 million 2023), lung can-

cer (from €58 million to €63 million 2023), and childhood asthma (from €49 million to €54 million 2023). Overall, the results suggest that while the mortality-related economic burden remains largely stable, the costs associated with several non-fatal outcomes are expected to increase towards 2030.

5 Conclusions

In this study, we have analysed the regional + urban background concentrations and the related average population exposure of NO₂, PM_{2.5}, PM₁₀ and ground-level O₃ in 2023 and a future 2030 scenario.

We find that NO₂ concentrations are projected to decrease from 2023 to 2030, while the concentrations and average exposure of PM₁₀ and PM_{2.5} are predicted to remain on a similar level or slightly increase. The average concentrations of ground-level O₃ experience a small decrease, while the average exposure remains almost unchanged. Considering the population exposure (of urban and regional background concentrations, not including street canyon effects) in relation to the air quality directive, 0 % are exposed to levels above the limit values of the current directive, and close to 0 % of the new directive. Compared to the WHO air quality guidelines, the most pronounced problem appears to be PM_{2.5}, where about 40 % of the population are exposed to concentrations above the recommended levels both in 2023 and the future scenario.

Children and the elderly are especially sensitive to the effects of air pollution. There are both geographic and demographic differences in the results, and these groups are shown to be less exposed to NO₂, PM₁₀ and PM_{2.5} than other adults in both 2023 and the 2030 scenario, while they are more exposed to ground-level O₃.

The decreased average NO₂ exposure is mainly a result of lower exhaust emissions in the 2030 vehicle fleet, due to an increasing proportion of electrical cars and stronger regulations on NO_x emissions from internal combustion vehicles. As this development is expected to continue, the exposure trend of NO₂ will likely be steadily decreasing. Fewer combustion engines further contribute to less VOC emissions, which next to NO_x is an important factor for ground-level O₃ production.

Internal combustion engines are also sources of air-borne particles, and such emissions are supposed to decline at the same rate as NO₂. However, a more important source of particles from vehicles, especially PM₁₀, is non-exhaust emissions, i.e. particles that are produced through road, tyre and brake wear. This is a particularly large problem in Sweden and other northern countries, that during the winter months have a high percentage of studded tyre usage and sanding/salting of roads. These emissions are not expected to decline like exhaust emissions, as the traffic volume is projected to increase and the vehicles are becoming heavier, which contributes to more wear.

The updated HIA shows that air pollution remains a substantial public health burden in Sweden, which indicates that mortality still accounts for the largest share of impacts and represents the dominant socio-economic cost, while morbidity, work loss, and school absence add considerable societal burden. The 2030 scenario suggests some overall improvement, but continued contributions from regional PM_{2.5} and non-exhaust traffic emissions mean that the health burden will remain important. Overall, the results support continued action to reduce both exhaust and non-exhaust emissions and to account for health impacts across the life course in future air-quality policy.

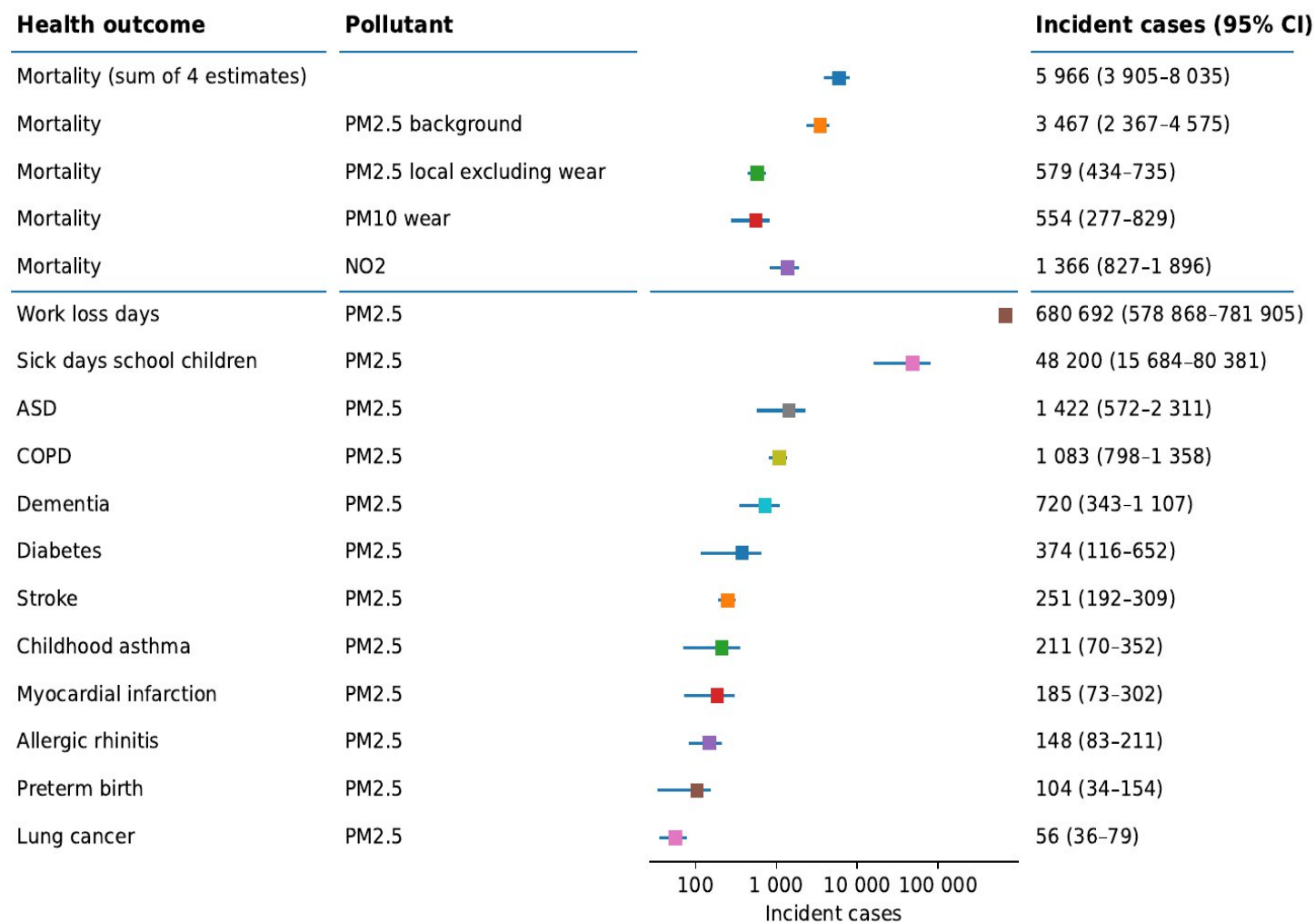


Figure 24. Estimated number of incident cases attributable to air pollution in 2023. The figure presents estimated incident cases and 95% confidence intervals for mortality and morbidity outcomes associated with long-term exposure to air pollution. Estimates are based on concentration–response functions from epidemiological studies and population exposure modelling. Results are shown separately by pollutant and source category where applicable. PM_{2.5} local is modelled urban emissions.

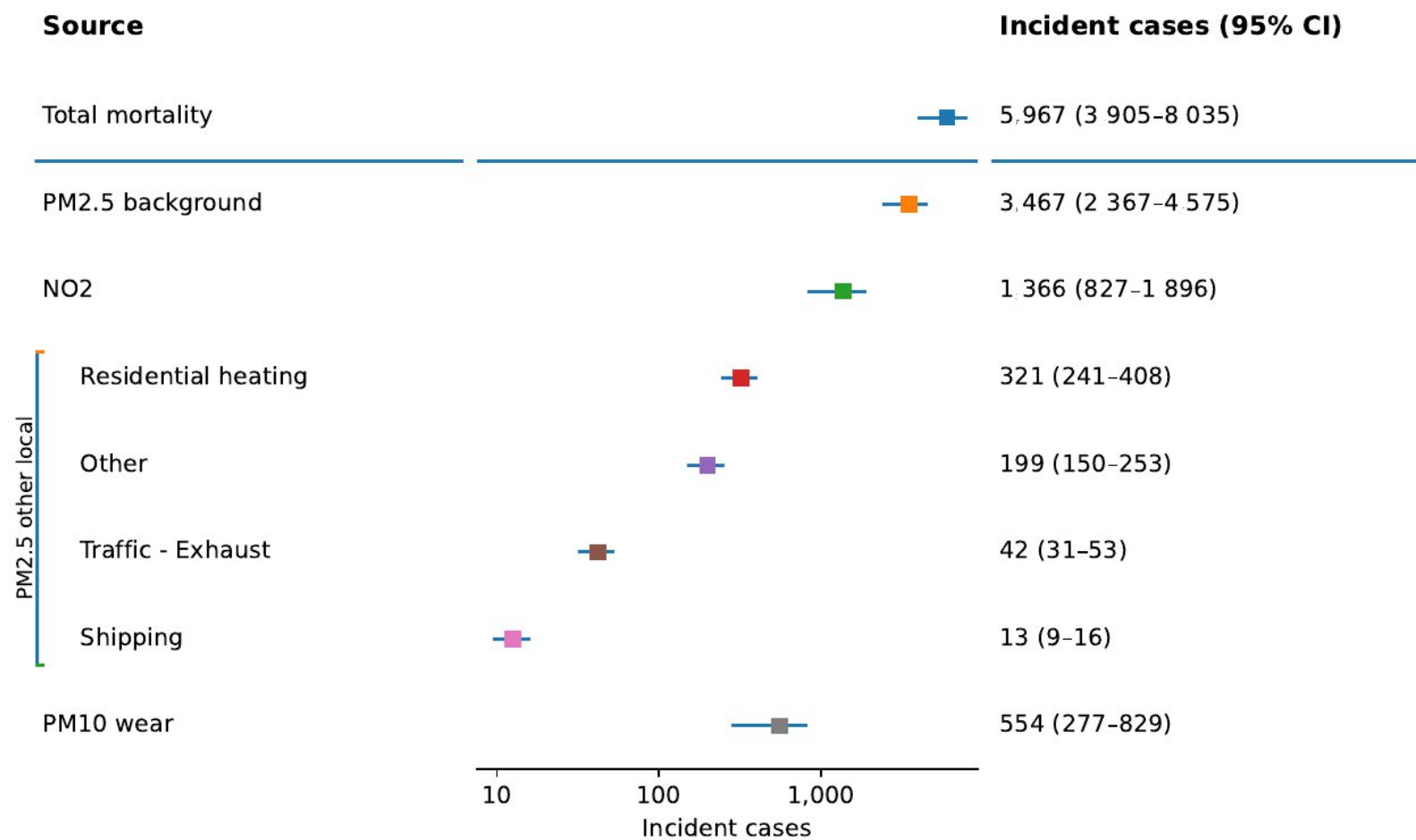


Figure 25. Estimated number of incident mortality cases attributable to air pollution by source (2023). The figure shows estimated incident mortality cases and corresponding 95% confidence intervals associated with long-term exposure to $PM_{2.5}$ (regional background and local sources), PM_{10} from wear particles, and NO_2 . Local $PM_{2.5}$ contributions are disaggregated into source categories (residential heating, other sources, traffic exhaust, and shipping), grouped for clarity. Results are not additive across pollutants or sources due to overlap in exposures and risk functions. The forest plot displays central estimates and 95% confidence intervals on a logarithmic scale. $PM_{2.5}$ local is modelled urban emissions.

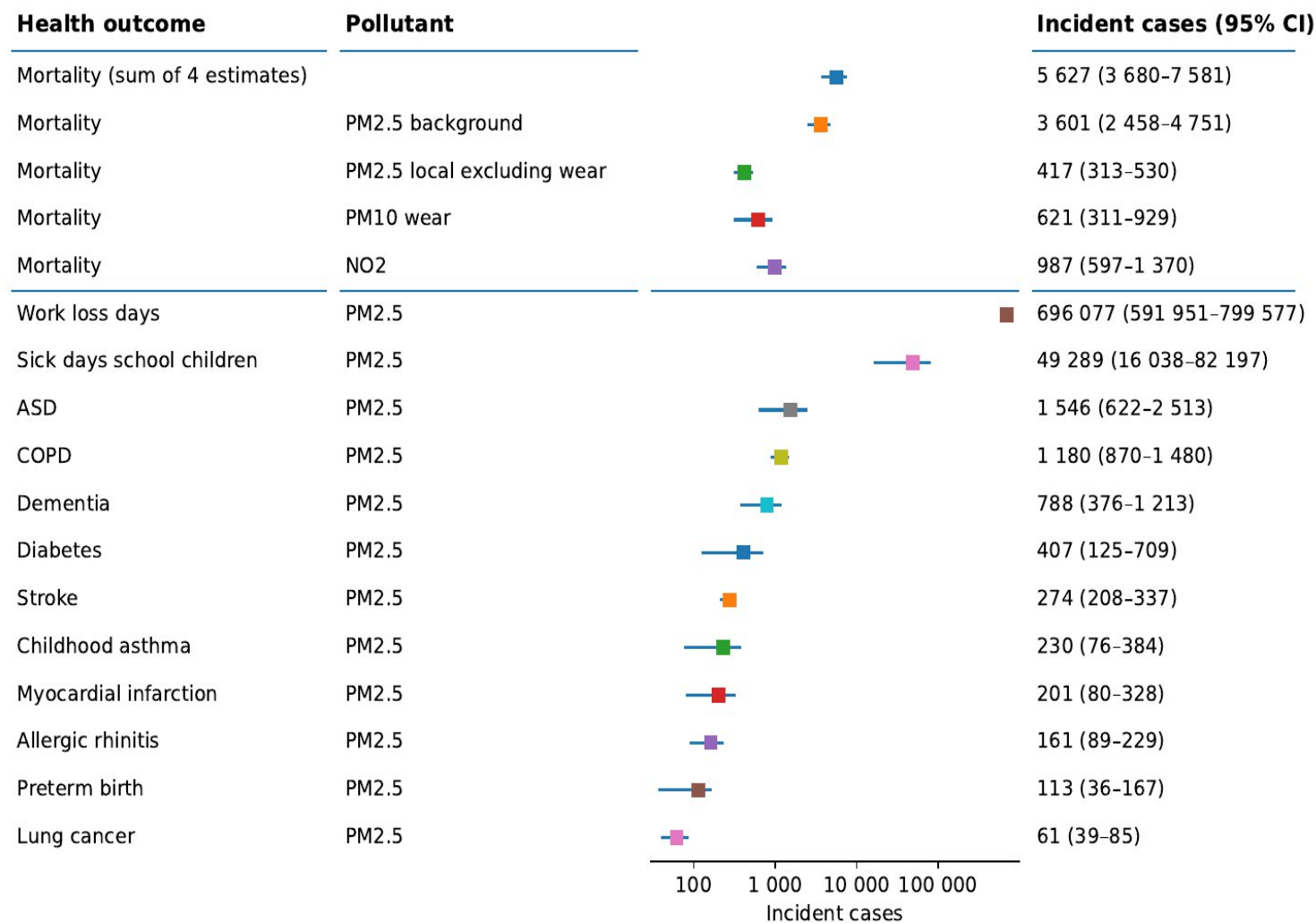


Figure 26. Estimated number of incident cases attributable to air pollution in 2030. The figure presents estimated incident cases and 95% confidence intervals for mortality and morbidity outcomes associated with long-term exposure to air pollution. Estimates are based on concentration–response functions from epidemiological studies and population exposure modelling. Results are shown separately by pollutant and source category where applicable. PM_{2.5} local is modelled urban emissions.

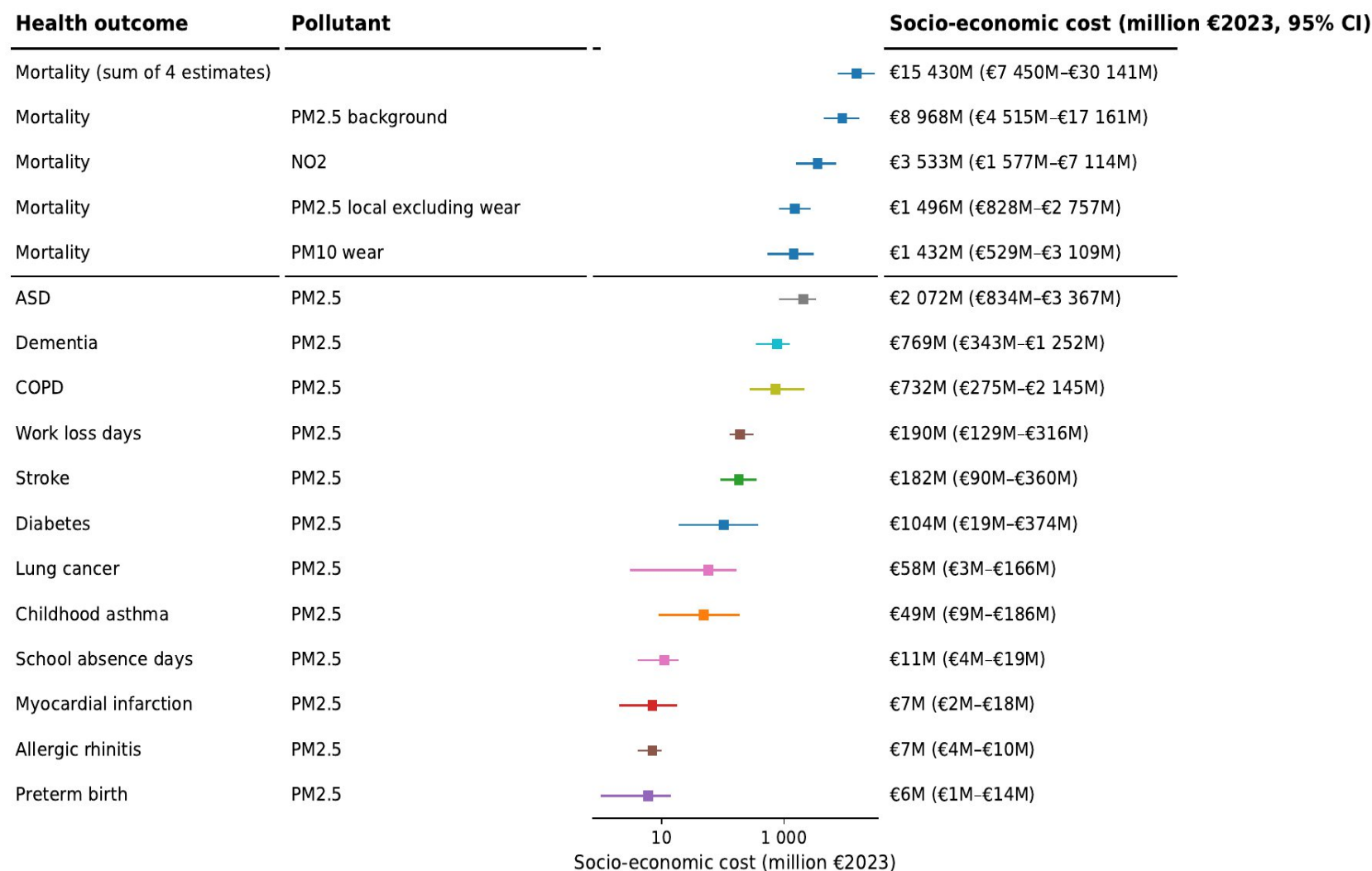


Figure 27. Estimated socio-economic costs attributable to air pollution in 2023. The figure presents total societal costs (in euros, 2023 price level) associated with mortality and selected morbidity outcomes attributable to air pollution exposure. Estimates are derived by combining incident cases with cost-per-case values obtained from the literature, including uncertainty intervals where available. PM_{2.5} local is modelled urban emissions.

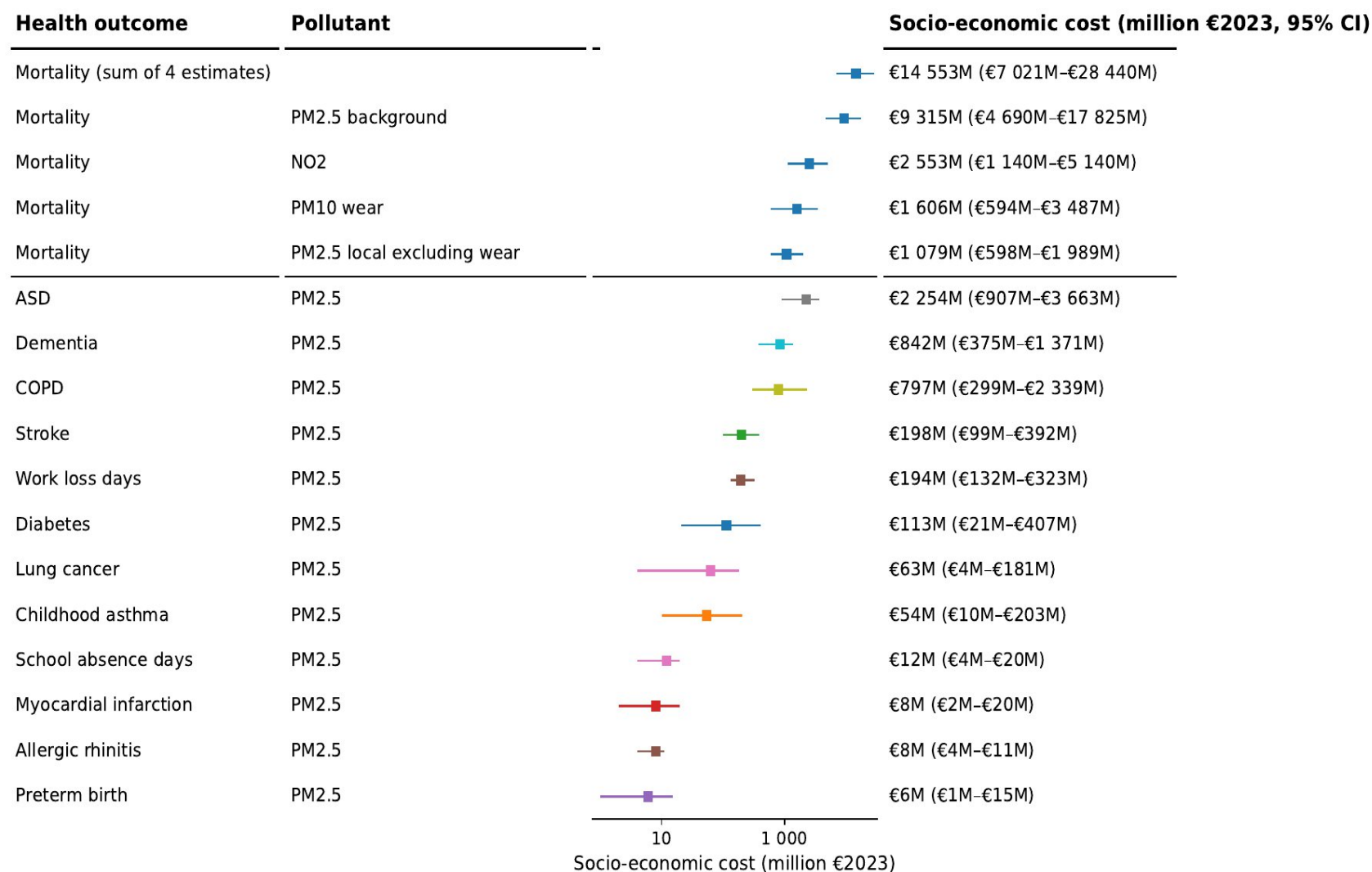


Figure 28. Estimated socio-economic costs attributable to air pollution in 2030. The figure presents total societal costs (in euros, 2023 price level) associated with mortality and selected morbidity outcomes attributable to air pollution exposure. Estimates are derived by combining incident cases with cost-per-case values obtained from the literature, including uncertainty intervals where available. PM_{2.5} local is modelled urban emissions.

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Appendix

1 Fraction of studded tyres per city

Table 11. Measured and assumed usage of studded tyres in 19 selected municipalities.

Municipality	2023	2030	2030/2023
Malmö	0.134	0.091	0.68
Växjö	0.450	0.296	0.66
Visby	0.860	0.862	1.00
Göteborg	0.340	0.183	0.54
Borås	0.533	0.465	0.87
Jönköping	0.477	0.438	0.92
Linköping	0.573	0.568	0.99
Södertälje	0.493	0.375	0.76
Stockholm	0.377	0.183	0.49
Nacka	0.440	0.282	0.64
Karlstad	0.713	0.722	1.01
Västerås	0.720	0.746	1.04
Uppsala	0.623	0.484	0.78
Borlänge	0.807	0.803	1.00
Sundsvall	0.930	0.983	1.06
Östersund	0.949	0.953	1.00
Umeå	0.947	0.930	0.98
Luleå	0.957	0.950	0.99
Kiruna	0.965	0.968	1.00

2 Concentrations and exposure of NO₂, PM_{2.5}, PM₁₀ and O₃

A. Concentrations and exposure of NO₂



Figure 29. Concentrations and exposure for Malmö of NO₂ for 2023 and 2030.

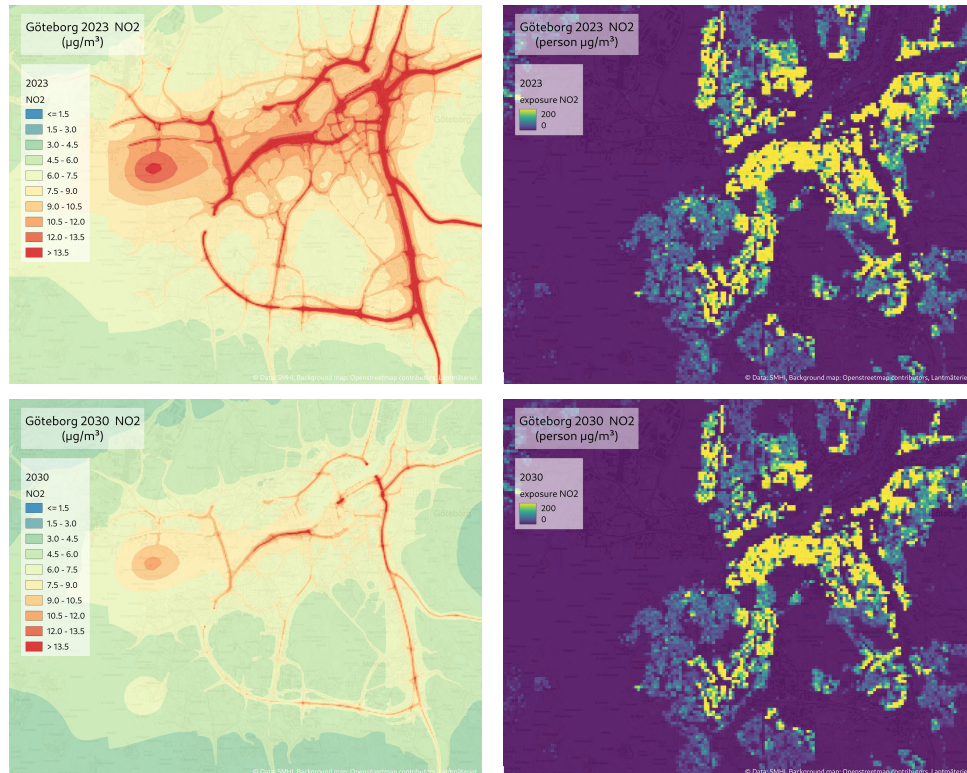


Figure 30. Concentrations and exposure for Gothenburg of NO₂ for 2023 and 2030..

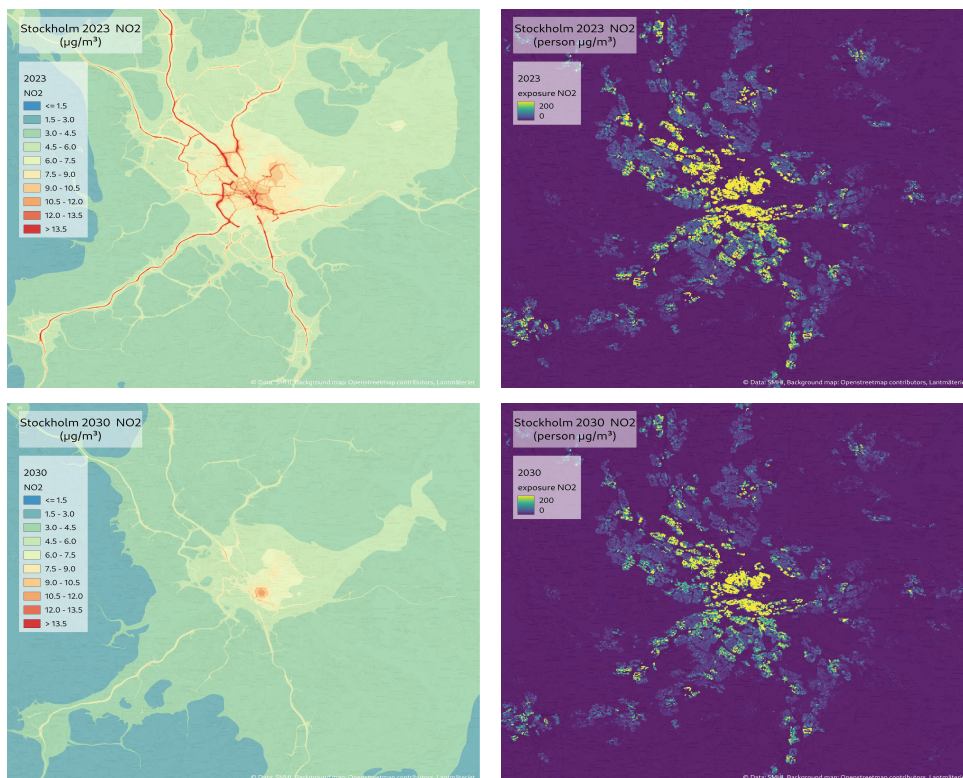


Figure 31. Concentrations and exposure for Stockholm of NO₂ for 2023 and 2030.

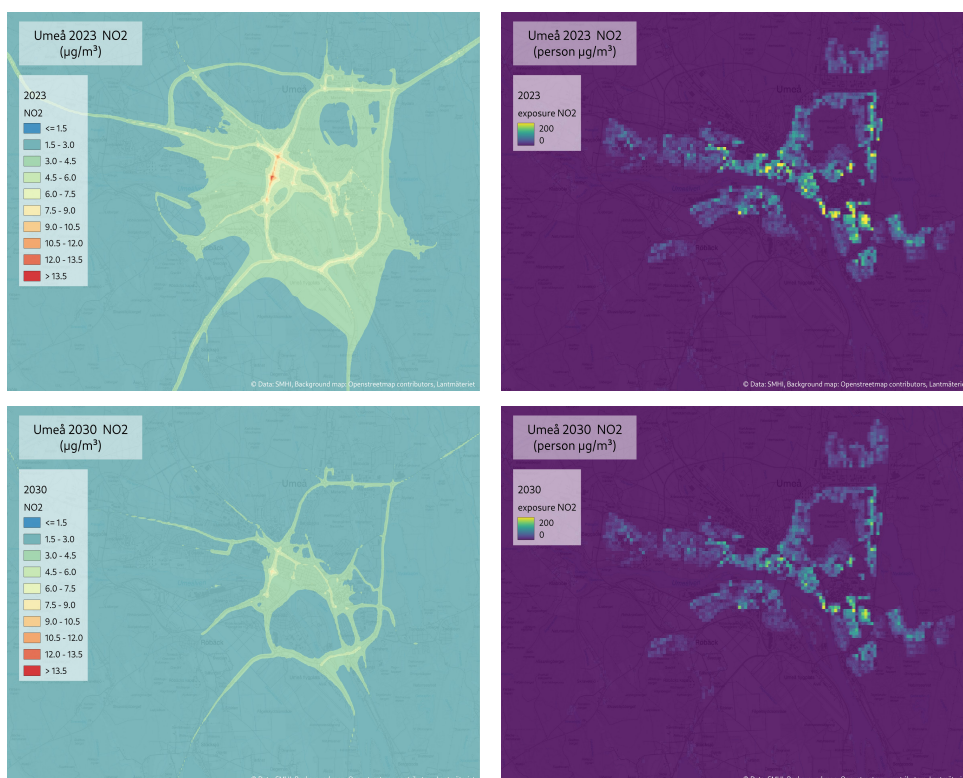


Figure 32. Concentrations and exposure for Umeå of NO₂ for 2023 and 2030.

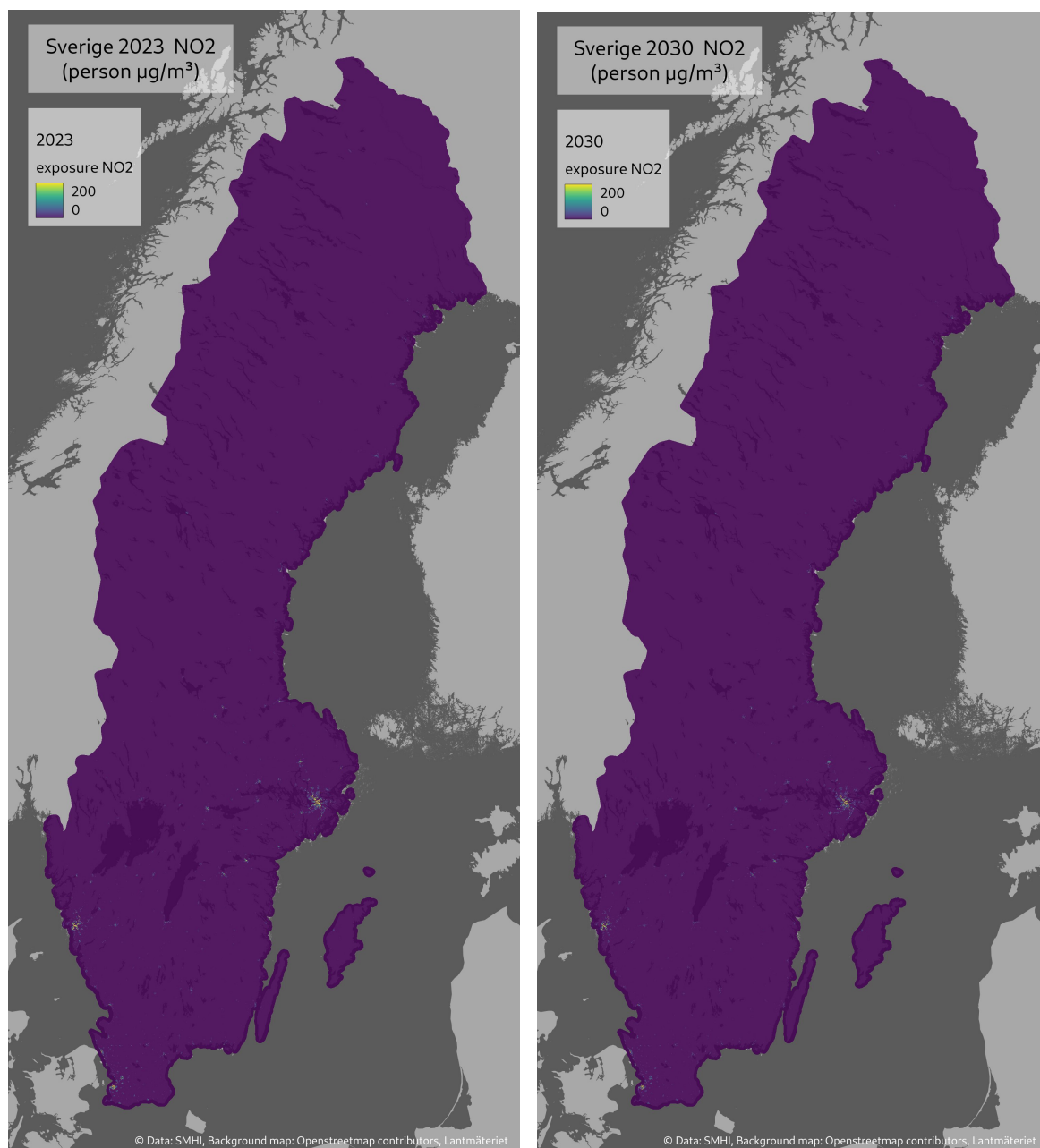


Figure 33. Population exposure for the whole country of NO₂ yearly mean for 2023 and 2030.

B. Concentrations and exposure of PM_{2.5}

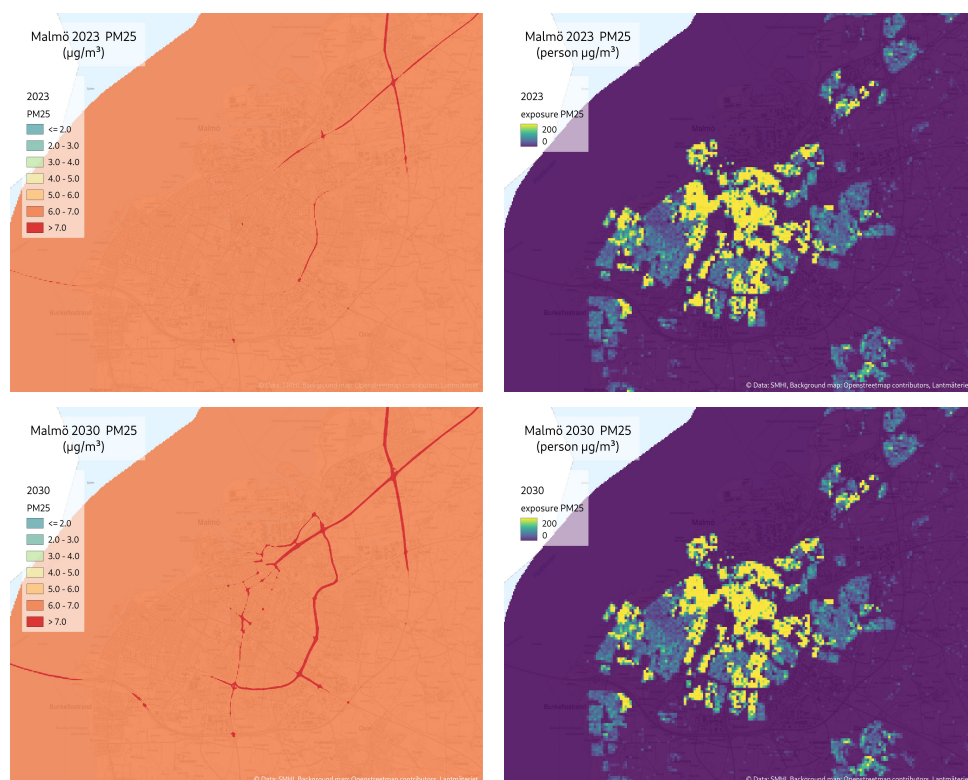


Figure 34. Concentrations and exposure for Malmö of PM_{2.5} for 2023 and 2030.

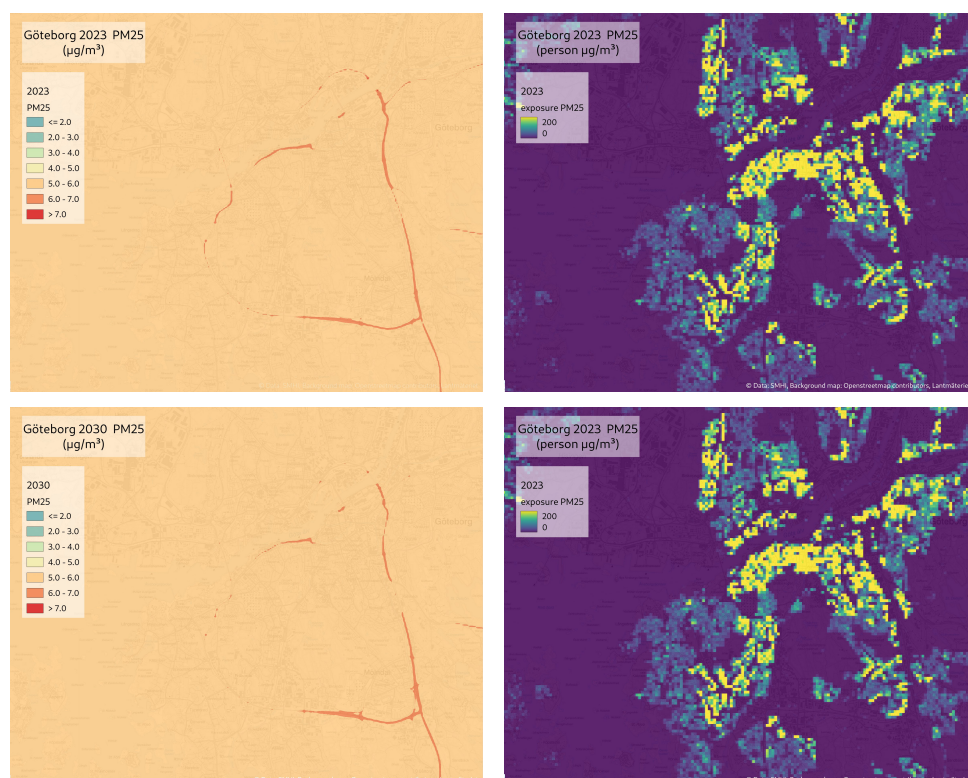


Figure 35. Concentrations and exposure for Gothenburg of PM_{2.5} for 2023 and 2030.

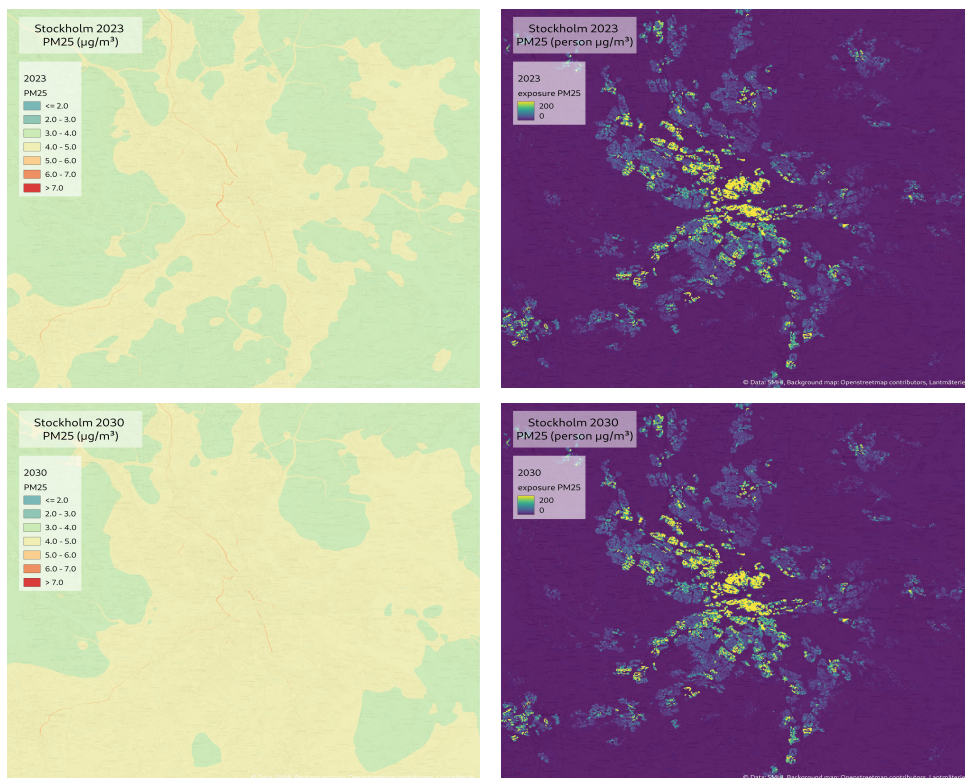


Figure 36. Concentrations and exposure for Stockholm of $\text{PM}_{2.5}$ for 2023 and 2030.

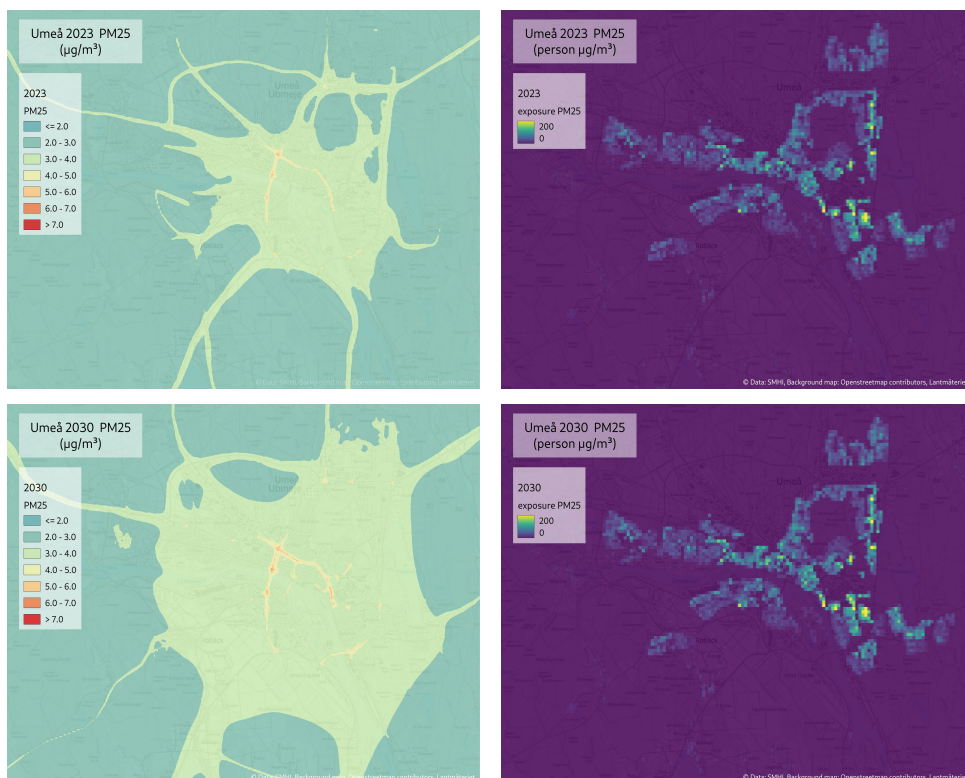


Figure 37. Concentrations and exposure for Umeå of $\text{PM}_{2.5}$ for 2023 and 2030.

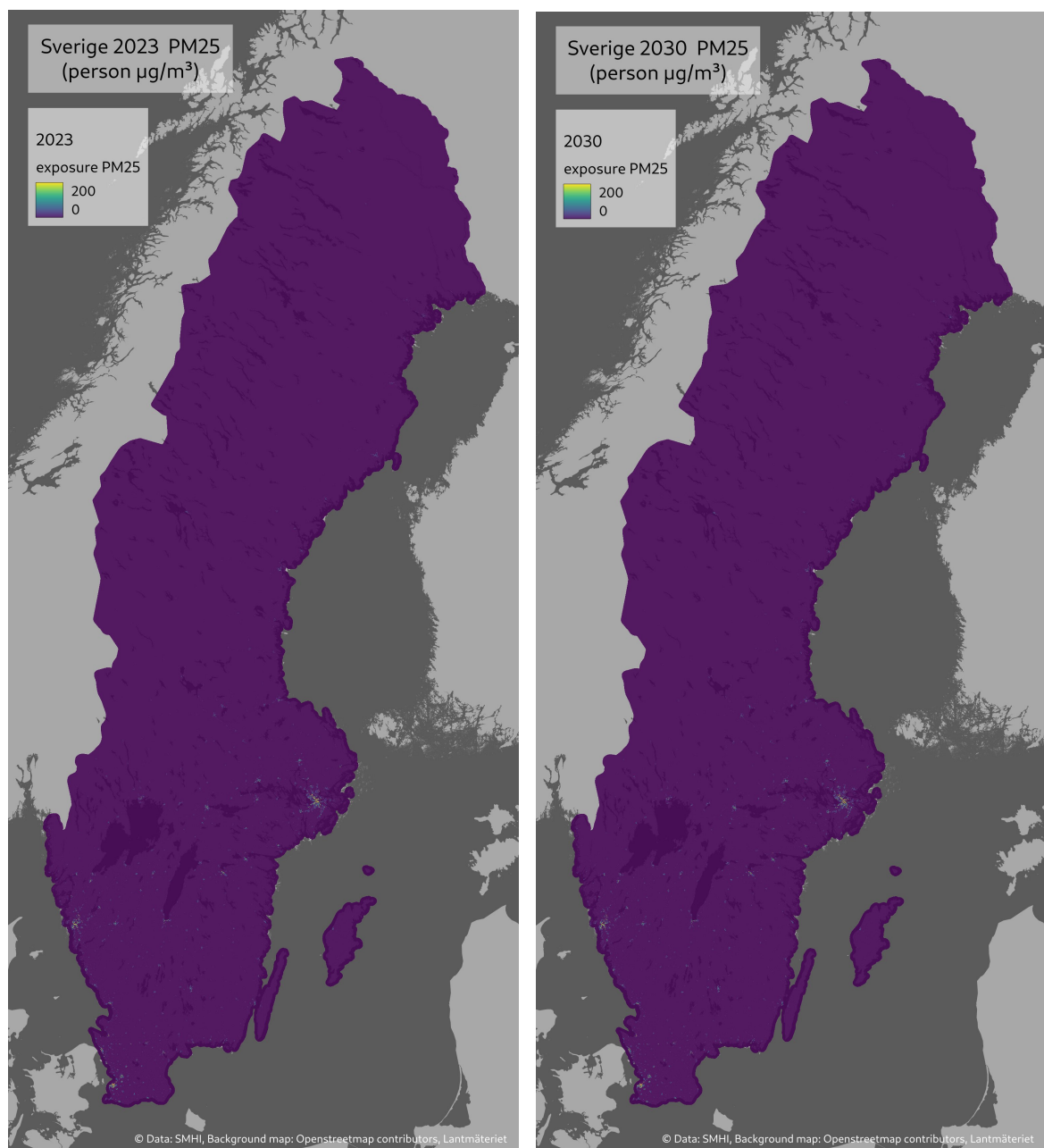


Figure 38. Population exposure for the whole country of PM_{2.5} yearly mean for 2023 and 2030.

C. Concentrations and exposure of PM₁₀

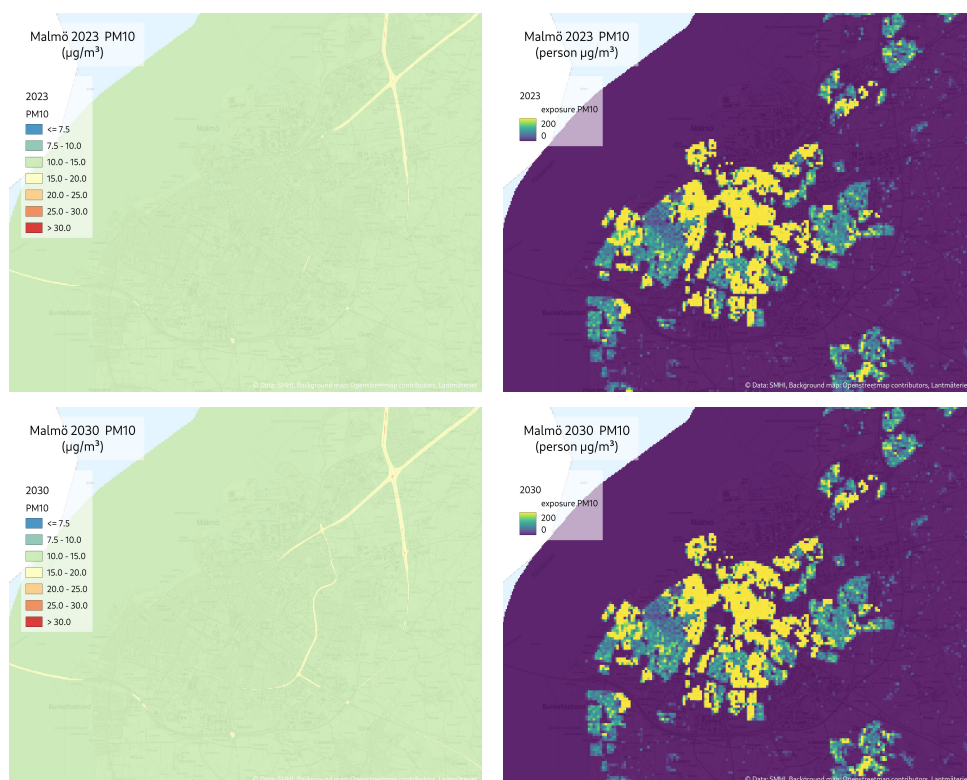


Figure 39. Concentrations and exposure for Malmö of PM₁₀ for 2023 and 2030.

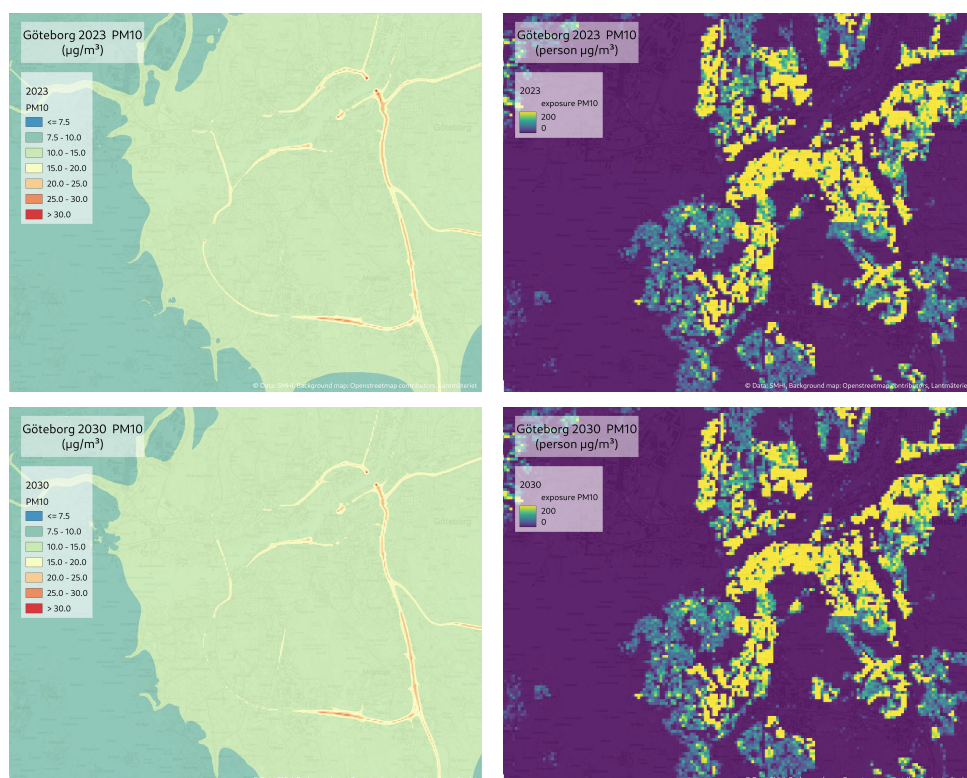


Figure 40. Concentrations and exposure for Gothenburg of PM₁₀ for 2023 and 2030.

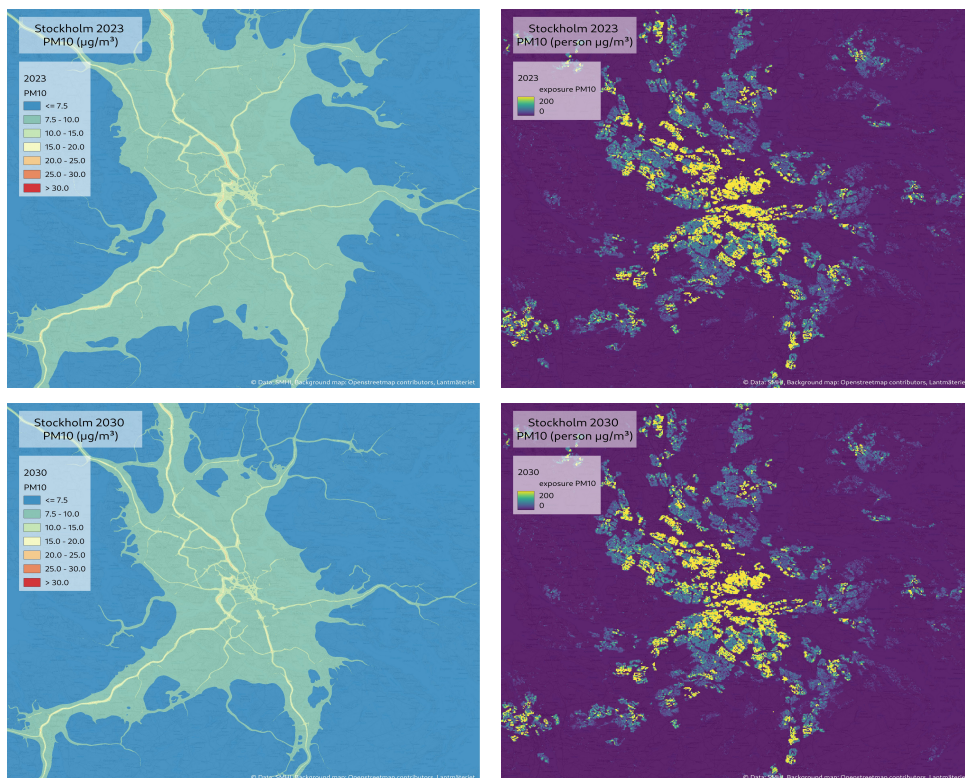


Figure 41. Concentrations and exposure for Stockholm of PM_{10} for 2023 and 2030.

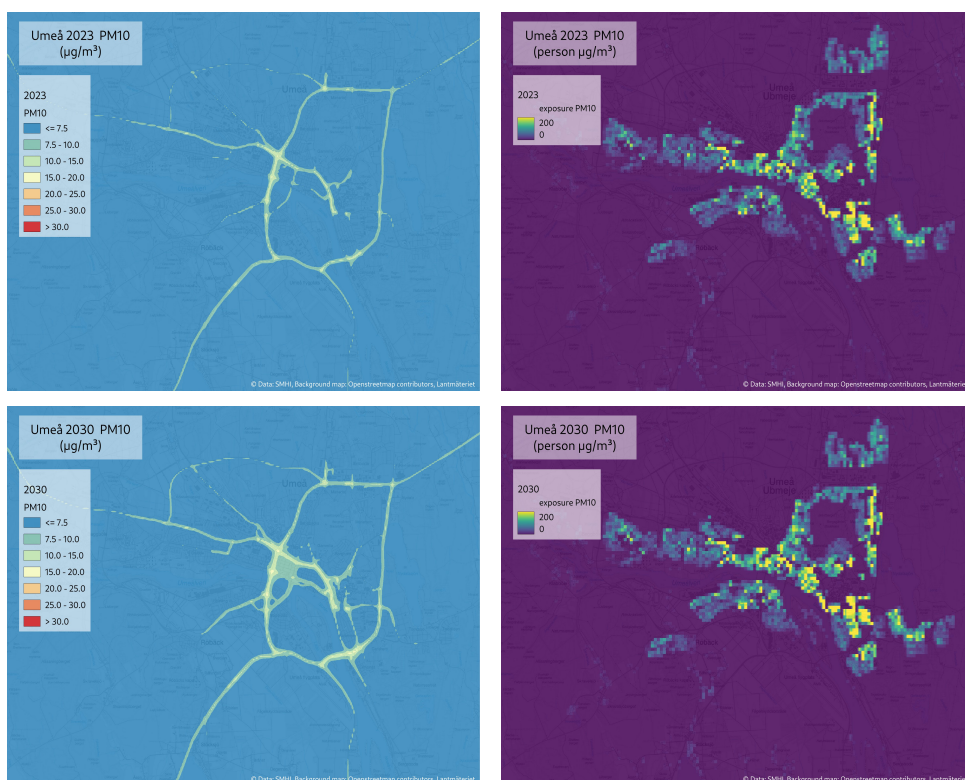


Figure 42. Concentrations and exposure for Umeå of PM_{10} for 2023 and 2030.

D. Concentrations and exposure of O₃

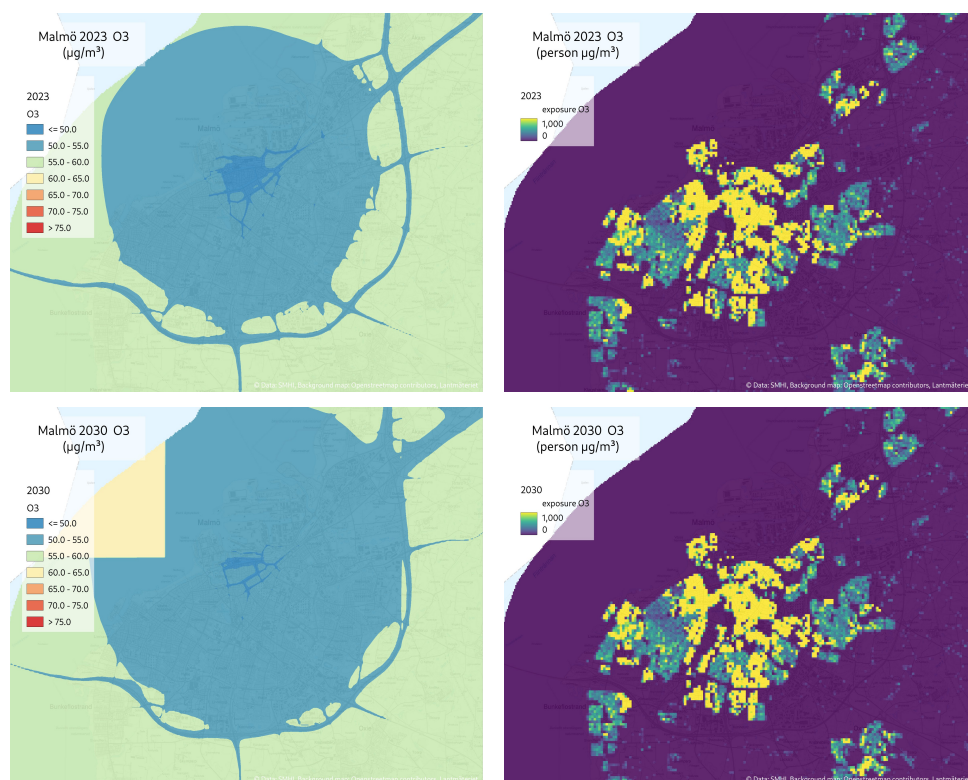


Figure 43. Concentrations and exposure for Malmö of O₃ for 2023 and 2030.

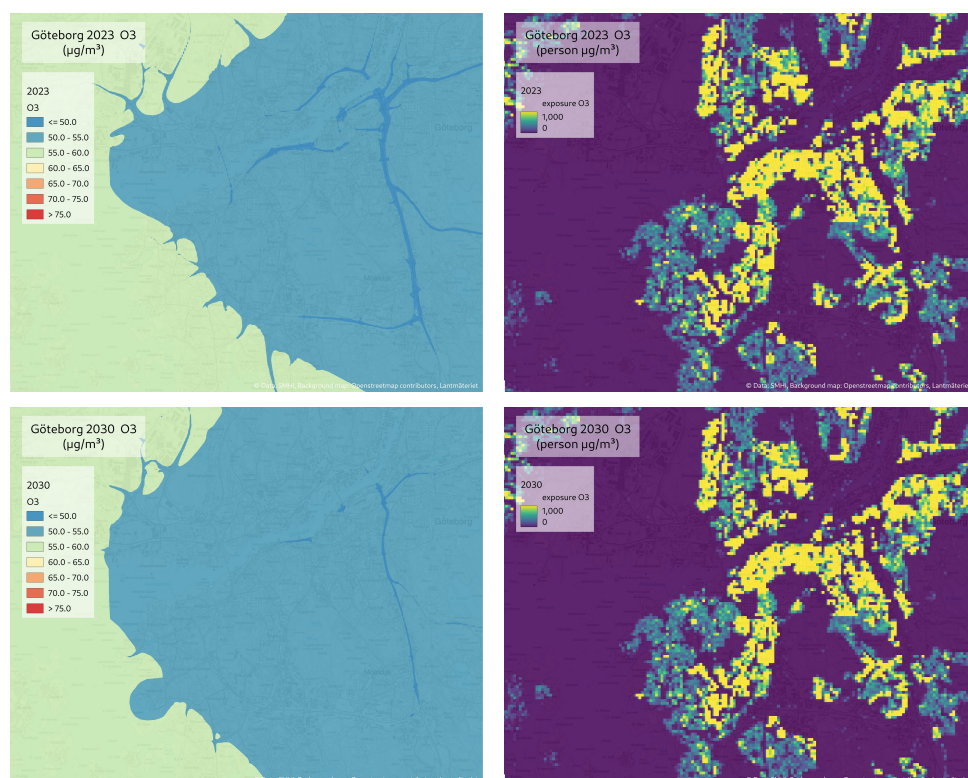


Figure 44. Concentrations and exposure for Gothenburg of O₃ for 2023 and 2030.

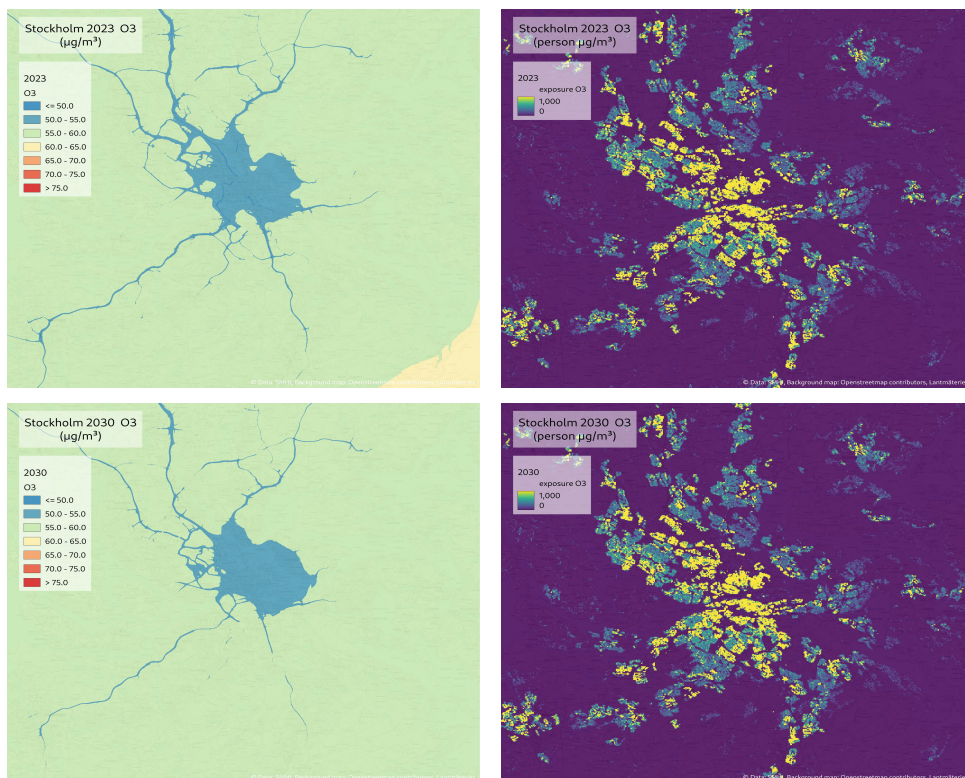


Figure 45. Concentrations and exposure for Stockholm of O₃ for 2023 and 2030.

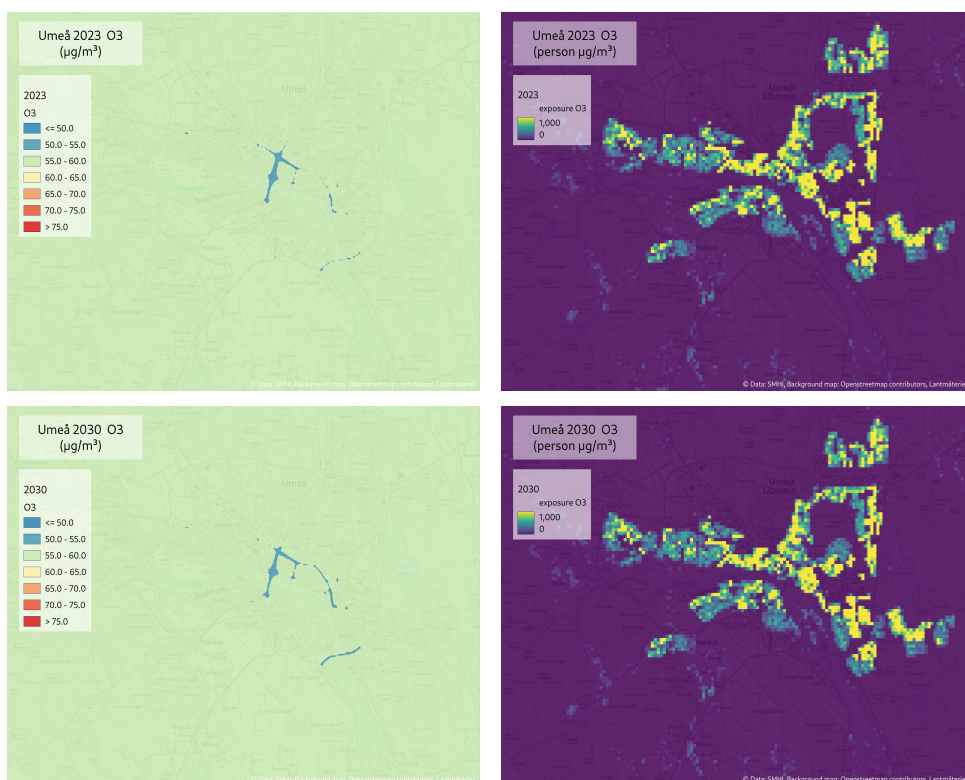


Figure 46. Concentrations and exposure for Umeå of O₃ for 2023 and 2030.

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