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Summary
In this study population exposure to annual mean concentrations of NO_2, PM_{10} and $PM_{2.5}$ in ambient air has been quantified, and the health and associated economic consequences have been calculated based on these results. To allow application of known exposure-response functions for assessment of health effects this study exclusively focus on regional and urban background concentrations.

The results from this study show that background concentrations of the examined pollutants in 2019 were overall low. Nearly the entire Swedish population was exposed to concentrations below the environmental standards, and 98%, 90% and 89% was exposed to concentrations below the environmental objectives for NO_2, PM_{10} and $PM_{2.5}$ respectively. Exposure to the highest concentrations was found in the most polluted central parts of our largest cities. However, comparing the results to the recently updated WHO 2021 Air Quality Guidelines our calculations indicate that 82% of the Swedish population is exposed to unacceptable concentrations of $PM_{2.5,}$ and 11 % to unacceptable levels of NO_2.

In the 2015 assessment we estimated a total burden of approx. 7600 deaths per year. In 2019 we estimate a total of approx. 6740 deaths per year, and there are several factors behind this reduction, both lower concentrations and, as a result of new reports, changes in both directions in the assumed relative risks.
Finally, the health impacts from exposure to NO_2 and $PM_{2.5}$ can be conservatively estimated to cause socio-economic costs of ~168 billion Krona in 2019. Just absence from work and studies can be estimated to cause socio-economic costs of ~0.02% of GDP in Sweden.

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Summary

Air pollution concentrations in Swedish cities are among the lowest in Europe. Despite this, health impacts due to exposure to ambient air pollution is still an important issue and the concentration levels, especially of nitrogen dioxide (NO_2) and particles (PM_{10} and $PM_{2.5}$), occasionally exceed the air quality standards at street level in many urban areas.

IVL Swedish Environmental Research Institute and the Section of Sustainable Health, Department of Public Health and Clinical Medicine at Umeå University have, on behalf of the Swedish EPA, performed a health impact assessment (HIA) for the year 2019. The population exposure to annual mean concentrations of NO_2, PM_{10} and $PM_{2.5}$ in ambient air has been quantified, and the health and associated economic consequences have been calculated based on these results.

To allow application of known exposure-response functions for assessment of health effects this study exclusively focus on regional and urban background concentrations. Roadside concentrations are not addressed here. The results from this study show that background concentrations of the examined pollutants in 2019 were overall low, well below the environmental standards in most parts of the country. The background concentrations were also below the environmental objective for all examined pollutants.

Nearly the entire Swedish population was exposed to concentrations below the environmental standards, and 98%, 90% and 89% was exposed to concentrations below the environmental objectives for NO_2, PM_{10} and $PM_{2.5}$ respectively. Exposure to the highest concentrations was found in the most polluted central parts of our largest cities.

Comparing the results from this study to the previous assessment shows a decrease in mean population exposure to NO_2 and PM. For NO_2, we find an overall downward trend in concentrations and exposure since 2005. The same trend can also be seen for both PM_{10} and $PM_{2.5}$, with a relatively strong decrease of 12% in population exposure to concentrations above the environmental objective between 2015 and 2019. In the recently updated WHO 2021 Air Quality Guidelines, the recommended maximum exposure is considerably lowered compared to the environmental objective for both NO_2 and $PM_{2.5}$. Based on these guidelines, our calculations indicate that 82% of the Swedish population is exposed to unacceptable concentrations of $PM_{2.5}$, and 11 % to unacceptable levels of NO_2.

The effect of fine particles ($PM_{2.5}$) on mortality has been the outstanding health impact from air pollution exposure in almost all health impact assessments. This association continues to be the major health problem according also to the new WHO Air Quality Guidelines. During the more than 15 years that have passed between the last and new air quality guidelines, new studies have reported adverse effects at much lower levels than previously reported, which now has resulted in more strict guidelines. Moreover, recent studies of mortality at rather low concentrations of $PM_{2.5}$ have reported higher relative risks per unit increase in concentration (steeper increase), especially for $PM_{2.5}$ from local sources. In addition, air pollutants are also associated with more health outcomes than before, which makes it even more difficult to compare old assessments and attributed number of cases and health costs with new assessments of the health burden.

Using the modelled exposure levels and presented assumptions and sources behind the health impact assessment, we have estimated the excess mortality associated with long-term exposure to urban (local source) $PM_{2.5}$ as vehicle exhaust and wear particles in the fine fraction to result in 268

(95% CI 197-283) and 488 (95% CI 359-515) deaths per year respectively, without assuming any threshold below which there is no association. The urban contribution of NO_2 is dominated by local emissions from motor vehicles and is estimated to result in 627 deaths per year (95% CI 312-1233). Together urban $PM_{2.5}$ and NO_2 mainly from road traffic is estimated to be associated with almost 1400 deaths per year in Sweden.

The modelled exposure to $PM_{2.5}$ from local residential wood burning is studied with the same assumptions as for local traffic-related $PM_{2.5}$. We have estimated the excess mortality associated with long-term exposure to $PM_{2.5}$ from local wood burning to result in 708 (95% CI 520-747) deaths per year. The modelled exposure to regional background and long-distance transported $PM_{2.5}$, particles not emitted from the local sources such as traffic and domestic heating, is estimated to result in 4652 (95% CI 3398-5033) deaths per year.

In the 2015 assessment we estimated a total burden of approx. 7600 deaths per year. We now estimate a total of approx. 6740 deaths per year, and there are several factors behind this reduction, both lower concentrations and, as a result of new reports, changes in both directions in the assumed relative risks.

With the Swedish age-specific baseline mortality the estimated number of years of life lost due to these deaths among persons aged 30+ years are approximately 10 years per preterm death.

Finally, the health impacts from exposure to NO_2 and $PM_{2.5}$ can be conservatively estimated to cause socio-economic costs of ~168 billion Krona in 2019. Just absence from work and studies can be estimated to cause socio-economic costs of ~0.02% of GDP in Sweden.

Sammanfattning

Halterna av luftföroreningar i svenska städer är bland de lägsta i Europa. Trots detta överskrider föroreningshalterna i gaturum, särskilt kvävedioxid (NO_2) och partiklar (PM_{10} och $PM_{2.5}$), i vissa fall de miljökvalitetsnormer (MKN) för människors hälsa som gäller för utomhusluft.

På uppdrag av Naturvårdsverket har IVL Svenska Miljöinstitutet och Avdelningen för hållbar hälsa, Institutionen för folkhälsa och klinisk medicin vid Umeå universitet kvantifierat den svenska befolkningens exponering för halter i luft av NO_2, $PM_{2,5}$ och PM_{10} för år 2019, beräknat som årsmedelkoncentrationer. Även de samhällsekonomiska konsekvenserna av de uppskattade hälsoeffekterna har beräknats.

För att kunna applicera kända dos-responsfunktioner för bedömning av hälsoeffekter från exponering för luftföroreningar har vi i den här studien fokuserat på halter i urban och regional bakgrundsmiljö. Halter i gaturum inkluderas inte. Resultaten visar att halter av de undersökta föroreningarna i bakgrundsluft år 2019 generellt var låga, med halter långt under respektive MKN i större delen av landet. Föroreningskoncentrationerna i bakgrundsluft låg också långt under preciseringarna i miljökvalitetsmålet *Frisk Luft* för alla undersökta föroreningar.

Nästan hela den svenska befolkningen exponerades för koncentrationer under MKN, med 98 %, 90 % och 89 % utsatta för koncentrationer även under miljökvalitetsmålets preciseringar för NO_2, PM_{10} och $PM_{2.5}$. Exponeringen för de högst koncentrationerna sker i de mest centrala delarna av våra största städer.

I jämförelse med bedömningen 2015 beräknas en minskning i medelexponeringen för NO_2 och PM för Sveriges befolkning. För NO_2 fann vi en nedåtgående trend i medelexponering sedan 2005, vilket indikerar att det trendbrott som beräknades 2015 sannolikt var tillfälligt. Motsvarande minskning beräknades för både PM_{10} och $PM_{2.5}$. Minskningen motsvarar en sänkning på upp till 12 % av andelen av befolkningen som exponerades för halter över miljökvalitetsmålets preciseringar. I de nyligen uppdaterade riktlinjer för luft, presenterade av WHO 2021, introducerades en kraftig sänkning av den rekommenderade maximala exponeringen för både NO_2 och $PM_{2.5}$. Utifrån dessa riktlinjer visar beräkningarna att 82 % av den svenska befolkningen utsätts för oacceptabla halter av $PM_{2.5}$ och 11 % för oacceptabla halter av NO_2.

Effekterna på dödligheten av partiklar i finfraktionen ($PM_{2.5}$) har varit den dominerande konsekvensen i nästan alla hälsokonsekvensanalyser om luftföroreningar. Detta samband står också i fokus för WHO:s nya Air Quality Guidelines. Under de 15 år som gått mellan utfärdandet av de senaste och dessa nya rekommendationer från WHO har nya studier rapporterat om negativa hälsoeffekter vid lägre nivåer än tidigare känt, vilket resulterat i mer stränga rekommendationer. Utöver detta har nya studier visat att sambanden vid låga halter ofta visar större riskökning per haltökning, särskilt för partiklar från lokala källor. Samtidigt visar nya studier om föroreningarnas effekter på allt fler hälsoutfall, vilket gör det svårt att jämföra äldre analyser och uppskattade antal fall med nya beräkningar av luftföroreningarnas hälsobörda.

Utifrån de modellberäknade exponeringsnivåerna och de antaganden för hälsokonsekvens-beräkningarna som gjorts utifrån redovisade källor har vi uppskattat att långtidsexponeringen för $PM_{2.5}$ från trafikavgaser respektive $PM_{2.5}$ från trafikens slitageemissioner kan förväntas orsaka 268

(95% KI 197-283) respektive 488 (95% KI 359-515) dödsfall per år i Sverige. De lokala utsläppen av NO_2, som huvudsakligen också kommer från trafiken, kan beräknas leda till ytterligare 627 dödsfall per år (95% KI 312-1233). Sammantaget kan därmed det lokala bidraget av $PM_{2.5}$ och NO_2 i huvudsak från trafik uppskattas ligga bakom nära 1400 dödsfall per år i Sverige.

De modellberäknade exponeringsnivåerna för $PM_{2.5}$ från lokal, småskalig vedeldning bedöms med samma riskantaganden som trafikens partiklar. Vi har beräknat att långtidsexponering för partiklar från småskalig vedeldning leder till drygt 700 dödsfall per år (95% KI 520-747) i Sverige. Den beräknade exponeringen för $PM_{2.5}$ från omgivande bakgrundsluft, inkluderande långdistans-transport, och oberoende av lokala källor som trafik och vedeldning, kan förväntas leda till ungefär 4650 dödsfall per år (95% KI 3398-5033) i Sverige.

I den nationella beräkningen avseende luftföroreningsexponeringen 2015 beräknades det totala antalet dödsfall per år på grund av långtidsexponeringen uppgå till cirka 7600. Vi beräknar nu motsvarande siffra till ungefär 6740 dödsfall per år. Det finns flera förklaringar bakom den lägre uppskattningen, dels beräknat lägre exponering, dels nya rapporter som motiverat förändrade antaganden om risökningens storlek i båda riktningarna.

Med den åldersspecifika dödligheten i Sverige som grund blir det genomsnittliga antalet förlorade levnadsår bland dödsfallen i åldrarna över 30 år ungefär 10 år per förtida dödsfall.

Hälsoeffekter från förhöjda halter av NO_2 och $PM_{2.5}$ kan med konservativa bedömningar skattas orsaka samhällsekonomiska kostnader på ca 168 miljarder svenska kronor år 2019. Enbart produktivitetsförluster från sjukfrånvaro kan uppskattas orsaka samhällsekonomiska kostnader på ca 0,02 % av BNP i Sverige.

1 Introduction

Despite the successful work to improve the outdoor air quality situation in Sweden by reducing emissions from both stationary and mobile sources (SOU 2016:47; Naturvårdsverket, 2018a), the health impacts from exposure to ambient air pollution is still an important issue. As shown in many studies during recent years, the air pollution concentrations, especially of nitrogen dioxide (NO_2) and particles (PM_{10} and $PM_{2.5}$), still exceed the air quality standards in many urban areas, and the impact on human health, due to exposure to these pollutants, is still significant (Grennfelt et al., 2017; Fredricsson et al., 2017; WHO, 2015; WHO, 2016a).

Within the framework of the health-related environmental monitoring programme, conducted by the Swedish Environmental Protection Agency (Swedish EPA), a number of different activities are performed to monitor health effects that may be related to environmental factors. As a part of this programme IVL Swedish Environmental Research Institute and the Department of Public Health and Clinical Medicine at Umeå University have quantified the population exposure to annual mean concentrations of NO_2, PM_{10} and $PM_{2.5}$ in ambient air in Sweden. In this study, health and associated economic consequences of the calculated exposure to air pollution have also been assessed. This is a recurring study, conducted on a five-year interval, with the last evaluation in 2015 (Gustafsson et al 2018). However, due to the irregularities in emissions caused by changed patterns in travel and other activities during the Covid-19 pandemic in 2020, this study is based on the year 2019 to represent a normal exposure situation.

2 Background

Emission reductions regarding both NO_2 and particles have been on the agenda for the past few decades and progress have been made, but urban areas are growing, and more people are moving to cities where the air pollution load in general is higher than in rural areas.

Environmental conditions and trends have been monitored for a long time in Sweden. Already in 1990/91 (winter half year, October-March) a study was performed, within the Swedish EPA´s investigation of the environmental status in the country, concerning the number of people exposed to ambient air concentrations of nitrogen dioxide (NO_2) in excess of the ambient air quality guidelines valid at that time (Steen and Cooper, 1992). Similar calculations were later made for the conditions during the winter half years 1995/96 and 1999/2000 using the same technique (Steen and Svanberg, 1997; Persson et al., 2001), and the results indicated a slight decrease in the excess exposure.

In 2007 a study of NO_2 exposure in Sweden for the year 2005 was conducted using a statistical model for air quality assessment, the so-called URBAN model, which can be used to estimate urban air pollution levels in Sweden and quantify population exposure to ambient air pollutants (Persson et al., 1999; Persson and Haeger-Eugensson, 2001; Haeger-Eugensson et al., 2002; Sjöberg et al., 2004; Sjöberg et al., 2007). Later the method was further developed to include the population exposure to PM_{10} and $PM_{2.5}$ (Sjöberg et al., 2009). Using the calculated population exposure to NO_2, PM_{10} and $PM_{2.5}$ the health consequences and socio-economic costs were calculated for 2005 (Sjöberg et al., 2007; Sjöberg et al., 2009).

The same basic method, using the URBAN-model, was used to calculate the exposure, health impact and socio-economic costs of NO_2, PM_{10} and $PM_{2.5}$ concentrations in Sweden for 2010

(Gustafsson et al., 2014) and 2015 (Gustafsson et al., 2018). In Table 1 the main results from the 2005, 2010 and 2015 studies are presented.

Table 1 **Main results from the 2005, 2010 and 2015 exposure studies (Sjöberg et al., 2007, Sjöberg et al., 2009, Gustafsson et al., 2014, Gustafsson et al., 2018).**

		2005	2010	2015
Total population (no. of inhabitants)		8 899 724	9 546 546	9 839 105
Mean population weighted exposure (µg/m³)	NO_2	6.3	6.2	6.4
	PM_{10}	13.0	12.0	12.5
	$PM_{2.5}$	9.8	8.6	8.3
Percentage of population exposed to concentrations above the environmental objective	NO_2 (20 µg/m³)	2.3%	2.7%	2.9%
	PM_{10} (15 µg/m³)	38%	25%	22%
	$PM_{2.5}$ (10 µg/m³)	49%	28%	23%
Percentage of population exposed concentrations above the environmental quality standard	NO_2 (40 µg/m³)	0%	0%	0%
	PM_{10} (40 µg/m³)	0.4%	0.3%	0.3%
	$PM_{2.5}$ (25 µg/m³)	0%	0.6%	0.6%

The studies showed that most of the country had concentrations of NO_2, PM_{10} and $PM_{2.5}$ in ambient air well below the environmental standards for annual means (Sjöberg et al., 2007; Sjöberg et al., 2009; Gustafsson et al., 2014, Gustafsson et al., 2018). Only in the larger urban centers, concentrations were reaching the same magnitude as the environmental standards. In parts along the west coast, concentrations approaching the long-term environmental objective were noted, especially for PM. The calculations showed that nearly the entire Swedish population was exposed to concentrations below the environmental standards. Regarding the environmental objectives, exposure to PM_{10} and $PM_{2.5}$ continually show a positive trend with a larger proportion of the population below the set objectives. For NO_2 a weak negative trend is presented. This was connected to that the great majority of the recent population growth has occurred in densely populated areas (SCB, 2015). This, in combination with an ongoing densification of existing urban spaces (e.g. Boverket, 2016; SKL, 2015), result in a larger proportion of the population being exposed to the higher urban NO_2 concentrations. Population weighted mean concentrations were found to remain relatively stable with a slight decrease in PM. Sjöberg et al (2007) also presented a trend analysis between 1990 and 2010 showing a continuous reduction in NO_2 exposure. During the same period the annual mean of NO_2 decreased by almost 40%, which was attributed to a reduction of the total NO_x emissions in Sweden (Naturvårdsverket, 2017).

2.1 Aim of this study

The aim of this study is to update the calculated exposure to yearly mean concentrations of NO_2, PM_{10} and $PM_{2.5}$ on a national scale for 2019, and to assess the associated long-term health impact as well as the related economic consequences. The results are also compared to earlier studies to assess trends. To enable a comparison with previously calculated numbers, the same basic calculation methods as described in Gustafsson et al. 2014, and Gustafsson et al. 2018 are applied in this study. Minor alterations to the method have been introduced when needed due to data availability, as described in chapter 3.

3 Methods

The method applied for calculation of ambient air concentrations and exposure to air pollutants has been described earlier (Sjöberg et al., 2007; Sjöberg et al., 2009). The empirical statistical URBAN model is used as a basis. Urban background monitoring data and a local ventilation index (calculated from mixing height and wind speed) are required as input information for calculating the air pollution levels in the urban background. To calculate the exposure across Sweden, regional background concentration of the NO_2, PM_{10} and $PM_{2.5}$, as well as population distribution, are needed in addition to the calculated urban background air concentrations. The concentration patterns of NO_2, PM_{10} and $PM_{2.5}$ over Sweden were calculated with a 1x1 km grid resolution (section 3.1, 3.2 and 3.3). PM_{10} and $PM_{2.5}$ were calculated both as total annual means and separated for different source contributions (section 3.4).

The quantification of the annual mean population exposure to NO_2, PM_{10} and $PM_{2.5}$ was based on comparisons between the pollution concentrations and the population density. Like the calculated air pollutant concentrations, the population density data had a grid resolution of 1x1 km (section 3.5). By over-laying the population grid to the air pollution grid the population exposure to a specific pollutant was estimated for each grid cell (section 3.6).

To estimate the health consequences, exposure-response functions for the long-term health effects were used, together with the calculated NO_2, PM_{10} and $PM_{2.5}$ exposure (section 3.7). For calculation of socio-economic costs, results from economic valuation studies and other cost calculations were used (section 3.8). These cost estimates were combined with the estimated quantity of health consequences performed in this study to give the related total socio-economic costs of NO_2, PM_{10} and $PM_{2.5}$ concentrations in ambient air during 2015.

3.1 NO_2 concentration calculations

The NO_2 concentration was calculated based on i) regional background levels, and ii) local source contributions to the urban background concentrations. For each urban area the contribution from regional background NO_2 concentration was calculated from the background grid and subtracted from the urban NO_2 concentration to avoid double counting. Hence, only the additional local NO_2 concentration (on top of the background levels) in urban areas was distributed.

3.1.1 Regional background

A national grid (1 x 1 km) representing the regional background concentration of NO_2 was calculated by interpolating measurement data from regional background sites. For 2019, 44 sites with monthly regional background data were used. 23 of these sites were part of the national air quality monitoring network within the Swedish environmental monitoring programme (Naturvårdsverket, 2018b), and the remaining 21 were part of The Swedish Throughfall Monitoring Network (http://krondroppsnatet.ivl.se).

The background grid was calculated for two-month periods during the year to account for seasonal variations in the NO_2 concentration. Dividing the year in two-month periods was deemed an appropriate time resolution as it gave a representation of the seasons without increasing the computational time for the calculations too much. At the end, an annual background map was compiled based on the results calculated from the 6 interpolated bimonthly maps, see Figure 1.

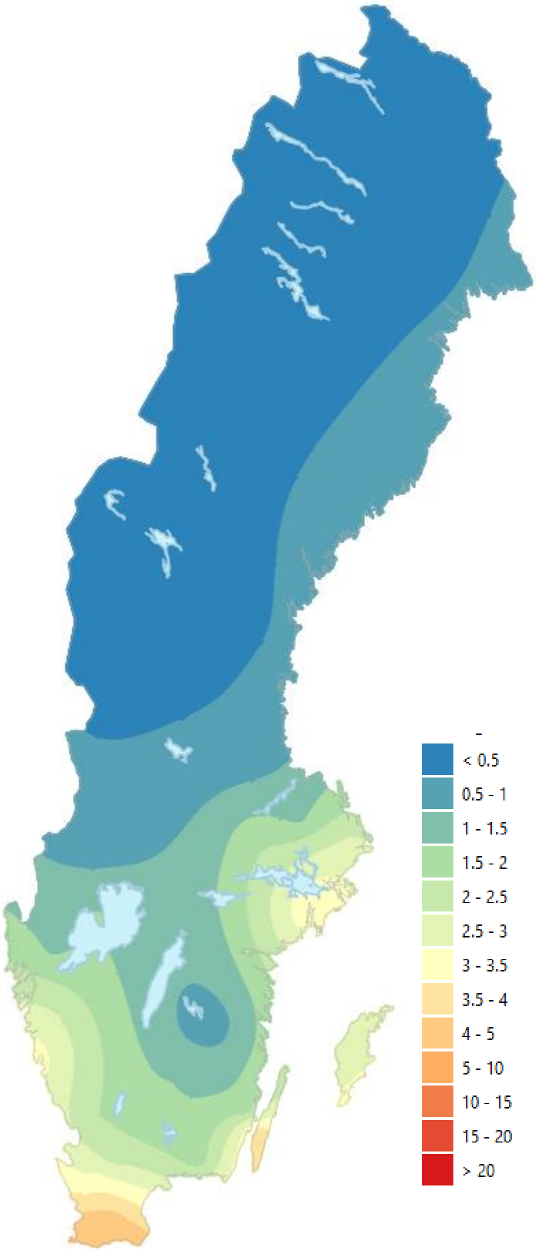


Figure 1 **2019 annual mean regional background concentrations of NO_2 in Sweden, unit µg/m^3.**

3.1.2 Urban background

The urban (local) contribution to NO_2 was calculated using the URBAN model, as described by Sjöberg et al. (2007). The distribution of the locally produced NO_2 in urban background air within cities was estimated based on the area of the city, where the grid cell within this area with the highest number of inhabitants was assigned the highest concentration of NO_2. Each grid cell within the city boundaries was then given a NO_2 concentration proportional to the number of inhabitants in each respective grid cell. The calculated concentrations of air pollutants are valid for the similar height above ground level as the input data (4-8 m) to describe the relevant concentrations for human exposure. The same method was used in the 2015 population exposure assessment, but differed slightly from the earlier assessments as explained in detail in Gustafsson et al. (2018)

The total NO_2 concentrations were then calculated by adding the urban contribution to the regional background NO_2 concentrations for each grid cell.

3.2 PM_{10} concentration calculations

3.2.1 Regional background

Monitoring of particles (PM_{10} and $PM_{2.5}$) in regional background air is carried out at five sites in Sweden in 2019, within the national environmental monitoring programme financed by the Swedish Environmental Protection Agency (www.naturvardsverket.se/data-och-statistik). In previous population exposure assessments, data from this sparse measurement station network has been complemented with calculated distribution patterns by the mesoscale dispersion model EMEP (www.emep.int). However, the EMEP model has undergone several steps of development since the 2015 evaluation, and now reflects not only the regional background concentrations but also the local contribution. It was thus no longer possible to use the previously applied combination of measurements and modelled data to obtain regional background concentrations without risking that the urban contribution is counted double. The 2019 evaluation was therefore calculated based on data from the five available measurement stations, see Figure 2.

When compared to regional background concentrations of PM in previous reports, this change in method generates similar concentrations in most of the inland parts of Sweden, but lower PM concentrations in the coastal areas, especially on the west coast. The rather high concentrations on the Swedish west coast in the previous reports were partly contributed to the high influence of sea-salt in these coastal areas. The suitability of including exposure to sea salt in exposure assessment has been questioned and was therefore discussed in the previous report (Gustafsson et al. 2018). Additionally, the high PM concentrations on the west coast in the 2015 evaluation could not be fully validated as suitable measurement data were not available, but crude comparisons with available data indicated that the 2015 west coast PM concentrations might be slightly overestimated (Linden et al. 2019). This potential over-estimation due to high coastal PM concentrations introduced by the EMEP will thus be avoided in the 2019 assessment.

To separate the regional and urban/local PM_{10} contributions, the regional background concentrations was separated into two-month periods to account for seasonal differences.

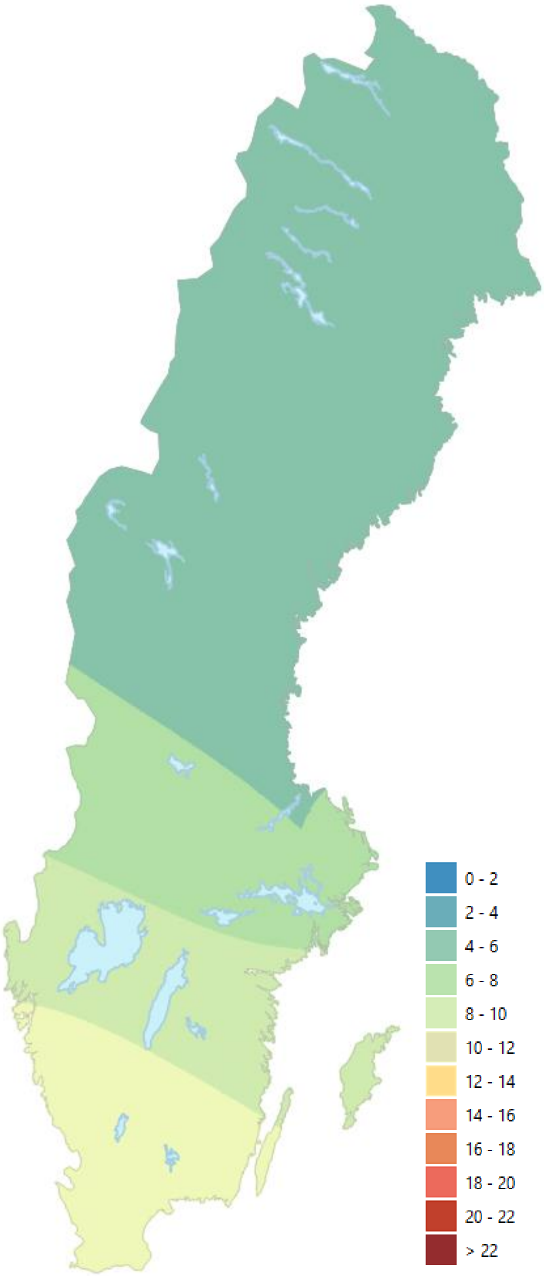


Figure 2 **Annual mean regional background concentrations of PM_{10} in Sweden in 2019, unit µg/m³.**

3.2.2 Urban background

The urban background concentration of PM_{10} was calculated by using the relationship NO_2/PM_{10} in urban background air for the year 2019 (see further Sjöberg et al., 2009; Chapter 3.1.2). To reflect the seasonal variation in the particle load the calculated yearly means were based on concentrations calculated with a bimonthly resolution.

To derive urban background concentrations of PM_{10}, the PM_{10}/NO_2 ratio for the stations providing data of both PM_{10} and NO_2 for the 2019 was used. For data from these stations, regional estimated background concentrations of NO_2 and PM_{10} were subtracted, and ratios of PM_{10}/NO_2 for the remaining local contribution were derived and analysed. In previous reports, this ratio has been found to vary depending on latitude, but in the 2019 assessment no latitudinal dependence was found. This may be caused by a decreasing high latitude coverage of measurement stations, thus reducing possibilities for identifying the latitudinal dependence of this ratio. It may also be caused by a changed relationship of PM_{10}/NO_2 concentrations over time. The true cause of this changed pattern could not be identified within the frame of this assessment and therefore the same PM_{10}/NO_2 ratio was used regardless of latitude. The resulting bi-monthly ratios are presented in Table 2.

Table 2 Bi-monthly PM_{10}/NO_2 ratios used derive urban background concentrations of PM_{10}. Ratios based on the locally developed contribution to the concentrations in urban background air.

Time period /Variable	Jan-Feb	Mar-Apr	May-Jun	Jul-Aug	Sep-Oct	Nov-Dec
PM_{10}/NO_2 ratio	0.64	1.06	0.51	0.51	0.24	0.51

3.3 $PM_{2.5}$ concentration calculations

Based on the calculated PM_{10} concentrations, $PM_{2.5}$ in regional background and local source contributions to the urban background concentrations were calculated. For each urban area the contribution from the regional background PM_{10} concentration was calculated and subtracted from the urban PM_{10} concentration to avoid double counting.

3.3.1 Regional and urban background

The estimation of the $PM_{2.5}$ concentrations in Sweden was performed using a ratio relation between monitored $PM_{2.5}/PM_{10}$ since 2000 (data from www.smhi.se). The ratio varies with type of site location, from lower values in city centers to higher values in regional background, where a large proportion of the PM_{10} concentration consists of $PM_{2.5}$. Three different ratios were based on monitoring data; for regional background, central urban background and suburban background (a mean between the two others) conditions (Table 3). This is a rough estimate as the ratio is likely to vary between years and with season. To enable comparison between the exposure assessment studies, the same urban background ratio as used in the 2005, 2010, and 2015 was also used in the 2019 calculations.

Table 3 Calculated ratios applied for different types of surroundings.

Type of area	Ratio ($PM_{2.5}/PM_{10}$)
Central urban background	0.6
Suburban background	0.7
Regional background	0.8

The ratios in Table 3 were allocated to the urban areas based on the population distribution pattern. For the three major cities (Malmö, Göteborg and Stockholm) 60% of the population was estimated to live in central urban areas and 40% in suburban areas. For the smaller cities, 45% of the population was estimated to live in central urban areas and 55% in suburban areas. These population distribution relations are based on information from cities in the eastern part of USA (Figure 3), as no similar studies of distribution patterns was found for European conditions.

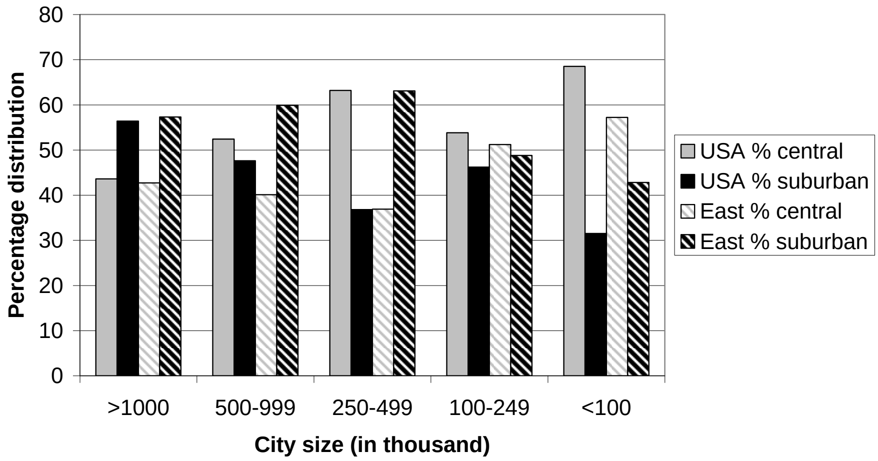


Figure 3 **Relations between distribution of population in central parts and suburban parts of cities, both for all cities in the USA and for cities located in the eastern part of the USA (developed in USA by Demographia, 2000, www.demographia.com/).**

The GIS-methodology applied to allocate the grid cells within each city into the different classes in Figure 3 consists of several steps: At first, the population size estimated to the central areas [pop_central] was identified (60 or 45% of the population depending on the size of the city). Secondly, the grid cell with the largest population [pop_large] in the city was identified and allocated to the central area. The population of that grid cell was then subtracted from the population size of the central area, i.e. [pop_central] – [pop_large]. Then the grid cell with the second largest population was identified. This loop was continued until the population in the central areas [pop_central] had been allocated to grid cells. The remaining grid cells were allocated to the suburban class, corresponding to the remaining 40 or 55% of the population.

When all grid cells had been allocated to the three classes (central urban, suburban and rural background), the ratio ($PM_{2.5}/PM_{10}$) in Table 3 was applied to the PM_{10} map to calculate the $PM_{2.5}$ map.

3.4 Separation of particle source contributions

Since it is assumed that the relative risk factors for health impact varies depending on the source of particles (WHO, 2013b) the total PM_{10} concentration was separated into different source contributions by using a multivariate method (see further Chapter 3.4.4). In the following sections calculations of different contributions of particles are described.

3.4.1 Small scale domestic heating

Small scale domestic wood fuel burning is an important contributor to particle emission in Sweden (Naturvårdsverket, 2018a). Specific information on the use of wood fuel on municipality level was not available for 2019. Therefore, in order to evaluate the proportion of PM from small scale domestic wood fuel burning, a relationship was established between total biofuel (of which wood fuel makes up a significant part) and wood fuel consumption on municipality level using data from 2003 (SCB, 2007). This relationship was then applied to the biofuel consumption data from 2019 to derive the wood fuel consumption (www.scb.se). Figure 4 and Figure 5 present the distribution of energy consumption on a county level. The proportion is governed by the air temperature and the supply of wood, as well as traditions in household fuel use in the area.

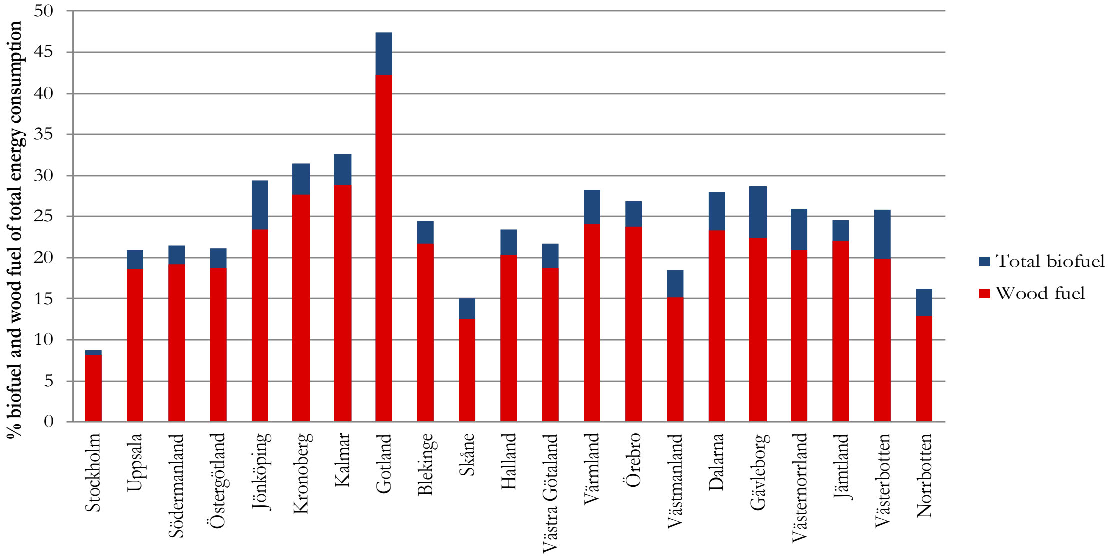


Figure 4 Percentage of total energy consumption from biofuels including wood fuel (blue bars), the percentage from wood fuel (red bars) and per county in 2019.

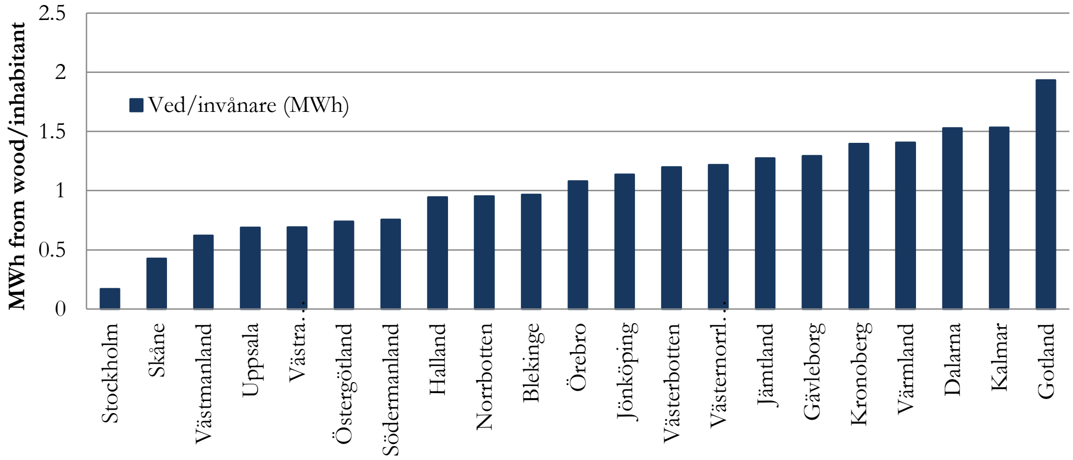


Figure 5 Yearly energy consumption from wood burning (MWh) per inhabitant in each county in 2019.

The energy consumption from wood burning for each municipality in Sweden was drawn from the information presented in Figure 6.

The outdoor air temperature is also an important parameter governing the use of wood for domestic heating. A method for describing the requirement of indoor heating is to calculate an energy index (I_e). The index is based on the principle that the indoor heating system should heat up the building to +17 °C, while the remaining part is generated by radiation from the sun and passive heating from people and electrical equipment. The calculation of I_e is thus the difference between +17 °C and the outdoor air temperature. For example, if the outdoor temperature is -5 °C the I_e will be 22. During spring, summer and autumn the requirement of indoor heating is less than during wintertime (November – March). Thus, during those months, the outdoor temperature is calculated with a baseline specified in Table 4. The energy index calculations were in previous studies based on monitored outdoor temperature as means for 30 years at 535 sites distributed over Sweden (www.smhi.se) and result in monthly national distribution of the energy indices.

As specified in Energimyndigheten (2019) the definition of a normal temperature year for Sweden is gradually changing. In calculations for the 2019 assessment are based on the period 2003- 2014 rather than the previously used period 1981-2010. The energy index is therefore reduced accordingly, while the distribution over the country remains the same as in previous assessments (e.g. Figure 8 in Gustafsson et al. 2018).

Based on these interpolated maps, bimonthly means of I_e were extracted for each of the 2016 towns in Sweden and used for calculation of a seasonal variation in the wood fuel consumption.

Table 4 **The base line for the outdoor temperature for calculation of I_e during April - October.**

Months	Baseline outdoor temperature (°C)
April	+ 12
May-July	+ 10
August	+ 11
September	+ 12
October	+ 13

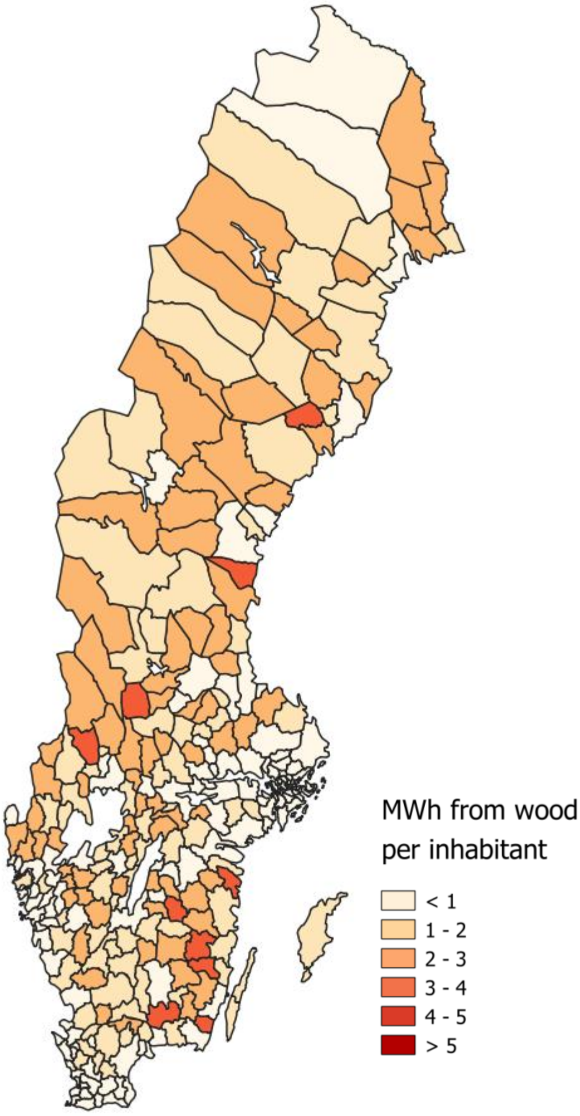


Figure 6 **Energy consumption from wood burning (MWh)/inhabitant in each municipality in Sweden in 2019.**

3.4.2 Traffic induced particles

Traffic contributes to the total concentration of PM_{10} both directly through exhaust emissions and to PM from brake, tyre and road ware from vehicles, and secondarily through re-suspension of dust from roads. Traffic related particle concentrations are associated with the NO_2 concentration in urban areas (Sjöberg et al., 2007). Therefore, the previously calculated NO_2 concentrations for all densely built-up areas in Sweden were used to include the direct emissions from traffic in the multivariate analysis to determine the contribution from this source.

Road dust arises mainly from wear of the road surface, brakes, and tyres, and in particular the use of studded tyres. It has been shown that the number of cars using studded tyres is a parameter that regulates the amount of road dust (Gustafsson et al., 2005). Therefore, the use of studded tyres was also included as a parameter in the multivariate analysis.

Re-suspension of road dust occurs mainly during late winter and spring, as a result of the drying of the road surfaces. The accumulated road dust goes into suspension in the air, as a result of traffic induced turbulence as well as wind. Suspension of dust and soil from non-vegetated land surfaces also occurs in springtime when soil surfaces dries up and before the vegetation season starts, mainly in the southern part of Sweden.

The use of studded tyres in January through March 2019 in six different road administration regions (Figure 7 and Figure 8) was obtained from The Swedish Transport Administration (Trafikverket, 2019). As this information is not available with a monthly resolution throughout the year, a monthly based usage of studded tyres in the road administration regions was established using the distribution pattern derived by Sjöberg et al. (2009).

From this information, bimonthly means of the percentage of studded tyres used were calculated for each densely built area in Sweden to be further used in the multivariate analysis.

In the previous assessments, the PM associated with road dust have been assumed to consist of larger PM, and thus assigned to the size category PM_{10} with fewer established exposure-response relations than $PM_{2.5}$. However, national emission data for 2015 and later have been size fractioned, reporting 20% within the smaller PM size $PM_{2.5}$ (Söderkvist et al, 2019). In this 2019 exposure assessment, 20 % of the road dust PM was assumed to fall within the size category $PM_{2.5}$.

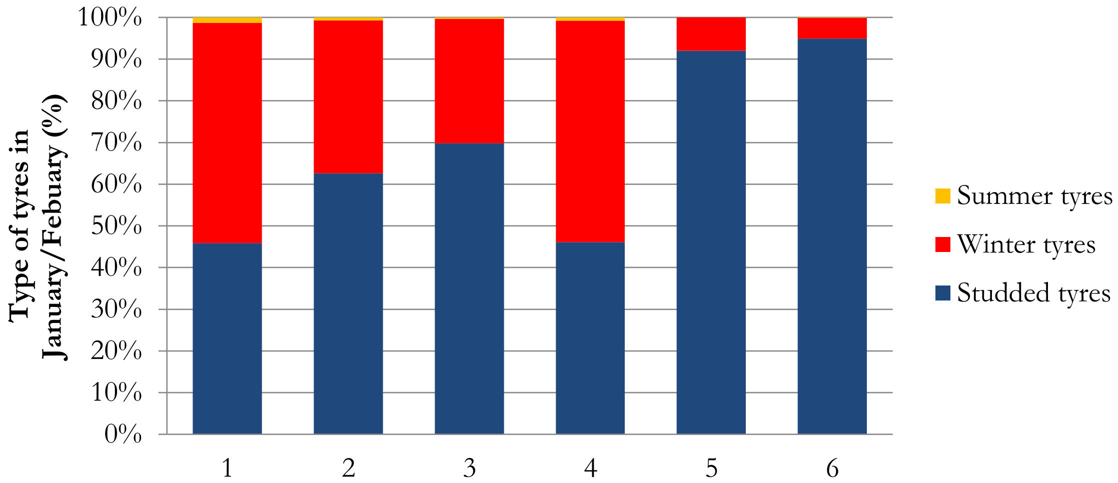
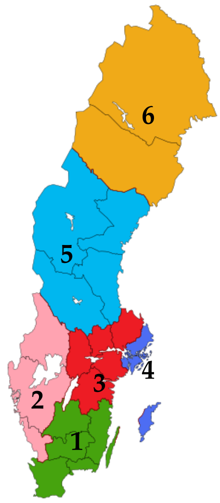


Figure 7 **The usage of different types of tyres in January/February within the six road administration regions in Sweden (visualized in Figure 8).**



Swedish road administration regions:

1. South
2. West
3. East
4. Stockholm and Gotland
5. Central north
6. North

Figure 8 **The six road administration regions of Sweden.**

3.4.3 Dispersion parameters

Meteorology also influences the air pollution concentrations. This can be defined in many ways, but a so-called mixing index (V_i) has been shown to capture both local (such as topographical and coastal effects) and regional variations (such as location of high/low pressures). V_i is determined by multiplying the mixing height and the wind speed. V_i's have been calculated for the whole of Sweden by using an advanced meteorological dispersion model, TAPM (see further Haeger-Eugensson et. al., 2002).

According to Chen (2000) the calculation of the mixing height and wind speed by the TAPM model is well in accordance with measurements. In Sweden, different weather systems are dominant in the northern and southern parts, especially during winter. This influences the V_i, and thus dispersion of air pollutants differs in the south and in the north. However, this latitudinal pattern is reduced during spring and summer, when other local differences, such as topographical effects, become more important to the dispersion pattern (see Sjöberg et al., 2007).

3.4.4 Multivariate data analysis

In this project, Multivariate data analysis (MVDA) has been used to separate different contributions to the total PM concentration based on six parameters which represent different sources as presented in the previous chapters. The data has been evaluated for 2016 communities in Sweden in 2019.

Typical examples of MVDA methods are principal component analysis (PCA) and partial least squares (PLS) (Martens and Naes, 1989; Wold et al., 1987; Geladi and Kowalski, 1986). For further description of MVDA and evaluation of model performance see Sjöberg et al. (2009).

In this project, the data was divided into six different bimonthly time periods. This is necessary to capture the use of studded tyres and the wood fuel burning, which contribute less to the PM_{10} content during the summer and more during the winter. Therefore, one generic model representing a whole year, would not give a good prediction of the PM_{10} content. This resulted in six different PLS models, one for each bimonthly period, predicting the PM_{10} content based on:

- urban background NO_2 concentration;
- usage of studded tyres;
- wood fuel burning;
- energy index;
- mixing index ;
- latitude for each community.

Three of the models (month 5-6, 7-8 and 9-10) do not have any contribution from the usage of studded tyres since these types of tyres are not used during the summer in any part of Sweden. This variable was therefore excluded in these three models.

All six models gave good predictions of the PM_{10} content. The maximum possible performance of a model is 100%, which is unrealistic to receive for a model since there are always contributions to the model that cannot be explained, the air does not behave exactly the same at all times. The model performance was here assessed by cross-validation[1], see Sjöberg et al. (2009).

The result presented in Table 5 shows the performance (Q^2)[2] of the models for each time period.

Table 5 **The performance of the models measured as cross validated explained variance for PM_{10}.**

Model	Performance (%)
Month 1-2	94.9
Month 3-4	97.9
Month 5-6	96.4
Month 7-8	97.2
Month 9-10	96.9
Month 11-12	96.3

Based on the prediction of PM_{10}, the proportional contribution from each parameter to the PM_{10} content was also calculated. The result presented in Table 6 shows the average contribution (in percent) from each parameter to the PM_{10} content for each specific time period, and have been further used for calculating the different source contributions (see further Chapter 4.2.2).

[1] **Cross validation:** Parameters are estimated on one part of a data matrix (observations) and the suitability of the parameters tested in terms of its success in the prediction of the rest of the data matrix (observations)

[2] **Q^2 :** Performance of model prediction of PM_{10} levels, describes the fraction of the total variation of the different parameters that can be predicted by the model according to cross validation (max 1) (in this case Q^2 = performance)

Table 6 Average contribution (%) to the PM10 content for each variable and time period normalised to sum up to 100. Other variables, not included in this analysis, are also affecting the PM10 content.

Time period /Variable	Wood fuel burning	Energy index	Studded tyres	Traffic content	Meteorological index	Latitude
Month 1-2	18	19	18	21	18	5
Month 3-4	14	14	35	22	14	0
Month 5-6	19	21	0	31	26	3
Month 7-8	1	1	0	51	43	5
Month 9-10	15	27	0	27	27	3
Month 11-12	8	21	21	24	21	5

3.5 Population distribution

The current population data applied for exposure calculations in this study were supplied by Statistics Sweden (www.scb.se). The population data used in the exposure assessment was divided into age categories (0-15, 0-20, 30+, 50+, 15-65 years of age and all ages) based on 2019 census, and in total, 10 309 699 inhabitants were recorded. The population data used in the exposure assessment had a resolution of 1 x 1 km. Exposure of woman giving birth was estimated as the mean exposure of children aged 0-15 years, because Statistics Sweden did not publish the actual number of births per grid cell.

3.6 Exposure calculation

The numbers and distributions of people in different age categories exposed to different levels of NO_2, PM_{10} and $PM_{2.5}$ concentrations were calculated by over-laying the population grid to the air pollution grid. The population exposure to a specific pollutant was estimated for each grid cell and transformed into the total population exposure to each pollutant and the mean exposure level totally and for different age categories.

3.7 Health impact assessment

Health impact assessments (HIA) are built on epidemiological findings; exposure-response functions and population relevant rates. A typical health impact function has four components: an effect estimate from a particular epidemiological study, a baseline rate for the health effect, the affected number of persons and the estimated "exposure" (here pollutant concentration).

The excess number of cases per year may be calculated as:

$$\Delta y = (y_0 \cdot pop)\,(e^{ß \cdot \Delta x} - 1)$$

where y_0 is the baseline rate, pop is the affected number of persons; ß is the exposure-response function (natural logarithm of relative risk (RR) per change in concentration), and x is the estimated (excess) exposure (WHO, 2016b). For the health effects of long-term exposure to air pollution a lower safe threshold has generally not been shown in epidemiological studies, why we here assume that all exposure has an effect on the risk as given by the published risk function that is selected.

The effect of particles (usually measured as $PM_{2.5}$) on mortality has been the outstanding health impact from air pollution exposure in almost all health impact assessments regardless if they estimated the national or global burden. The association between fine particles and mortality continue to be the major health problem according also to the new WHO Air Quality Guidelines (WHO, 2021). During the more than 15 years that have passed between the last and new air quality guidelines, new studies have reported adverse effects at much lower levels than previously reported, which has resulted in more strict guidelines. Moreover, recent studies of mortality with rather low concentrations of $PM_{2.5}$ have reported higher relative risks per unit increase in concentration (steeper increase), especially for $PM_{2.5}$ from local sources (Segersson et al, 2021).

3.7.1 Exposure-response functions for mortality

3.7.1.1 Earlier exposure-response assumptions for mortality

At the time of our 2015 Swedish assessment (Gustafsson et al., 2018) the WHO report *Review of evidence on health aspects of air pollution*, REVIHAAP, (WHO, 2013a), had been used as a basis for the WHO Project *Health risks of air pollution in Europe*, HRAPIE, (WHO, 2013b).

From the WHO HRAPIE impact assessment report (WHO, 2013b) it was recommended to use the exposure-response function for long-term exposure to $PM_{2.5}$ and all cause (natural) mortality in ages 30+ from a literature review with a meta-analysis of 13 cohort studies (Hoek et al., 2013). The RR for $PM_{2.5}$ from this meta-analysis was 1.062 (95% CI 1.040-1.083) per 10 $\mu g/m^3$. However, since many years the research community has meant that it is likely that particles of different types have different effects on mortality and other health outcomes (WHO, 2007; WHO, 2013a). One example is ExternE (2005) with assumptions about the toxicity of different types of PM. ExternE treats nitrates as equivalent to half the toxicity of PM_{10}; sulfates as equivalent to PM_{10}; primary particles from power stations as equivalent to PM_{10}; primary particles from vehicles as equivalent to 1.5 times the toxicity of $PM_{2.5}$.

For elemental carbon (EC) the review by Hoek et al (2013) in a meta-analysis estimated a combined RR about 10 times higher than for $PM_{2.5}$, 1.061 per 1 $\mu g/m^3$ EC (95% CI 1.049-1.073). The conversion from exhaust particles to EC is complicated. Measurements performed 2013 by Stockholm City Environment Administration in the tunnel Söderledstunneln suggest that EC represents 30% of $PM_{2.5}$ from exhaust (Krecl et al, 2011). Other studies have indicated similar results, and confirm that the RR for background $PM_{2.5}$ becomes too low for PMexhaust. With the RR for EC (1.061 per 1 $\mu g/m^3$), the assumption that around 30% of PMexhaust is EC, the RR for PMexhaust would become 1.18 per 10 $\mu g/m^3$ if the rest of the particle mass had no effect, which however is unlikely. However, 1.18 per 10 $\mu g/m^3$ comes very close to a RR of 1.17 (95% CI = 1.05–1.30) per 10 $\mu g/m^3$ found when ACS CPS II study subjects from the Los Angeles basin were assigned exposure estimates at the zip code area level (Jerrett et al., 2005). Acknowledging the indications of a stronger effect on mortality of particles from sources within a city, we in our 2015 Swedish assessment for $PM_{2.5}$ in general adopted the exposure-response coefficient 1.062 per 10 $\mu g/m^3$ from HRAPIE (WHO, 2013b), but in addition applied the exposure-response coefficient 17% per 10 μg/m³ for $PM_{2.5}$ from local sources. An alternative would have been to assume 26% (95% CI 19–34) per 10 μg/m³ for near-source $PM_{2.5}$ from a much larger study also using cohort data from ACS CPS II (Turner et al., 2016). However, the estimate for NO_2 from the same multipollutant model was very small compared to Hoek et al (2013) and the recommendation from WHO HRAPIE (WHO, 2013b). Instead we estimated mortality effects associated vehicle exhaust using the exposure to NO_2 from local sources and epidemiological results from Denmark, with similar conditions as in Sweden, presented by Raaschou-Nielsen et al. (2012) with a RR of 1.08 per 10 μg/m³ (95% CI 1.01–1.14%) for all-cause mortality. Regarding long-term exposure to NO_2 and mortality the WHO HRAPIE report (WHO, 2013b) recommended a RR of 1.055 (95% CI 1.031-1.08). Because of the potential confounding and double counting of mortality effects from $PM_{2.5}$, the HRAPIE report stressed more uncertainty about quantification of NO_2 effects from single-pollutant models. The HRAPIE report also recommended to use the RR from Hoek et al only above the annual mean 20 μg/m³, a recommendation later seen as too conservative by the same group of experts (Heroux et al., 2015).

In our 2015 Swedish assessment (Gustafsson et al., 2018) road dust was assumed to only have a short-term effect on mortality, applying a RR estimated in a study of coarse PM, road dust and daily number of deaths in Stockholm (Meister et al., 2012).

3.7.1.1 Available exposure-response functions for mortality

Fine particles have different origin, composition and properties in different places. Particles may be primary or secondary, natural or anthropogenic, and consist more or less from minerals, soot, metals, salts etcetera. Thus, depending on the location, spatial resolution and the particle exposure variables, studies have a different potential to reflect the association between a specific type of exposure and mortality.

A systematic review of the evidence of associations between long-term exposure to particulate matter ($PM_{2.5}$ and PM_{10}) was recently published with the objective to support the new WHO Air Quality Guidelines (Chen & Hoek, 2020). For natural-cause mortality, the combined effect estimate across 25 studies was 1.08 (95% CI 1.06 - 1.09) per 10 μg/m³. The authors comment that "the large heterogeneity of effect estimates across studies suggests that health impact assessments in specific locations may have fairly large uncertainty". Notable, for the five studies with the lowest mean concentration, all below 10 μg/m³ and relevant for Sweden, the combined effect estimate was more than doubled, 1.17 per 10 μg/m³ (95% CI 1.12 - 1.23). This higher relative risk could perhaps be a result of less variation in exposure explained by the regional background in the underlying studies, and more variation in exposure related to local sources and within-city patterns.

A recent Swedish health impact assessment questions the use of the same exposure-response function for $PM_{2.5}$ and mortality regardless of the exposure level (Segersson et al, 2021). It is now well documented that the increase in risk per μg/m³ is bigger at low total $PM_{2.5}$ concentrations and for the local sources such as traffic, than for the regional background of $PM_{2.5}$. It has been shown, even within large cohort studies, that the scale of spatial variability in concentrations is important for the estimated increase in mortality per μg/m³ (Segersson et al, 2021). Turner et al (2016) in their multiple pollution model observed a more than six times higher relative risk (1.26 per 10 μg/m³) per absolute increase in concentration for near-source $PM_{2.5}$ in comparison with regional $PM_{2.5}$ (1.04 per 10 μg/m³). Lefler et al (2019) found similar patterns and concluded that regressions using spatially decomposed $PM_{2.5}$ suggest that more spatially variable components of $PM_{2.5}$ may be more toxic. An important review used meta-regression techniques to test whether study population or analytic characteristics modify the $PM_{2.5}$ -mortality association and to estimate the shape of the concentration-response curve (Vodonos et al, 2018). The authors found the $PM_{2.5}$ coefficient to decrease inversely proportional to the mean concentration. For all-cause all-age mortality, a 1 μg/m^3 increase in $PM_{2.5}$ was associated with a 1.29% increase in all-age all-cause mortality at a mean exposure of 10 μg/m^3, which decreased to 1.03% at a mean exposure of 15.7 μg/m^3 (the mean level across all studies). Restricted to studies with mean $PM_{2.5}$ concentrations below 10 μg/m^3, the increase was 2.4% (95% CI 0.8 - 4.0) increase per 1 μg/m^3.

A study of the Canadian community health survey cohort (Pinault et al, 2016) is interesting because of the low $PM_{2.5}$ concentrations with a mean = 6.3 μg/m^3 and the detailed data on life style. For non-accidental mortality, the relative risk was 1.26 (95% CI: 1.19 - 1.34) per 10 μg/m^3, and although the lowest measured concentration of $PM_{2.5}$ was 1 μg/m^3, the authors found no lower threshold for response.

A new important European study is the pooled analysis of eight cohorts in the multi-centre project Effects of Low-Level Air Pollution: A Study in Europe (ELAPSE). In this study 325 367 adults were followed-up for an average of 19.5 years, with 47 131 observed deaths. An increase of 5 μg/m^3 in $PM_{2.5}$ was associated with 13% (95% CI 10.6 - 15.5) increase in natural deaths (Strak et al, 2021). Associations tended to be steeper at low concentrations, levelling off at high concentrations. For participants with exposures below the US standard of 12 μg/m^3 an increase of 5 μg/m^3 in $PM_{2.5}$ was associated with 29.6% (95% CI 14% - 47.4%) increase in natural deaths.

A recent Swedish multi-cohort study within the Swedish Clean Air and Climate Research Program (SCAC) high-resolution dispersion models were used to estimate annual mean concentrations of PM_{10}, $PM_{2.5}$ and BC at individual addresses during each year of follow-up, 1990-2011 (Nilsson Sommar et al, 2021). Moving averages were calculated for the time windows 1-5 years (lag1-5) and 6-10 years (lag6-10) preceding the outcome. For $PM_{2.5}$ (range: 4.0 - 22.4 μg/m^3), the estimated increase in SCAC was 13% per 5 μg/m^3, as in a Danish cohort (Hvidfeldt et al, 2019) and the European multi-cohort study ELAPSE (Strak et al, 2021), but less precise (95% CI -9 - 40%). However, for cardiovascular mortality the increase in SCAC was bigger and statistically significant, 23% per 5 μg/m^3 $PM_{2.5}$ (95% CI 3 - 48) for lag1–5 years.

Since a large part of PM_{10} is $PM_{2.5}$, and the finer fraction generally is found to be more toxic, it is not possible to estimate the long-term effects on mortality of both $PM_{2.5}$ and PM_{10} as they were independent, even if some relative risks associated with PM_{10} exposure have been published.

A systematic review of the evidence of associations between long-term exposure to NO_2 and mortality was recently published with the objective to support the new WHO Air Quality Guidelines (Huangfu & Atkinson, 2020). For natural-cause mortality, the combined effect estimate across 24 studies was 1.02 (95% CI 1.01 - 1.04) per 10 μg/m^3. In this case the evidence was considered moderate, but this estimate from single pollutant models was used as basis for the guideline development (WHO, 2021). However, associations between NO_2 and mortality have been sensitive to adjustment for particles. Results from multiple pollution models report relative risks more in line with the lower end of the 95% CI from Huangfu & Atkinson (Turner et al, 2016; Stieb et al, 2021).

3.7.1.1.1 Assumed functions and baseline for mortality

Recent studies investigating the associations between the local variation in long-term levels of $PM_{2.5}$ and mortality in adult cohorts consistently find much higher relative risks than typical when all types of studies are considered. High meta estimates are also obtained when results from studies with low levels are combined, but this is likely due to the fact that these results are more than other studies influenced by local sources and contrasts within cities, and less by variation in the regional background concentrations. As the most relevant relative risk assumption for local sources, traffic and domestic wood burning, we see the original results reported by Turner et al. (2016), Pinault et al. (2016), Hvidfeldt et al. (2019), Strak et al. (2021) and Nilsson Sommar et al. (2021), which also were supported by the analyses by Vodonos et al. (2018) and Lefler et al. (2019). We here apply a relative risk of 1.26 per 10 μg/m^3.

For the effect on mortality from the regional background concentration of $PM_{2.5}$, mainly long-distance transported particles, and for the urban contribution of NO_2 we apply the overall relative risks assumed in the WHO Air Quality Guidelines (WHO, 2021): 1.08 and 1.02, respectively per 10 μg/m^3. Given the low regional background levels of NO_2 in Sweden in relation to levels in the epidemiological studies, and the before commonly assumed thresholds, no impacts are assumed from the regional background of NO_2.

The cited studies of long-term exposure and mortality all include only adults, typically 30 years or older when the follow up started. As in the previous national assessments we here estimate the effect of long-term exposure on persons 30 years or older. According to The National Board for Health and Welfare (Socialstyrelsen) the number of deaths in Sweden 2019 was 1338 per 100 000 persons according to the national Cause of Death Register (https://www.socialstyrelsen.se/statistik-och-data/statistik/statistikdatabasen/).

3.7.2 Exposure-response functions for morbidity

According to international scientists air pollution increases morbidity from a wide range of diseases (Thurston et al., 2017; Schaffer et al., 2019, Perera et al., 2019). According to WHO there is strong evidence for ischaemic heart disease, stroke, chronic obstructive pulmonary disease (COPD), lung cancer and acute respiratory infections, and a growing body of evidence suggesting causal relationships for type II diabetes, short gestation and low birth weight and neurological diseases (WHO, 2021). Besides the trend of a growing list of morbidity outcomes associated with air pollutants, especially $PM_{2.5}$, the strength of evidence is judged a bit different by different experts and organisations.

Since a large part of PM_{10} is $PM_{2.5}$, and the finer fraction generally is found to be more toxic, it is as for mortality not possible to estimate the long-term effects on morbidity of both $PM_{2.5}$ and PM_{10} as they were independent, even if some relative risks associated with PM_{10} exposure have been published.

3.7.2.1 Selected exposure-response functions and baselines for morbidity

For this health impact assessment we chose to include the same morbidity outcomes and relative risk functions for PM as were selected recently after a literature review initiated by the Swedish Transport Administration for health cost calculations related to local emissions from road traffic (Söderkvist et al., 2019; Forsberg et al., 2021), except for stroke where we apply the relative risk from a more recent European multi-cohort study (Wolf et al, 2021). These relative risks were recommended to apply when the impact of local $PM_{2.5}$ emissions are studied, which means usually upon an annual background mean of more than 5 µg/m³. Since many morbidity studies build on within-city variations in exposure and the lower side of the exposure range usually is above 5 µg/m³, it is not shown that these risk functions exist under the new WHO Air Quality Guideline for $PM_{2.5}$ of 5 µg/m³ as annual mean.

We have not judged it possible to also calculate corresponding health impacts related to NO_2 itself because of the correlation between the pollutants and lack of adjusted risk estimates. The selected risk estimates are listed in table 7 together with the corresponding baseline frequencies that are applied. National health register data were obtained from The National Board for Health and Welfare (Socialstyrelsen) and the frequency usually represents 2017 (https://www.socialstyrelsen.se/statistik-och-data/statistik/statistikdatabasen/). For preterm birth and childhood asthma the baseline rate is estimated from prevalence data in order to calculate a number of attributed cases per year.

Table 7 Applied relative risks and the corresponding baseline frequencies.

Outcome/considered at risk	RR from (source)*	RR per 10 µg/m³	% per 1 µg/m³	Freq/pers year	Freq from (source)*
Myocardial infarction/30+ yrs	Cesaroni G, 2014	1.12	1.1	0.00246	National health register
Stroke/30+ yrs	Wolf et al, 2021**	1.44	3.7	0.00253	National health register
Lung cancer/30+ yrs	Hvidtfelt U, 2021	1.28	2.5	0.00042	National health register
Dementia/50+ yrs	Yu X, 2020	1.17	1.6	0.00577	Van Bussel, 2017
Diabetes/15+ yrs	He D, 2017	1.25	2.3	0.00400	Norhammar, 2016
COPD/50+ yrs	Weichentahl S, 2017	1.20	1.8	0.00157	Lindberg, 2006
Childhood asthma/-18 yrs	Khreis H, 2017	1.34	3.0	*0.075 prev*	Oudin, 2017
Preterm birth	Klepac P, 2018	1.24	2.2	*0.058 prev*	National health register
Work loss days/15-64 yrs	Ostro B, 1987	1.05	0.5	11.9	National health register

** For details and references see Söderkvist et al. (2019) and Forsberg et al. (2021)*
*** <15 µg/m³*

The selection of risk functions takes into account how relevant and established the presented results are, where meta-estimates (from a literature review) and European multi-cohort results with relevant exposures were seen as a goal. European multi-cohort results are used for myocardial infarction (ESCAPE), stroke (ELAPSE) and lung cancer (ELAPSE). The relative risks applied for $PM_{2.5}$ and diabetes, childhood asthma, dementia and preterm birth are all from review papers with a calculated meta-estimate. It is only for COPD the applied risk function comes from one single large study that was conducted in Toronto. The selection of these risk functions is further discussed in reports from the Swedish Transport Administration (Söderkvist et al., 2019; Forsberg et al., 2021).

The literature on sick leave is very poor. Six consecutive years (1976–1981) of the US Health Interview Study (HIS) were used to study restricted activity days (RADs) in adults aged 18–64 (Ostro, 1987; Ostro and Rothschild, 1989). In the multi-stage probability sample of 50,000 households from metropolitan areas of all sizes and regions severity was classified as (i) bed disability days; (ii) work or school loss days and (iii) minor restricted activity days (MRADs), which do not involve work loss or bed disability but do include some noticeable limitation on 'normal' activity. The weighted mean pollutant coefficient for RADs was linked to estimated background rates of, on average, 19 RADs per person per year. From this study came an exposure-response function of 902 RADs (95% CI 792, 1013) per 10 μg/m³ $PM_{2.5}$ per 1,000 adults at age 15–64, or 0.092 RADs for a change of 1 μg/m³*person and year. In this age group we may see this as work loss days. In HRAPIE (WHO, 2013b) this RR is expressed as 1.046 per 10 μg/m³ $PM_{2.5}$, giving almost the same number of RADs for a change of 1 μg/m³*person and year.

3.7.3 Health impact calculations

Our health impact assessment is organized to make it possible to add the impacts and still avoid double counting.

Impact of regional background $PM_{2.5}$ on mortality

We estimate the regional background $PM_{2.5}$ impact on all-cause mortality in ages 30+. In lines with epidemiological results we assume a linear risk function without any cut off, the same effect in all ages and for all sources.

Impact of local source $PM_{2.5}$ on mortality

We estimate the local (urban) source $PM_{2.5}$ impact on all-cause mortality in ages 30+. We study the exposure originating from local emissions of vehicle exhaust, traffic wear particles (road dust) and domestic wood burning. We assume a linear risk function without any cut off, the same effect in all ages and for these local sources.

Impact of local source NO_2 on mortality

We estimate an impact of NO_2 itself on all-cause mortality in ages 30+ only from the local (urban) contribution. Since the smooth risk slope is steeper at low concentrations, and we consider only the local contribution added upon the regional background and thus we use no cutoff.

Impact of regional and local PM on morbidity

We estimate the impact only of $PM_{2.5}$ on morbidity, applying the same risk functions regardless of type of particle source. For diseases and preterm birth we assume a cut off of 5 μg/m³, which means that more than 75% of the estimated population exposure from local traffic and domestic wood burning occur above such a background, whereas only just over one fourth of the estimated total population exposure to regional background $PM_{2.5}$ occur over this threshold.

For restricted activity days the exposure-response relation we previously have applied is based on old cross-sectional and self-reported results from USA. This is a weak study design, but the results have been widely used, e.g. by WHO in HRAPIE (WHO, 2013a). Even if our previous calculations built on the total exposure range must be interpreted with caution, we in this assessment repeat the same type of estimation.

3.8 Socio-economic valuation

In brief, socio-economic valuation of health impacts from air pollution should include all welfare parameters of relevance for health effects related to air pollution. The valuation allows for consideration of all economic decision makers in society; individuals (households), firms and government, and should include direct and indirect use costs as well as non-use (intangible) costs of poor air quality (Figure 9). Ideally, all these cost parameters should be taken under consideration during the valuation of health impacts, but it is sometimes difficult to measure and calculate reliable estimates of them. It can also be that some methods of valuation aggregate the parameters, thereby making it difficult to distinguish between them.

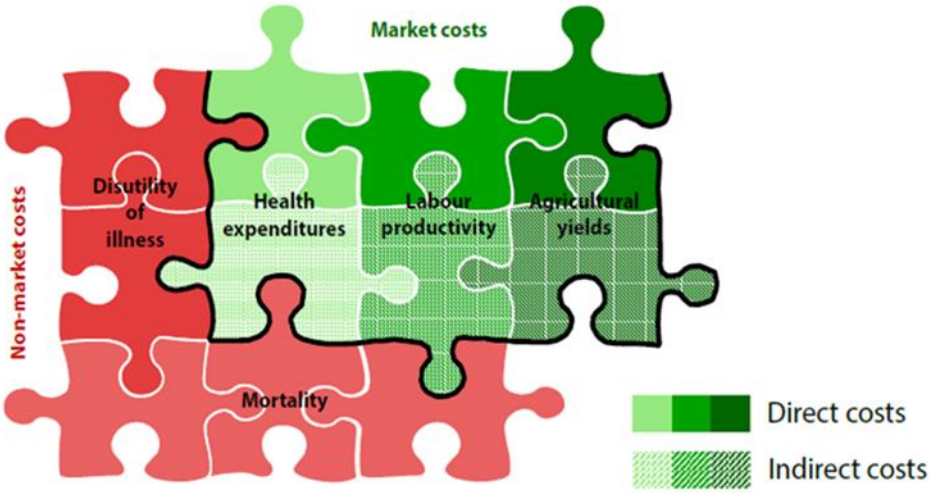


Figure 9 **Market and non-market costs of air pollution damage, split into their main categories. Figure copied from OECD, 2016.**

The method and data used for socio-economic valuations of mortality and morbidity are based on the latest published Swedish assessment made within the Swedish Road Administrations' ASEK work and newer publications. For cancer, data was not available, so we search in google and google scholar (search term: "Economic cost of lung cancer per incidence") and filter the results with respect to European and North American studies published after 2002. All identified values are converted to Swedish Krona (SEK) using purchase power parities from OECD[3] for the study year and then inflated with respect to changes in all-goods consumer price indices (CPI) between the study year and 2019. Data on CPI was also taken from OECD[4].

The latest Swedish Road Administration values of statistical life are based on a 2016 study on willingness to pay to reduce deadly outcomes of road traffic accidents and gives at hand a value of 40.5 million SEK_{2014} per statistical life and a range of 29.8-58.6 million SEK_{2014} (Olofsson et al., 2016, Swedish Road Administration, 2018). Nerhagen et al. (2015) further specifies that with a standard 3.5% discount rate, this value of statistical life corresponds to a value of a life year lost of 1.9

[3] https://data.oecd.org/conversion/purchasing-power-parities-ppp.htm?fbclid=IwAR26duaP4UjuXHHgGg8VJUITnrqeAai_xdJlMyDJ1r4qG2W3EJ-5z8g0fec
[4] https://data.oecd.org/price/inflation-cpi.htm

million SEK_{2014} (1.4 – 2.7 million SEK_{2014}). Inflated with respect to CPI, the central VOLY estimate becomes 2.2 million SEK_{2019}.

For the morbidity outcomes, values were taken from ASEK support material and newer valuation studies. The present value of one case of myocardial infarction in Sweden, including effects on quality of life, is estimated to be 24 000 $€_{2016}$ per case (16 000 – 38 000) in a recently submitted manuscript by Kriit, Sommar & Åström. The central estimate corresponds to 323 000 SEK_{2019}. The same study also presents values for one case of stroke (central estimate 460 000 $€_{2016}$, 6.2 million SEK_{2019}) and one case of preterm birth (central estimate 34 000 $€_{2016}$, 457 000 SEK_{2019}).

Not all morbidity values in the literature includes the values from changes in quality of life. We therefore adjusted some values. In this adjustment, we used a quality-of-life value of 106 000 $€_{2016}$ per life year (53 000 – 250 000 $€_{2016}$) as reported in Kriit, Sommar & Åström (submitted) and complementary information on changes in quality of life from the ASEK support material (Söderquist et al 2019). The changes in quality of life from diabetes, COPD, and childhood asthma is expected to last 9, 19, and 17 years respectively. And in total over these years, the quality-adjusted life years affected are 1.55, 5.51, and 1.75 years respectively (Söderquist et al., 2019). Table 8 shows the present value of quality-of-life effects for the outcomes, calculated with a 3.5% discount rate

Table 8 **Quality-adjusted life years and present value socio-economic costs calculated with a 3.5% discount rate for diabetes, COPD, and childhood asthma.**

	Duration [years]	Total loss in QALY	QALY/year	Present value [SEK_{2017}]
Diabetes	9	1.55	0.17	1 797 000 (899 000 – 4 239 000)
COPD	19	5.51	0.29	3 026 000 (1 513 000 – 7 138 000)
Childhood asthma	17	1.75	0.10	1 074 000 (537 000 – 2 534 000)

Together with the cost-of-illness-related costs of diabetes, COPD, and childhood asthma from Söderquist et al., 2019, the total socio-economic costs including QALY aspects are 2 366 000 SEK_{2019} (1 433 000 – 4 899 000), 3 251 000 SEK_{2019} (1 681 000 – 7 518 000), and 1 257 000 SEK_{2019} (699 000 – 2 771 000) respectively for diabetes, COPD, and childhood asthma. Costs of dementia are taken from Kriit et al., 2021 and corresponds to 9 130 000 SEK_{2019} (8 535 000 – 9 659 000) per case.

The effect with readily available socio-economic costs is work loss, which is valued based on average daily salary Sweden with salaries for low- and high education as boundaries. To this we also add inconvenience valued at 550 SEK_{2019} per day (Söderquist et al., 2019). In total, the socio-economic costs of one day of work loss due to air pollution is estimated to 2 380 SEK_{2019} (1 910 – 3 450).

3.8.1 Socio-economic costs of lung cancer

In the Swedish literature we can't find any studies clarifying the per case costs of lung cancer from air pollution. We therefore made a quick overview of European and North American publications and included results published after 2001. The search results are scarce and only three studies are suitable as references. Kutikova et al. (2005) presents results for the United States. Inter alia, over the two-year study period the costs for patients receiving only initial treatment was US$$_{2000}$ 45 953 while the costs for the control group was US$$_{2000}$ 8 136. Demeter et al. (2007) presents health care costs for nonsmall cell lung cancer and small cell lung cancer to be CAN$$_{1998}$ 10 928 and CAN$$_{1998}$ 15 350 respectively for incidences reported in 1998. Of these costs, 76% were associated with patient admissions and therapy. For stage I nonsmall cell lung cancer, the costs were CAN$$_{1998}$ 13 380 (1 207 – 25 553). Finally, Cicin et al. (2021) report inpatient and outpatient costs as well as indirect costs for 50 000 lung cancer patients in 2018. For nonsmall cell lung cancer the direct costs are €$_{2018}$ 10 167 per year, and for 50 000 patients the total indirect costs is €$_{2018}$ 1 billion per year out of which €$_{2018}$ 0.5 billion is driven by work losses due to premature fatality.

For the cost assessment we make the following assumptions:

- lung cancer caused by air pollution is less severe (Stage I) and,
- of a nonsmall cell lung cancer type, and
- costs over two years are considered,
- costs only for successful initial treatment are considered,
- The cost range in Kutikova et al. (2005) and Cicin et al. (2021) is identical to the reported range in Demeter et al (2007),
- The reported exchange rate between Turkisk Lira in and euro in Cicin et al. (2021) is erroneously reported (they report 5.555, €/TL but the actual exchange rate in 2019 was ~5.5 TL/€).

Given these assumptions the health care costs in Kutikova et al. (2005) correspond to 444 164 SEK$_{2019}$ per case (40 068 - 848 260), Demeter et al. (2007): SEK$_{2019}$ 137 819 per case (12 433 - 263 205), and Cicin et al. (2021) which includes indirect costs: SEK$_{2019}$ 1 306 600 (117 867 - 2 495 332). For our calculations we use the average of these studies: SEK$_{2019}$ 629 527 per case (56 789 - 1 202 266).

The quality-of-life score associated with lung cancer has been reported by Pickard et al. (2007). Following the Eastern Cancer Oncology Group (ECOG) grade ranges, lung cancer patients with the lowest ECOG range (0) had EQ-5D index-based utility scores of 0.78 (SD ±0.15) and 0.83 (SD ±0.11) for UK and US patients respectively. The EQ-5D mean utility score for a healthy person has been reported by Burström et al. (2014) as 0.97, from which we subtract the average UK and US QALY value. As for the above mentioned morbidity effects, we assume the value of a quality-adjusted-life-year to be €$_{2016}$ 106 000 (53 000 - 250 000). The resulting direct health care costs, indirect costs and quality-of-life losses is valued to SEK$_{2019}$ 666 478 (60 708 - 1 358 077).

4 Results

4.1 Calculation of air pollutant concentrations

4.1.1 National distribution of NO_2 concentrations

The annual mean concentration of NO_2 for 2019, calculated with the URBAN model, is presented in Figure 10. The result is based on calculated bimonthly means in order to capture the seasonal variation, where higher concentrations usually occur during winter.

As presented in Figure 10, calculations indicate annual mean background NO_2 concentrations for 2019 below 5 µg/m^3 in all rural areas. Urban background concentrations in small to medium sized cities reached NO_2 concentrations of up to 15 µg/m^3, and concentrations exceeds 20 µg/m^3 only in the central parts of the three largest cities in Sweden; Stockholm, Gothenburg and Malmö. The calculated NO_2 concentrations were thus below the environmental standard for the maximum annual mean value (40 µg/m^3). The long-term environmental objective of concentrations below 20 µg/m^3 as an annual mean for the whole country was, however, exceeded in the larger urban areas.

Based on the calculated results, no 1 x 1 km grid cell exceeded the annual air quality standard for NO_2 concentrations for 2019. However, the standards are also valid for road-side concentrations in street canyons. A study by Persson and Haeger-Eugensson (2006) showed that road-side concentrations in Swedish cities were generally around 1.5 times higher than the urban background, although in poorly ventilated urban streets with dense traffic, much higher concentrations could be found. Thus, there are likely additional exceedances of the air quality standard at road-side locations, which is not considered in this study. This is also reflected in Air Quality Plans submitted by municipalities where violation of the limit values remains at some locations.

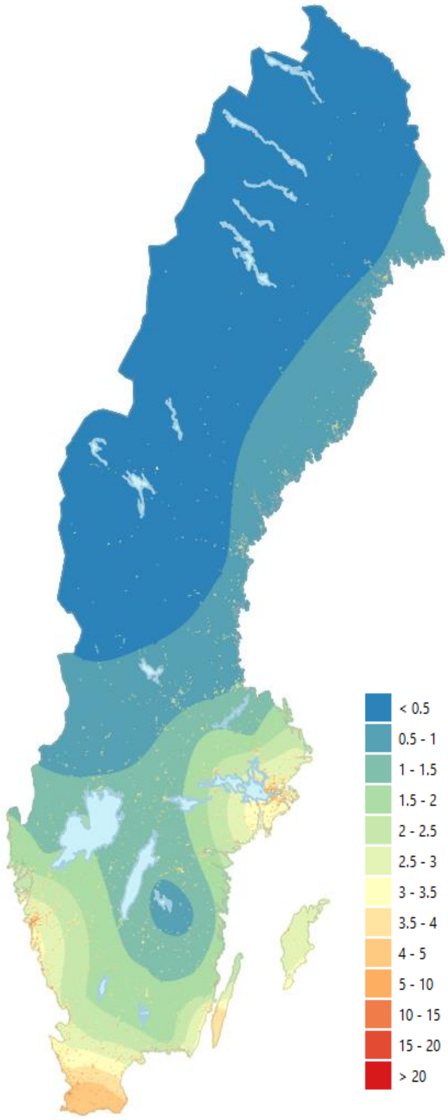


Figure 10 **NO_2 concentrations, as annual mean, for 2019 in Sweden, unit µg/m^3.**

4.1.2 National distribution of PM_{10} concentrations

The annual mean concentrations of PM_{10} for 2019, calculated with the URBAN model, are presented in Figure 11. The result is based on calculated bi-monthly means in order to capture the seasonal variations, where higher concentrations of PM_{10} usually appear during late winter-spring depending on the location in the country.

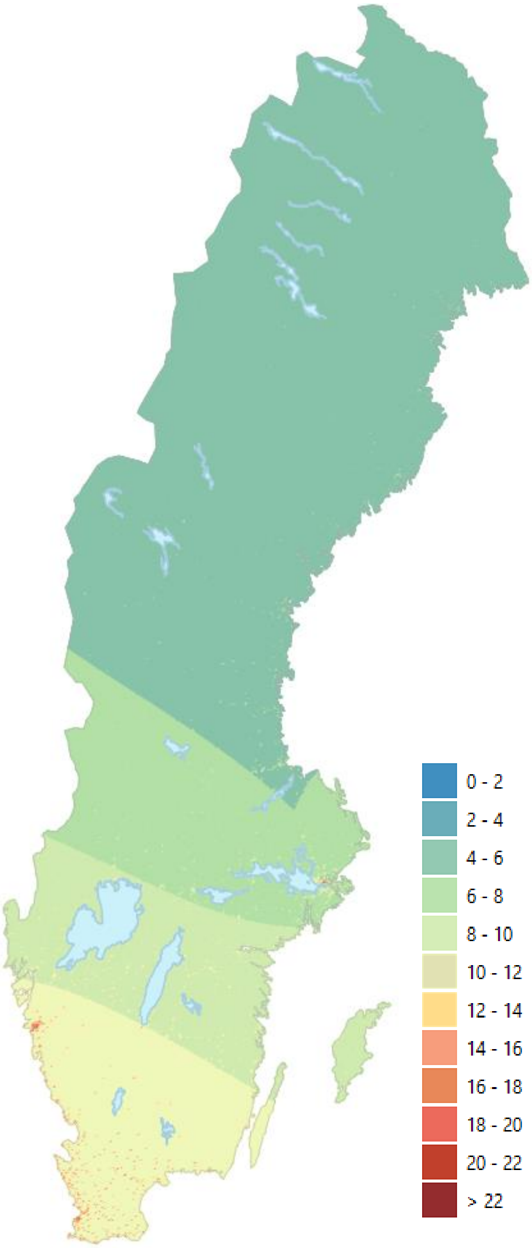


Figure 11 PM_{10} concentrations, as annual mean, for 2019 in Sweden, unit μg/m³.

As shown in Figure 11, the PM_{10} concentrations as yearly mean are primarily governed by the regional background concentrations. Compared to the 2015 assessment, the calculated background PM concentrations are considerably lower in coastal areas in 2019, particularly along the west coast. This change is attributed to the method change in this assessment, where regional background PM concentrations are now based only on data from measurements stations. The previous inclusion of background concentrations calculated with the EMEP model, generated considerably higher PM concentrations in coastal areas. The implications of this method change for trend assessments will be discussed in chapter 5.

Due to the strong influence from the long-range transport originating from continental Europe, there is a considerable latitudinal decrease to the north in the regional background concentrations. The calculated urban background PM_{10} concentrations in small and medium sized cities were around 12 in the south and below 6 $\mu g/m^3$ in the north. Concentrations over 15 $\mu g/m^3$ were only found in the central parts of the medium and larger cities.

Concentrations were thus well below the environmental standard for the annual mean value (40 $\mu g/m^3$) in both urban and rural background air in Swedish towns on the 1 x 1 km resolution in 2019. The long-term environmental objective of PM_{10} annual mean concentrations below 15 $\mu g/m^3$ was only exceeded in medium and larger urban areas in 2019. This result differs from previous years, when the environmental objective was exceeded in rural areas along the west coast. This change is at least partially connected to a change in methods as described in section 3.2, and its impact on the assessment will be discussed further in section 6.

4.1.3 National distribution of $PM_{2.5}$ concentrations

The annual mean concentrations of $PM_{2.5}$ for 2019 are presented in Figure 12. The result is based on the earlier calculated PM_{10} concentrations and the empirical $PM_{10}/PM_{2.5}$ ratio.

Calculated concentrations were well below the environmental standard for the annual mean value (25 $\mu g/m^3$) in both urban and rural background air in Swedish towns on the 1 x 1 km resolution in 2019. The long-term environmental objective of $PM_{2.5}$ annual mean concentrations below 10 $\mu g/m^3$ was only exceeded in the central parts of Malmö, Gothenburg and Stockholm.

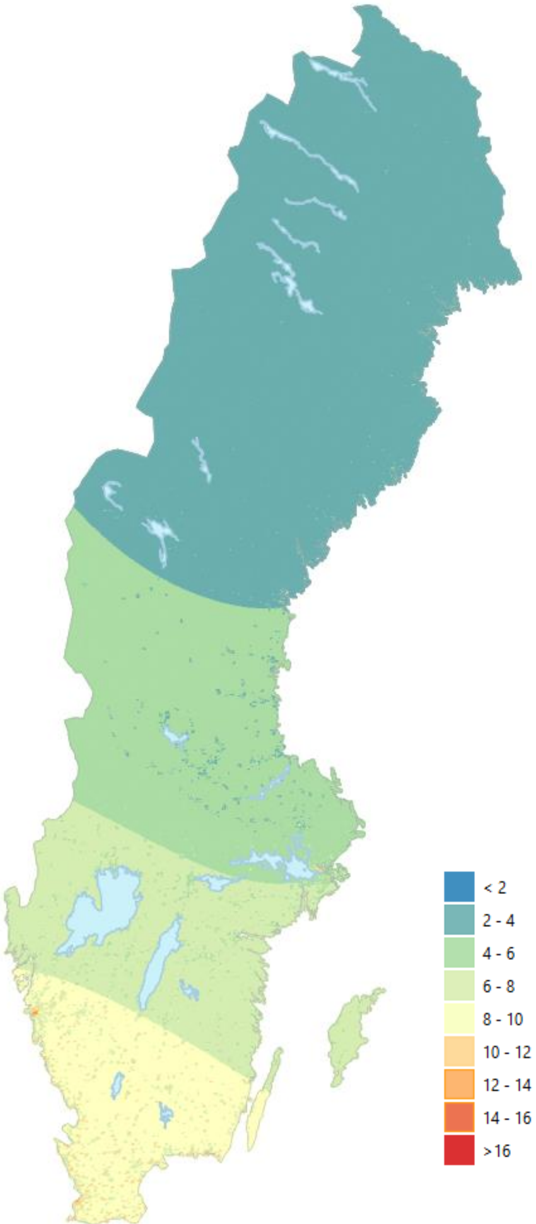


Figure 12 **$PM_{2.5}$ concentrations, as annual mean for 2019, in Sweden, unit μg/m^3.**

4.2 Population exposure

The population exposure to different NO_2 and particle concentrations has been calculated based on the calculated air concentrations.

4.2.1 Exposure to NO_2

Studies providing dose-response relationship for calculations of health impact from air pollution exposure are almost exclusively based on urban background air pollutant concentrations. In order to allow application of known relationships, this study is therefore based on urban background concentrations. As previously mentioned, higher NO_2 concentration will normally be found in roadside locations compared to urban background, due to emissions from, for example, traffic within street canyons. Consequently, a slightly higher exposure would likely have been found if roadside concentrations were used instead of background in the exposure calculations. However, very few dose-response functions are based on roadside concentrations and exposure studies such as this one can therefore not rely on roadside concentrations.

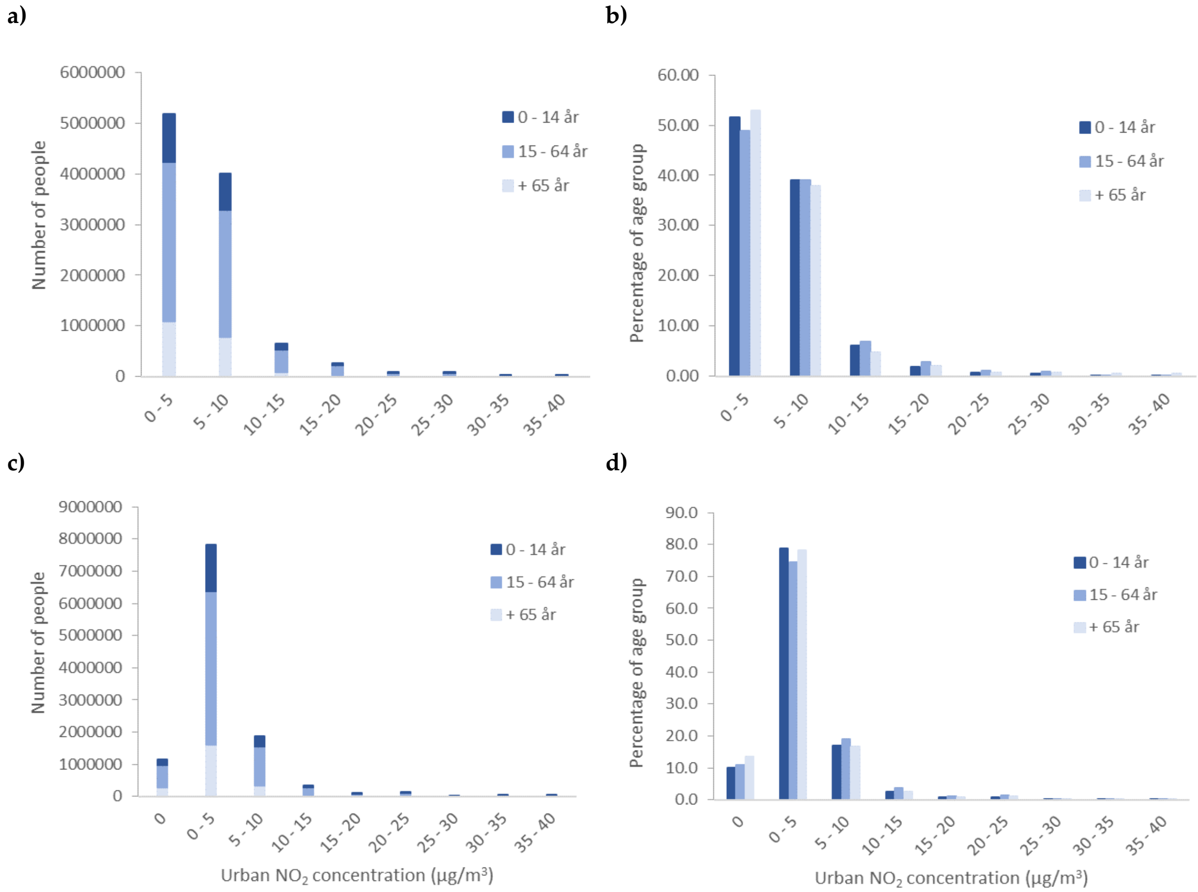


Figure 13 **Population exposure to total NO_2 annual mean concentrations in Sweden in 2019 expressed in a) number of inhabitants and b) percentage of population, divided into the age categories 0 – 14 (dark blue), 15-64 (blue), and 65+ years of age (light blue). Population exposure to mean NO_2 contribution from local urban sources in Sweden in 2019 expressed in c) number of inhabitants and d) percentage of population, divided into the same age categories.**

The population exposure to NO_2 annual mean concentrations in Sweden in 2015 is shown in Table 9 and Figure 20. In 2019, the annual mean population weighted exposure to NO_2 was 5.9 µg/m³, of which the urban contribution was 3.6 µg/m³. The largest group in all age classes, around 50 %, was exposed to annual mean concentrations of NO_2 below 5 µg/m³ (Figure 13). Approximately 40% were exposed to NO_2 concentration levels between 5-10 µg/m³, and less than 5% to levels of NO_2 above 15 µg/m³. The population exposed to NO_2 from local urban sources are presented in Figure 13c and 13d. According to these calculations 12% of the Swedish population lives in areas without any urban NO_2 contribution.

Our calculations also show that compared to the population as a whole, children and elderly (age categories 0-14 and 65 +) were slightly overrepresented in the lower exposure concentration categories, and slightly underrepresented in the higher, with the opposite pattern in the age category 15-64 years of age.

4.2.2 Exposure to PM_{10} and $PM_{2.5}$

As for NO_2, exposure to PM_{10} and $PM_{2.5}$ are based on calculations of urban and regional background concentrations to allow application of known dose-response functions for health effects. Higher particle concentrations, especially PM_{10}, and consequently higher exposure, would likely have been found if roadside concentrations were used instead of background in the exposure calculations. However, as very few dose-response functions are based on roadside concentrations exposure studies such as this one can therefore not rely on roadside concentrations.

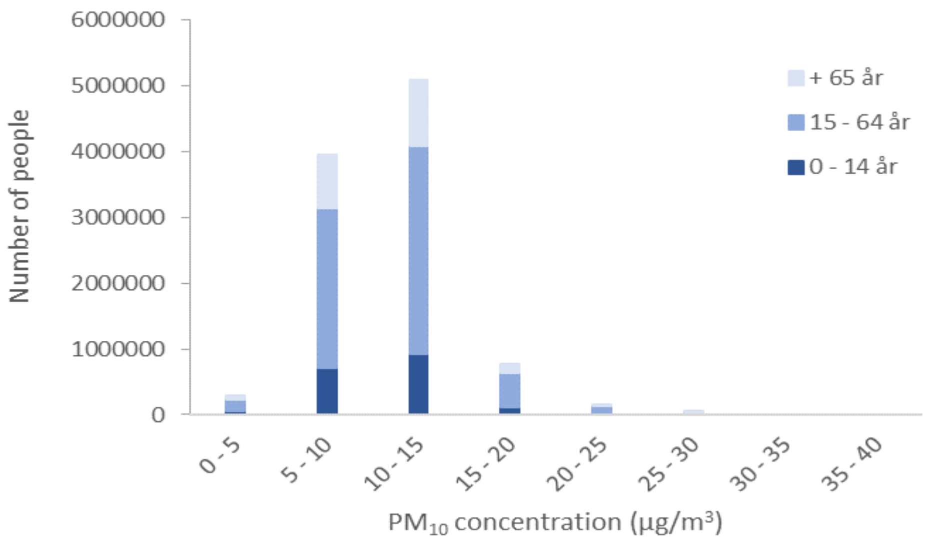


Figure 14 **Number of inhabitants exposed to total PM_{10} annual mean concentrations in Sweden in 2019, divided into the age categories 0 – 14 (dark blue), 15-64 (blue), and 65+ years of age (light blue).**

The exposure distribution of the Swedish population to annual mean PM_{10} concentrations in 2019 is shown in Figure 14. Less than 3% of the population was exposed to concentrations below 5 µg/m³, with a minimum around 3.5 µg/m³. Approximately 88 % of the population was exposed to PM_{10} concentrations between 5 and 15 µg/m³. That leaves 18 % of Swedish inhabitants exposed to PM_{10}

levels higher than the environmental objective for PM_{10} (15 µg/m³). However, no one is exposed to concentrations exceeding the environmental air quality standard of 40 µg/m³. As for NO_2, children and elderly (age categories 0-14 and 65 +) were slightly overrepresented in the lower exposure concentration categories, and slightly underrepresented in the higher, with the opposite pattern in the age category 15-64 years of age.

The estimated exposure to the total annual mean concentrations of $PM_{2.5}$ is shown in Figure 15. The majority of the population, almost 90%, was exposed to $PM_{2.5}$ annual mean concentrations below the environmental objective (10 µg/m³), with a minimum of 3 µg/m³. Approximately 10% of the people in Sweden were exposed to levels between 10 and 20 µg/m³. No one is exposed to concentrations exceeding the environmental air quality standard of 25 µg/m³.

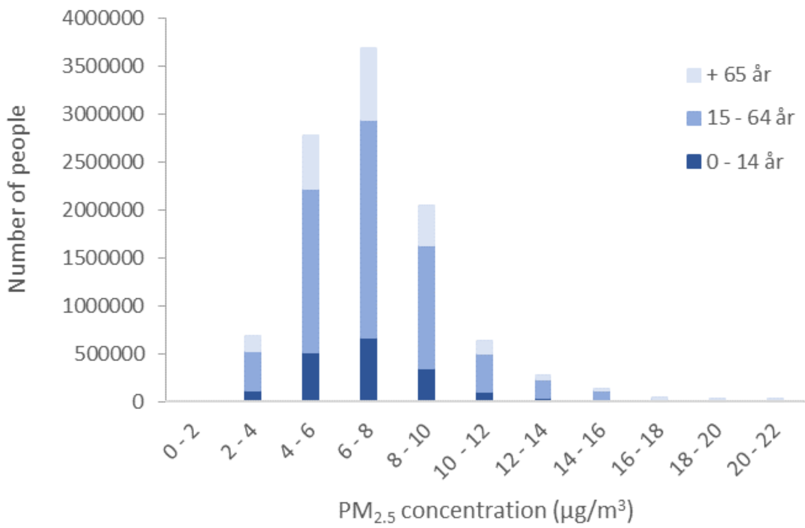


Figure 15 **Number of inhabitants exposed to total $PM_{2.5}$ annual mean concentrations in Sweden in 2019, divided into the age categories 0 – 14 (dark blue), 15-64 (blue), and 65+ years of age (light blue).**

As described earlier the particle contribution from different sources (road dust, traffic exhaust, wood burning and regional background) to the particle levels was calculated. The number of people exposed to different PM_{10} concentrations from road dust is presented in Figure 16. Particles from traffic exhaust, wood burning, and long-range transport were assumed to all belong to the $PM_{2.5}$ fraction and are presented in Figure 17 - 19.

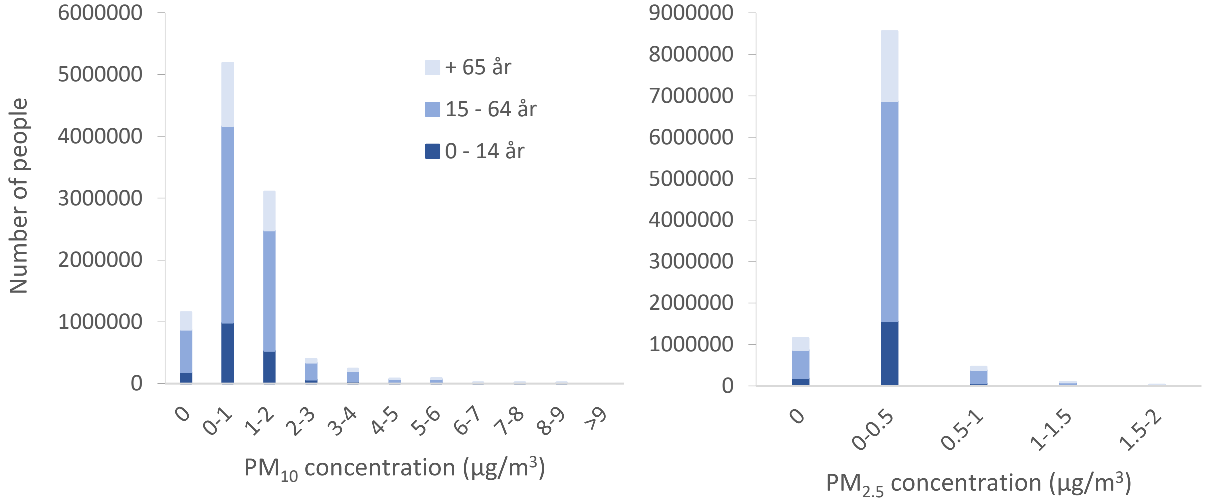


Figure 16 **Number of inhabitants exposed to PM_{10} (left) and $PM_{2.5}$ (right) annual mean concentrations from *road dust* in Sweden in 2019, divided into the age categories 0 – 14 (dark blue), 15-64 (blue), and 65+ years of age (light blue).**

Road dust contributed on average 0.9 µg/m³ to the annual mean population weighted exposure of PM_{10}, and 0.2 µg/m³ to the annual mean population weighted exposure of $PM_{2.5}$ in 2019. Approximately 98% of the population were exposed to less than 1 µg/m³ $PM_{2.5}$ from *road dust* (Figure 16). In the previous assessments, health impacts connected to *road dust* were based on PM_{10}. In the 2019 exposure assessment, health impacts connected to *road dust* are instead based on $PM_{2.5}$. The reason for this change is that $PM_{2.5}$ is now used in ASEK, which is built on the national reporting (Söderkvist et al, 2019) and health impact assessments of traffic emissions (Segersson et al, 2017; Segersson et al, 2021).

According to the calculations the contribution from traffic exhaust to the total $PM_{2.5}$ concentration was 0.1 µg/m³ (Figure 17). 98% of the population was exposed to less than 0.5 µg/m³ of $PM_{2.5}$ from *traffic exhaust*. As this does seem unrealistically low, the NO_2 concentration may be a better indicator of traffic exhaust pollution.

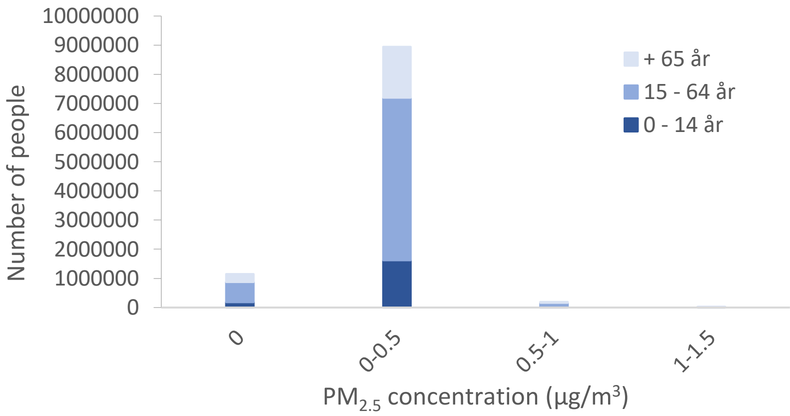


Figure 17 **Number of inhabitants exposed to $PM_{2.5}$ annual mean concentrations from *traffic exhaust* in Sweden in 2015, divided into the age categories 0 – 14 (dark blue), 15-64 (blue), and 65+ years of age (light blue).**

According to the calculations the contribution from *wood burning* to the total $PM_{2.5}$ concentration was 0.3 µg/m^3, and approximately 85 % of the population were exposed to less than 0.5 µg/m^3 $PM_{2.5}$ from *wood burning* (Figure 18). This is a considerable reduction compared to the 2015 exposure assessment, where the contribution from *wood burning* was 0.8 µg/m^3. This reduction was found to originate in a small error in the calculations in the 2015 assessment, which was corrected in this study. The error only affected the contribution from *wood burning* to the total $PM_{2.5}$ concentrations, not the overall exposure situation. The possible implications of this error for assessment of trends will be discussed in chapter 5.

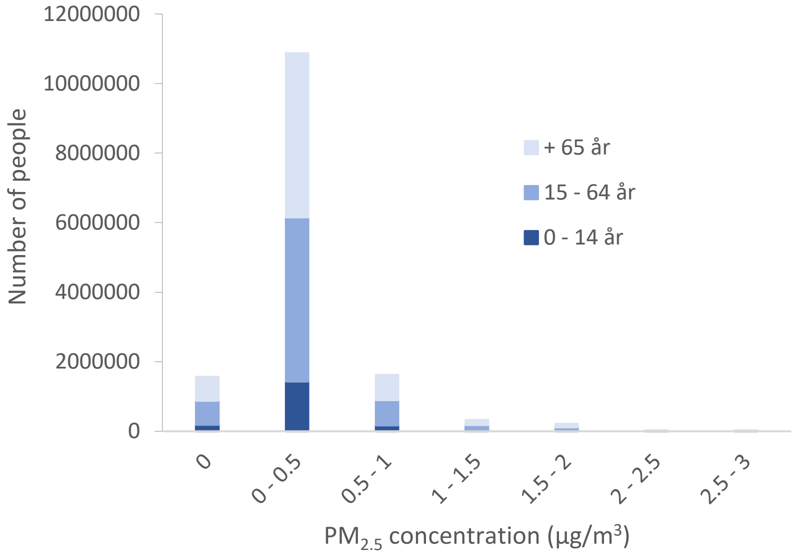


Figure 18 **Distribution of exposure levels to $PM_{2.5}$ annual mean concentrations from *wood burning* in the Swedish population in 2019, divided into the age categories 0 – 14 (dark blue), 15-64 (blue), and 65+ years of age (light blue).**

In order to assess the exposure and health effects from long distance transported particles, these were assumed to be represented by the *regional background* $PM_{2.5}$ concentrations. This was the category that contributed the most to the total $PM_{2.5}$ concentration with an average of 6.7 µg/m³ in 2019 (Figure 19).

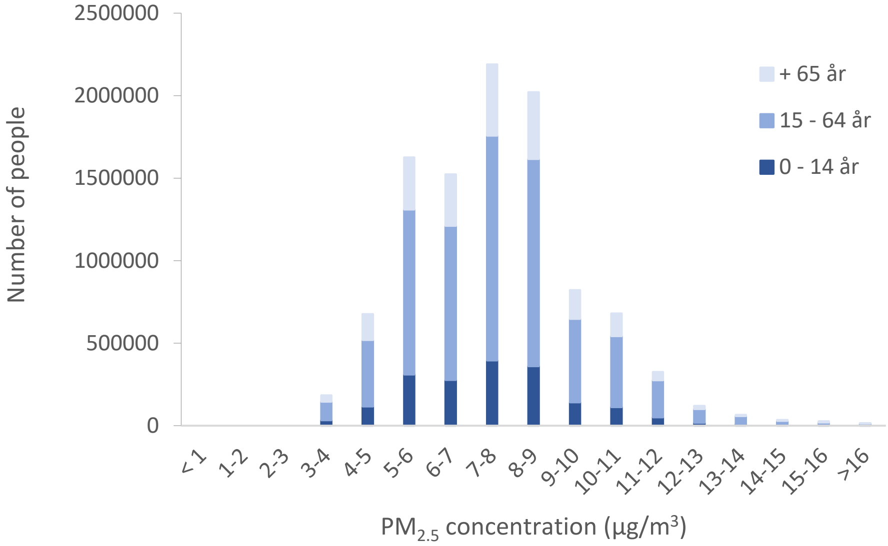


Figure 19 **Distribution of exposure levels to annual mean *regional background* $PM_{2.5}$ concentrations in the Swedish population in 2019, divided into the age categories 0 – 14 (dark blue), 15-64 (blue), and 65+ years of age (light blue).**

4.3 Trends in population exposure

In Table 8 the population exposure to NO_2 and particles in ambient air calculated for the years 2005, 2010, 2015 and 2019 respectively are summarized. Percentage of the population exposed above the environmental objective as well as the environmental standards is specified for the different years. In addition, exposure above the recently presented WHO air quality guidelines are presented for the 2019 evaluation.

Table 9 **Calculated population exposure to NO_2 and particles in ambient air in 2005, 2010, 2015 and 2019 respectively.**

		2005	2010	2015	2019
Total population (no. of inhabitants)		8 899 724	9 546 546	9 839 105	10 309 699
Mean population weighted exposure (µg/m³)	NO_2	6.3	6.2	6.4	5.9
	PM_{10}	13.0	12.0	12.5	10.9
	$PM_{2.5}$	9.8	8.6	8.3	7.2
Percentage of population exposed to concentrations above the environmental objective	NO_2 (20 µg/m³)	2.3%	2.7%	2.9%	2.1%
	PM_{10} (15 µg/m³)	38%	25%	22%	10 %
	$PM_{2.5}$ (10 µg/m³)	49%	28%	23%	11 %
Percentage of population exposed to concentrations above the environmental quality standard	NO_2 (40 µg/m³)	0%	0%	0%	0 %
	PM_{10} (40 µg/m³)	0.4%	0.3%	0.3%	0 %
	$PM_{2.5}$ (25 µg/m³)	0%	0.6%	0.6%	0 %
Percentage of population exposed to concentrations above the WHO 2021 air quality guidelines	NO_2 (10 µg/m³)	-	-	-	11 %
	PM_{10} (15 µg/m³)	-	-	-	10 %
	$PM_{2.5}$ (5 µg/m³)	-	-	-	82 %

The calculated exposure indicates that the percentage of the population exposed to concentrations above the environmental quality standards is now zero, and that exposure above the environmental objective has decreased to 2.1 %, 10 % and 11 % for NO_2, PM_{10} and $PM_{2.5}$ respectively. In the recently updated WHO 2021 Air Quality Guidelines, the recommended maximum exposure is considerably lowered compared to the environmental objective for both NO_2 and $PM_{2.5}$. Our calculations indicate that 82 % of the Swedish population is exposed to concentrations exceeding the WHO guidelines for $PM_{2.5}$, and 11 % to concentrations exceeding the WHO guidelines for NO_2.

4.3.1 NO_2

Figure 20 illustrates the percentage of the population exposed to NO_2, divided into concentration classes of 5 µg/m³, in the seven studied years. A trend towards an increasing part of the population exposed to lower concentration levels can be observed, as the proportion of the population exposed to NO_2 concentrations in the two lowest categories gradually increase while exposure in the higher categories decrease.

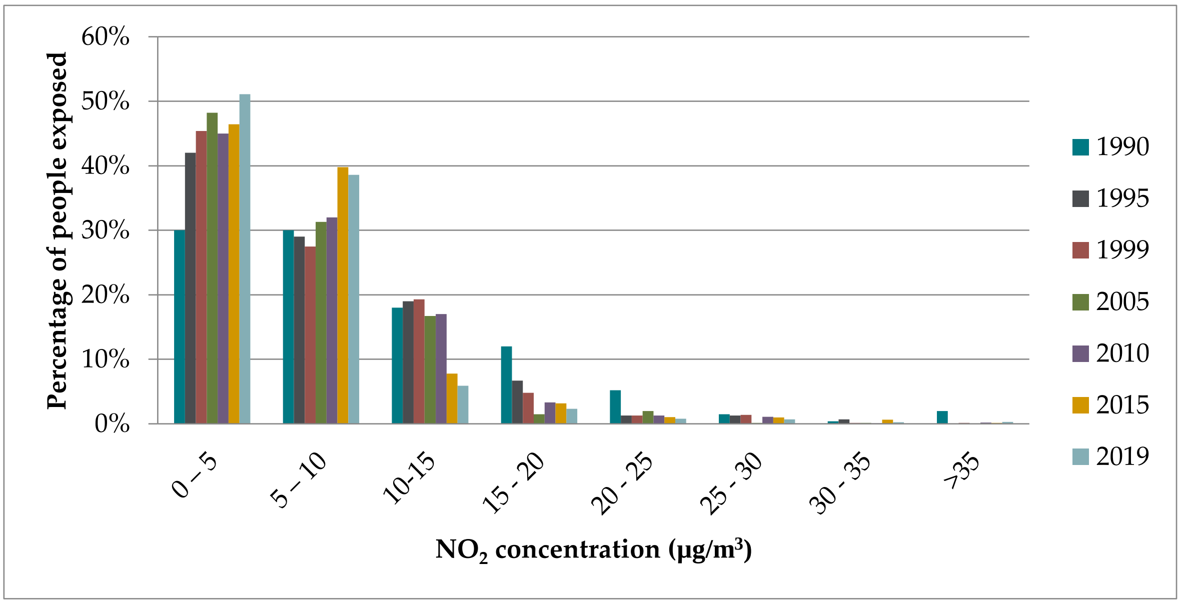


Figure 20 Percentage of the population exposed to NO_2 (μg/m³) annual mean concentrations in 1990, 1995, 1999, 2005, 2010, 2015, and 2019.

4.3.2 Particles

Exposure to PM_{10} shows similar trends as NO_2, with increasing exposure in the lower concentration categories, and decreasing exposure in the higher categories (Figure 21).

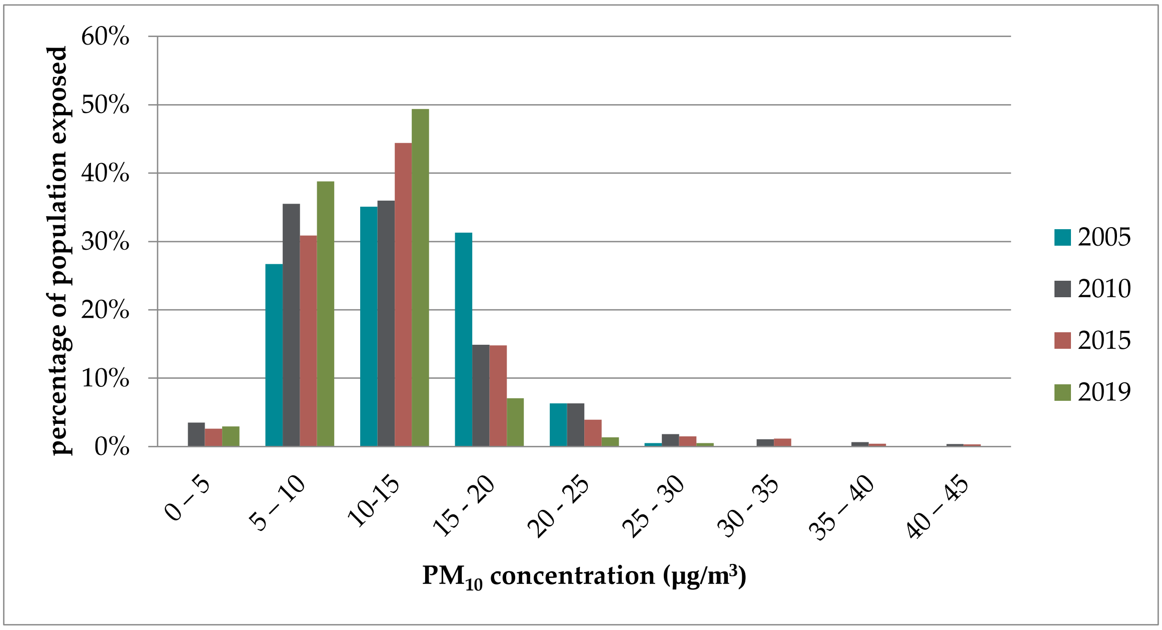


Figure 21 Percentage of the population exposed to PM_{10} (μg/m³) annual mean concentrations in 2005, 2010, 2015, and 2019.

The comparison for $PM_{2.5}$ yielded similar results to PM_{10} (Figure 22), though for $PM_{2.5}$, both the mean population weighted exposure and the percentage of the population exposed to concentrations above the environmental objective decreased since the 2010 report.

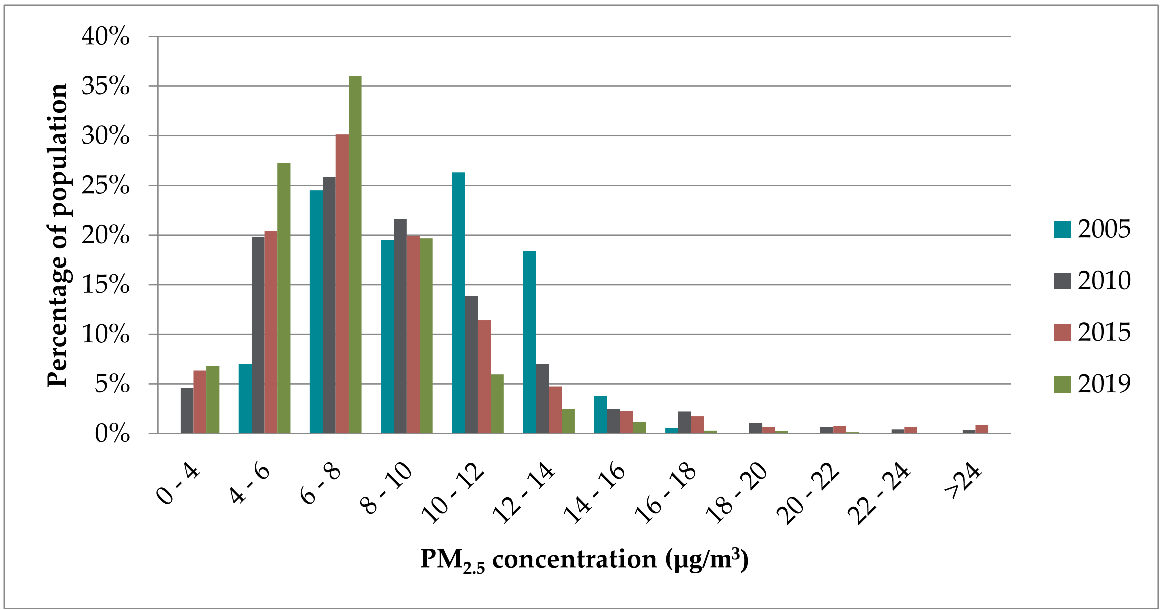


Figure 22 **Percentage of the population exposed to $PM_{2.5}$ (µg/m³) annual mean concentrations in 2005, 2010, 2015, and 2019.**

4.4 Estimated health impacts

4.4.1 Mortality

4.4.1.1 Effects associated with exposure to local traffic $PM_{2.5}$ and urban NO_2

Using the modelled exposure levels and presented assumptions for the health impact assessment, we have estimated the excess mortality associated with long-term exposure to urban (local source) $PM_{2.5}$ as vehicle exhaust and wear particles in the fine fraction to result in 268 (95% CI 197-283) and 488 (95% CI 359-515) deaths per year, without assuming any threshold below which there is no association. The urban contribution of NO_2 is dominated by local emissions from motor vehicles, and is estimated to result in 627 deaths per year (95% CI 312-1233). Together urban $PM_{2.5}$ and NO_2 mainly from road traffic is estimated to be associated with almost 1400 deaths per year in Sweden. This estimated impact on mortality is independent of the impact resulting from regional background levels of $PM_{2.5}$, and not including any potential short-term effect associated with the coarse fraction of road dust ($PM_{2.5-10}$).

With the Swedish age-specific baseline mortality the estimated number of years of life lost (YLL) due to these deaths among persons aged 30+ years are approximately 10 years per preterm death.

4.4.1.2 Effects associated with exposure to $PM_{2.5}$ from residential wood burning

The modelled exposure to $PM_{2.5}$ from local residential wood burning is studied with the same assumptions as for local traffic-related $PM_{2.5}$, without assuming any threshold below which there is no association. We have estimated the excess mortality associated with long-term exposure to $PM_{2.5}$ from local wood burning to result in 708 (95% CI 520-747) deaths per year.

With the Swedish age-specific baseline mortality the estimated number of years of life lost (YLL) due to these deaths among persons aged 30+ years are approximately 10 years per preterm death.

4.4.1.3 Effects associated with exposure to the regional background level of $PM_{2.5}$

The modelled exposure to regional background and long-distance transported $PM_{2.5}$ represents particles not emitted from the local sources such as traffic and domestic heating. For this exposure the long-term effect on mortality is estimated applying no threshold and the combined (total) relative risk estimate from WHO Air Quality Guidelines 1.08 (95% CI 1.06 - 1.09) per 10 µg/m³. This way we have estimated the regional background $PM_{2.5}$ exposure to result in 4652 (95% CI 3398-5033) deaths per year.

With the Swedish age-specific baseline mortality the estimated number of years of life lost (YLL) due to these deaths among persons aged 30+ years are approximately 10 years per preterm death.

4.4.2 Morbidity effects

4.4.2.1 Morbidity associated with exposure to $PM_{2.5}$

We have not assumed that the selected risk functions for diseases and preterm birth are valid below the new WHO air quality guideline for the annual mean $PM_{2.5}$ concentration of 5 µg/m³ when the impact on morbidity estimated. In table 10 (below) the yearly numbers attributed to studied exposure are shown. The asthma cases could be seen as the yearly number of childhood onset cases attributed to exposure that among 18 years old persons still are prevalent.

Table 10 The estimated yearly number of cases associate with PM2.5 exposure above the WHO annual guideline.

Outcome/considered at risk	Cases/year	95% LCL	95% UCL
Myocardial infarction/30+ yrs	455	38	855
Stroke/30+ yrs	1715	308	2584
Lung cancer/30+ yrs	180	64	297
Dementia/50+ yrs	841	300	1251
Diabetes/15+ yrs	1983	760	2889
COPD/50+ yrs	275	170	335
Childhood asthma/-18 yrs	659	191	957
Preterm birth	355	114	518

In the age group 15-64 we estimate 4 223 185 RADs or work loss days per year.

4.5 Socio-economic costs

In the most conservative estimate, the socio-economic costs in Sweden 2019 caused by health effects linked to elevated levels of $PM_{2.5}$ and NO_2 are in the central estimate ~170 billion SEK_{2019} if valuing shortened life expectancy and assuming an effect cut-off of local $PM_{2.5}$ pollution. Out of these ~170 billion, 77% are due to shortened life expectancy, 16% due to morbidity, and 7% due to income losses from work absenteeism. (Table 11).

Table 11 Annual socio-economic costs of high long term air pollution levels in Sweden, 2019. Economic costs calculated based on shortened life-expectancy (VOLY) and a cut-off of effects from local traffic $PM_{2.5}$ exposure.

	Socio-economic cost per outcome (low-high) [SEK_{2019} / case]	Number of health effects (low-high) [years, cases)	Socio-economic cost (low – high) [million SEK_{2019}]
Total Sweden			168 000 (80 000 – 295 000)
Out of which:			
Reduced life expectancy due to urban $PM_{2.5}$ (with cutoff)	2 020 000 (1 490 000 – 2 930 000)	2 680 (1 970 – 2 830)	5 350 (2 930 – 8 280)
Reduced life expectancy due to urban NO_2	2 020 000 (1 490 000 – 2 930 000)	6 270 (3 120 – 12 300)	15 500 (4 650 – 36 100)
reduced life expectancy due to residential wood combustion $PM_{2.5}$ (with cutoff)	2 020 000 (1 490 000 – 2 930 000)	7 080 (5 200 – 7 470)	14 100 (7 750 – 21 900)
reduced life expectancy regional background $PM_{2.5}$ (with cutoff)	2 020 000 (1 490 000 – 2 930 000)	46 500 (34 000 – 50 300)	93 600 (50 600 – 147 000)
Myocardial infarction	322 000 (215 000 – 510 000)	455 (38 - 855)	157 (8.16 - 436)
Stroke	6 180 000 (4 030 000 – 9 940 000)	1 720 (308 – 2 580)	10 300 (1 240 – 25 700)
Lung Cancer	8 820 000 (803 000 – 18 000 000)	180 (64 - 297)	1 660 (51.4 – 5 340)
Dementia	9 130 000 (8 530 000 – 9 660 000)	841 (300 – 1 250)	7 260 (2 560 – 12 100)
Diabetes	2 370 000 (1 430 000 – 4 900 000)	1 980 (760 – 2 890)	5 440 (1 090 – 14 200)
COPD	5 770 000 (2 940 000 – 13 500 000)	275 (170 - 335)	1 920 (500 – 4 510)
Childhood asthma	2 000 000 (1 070 000 – 4 510 000)	659 (191 - 957)	1 520 (204 – 4 320)
Preterm Birth	457 000 (255 000 – 765 000)	355 (114 - 518)	162 (29.1 - 397)
Work loss days	2 380 (1 910 – 3 450)	4 220 000 (-)	10 900 (8 070 – 14 600)

Cost calculations were also made for cases where valuation of life-loss was made by valuing each fatality affected with the value of statistical life, and for situations were no cutoff was considered. The results from these calculations showed that if removing the assumed cutoff of local traffic

$PM_{2.5}$ exposure, the total damage costs would be 172 billion SEK instead of the 168 billion SEK reported above. The effect of basing the cost estimates on value of statistical life would increase the total damage costs to 314 billion SEK with cutoff and ~323 billion without cutoff.

5 Discussion

The exposure of Sweden's population to NO_2, PM_{10} and $PM_{2.5}$ has been calculated using the URBAN model. This is a recurring study presented every four to five years. In the following chapter, the resulting pollutant concentrations will be compared to previous calculations as well as other studies, followed by a discussion of the expected exposure-related health effects and the resulting costs.

5.1 Pollutant concentrations

The calculated pollutant concentrations in background air in 2019 were overall considerably lower than the environmental standard for the annual mean, 40 μg/m³ for both NO_2 and PM_{10}, and 25 μg/m³ for $PM_{2.5}$. In all rural areas, concentrations were also lower that the environmental objectives of 20 μg/m³ for NO_2, 15 μg/m³ for PM_{10}, and 10 μg/m³ for $PM_{2.5}$. However, the environmental objective for NO_2 was exceeded in the central parts of the larger urban areas. For PM_{10}, and $PM_{2.5}$, concentrations in the southern part of Sweden exceeded the environmental objective in the central parts of most medium and large urban areas. The average exposure concentrations were lower than the measured urban background concentrations but higher than the measured average regional background concentration. This is reasonable as more than 1 million people live in the countryside and the method used in this study assumes a reduction in concentration moving towards the edges of urban towns and cities.

In the recently updated WHO 2021 Air Quality Guidelines, the recommended maximum exposure is considerably lowered compared to the environmental objective for both NO_2 (10 μg/m³) and $PM_{2.5}$ (5 μg/m³). While these recommendations are not legally binding, they may influence the long-term development of the Swedish environmental standards and objectives, and a comparison was therefore done. Our calculations indicate that 82 % of the Swedish population is exposed to concentrations exceeding the WHO guidelines for $PM_{2.5,}$ and 11 % to concentrations exceeding the WHO guidelines for NO_2. While this may be a striking exceedance for $PM_{2.5,}$ the great majority of the population is exposed to concentrations only slightly higher than this, and the population weighted average only reached 7.2 μg/m³.

The most pronounced difference compared to previous assessments in 2010 and 2015 is the reduced background PM concentrations along the west coast. As described in chapter 3.2, this difference is connected to the method change, where the background PM concentrations are based on measured concentrations rather than a combination of measured concentrations and concentrations calculated with the EMEP model. As the EMEP model has undergone several steps of development since the 2015 evaluation, it now reflects not only the regional background concentrations but also the local contribution. To avoid double counting the urban contribution in this assessment, the background concentrations were calculated only based on regional background measurements of PM. Since there are only five sites in Sweden where PM is measured in regional background it is difficult to, with certainty, estimate the regional background concentrations, which impose an uncertainty in the calculations. However, the calculated concentrations compared well with these measurements, which ranged from 11 to 15 μg/m^3 for PM_{10}, and from 7 to 10 μg/m^3 for $PM_{2.5}$ in background air (datavardluft.smhi.se/portal). The slight overestimation that was indicated along the coastal areas in 2015 (Gustafsson et al. 2018, Lindén et al 2019), does thus not appear to be an issue in the 2019 assessment.

Measurements of air quality have over the past decades shown reduced NO_2 and PM concentrations (Olstrup et al., 2018), however since 2004 this trend has leveled out for NO_2 due to increasing traffic and the use of diesel vehicles (Fredriksson et al., 2016, Naturvårdsverket 2017). The population weighted exposure to NO_2 calculated for 2019 indicated a reduced population weighted exposure of up to 0.5 μg/m^3 compared to previous assessments. This deviates from the 2010 and 2015 assessment, where a stagnation of the improved exposure situation trend was indicated. The 2019 reduction is likely partially connected to the meteorological conditions, which in 2019 were overall warmer and wetter and in the south also slightly windier (SMHI 2021), which is generally favorable for lower concentrations. If the reduced 2019 concentrations also reflect reduced emissions and thus a continued trend towards improved air quality requires recurring future assessments.

A clear reduction was also found in the percentage of the population exposed to concentrations above the environmental objective. The 2010 and 2015 assessment indicated an increase in this percentage, which was explained by urbanization causing an increased exposure to the higher urban concentrations (Gustafsson et al. 2018). The urbanization trend appears to continue unchanged (SCB 2022), and the reduced percentage of the population exposed to concentrations above the environmental objective in 2019, is likely connected to lower concentrations in both urban and rural areas rather than changes in the spatial population distribution.

The mean population weighted exposure to PM for 2019 also showed similar trend as NO_2, although the reduction is more pronounced. This is reflected in a considerable drop in the percentage of the population exposed to concentrations above the environmental objectives for $PM_{2.5}$ and PM_{10}. A reduction of similar magnitude was found between the 2005 and 2010 assessment. Warm, wet, and windy meteorological conditions are favorable for reduced PM concentrations, and likely account for part of this reduction. As with NO_2, recurring future assessments are required to determine if the reduced 2019 concentrations also reflect reduced emissions and thus a continued trend towards improved air quality. However, for PM the reduction is likely also partially caused by the method change in calculating PM, as discussed above, where regional background concentrations are based only on measurements instead of a combination of measurements and data from the EMEP model.

The results from this study indicate that Sweden has a very good air quality in comparison with the average exposure situation in urban Europe presented in a report by the EEA (2017). The EEA report indicated that around 8% of the European population is exposed to both NO_2 and $PM_{2.5}$ concentrations exceeding the environmental standard for the annual mean. In this exposure assessment, none of the Swedish population were exposed to concentrations exceeding the environmental standard. However, exposure in both this and the EEA study are estimated based on background concentrations to allow application of dose-response functions for the general population. As previously mentioned, higher concentrations are often found at roadside locations due to emissions from traffic. However, as very few dose-response functions are based on roadside concentrations it is not possible to evaluate this in exposure assessments.

The contribution of different sources was overall similar to that obtained in the previous exposure assessments. A noticeable reduction was however found in the contribution of wood burning to total $PM_{2.5}$ exposure compared to the 2015 assessment (Gustafsson et al. 2018). The reduction was found to originate in a small error in the calculations in the 2015 assessment. This only affected calculation of the contribution from wood burning and not the overall exposure. Hence, the influence of this error will only affect trend analysis of health effects and costs related to exposure to $PM_{2.5}$ from wood burning, where the true change is smaller than it looks because of the earlier overestimation.

The exposure to $PM_{2.5}$ from traffic exhaust was very similar to that obtained in the 2015 assessment, and exposure to regional background concentrations was reduced proportionally to the reduced overall exposure in this assessment compared to previous. Evaluation of health effects and costs related to PM exposure from road dust has in previous assessments been based on PM_{10}. The current assessment is instead based on the proportion of PM_{10} that falls within $PM_{2.5}$, and direct comparison with previous years is thus not possible.

As in all model calculation, the method used to determine concentrations and exposure contains uncertainties. One uncertainty in this study is that the empirical model used for calculating pollution concentrations requires a reliable and relatively dense monitoring network providing measurement data in urban and regional background. As addressed more in-depth in the previous assessment report, the Swedish measurement stations is reliable, but the density of the stations has been gradually reduced over the last decades. This limits the possibility to use measurement data as a base for calculating exposure as well as for validation of model-based calculation of exposure.

The assumption that the NO_2 and PM concentrations are proportional to the number of people in a grid cell fails to capture the spatial patterns of roads. However, a comparison between this approach and modelling with a higher spatial resolution showed similar population exposure results (Sjöberg et al., 2009; SLB, 2007). Thus, the assumption is therefore considered appropriate when calculating the PM exposure at a national level and in the resolution of 1*1 km grid cells. Future development of the modelling methodology would be possible by incorporating an improved spatial pattern of emissions. It might also be possible to use concentration maps available for larger cities, and apply the dispersion pattern to the URBAN model.

5.2 Health effects

Time-trends in estimated health impacts of air pollution exposure are driven by many other factors than changes in concentrations or population exposure. The size of the population and the base-line frequency of the studied outcome are both important for the attributed number of cases. The among several potential alternatives actually applied risk function and any assumed low threshold below which no effects are expected, are also important factors that will has an effect on the estimate health impact.

In the 2015 assessment we estimated a total burden of approx. 7600 deaths per year, and noted that assuming a cut off would have resulted in result in lower estimates. We now estimate a total of 6743 deaths per year, and there are several factors behind this reduction, both changed exposures and changes in both directions in the assumed relative risks as a result of new reports.

Referring to a number of studies with similar results, including from Sweden, we apply a relative risk of 1.26 per 10 $\mu g/m^3$ as the most relevant relative risk assumption for the local sources traffic and domestic wood burning. As no meta-analysis was available, we apply the 95% confidence interval around 1.26 from the original results reported by Turner et al. (2016). It is entirely the growing number of relevant studies that has led to the use of this one and a half times higher relative risk per μg/m³ compared to in our 2015 assessment. On the other hand, for the urban contribution of NO_2 we apply the new meta-estimate referred to in the WHO Air Quality Guidelines, 1.02 per 10 μg/m³, which is only one fourth of the relative risk applied in our previous assessment.

In the earlier assessments we have estimated the impact of wear particles from traffic on mortality assuming a short-term effect on PM_{10} road dust on mortality. Now we instead estimate a long-term effect on mortality from the fine fraction wear only, using the same relative risk as for exhaust particles, since epidemiological studies have not been able to distinguish the effects. This assumption has made wear particles even more important as a local air pollution problem.

For the effect on mortality from the regional background concentration of $PM_{2.5}$, mainly long-distance transported particles we apply the overall relative risks assumed in the WHO Air Quality Guidelines, 1.08 per 10 μg/m³, which is 29% higher than the relative risk 1.062 per 10 μg/m³ in 2013 recommended by WHO (HRAPIE) and applied in our 2015 assessment. The number of deaths attributed to the regional background level of $PM_{2.5}$ would be reduced if we used a cutoff level below which no effect is assumed, However, there is no evidence for such a threshold. Instead, the relative increase in mortality per increase in $PM_{2.5}$ concentration is bigger at the lowest levels studied.

We have in comparison with the corresponding 2015 national assessment added more morbidity outcomes this time. Risk functions for morbidity were as far as possibly selected based on how relevant and established they are. Our goal was to apply meta-estimates (from a literature review) and European multi-cohort results with relevant exposures. European multi-cohort results are used for myocardial infarction (ESCAPE), stroke (ELAPSE) and lung cancer (ELAPSE). The functions selected for $PM_{2.5}$ and diabetes, childhood asthma, dementia and preterm birth were all from literature reviews with a calculated meta-estimate.

Ozone has not been included in this study, but has also an impact on preterm deaths and causes also other adverse health effects.

5.3 Socio-economic costs

This study reports socio-economic costs of 2019 levels of $PM_{2.5}$ and NO_2-pollution to be SEK_{2019} 168 billion per year (80 – 295). These costs are higher than the SEK_{2017} 56 billion per year reported in Gustafsson et al. (2018). As can be seen in Table 12, the key reason for the difference is the value used to estimate socio-economic costs of reduced life expectancy. In the current study we have updated the value with values from a 2016 Swedish valuation study, at the expense of loosing comparability of our results with results made by the European Commission and other international organisations.

Table 12 Comparisons of mid-estimate health damage costs in this study and Gustafsson et al., 2018.

	Socio-economic cost per outcome [SEK / case]		Number of health effects [years, cases)	Socio-economic cost [million SEK]	
	Current study	Corresponding in Gustafsson et al., 2018		Current study	Corresponding in Gustafsson et al., 2018
Total Sweden				**168 000**	**47 200**
Out of which:					
Reduced life expectancy	2 020 000	5 150 000*	62 550 / 6 255**	128 580	32 200
Myocardial infarction	322 000	1 930 000	455	157	878
Stroke	6 180 000	2 790 000	1 720	10 300	4 790
Lung Cancer	8 820 000	-	180	1 660	-
Dementia	9 130 000	-	841	7 260	-
Diabetes	2 370 000	-	1 980	5 440	-
COPD	5 770 000	-	275	1 920	-
Childhood asthma	2 000 000	-	659	1 520	-
Preterm Birth	457 000	-	355	162	-
Work loss days	2 380	1500	4 220 000	10 900	6 330
Not assessed in current study but included in 2018 totals for completeness of comparison					
Restricted activity days (age group 0-14, 65-)					1 380
Chronic Bronchitis					1 590

*Life-expectancy-adjusted VOLY over 10 years, adjusted from 11 years in Gustafsson et al. (2018)
**Value used for recalculation of Gustafsson et al. (2018)-values

When comparing the socio-economic costs of air pollution as if we would have used the same values as in Gustafsson et al. (2018), we can see that socio-economic costs have declined from SEK_{2017} 56 to 47 billion between 2015-2019, a larger improvement than between 2010-2015 as reported in Gustafsson et al. (2018).

We have also made a back-of-the envelope valuation of lung cancer cases caused by air pollution. This valuation is based on few studies, out of which two are old, and the valuation technique has been direct value transfer, which can be criticized. However, the project resources could not accommodate larger valuation efforts. And even though the value is uncertain, it is a more reasonable value than the hitherto used economic value for air pollution-caused lung cancer (0 SEK/case).

Further, we calculate socio-economic costs for all reported uncertainty ranges in the literature and the uncertainty ranges of the health impact assessment. From the uncertainty analysis it is important to stress that the lowest of the low estimates, still presents socio-economic costs of air pollution to be SEK_{2019} ~80 billion per year, a substantial number. Also, just the value of work losses, which to 75% consists of foregone salaries, has a value corresponding to 0.02% of the Swedish GDP in 2019.

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