REPORT

Quantification of population exposure to NO_2, $PM_{2.5}$ and PM_{10} and estimated health impacts in Sweden 2010

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Summary

Sweden is one of the countries in Europe which experiences the lowest concentrations of air pollutants in urban areas. Despite this, health impacts of exposure to ambient air pollution is still an important issue in the country and the concentration levels, especially of nitrogen dioxide (NO_2) and particles (PM_{10} and $PM_{2.5}$), exceed the air quality standards at street level in many urban areas.

IVL Swedish Environmental Research Institute and the 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 2010. 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.

Environmental standards as well as environmental objectives are to be met everywhere, also at the most exposed kerb sides. However, for exposure calculations it is more relevant to used urban background data, on which also most available exposure-response functions are based. The results show that in 2010 most of the country had rather low NO_2 urban background concentrations in comparison to the environmental quality standard for the annual mean (40 $\mu g/m^3$) and the population weighted average exposure to NO_2 was 6.2 $\mu g/m^3$. Likewise the PM_{10} urban background concentrations, compared to the environmental quality standard for the annual mean (40 $\mu g/m^3$), were also low in most parts of the country. However, in some parts, mainly in southern Sweden the concentration levels were of the same magnitude as the environmental objective (20 $\mu g/m^3$ as an annual mean) for the year 2010. The majority of people, 90%, were exposed to annual mean concentrations of PM_{10} less than 20 $\mu g/m^3$. Less than 5% of the Swedish inhabitants experienced exposure levels of PM_{10} above 25 $\mu g/m^3$.

The modelling results for $PM_{2.5}$ show that the urban background concentration levels in 2010 were of the same order of magnitude as the environmental objective (12 $\mu g/m^3$ as an annual mean for the year 2010) in a quite large part of the country. About 70% of the population was exposed to $PM_{2.5}$ annual mean concentrations lower than 10 $\mu g/m^3$, while less than 15% experienced levels above 12 $\mu g/m^3$.

There is currently within the research community a focus on the different types of particles and more and more indications that their impact on health and mortality differ. Yet a common view is still that current knowledge does not allow precise quantification of the health effects of PM emissions from different sources. However, when the impact on mortality from PM_{10} is predicted, exposure-response functions obtained using $PM_{2.5}$ are usually reduced using the $PM_{2.5}/PM_{10}$ concentration ratio.

Assessment of health impacts of particle pollution is thus difficult. Even if WHO in HRAPIE and others assessments still choose to recommend the same relative risk per particle mass concentration regardless of source and composition, we find this a too conservative approach. Therefore we applied different exposure-response functions for primary combustion generated particles (from motor vehicles and residential wood burning), for road dust and for other particles (the regional background of mainly secondary particles).

For primary combustion particles we have in this study applied the exposure-response coefficient 17% per 10 μg/m³ for mortality. For other $PM_{2.5}$ sources and for $PM_{2.5}$ totally, we applied the 6.2% per 10 μg/m³ as was recently recommended by WHO. For road dust we here assumed only a "short-term" effect on mortality as has been done for PM_{10} in general.

We estimated approximately 3 500 preterm deaths per year from $PM_{2.5}$ without any division between sources and using the exposure-response coefficient 6.2% per 10 μg/m³. Assuming a division between sources we estimated that non-local sources caused just over 3 000 preterm deaths per year (exposure-response coefficient 6.2% per 10 μg/m³), and residential wood burning caused just over 1 000 preterm deaths per year (exposure-response coefficient 17% per 10 μg/m³). In addition, we estimated approximately 1 300 preterm deaths per year from locally generated vehicle exhaust using NO_2 as an indicator (exposure-response coefficient 17% per 10 μg/m³ and a 5 μg/m³ cut-off). Preterm mortality related to short-term exposure to road dust PM, estimated to over 200 deaths per year (exposure-response coefficient 1,7% per 10 μg/m³), should probably be added to the impact of local traffic in Sweden. In summary, the total number of preterm deaths can be estimated to approximately 5 500 per year when taking into account differences in exposure-response for different PM sources. Note that the ground-level ozone has not been taken into account in this study, but can still cause premature deaths and other health issues.

For morbidity we have in this study included only some of the potentially available health endpoints to be selected. Only a few important and commonly used endpoints were included to allow comparisons with other health impact assessments and health cost studies.

The estimated respiratory and cardiovascular hospital admissions due to the short-term effects of air pollution may seem to be low in comparison with the estimated number of deaths, new chronic bronchitis cases and restricted activity days. However, for hospital admissions we can only estimate the short-term effect (acute effect) on admissions, not the whole effect on hospital admissions following morbidity induced by the air pollution exposure.

The socio-economic costs (welfare losses) related to population exposure to air pollutants as indicated by NO_2 were calculated both with and without a threshold of 5 μg/m³. The results suggest that the health effects related to annual mean levels of NO_2 can be valued to between 7 and 25 billion Swedish crowns (SEK_{2010}) during 2010 depending on if a threshold of above 5 μg/m³ is included or not.

Moreover, welfare losses resulting from exposure to PM pollutants from road dust, domestic heating and other sources can be valued to annual socio-economic costs of about 35 billion SEK_{2010} during 2010. Approximately 6.5 of these 35 billion SEK_{2010} are from productivity losses in society. Furthermore, the amount of working and studying days lost constitutes about 0.3% of the total amount of working and studying days in Sweden during 2010. Using the division between PM sources and NO_2 (with a 5 μg/m³ cut-off) as an indicator of traffic combustion the total socio-economic cost would be approximately 42 billion SEK_{2010}.

In a counterfactual analysis, impacts of a hypothetical large scale introduction of electric passenger vehicles in the Stockholm, Göteborg, and Malmö regions were studied. The results from this analysis indicated that the health benefits from introducing ~10% electric vehicles in these regions would motivate 13 – 18% of the investment.

Sammanfattning

Befolkningens exponeringen både för partiklar (PM) och kvävedioxid (NO_2) har minskat mellan 2005, då den föregående beräkningen genomfördes, och 2010. Knappt 10 % av Sveriges befolkning utsätts för bakgrundshalter av PM_{10} (partiklar mindre än 10 μm) högre än 20 μg/m^3 i den allmänna utomhusluften. Denna halt motsvarar miljömålet för år 2010, men nivån skall även klaras i mer belastade områden såsom gaturum. För mindre partiklar ($PM_{2,5}$, partiklar mindre än 2,5 μm) visar motsvarande jämförelse med miljömålet (12 μg/m^3 för år 2010) på att knappt 15 % av landets invånare exponeras för halter över denna nivå.

Om man utgår från att $PM_{2.5}$ har samma farlighet oavsett ursprung så uppskattar vi att cirka 3 500 förtida dödsfall årligen inträffar i Sverige på grund av den totala exponeringen. Troligtvis är det inte tillräckligt att göra beräkningar utifrån den totala halten, eftersom partiklar av olika ursprung tycks ha olika farlighet. Med separata bedömningar för olika källor till PM uppskattar vi att det årligen rör sig om cirka 3 000 förtida dödsfall från partiklar som inte genererats lokalt. Förbränningspartiklar från vedeldning uppskattas orsaka ytterligare drygt 1 000 förtida dödsfall per år. Utöver dessa dödsfall uppskattar vi, utifrån exponeringen för NO_2, att lokalt genererade avgaser leder till ytterligare minst 1 300 förtida dödsfall per år, samt vägdamm till ytterligare drygt 200 dödsfall orsakade av kortvarig exponering. Sammanfattningsvis kan antalet förtida dödsfall uppskattas till cirka 5 500 per år på grund av dessa exponeringar, när beräkningarna tar hänsyn till olika exponering-responsvärden för olika källor.

De samhällsekonomiska kostnaderna (välfärdsförluster) relaterade till exponering för luftföroreningar mätt som NO_2 beräknades både för effekter över 5 μg/m^3 och utan tröskel. Resultaten tyder på att de hälsoeffekterna, relaterade till årsmedelhalten av NO_2, kan värderas till mellan 7 och 25 miljarder kronor under 2010, beroende på om en tröskel på över 5 μg μg/m^3 ingår eller ej.

Resultaten från vår studie visar att negativa hälsoeffekter relaterade till förorenad luft med höga nivåer av PM kan värderas till årliga samhällsekonomiska kostnader (välfärdsförluster) på ca 35 miljarder svenska kronor under 2010. Ungefär 6,5 av dessa 35 miljarder utgörs av produktivitetsförluster i samhället. Detta motsvarar en förlust i antalet arbets- och studiedagar motsvarande drygt 0,3 % av den totala mängden arbets- och studiedagar under 2010.

Haltnivåerna av partiklar (PM) i omgivningsluften har fortfarande en betydande hälsopåverkan, trots att det under de senaste årtiondena har införts ett flertal åtgärder för att minska utsläppen. Miljökvalitetsnormerna för utomhusluft överskrids på många håll, och tidigare studier har uppskattat att höga partikelhalter orsakar upp till 5 000 förtida dödsfall i Sverige per år (Forsberg et al., 2005; Sjöberg et al., 2009).

På uppdrag av Naturvårdsverket har IVL Svenska Miljöinstitutet och Yrkes- och miljömedicin vid Umeå universitet kvantifierat den svenska befolkningens exponering för halter i luft av kvävedioxid (NO_2), $PM_{2,5}$ och PM_{10} för år 2010, beräknat som årsmedelkoncentrationer. Även de samhällsekonomiska konsekvenserna av de uppskattade hälsoeffekterna har beräknats.

Angivna miljökvalitetsnormer och miljömål skall klaras överallt, även i de mest belastade gaturummen. För exponeringsberäkningar är det dock mest relevant att använda urbana bakgrundshalter, som även tillgängliga exponerings/respons-samband baseras på. Resultaten visar att den urbana bakgrundshalten av NO_2 i merparten av landet var relativt låg i förhållande till miljökvalitetsnormen för årsmedelvärde (40 µg/m³). Likaså var den urbana bakgrundshalten av PM_{10} relativt låg i förhållande till miljökvalitetsnormen för årsmedelvärde (40 µg/m³) i stora delar av landet. I vissa områden, huvudsakligen i södra Sverige, var haltnivåerna i samma storleksordning som miljömålet (20 µg/m³ som årsmedelvärde) för år 2010. Merparten av befolkningen, 90 %, exponerades för årsmedelhalter av PM_{10} lägre än 20 µg/m³. Mindre än 5 % av landets invånare utsattes för exponeringsnivåer av PM_{10} över 25 µg/m³.

Beträffande $PM_{2,5}$ var den urbana bakgrundskoncentrationen år 2010 i samma storleksordning som miljömålet (12 µg/m³ som årsmedelvärde för år 2010) i en stor del av landet. Ungefär 70 % av befolkningen exponerades för årsmedelhalter av $PM_{2.5}$ lägre än 10 µg/m³, medan knappt 8 % utsattes för halter över 14 µg/m³.

Eftersom forskningsresultat tyder på att de relativa riskfaktorerna för hälsoeffekter är högre för förbränningsrelaterade partiklar än för partiklar från andra källor så separerades också den totala PM_{10}-halten på olika källbidrag med hjälp av en multivariat analysmetod. Även om projekt som HRAPIE fortfarande väljer att rekommendera att samma relativa risk används för alla partikelar oavsett källa och sammansättning, anser vi att detta är en alltför konservativ strategi. Därför använder vi olika dos-respons funktioner för primära förbränningspartiklar (från motorfordon och vedeldning), vägdamm samt övriga partiklar (regional bakgrund av främst sekundära partiklar). För primära förbränningspartiklar har vi i denna studie tillämpat dos-responssambandet 17 % ökad dödlighet per 10 µg/m³ årsmedelhalt. För andra $PM_{2,5}$ källor, tillämpat vi dos-respons sambandet 6,2 % per 10 µg/m³ som nyligen rekommenderats av WHO. För vägdamm har vi här antagit endast en "kortvarig" effekt på dödlighet orsakad av PM_{10}.

Vi uppskattar att cirka 3 500 förtida dödsfall årligen inträffar i Sverige på grund av den totala exponeringen för $PM_{2.5}$ uppskattad med dos-responskoefficient på 6,2 % per 10 µg/m³. Troligtvis är det inte tillräckligt att göra beräkningar av hälsopåverkan utifrån den totala halten, eftersom partiklar av olika ursprung tycks ha olika farlighet. Om vi begränsar beräkningen till partiklar som inte genererats lokalt, uppskattar vi att det årligen rör sig om cirka 3 000 förtida dödsfall. Förbränningspartiklar från vedeldning uppskattas orsaka drygt 1 000 förtida dödsfall per år. Utöver dessa dödsfall uppskattar vi utifrån exponeringen för kvävedioxid att lokalt genererade avgaser leder till ytterligare minst 1 300 förtida dödsfall per år, samt vägdamm till ytterligare drygt 200 dödsfall orsakade av kortvarig exponering. Sammanfattningsvis kan antalet förtida dödsfall uppskattas till cirka 5 500 per år på grund av dessa exponeringar. Notera att marknära ozon inte har beaktats i denna studie, men orsakar också förtida dödsfall och andra hälsokonsekvenser.

Både hälsoeffekter, orsakade av höga halter av luftföroreningar, och åtgärder för att minska dessa halter är oundvikligen kopplade till samhällskostnader. Eftersom det är viktigt för beslutsfattare att använda skattepengar och andra finansiella resurser effektivt är det även viktigt att göra bedömningar av värdet för samhället av att bland annat undvika hälsoeffekter orsakade av höga halter av luftföroreningar. I den ekonomiska delen av denna rapport har genomförts separata ekonomiska värderingar av de hälsoeffekter som orsakas av höga halter av NO_2 och PM i luft.

Internationellt har det skett mycket arbete kring värdering av hälsoeffekter och vi har i denna studie valt att använda de värderingar som gjorts i tidigare internationella studier som grund för värdering av svenska samhällskostnader kopplade till höga halter av NO_2 och PM. Detta gynnar jämförelse med andra resultat inom området kring ekonomisk värdering av hälsoeffekter.

De samhällsekonomiska kostnaderna (välfärdsförluster) relaterade till NO_2 beräknades både för effekter över 5 µg/m³ och utan tröskel. Resultaten tyder på att de hälsoeffekterna, relaterade till årsmedelhalten av NO_2, kan värderas till mellan 7 och 25 miljarder kronor under 2010, beroende på om en tröskel på över 5 µg/m^3 ingår eller ej.

Resultaten från vår studie visar att negativa hälsoeffekter relaterade till höga nivåer av PM kan värderas till årliga samhällsekonomiska kostnader (välfärdsförluster) på ca 35 miljarder svenska kronor under 2005. Ungefär 6,5 av dessa 35 miljarder utgörs av produktivitetsförluster i samhället. Detta motsvarar en förlust i antalet arbets- och studiedagar motsvarande drygt 0,3 % av den totala mängden arbets- och studiedagar under 2010. Används källfördelningen av PM och NO_2 med tröskelvärdet 5 µg/m^3 som en indikator för trafikavgaser blir de totala samhällsekonomiska kostnaderna cirka 42 miljarder SEK_{2010}.

I en alternativ analys studerades effekter av en hypotetisk storskalig introduktion av elbilar i Stockholms-, Göteborgs-, och Malmöregionerna. Resultaten indikerade att kostnaderna för att introducera ungefär 10 % elbilar skulle till ungefär 13 – 18 % kunna motiveras endast av denna åtgärds påverkan på luftföroreningsrelaterade hälsoeffekter år 2010.

Contents

1 Introduction

Despite the successful work to improve the outdoor air quality situation in Sweden (SOU 2011:34; Naturvårdsverket, 2014) by reducing emissions from both stationary and mobile sources, the health impacts of exposure to ambient air pollution is still an important issue. As shown in many studies during recent years, the concentration levels, especially of nitrogen dioxide (NO_2) and particles (PM_{10} and $PM_{2.5}$), in many areas exceed the air quality standards and health effects of exposure to air pollutants (Grennfelt et al., 2013; Persson et al., 2013; WHO, 2013a).

The concentration of particulate matter (PM) in ambient air still has significant impact on human health, even though a number of measures to reduce the emissions have been implemented during recent decades (Sjöberg et al., 2009; Miljömålsrådet, 2008; Persson et al., 2013). The air quality standards are exceeded in many areas, and a previous study estimated that more than 5 000 premature deaths in Sweden per year were due to PM exposure (Forsberg et al., 2005).

Within the framework of the health-related environmental monitoring programme, conducted by the Swedish Environmental Protection Agency, a number of different activities are performed to monitor health effects that may be related to environmental factors.

On behalf of the Swedish Environmental Protection Agency, 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 for the year 2010. Based on these results the health and associated economic consequences have been calculated.

2 Background and aims

The highest concentrations of nitrogen dioxide (NO_2) and particles (PM_{10} and $PM_{2.5}$) in a city are normally found in street canyons. However, for studies of population exposure to air pollution it is customary to use the urban background air concentrations, since these data are used in dose-response relationship studies and health consequence calculations.

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 the 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 have later been 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). The results indicated a slight decrease in the excess exposure (hourly NO_2 concentrations above 110 $\mu g/m^3$ as a 98 percentile) in 1999/2000 compared to the earlier years, from roughly 3% of the population in 1990/91 to about 0.3% in 1999/2000.

In 2007 a study of NO_2 exposure in Sweden was conducted using the 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). The results from the urban modelling showed that in 2005 most of the country had low NO_2 urban background concentrations compared to the environmental standard for the annual mean (40 $\mu g/m^3$). In most of the small to medium sized cities the NO_2 concentration was less than

15 µg/m³ in the city centre. In the larger cities and along the Skåne west coast the concentrations were higher, up to 20-25 µg/m³, which is of the same magnitude as the long-term environmental objective (20 µg/m³ as an annual mean). Almost 50% of the population was exposed to annual mean NO_2 concentrations of less than 5 µg/m³. A further 30% of the population was exposed to concentration levels between 5-10 µg/m³ NO_2, and only about 5% of the Swedish inhabitants experienced exposure levels above 15 µg/m³ NO_2. The concentrations of NO_2 in urban air were estimated to result in more than 3 200 excess deaths per year in the 2005 study (Sjöberg et al., 2007). In addition they estimated more than 300 excess hospital admissions for all respiratory diseases and almost 300 excess hospital admissions for cardiovascular diseases due to the short-term effect of levels above 10 µg/m³. The total annual socio-economic costs related to health effects associated with mean levels of NO_2 higher than 10 µg/m³ were valued to 18.5 billion Swedish crowns (SEK_{2005}).

The trend analysis between 1990 and 2005 clearly showed an increasing number of people exposed to lower NO_2 concentration levels (Sjöberg et al., 2007). During the same period the population weighted annual mean of NO_2 decreased by almost 40%, accordingly to the 35% reduction of total NO_X emissions in Sweden (www.naturvardsverket.se).

In contrast to NO_2, measurements of PM_{10} have only been carried out over the past decade. The available data on $PM_{2.5}$ in urban areas is even more limited. Today the monitoring of NO_2, PM_{10} and $PM_{2.5}$ in smaller municipalities is undertaken within the framework of the urban air quality network, a co-operation between local authorities and IVL Swedish Environmental Research Institute (Persson et al., 2013). In larger municipalities the local authorities conduct their own air quality monitoring.

A study, using the URBAN model, was conducted in 2009 to estimate the population exposure to PM_{10} and $PM_{2.5}$ in 2005 (Sjöberg et al., 2009). The results from the urban modelling showed that the majority of people in 2005, 90%, were exposed to annual mean concentrations of PM_{10} less than 20 µg/m³. Less than 1% of the Swedish inhabitants experienced exposure levels of PM_{10} above 25 µg/m³. The modelling results regarding $PM_{2.5}$ showed that the urban background concentration levels in 2005 were in the same order of magnitude as the environmental objective (12 µg/m³ as an annual mean for the year 2010) in a quite large part of the country. About 50% of the population was exposed to $PM_{2.5}$ annual mean concentrations less than 10 µg/m³, while less than 2% experienced levels above 15 µg/m³.

In the 2005 study the calculated concentrations of PM_{10} were estimate to lead to approximately 3 400 premature deaths per year. Together with 1 300 - 1 400 new cases of chronic bronchitis, around 1 400 hospital admissions and some 4.5-5 million reduces activity days (RADs), the societal costs for health impacts were estimated to approximately 26 billion SEK_{2005} per year. For $PM_{2.5}$ somewhat lower numbers were estimated, for example approximately 3 100 premature deaths per year.

Even though emission reductions regarding both NO_2 and particles have been on the agenda for the past few decades and progress have been made, urban areas are growing and more people are moving to city areas where air quality is poor. The purpose of this study was to calculate the exposure to yearly mean concentrations of NO_2, PM_{10} (total, as well as, different source contributions) and $PM_{2.5}$ on a national scale, 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 possible existing trends.

3 Methods

The method applied for calculation of ambient air concentrations and exposure to air pollutants has been described earlier (Sjöberg et al. 2007). 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 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.

The quantification of the annual mean of population exposure to NO_2, PM_{10} and $PM_{2.5}$ (PM_{10} and $PM_{2.5}$ was calculated annual means as well as separated for different source contributions) was based on a comparison between the pollution concentration and the population density. Like the calculated air pollutant concentrations the population density data had a grid resolution of 1x1 km. By over-laying the population grid to the air pollution grid the population exposure to a specific pollutant was estimated for each grid cell.

To estimate the health consequences, exposure-response functions for the long-term health effects were used, together with the calculated NO_2 and PM exposure. For calculation of socio-economic costs, results from economic valuation studies and other cost calculations were used. These cost estimates were combined with the estimated quantity of health consequences performed in this study to give the total socio-economic costs of NO_2 and PM concentration in ambient air during 2010.

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 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 concentrations of NO_2 was calculated by interpolating measurements from regional background sites. For 2010, 17 sites with monthly background data were used. The background grid was calculated for 2-month periods during the year to account for seasonal variations in the NO_2 concentration. Dividing the year up in 2-month periods was deemed an appropriate time resolution as it gave a representation of the seasons without increasing the computational time too much. Finally the annual background map was calculated from the 6 interpolated bi-monthly maps, see Figure 1.

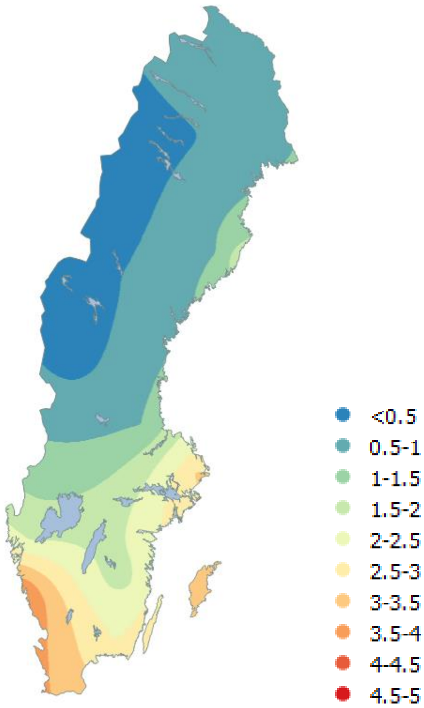

Figure 1 Annual mean of regional background concentrations of NO_2 in Sweden in 2010, unit µg/m^3.

3.1.2 Urban background

The urban background concentrations of NO_2 were calculated using the URBAN model, the method is described in Sjöberg et al. (2007). The concentration distribution of air pollutants in urban background air within cities was estimated assuming a decreasing gradient from the town center towards the regional background areas. The calculated concentrations of air pollutants are valid for the similar height above ground level as the input data (4-8 m) in order to describe the relevant concentrations for exposure. For the larger cities (>10 000 inhabitants) the area of the city was defined and 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.

3.2 PM_{10} concentration calculations

3.2.1 Regional background

Monitoring of PM_{10} and $PM_{2.5}$ in regional background air is carried out at four sites in Sweden, within the national environmental monitoring programme financed by the Swedish Environmental Protection Agency (data hosted by www.ivl.se). The basis for calculating a reasonable realistic geographical distribution of PM_{10} and $PM_{2.5}$ concentrations over Sweden is thus limited. Therefore, calculated distribution patterns by the mesoscale dispersion model EMEP on a yearly basis were used, in combination with the existing monitoring data (EMEP, 2012).

As for NO_2, to separate the regional and local PM_{10} contributions it was necessary to divide the regional background concentrations into two-month periods. This was done by using data for the four monitoring sites, and applying similar conditions between the annual and monthly distribution of the calculated PM_{10} concentrations from the EMEP model. The annual background map of PM_{10} was calculated from the 6 bi-monthly interpolated maps, see Figure 2.

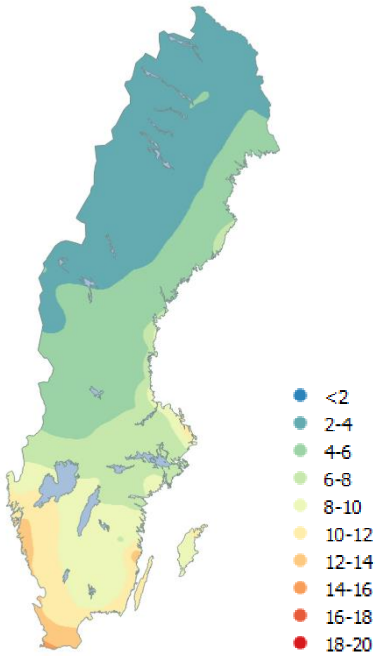

Figure 2 Annual mean regional background concentrations of PM_{10} in Sweden in 2010 (the EMEP model in combination with monitoring data), unit µg/m^3.

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 2010 (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 resolution of 2 months.

Two-month means were calculated for the urban areas where data were available for both PM_{10} and NO_2 for the years 2000-2010. The regional estimated background concentrations of NO_2 and PM_{10} were subtracted, and seasonal ratios of PM_{10}/NO_2 for the remaining local contribution were derived and analyzed with respect to the latitude, see Figure 3. Thus, different equations for each season were derived for the graphs presented in Figure 3. It was not statistically relevant to calculate a standard deviation of the ratios for each season since there were not enough data.

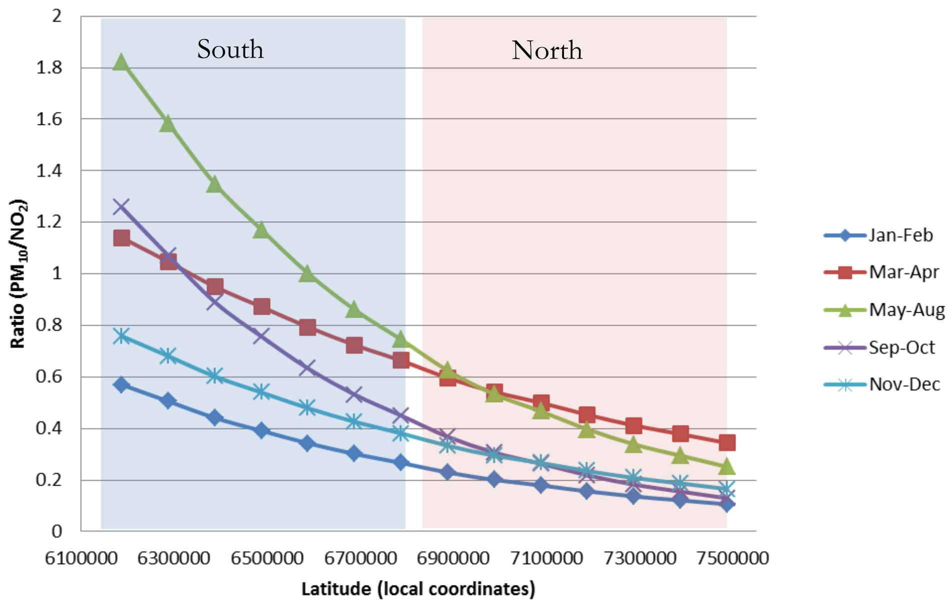

Figure 3 Latitudinal and seasonal variation of the functions based on the locally developed ratios (PM_{10}/NO_2) in urban background air.

According to the calculated functions of the ratio (PM_{10}/NO_2) there are large seasonal differences both in the northern and southern part of Sweden. For the southern part the largest difference was found in May-June and the smallest in January-February. In the north the differences were very small compared to the situation in the south. Due to a limited number of data in July–August, the function for May-June was also applied for those months.

When comparing the national annual means of calculated and monitored urban background concentrations of PM_{10} it becomes clear that the calculated concentrations are overestimated by about 5%. Variations between measurements and the calculated concentrations were large in a few places, for example in Kiruna where the calculated concentration was almost 60% lower than the measured annual mean. This variation most likely depends on that the URBAN model is based on population data and does not include specific local emission sources, such as in the case of Kiruna a large open mine located close to the city. Variations in other parts of the country are most likely associated with the interpolation of the regional background concentrations. Since the urban background concentration constitutes between 50-70% of regional background concentration an error in this calculation can cause rather large overall errors. In spite of this uncertainty the validation shows a reasonably good agreement between measured and calculated urban background concentrations.

3.2.3 Separation of particle source contributions

Since it is assumed that the relative risk factors for health impact are higher for combustion related particles (WHO, 2013b) the total PM_{10} concentration was also separated into different source contributions by using a multivariate method (see further Chapter 3.2.3.4). In the following sections different calculated contributions of particles are described.

3.2.3.1 Small scale domestic heating

Small scale domestic wood fuel burning is an important contributor to particle emission in Sweden (Naturvårdsverket, 2014). Specific information on the use of wood fuel on municipality level was not available for 2010. Therefore, in order to evaluate the proportion of PM_{10} 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 2010 to derive the wood fuel consumption (www.scb.se). Figure 4 – 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.

The energy consumption from wood burning for each of the densely built-up areas in Sweden was drawn from the information presented in Figure 6.

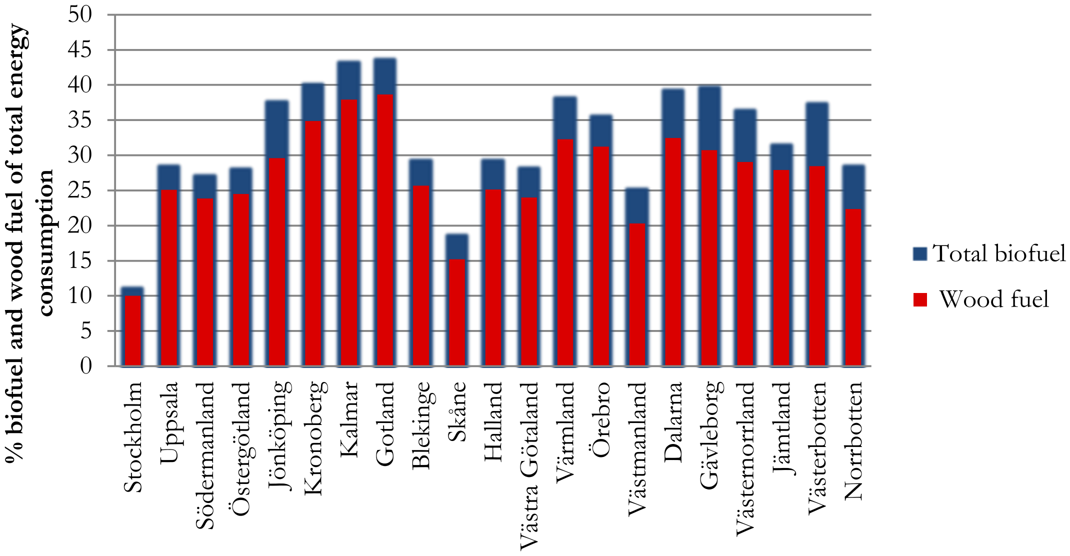

Figure 4 Percentage of total energy consumption from biofuels including wood fuel (blue bars), the percentage from wood fuel (red bars) and per county in 2010.

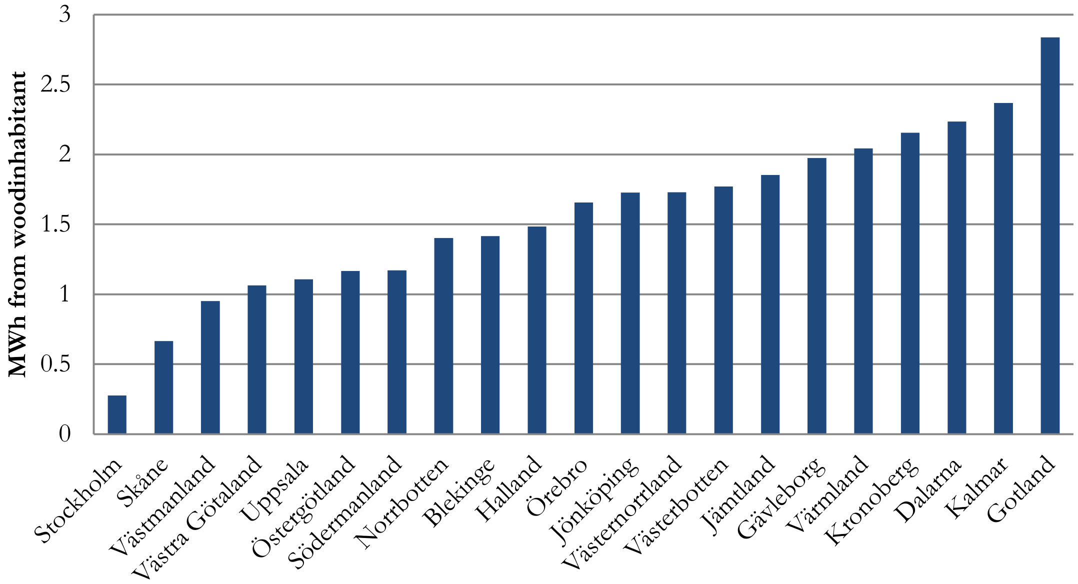

Figure 5 Energy consumption from wood burning (MWh) per inhabitant in each county in 2010.

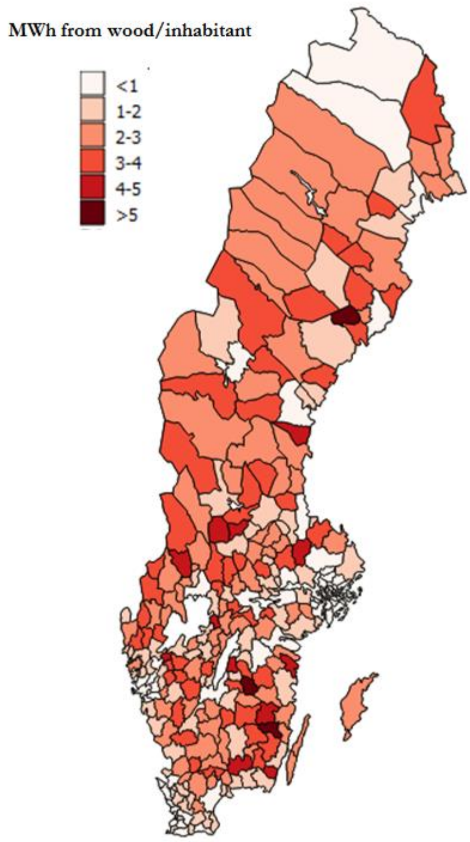

Figure 6 Energy consumption from wood burning (MWh)/inhabitant in each municipality in Sweden in 2010.

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 1. The energy index calculations are based on monitored (by SMHI) outdoor temperature as means for 30 years at 535 sites located all over Sweden and result in monthly national distribution of the energy indices, see Figure 7.

Table 1 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

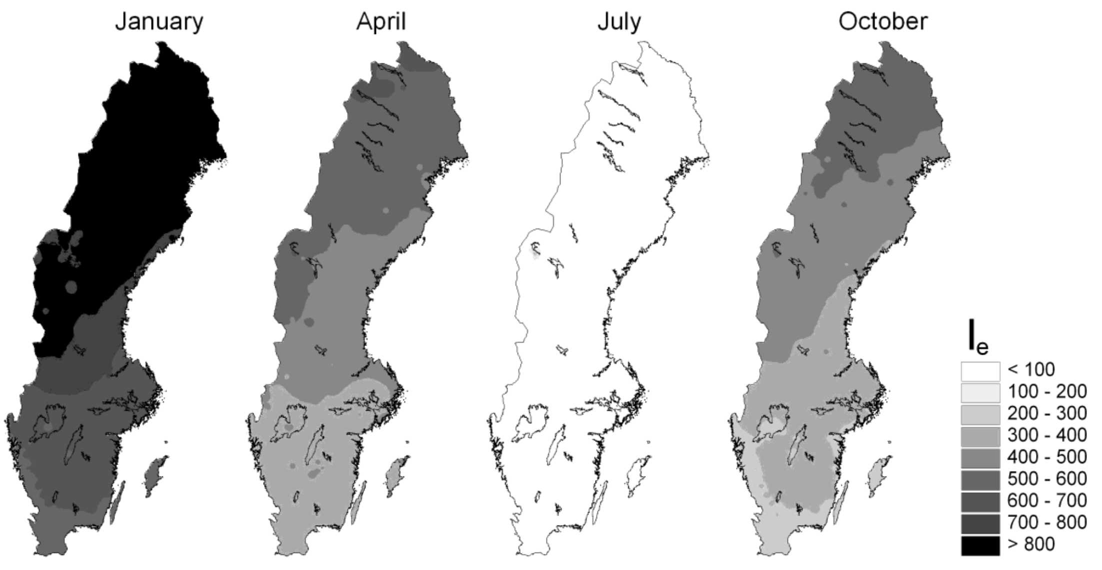

Figure 7 The calculated energy index (I_e) for Sweden i January, April, July, October.

Based on these interpolated maps, two-month means of I_e were extracted for each of the 2 051 towns in Sweden. Based on the I_e the seasonal variation in wood fuel consumption were calculated.

3.2.3.2 Traffic induced particles

Traffic contributes to the total concentration of PM_{10} both directly through exhaust emissions from vehicles and secondarily through re-suspended dust from roads. Traffic related particle concentrations are associated with the NO_2 concentration in urban areas, why the earlier calculated NO_2 concentrations for all densely built-up areas (Sjöberg et al., 2007) were used in the multivariate analysis to determine this source.

Road dust arises mainly from wear of the road surface, from brakes and tyres, and in particular 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, evaluation of the use of studded tyres was also included as a parameter (see below) analysed with the multivariate method.

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 vegetation season starts, mainly in the southern part of Sweden.

The Swedish Transport Administration (Trafikverket, 2010) supplied information on the use of studded tyres in January/February 2010 in the seven different road administration regions (Figures 8 and 9). Unfortunately, there is no such information available with a monthly resolution throughout the year. A monthly based usage of studded tyres in the seven road administration regions were established using the distribution pattern derived by Sjöberg et al. (2009).

From this information two-month means of the percentage use of studded tyres were calculated for each densely built-up area in Sweden to be further used in the multivariate analysis.

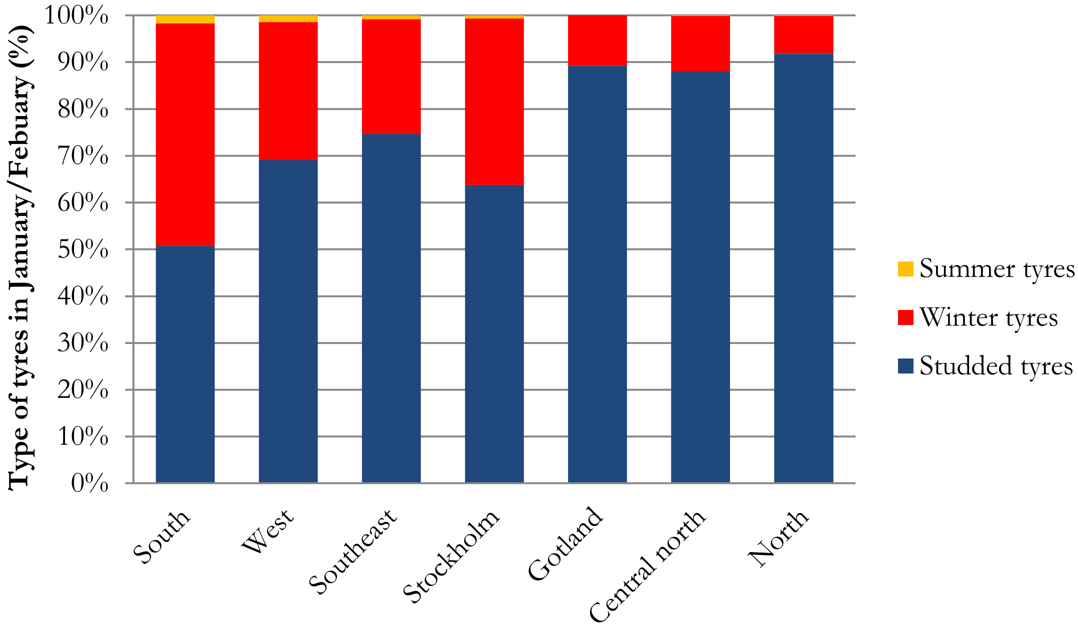

Figure 8 The usage of different types of tyres in January/February within the seven road administration regions in Sweden (visualized in Figure 9).

The different road administration regions:

1. South
2. West (Väst)
3. East (Öst)
4. Stockholm
5. Gotland
6. Central north
7. North

Figure 9 The seven road administration regions of Sweden.

3.2.3.3 *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). The mean values of V_i have in Figure 10 been calculated in groups of every 1000 steps of the local coordinates.

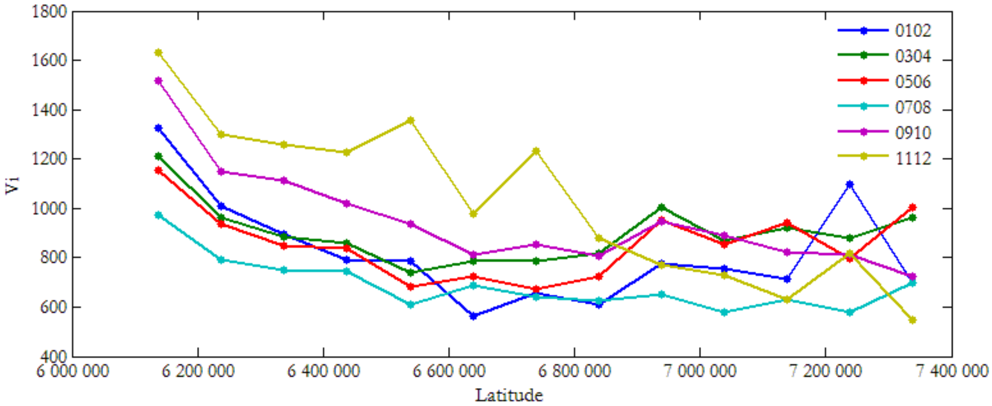

Figure 10 Two-month means of V_i calculated in groups of every 1000 steps of the local coordinates (from south to north) in all towns in Sweden.

According to results presented in Chen et. al (2000) the calculation of the mixing height and wind speed by the TAPM model is well in accordance with measurements. During winter V_i decreases with latitude from V_i about 1500 in the south to about 7000 at the level of about Gävle (between 6838000 and 6938000 in Figure 10), indicating better dispersion facilities in the south. In Sweden different weather systems are dominant in the northern and southern parts during winter, influencing the V_i, and thus the dispersion of air pollutants, differently. However, this latitudinal pattern is very much levelled out during spring and summer, whereas other local differences, such as topographical effects, become more important to the dispersion pattern (see Sjöberg et al., 2007).

3.2.3.4 Multivariate data analysis

In this project Multivariate data analysis (MVDA) has been used to separate different contribution of the total PM_{10} concentration based on six parameters which represent different sources. The six parameters are presented in Chapter 3.2.3. The data has been evaluated for 1881 communities in Sweden.

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 time periods (two months per period), based on the fact that the use of studded tyres and the wood fuel burning 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 2-month 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 is therefore excluded in these three models.

All six models give 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 is here assessed by cross-validation[1], see Sjöberg et al. (2009).

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

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

2 **Q^2 :** Goodness of prediction, describes the fraction of the total variation of the Y:s that can be predicted by the model according to cross validation (max 1) (in this case Q^2 = performance)

Table 2 The performance of the models, measured as cross validated explained variance for PM_{10}.

Model	Performance (%)
Month 1-2	94.9
Month 3-4	82.1
Month 5-6	84.5
Month 7-8	84.6
Month 9-10	80.2
Month 11-12	88.3

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 3 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 4.3.1).

Table 3 Average contribution (%) to the PM_{10} content for each variable and time period normalised to sum up to 100. Other variables, not measured and not presented here, are also affecting the PM_{10} content.

Time period /Variable	Wood fuel burning	Energy index	Studded tyres	Traffic content	Meteorological index	Latitude
Month 1-2	17	6	15	29	17	16
Month 3-4	10	4	40	32	12	2
Month 5-6	20	20	0	30	24	6
Month 7-8	4	4	0	46	40	6
Month 9-10	8	22	0	38	27	5
Month 11-12	4	19	20	25	21	11

3.3 $PM_{2.5}$ concentration calculations

Similar to PM_{10} the $PM_{2.5}$ concentrations were calculated based on i) regional background levels and ii) local source contributions to the urban background concentrations. 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 by using a ratio relation between monitored $PM_{2.5}/PM_{10}$ on a yearly basis (data from www.ivl.se). This is somewhat rough, since the ratio is likely to vary with season, but as the available monitoring data was very limited it was not possible to adjust for this.

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 calculated based on monitoring data; for regional background, central urban background and suburban background (a mean between the two others) conditions (Table 4).

Table 4 Calculated ratios applied for different types of surroundings, based on monitoring data.

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

The different ratios in Table 4 were allocated to different city areas based on the population distribution pattern of cities. 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 11), as no similar distribution pattern was found for European conditions.

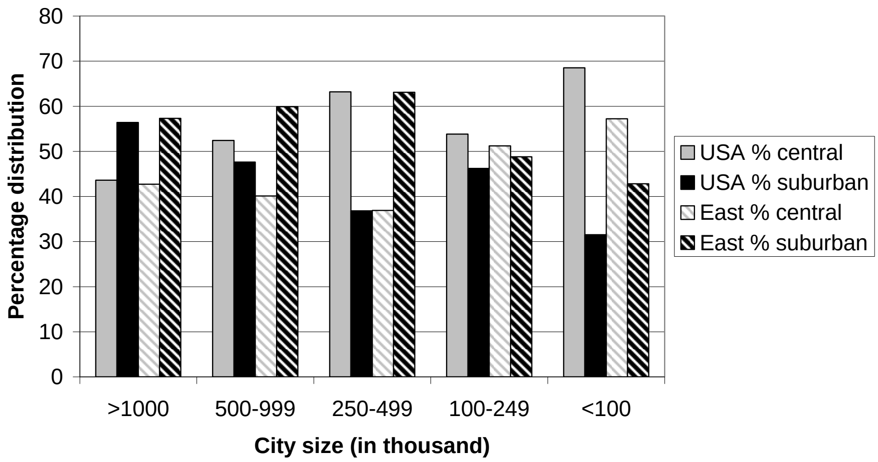

Figure 11 Relations between distribution of population in central parts and suburban parts of cities, both for all cities in the USA and for cities located in the eastern part of the USA (developed in USA by Demographia, 2000, www.demographia.com/).

The GIS-methodology applied to allocate the grid cells within each city into the different classes in Figure 11 consists of several steps: At first, the population size estimated to the central areas [pop_central] was identified (40 or 55% 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 areas, 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 60 or 45% 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 4 was applied to the PM_{10} map to calculate the $PM_{2.5}$ map.

3.4 Population distribution

The current population data applied for exposure calculations in this study were supplied by Statistics Sweden (www.scb.se). The population dataset was based on 2010 consensus, and in total, 9 546 546 inhabitants were recorded within the Swedish borders. The population data had a resolution of 100 x 100 m, but for the purpose of this study it was aggregated into a 1 x 1 km grid resolution.

3.5 Exposure calculation

The distribution of the NO_2, PM_{10} and $PM_{2.5}$ concentrations in the urban areas was added to the maps of the background concentration levels to arrive at the final concentration maps. The number of people exposed to different levels of NO_2, PM_{10} and $PM_{2.5}$ concentrations were then 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.

3.6 Health impact assessment (HIA)

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:

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

where y_0 is the baseline rate, pop is the affected number of persons; ß is the exposure-response function (relative risk per change in concentration), and x is the estimated excess exposure.

The number of Years of Life Lost (YoLL) and decrease of life expectancy was calculated using "life-tables" methodology, where the hypothetical life expectancy is compared with the life expectancy affected by air pollution. The calculation of YoLL and changes in life expectancy were facilitated by a WHO Centre for Environment and Health developed software AirQ 2.2.3 (Air Quality Health Impact Assessment Tool) (WHO, 2004).

3.6.1 Exposure-response functions (ERFs) for mortality

It has long been recognized that particle concentrations correlate with mortality, both temporally (short-term fluctuations) and spatially based on mortality and survival (WHO, 2003; WHO, 2006a).

The recent WHO Review of evidence on health aspects of air pollution, REVIHAAP, (WHO, 2013a), concludes that recent long-term studies are showing associations between PM and mortality at levels well below the current annual WHO air quality guideline level for $PM_{2.5}$ (10 $\mu g/m^3$). The WHO expert panel thus concluded that for Europe it is reasonable to use linear exposure-response functions, at least for particles and all-cause mortality, and to assume that any reduction in exposure will have benefits. The findings from REVIHAAP are used as a basis for the WHO Project Health risks of air pollution in Europe – HRAPIE (WHO, 2013b).

The REVIHAAP report also concludes that more studies have now been published showing associations between long-term exposure to NO_2 and mortality (WHO, 2013a). This observation makes the situation a bit more complicated when it comes to impact assessments for vehicle exhaust particles, where the close correlation between long-term concentrations of NO_2 and exhaust particles may be confounding (both pollutants cause similar disease and overestimation might appear) in epidemiological studies.

The potential confounding problem in studies of effects from NO_2 and $PM_{2.5}$ on mortality was dealt with in a recent review paper focusing on 19 epidemiological long-term studies of mortality using both pollutants as exposure variable (Faustini et al, 2014). In their analysis, studies with bipollutant analyses ($PM_{2.5}$ and NO_2) in the same models showed decrease in the effect estimates of NO_2, but still suggesting partly independent effects. The greatest effect on natural or total mortality was observed in Europe for both NO_2 and $PM_{2.5}$. In Europe, there was a 7% increase in total mortality for both NO_2 and fine particles, the relative risk (RR) for NO_2 was 1.066 (95% CI 1.029-1.104) per 10 $\mu g/m^3$ and RR for $PM_{2.5}$ was 1.071 (95% CI 1.021-1.124) per 10 $\mu g/m^3$.

One relevant study of NO_2 and mortality not included in the meta-analysis followed up 52 061 participants in a Danish cohort for mortality from enrollment in 1993–1997 through 2009, traced their residential addresses from 1971 onwards and used dispersion-modelled concentration of nitrogen dioxide (NO_2) since 1971 to estimate mortality rate ratios with adjustment for potential confounders (Raaschou-Nielsen et al, 2012). The mean NO_2 concentration at the residences of all participants after 1971 was 16.9 $\mu g/m^3$ (median 15.1 $\mu g/m^3$). The modelled NO_2 concentration at home was associated with a RR of 1.08 (95% CI 1.01–1.14%) higher all-cause mortality per 10 $\mu g/m^3$.

For the WHO HRAPIE impact assessment (WHO, 2013b) it was for long-term exposure to $PM_{2.5}$ and all cause (natural) mortality in ages 30+ recommended to use the exposure-response function from 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 $\mu g/m^3$. This is a coefficient very close to the long-term effect on mortality of $PM_{2.5}$ from the American Cancer Society Cohort Study (Pope et al., 1995) reported to be 1.06 per 10 $\mu g/m^3$ increment of the annual average $PM_{2.5}$. This assumption, 6% per 10 $\mu g/m^3$, has been used in many health impact assessments including our previous report (Sjöberg et al, 2007).

There is now within the research community a focus on the different types of particles and a reasoning that it is likely that their impacts on mortality differ (WHO, 2007, WHO 2013a). However, a common view is that current knowledge does not allow precise quantification of the health effects of PM emissions from different sources; "Thus current risk assessment practices should consider particles of different sizes, from different sources and with different composition as equally hazardous to health" (WHO, 2007). The practice has also been to treat both PM_{10} and the fine fraction $PM_{2.5}$ (quite often considered to be more detrimental to health than the coarse fraction of PM_{10}) as being equally toxic by mass, irrespective of the origin. This means that it has been common to convert exposure-response functions obtained using urban background $PM_{2.5}$ as the exposure indicator to be used for PM_{10}. The factor used has often been their mass relation, but in the new impact assessment HRAPIE no such conversion is recommended for PM_{10} and mortality.

Usually the short-term associations are seen as included in the long-term effects when the number of excess deaths is estimated. In addition, the potential years of life lost (PYLL or YoLL) due to excess mortality can only be directly calculated from the long-term (cohort) studies. However, because of the different sources it is likely that there in addition to the effects of background $PM_{2.5}$ is a short-term effect on mortality of road dust and coarse wear particles measured as PM_{10} (Meister et al., 2012). In this study from Stockholm, the estimated short-term (lag01) RR was 1.017 per 10 $\mu g/m^3$ increase (95% CI 1.002-1.032), with a somewhat smaller effect for $PM_{2.5}$, RR was 1.015 (95% CI 1.007-1,028) per 10 $\mu g/m^3$ increase in $PM_{2.5}$.

However, for example ExternE[3] (2005) includes assumptions about the toxicity of other different types of PM, which reflect results that indicates a higher toxicity of combustion particles and especially of particles from internal combustion engines. 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 $PM_{2.5}$.

The effects of combustion-related particles have also been studied using black smoke, black carbon or elemental carbon as the exposure variable. The WHO Project REVIHAAP (WHO 2013a) recommended that black carbon should be used as exposure variable in more studies, but did not recommend it to be used for the HRAPIE impact calculations (WHO 2013b).

A recent review collected information on studies of mortality and long-term exposure to the combustion-related particle indicators (Hoek et al., 2013). The included studies used different methods, and their relation and conversion factors have been described before (Janssen et al., 2011) All-cause mortality was significantly associated with elemental carbon, the meta-analysis resulted in a RR of 1.061 per 1 $\mu g/m^3$ EC (95% CI 1.049-1.073), with highly non-significant heterogeneity of effect estimates. Most of the included studies assessed EC exposure without accounting for small-scale variation related to proximity to major roads.

The conversion from $PM_{exhaust}$ to EC is complicated. The vehicle emission model HBEFA gives the emissions of NO_X and $PM_{exhaust}$ from the vehicle fleet and this way also measurements performed 2013 by Stockholm City Environment Administration in the tunnel Söderledstunneln suggests that EC represents 30% of exhaust PM_{avgas}. Other studies have indicated similar results, and confirm that the RR for background $PM_{2.5}$ becomes too low for $PM_{exhaust}$. With the RR for EC (1.061 per 1 $\mu g/m^3$) and the assumption that 30% of $PM_{exhaust}$ is EC, the RR for $PM_{exhaust}$ would become 1.183 per 10 $\mu g/m^3$.

The calculated RR for $PM_{exhaust}$ of 1.183 per 10 $\mu g/m^3$ comes very close to a RR found for a subset of ACS subjects all from Los Angeles County (Jerrett et al, 2005). The authors extracted health data from the ACS survey for metropolitan LA on a zip code-area scale. Using kriging and multiquadric models and data from 23 state and local district monitoring stations in the LA basin they then assigned exposure estimates to 267 zip code areas with a total of 22 905 subjects. For all-cause mortality with adjustments for 44 individual confounders the RR was 1.17 (95% CI = 1.05–1.30) per 10 $\mu g/m^3$. These results suggest that the chronic health effects associated with $PM_{2.5}$ from local sources, mainly traffic and heating, is much larger than reported for metropolitan areas. The direct comparison with the ACS main results show effects that are nearly 3 times larger than in models relying on inter-community exposure contrasts.

Coarse ($PM_{10-2.5}$) and crustal particles have not been associated with long-term mortality in the cohort studies, and have often shown less evident short-term effects on mortality (Brunekreef & Forsberg, 2005; WHO, 2006b; WHO 2013a).

Selected exposure-response functions

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[3] The ExternE project (www.externe.info, ExternE 2005) is a long lasting research project funded by the European Commission's Directorate-General XII (Science, Research and Development) initiated in 1991. The main purpose of the project was to provide knowledge concerning the external costs of energy production in Europe. The first series of reports were published in 1995, with updates in 1998 and 2005.

Despite the fact that usually, as in HRAPIE (WHO, 2013b), all PM regardless of source had to be considered as having the same effect per mass concentration, we have in this study for $PM_{2.5}$ and mortality used a less conservative approach. We have chosen to assume that road dust, as mainly coarse crustal particles have a smaller effect than the typical, total bulk of particles in the cohort studies, that was largely built up by secondary particles. We also assume that primary combustion PM has a larger effect than the typical, total mix of particles.

For $PM_{2.5}$ in general we have adopted the exposure-response coefficient from HRAPIE (WHO, 2013b) coming from the meta-analysis by Hoek et al. (2013), assuming the RR to be 1.062 (95% CI 1.040-1.083) per 10 µg/m3.

Primary combustion particles from motor vehicles and domestic heating are found in the fine fraction ($PM_{2.5}$). Acknowledging the indications of a stronger effect of such particles, we have in this study applied the exposure-response coefficient 17% per 10 µg/m^3 from the intraurban Los Angeles analysis of ACS data (Jerrett et al., 2005). This RR is also supported by the calculations using results for EC.

More than half of road dust PM is in the coarse fraction. Since there is in principle no evidence from the cohort studies for an effect of coarse particles ($PM_{2.5-10}$) on mortality, and weak support for any effect of the crustal fraction, road dust will here be assumed to only have a short-term effect on mortality on the scale that PM_{10} has in general. Since the study of coarse PM and road dust from Stockholm (Meister et al., 2012) indicated very similar effects for the coarse and fine fraction, we will for road dust (PM_{10}) use the RR for the coarse fraction, 1.017 per 10 µg/m^3 increase (95% CI 1.002-1.032).

It is not clear if the association between long-term concentrations of NO_2 and mortality are driven by NO_2 itself, or partly related to correlated exposure to exhaust particles. When we estimate the effects on mortality from NO_2 exposure, we select the results from Denmark presented by Raaschou-Nielsen et al. (2012) with a RR of 1.08 per 10 µg/m^3 (95% CI 1.01–1.14%) for all-cause mortality.

3.6.2 Exposure-response functions (ERFs) for morbidity

In the recent work on the Thematic Strategy on Air Pollution of European Union about 50 different strategy options are compared, each requiring substantial modelling effort (WHO, 2013b). For practical reasons the number of recommended exposure-response functions must be kept to a minimum. For a national health impact assessment, as for cost-benefit analyses, the set of exposure-response functions selected should be more complete. For morbidity we have in this study included only some of the potentially available health endpoints to be selected. We have decided to include some important and commonly used endpoints that allow comparisons with other health impact assessments and health cost studies.

3.6.2.1 ERF for hospital admissions

The meta-coefficients for PM_{10} and hospital admissions have been described for the European APHEIS and APHEKOM Projects, and are used in the APHEKOM HIA (Pascal et al., 2013). The exposure-response function (ERF) for the short-term effect of PM_{10} on total respiratory hospital admissions has been estimated to 1.0062 per 10 µg/m^3 (95% CI 1.0114-1.0167) and for total cardiovascular hospitalizations 1.003 per 10 µg/m^3 (95% CI 1.006-1.009).

In the HRAPIE Project, the short-term effects are described for $PM_{2.5}$ based on a meta-analysis performed using the UK APED database (WHO, 2013b). For all respiratory hospital admissions the RR was 1.0190 per 10 µg/m³ $PM_{2.5}$ (95% CI 0.9982–1.0402), and for cardiovascular hospital admissions the RR was 1.0091 per 10 µg/m³ $PM_{2.5}$ (95% CI 1.0017–1.0166).

The HRAPIE Project also gives the short-term effects on respiratory hospital admissions for NO_2 based on a meta-analysis performed using the UK APED database (WHO, 2013b), the RR was 1.0180 per 10 µg/m³ NO_2 24h mean (95% CI 1.0115–1.0245).

3.6.2.2 Exposure-response function for chronic bronchitis

There is very limited data regarding new cases of chronic bronchitis and long-term exposure to PM. The Seventh Day Adventist Study (AHSMOG: Adventist Health Smog; Abbey, 1999) conducted in the US examined people on two occasions approximately 10 years apart, in 1977 and again in 1987-88. In this study chronic bronchitis was defined with the common definition of reporting chronic cough or sputum on most days, for at least three months of the year, for at least two years. New cases were defined as those which met the criteria at the follow up in 1987-88 but not when included in 1977. Assuming the RR from Abbey at al. (1995) and a background incidence rate (adjusted for remission of chronic bronchitis symptoms) of 0.378% estimated from Abbey et al. (1993, 1995), Hurley et al. (2005) for the CAFE programme has derived an estimated exposure-response function for new cases of chronic bronchitis in the population aged 27 years or older of 26.5 (95% CI -1.9-54.1) per 10 µg/m³ PM_{10} per year per 100 000 adults, or 0.0000265 new cases for a change of 1 µg/m³*person and year. Airport visibility data was used to estimate $PM_{2.5}$ (Abbey et al, 1995) resulting in 14% (95% CI = −0.45–26.2%) change in new cases per 10 µg/m³ of $PM_{2.5}$. The HRAPIE (WHO, 2013b) preferred this second calculation finding it much more uncertain to estimate PM_{10} in multiple cities from total suspended particles than in directly estimating $PM_{2.5}$ from airport visibility in each city where good model fits were obtained.

A Swiss study examined relationships between chronic bronchitis and the change in modelled concentrations of PM_{10} in the residence in about 7000 adults aged 16–60 years, at first survey residing in eight communities in Switzerland (Schindler et al, 2009). This study estimated an odds ratio of 0.78 (95% CI = 0.62–0.98), equivalent to a decrease of risk of new reports of chronic bronchitis by 22% (95% CI = 2–38%) per 10 µg/m³ decrease in PM_{10}.

In HRAPIE (WHO, 2013b) the results of the Abbey et al. (1995) study was converted to PM_{10} units (assuming $PM_{2.5}/PM_{10}$ = 0.65) and using an inverse-variance weighted average of that study with the results of the Schindler et al. (2009) study resulted in an RR for chronic bronchitis of 1.117 (95% CI = 1.040, 1.189) per 10 µg/m³ PM_{10}.

3.6.2.3 Exposure-response function for restricted activity days

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 throughout 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 for a change of 1 µg/m³*person and year. According to the experts in HRAPIE similar or greater effects of PM should be expected in older and younger persons (WHO, 2013b).

For PM_{10} the relation used in APHEIS (Medina et al., 2004) for long-term effects on mortality was RR for $PM_{10} \approx 0.72$ RR for $PM_{2.5}$), that means for PM_{10} close to 0.065 RADs for a change of 1 µg/m³*person and year.

3.6.3 Selected base-line rates for mortality and morbidity

In order to estimate how many deaths and hospital admissions that depend on elevated air pollution exposure we need to use a base-line rate. We collected the base-line rates 2010 for Sweden from the Swedish National Board of Health and Welfare.

For mortality we calculated the following rates: total mortality (all causes) all ages: 965.2 per 100 000 persons; external causes, all ages: 49.7 per 100 000 persons; total mortality in age group 30+: 1489.8; external causes in age group 30+: 69.4.

For hospital admissions all ages we calculated for cardiovascular disease 2540.1 per 100 000 persons, and for respiratory disease 1066.2 per 100 000 persons. We adjusted these for planned admissions to have the rates for emergency hospital admissions for cardiovascular disease 2014 per 100 000 persons, and for respiratory disease 987 per 100 000 persons.

As we have no country-specific baseline rates for chronic bronchitis, we used HRAPIE experts (WHO, 2013b) recommended estimates based on AHSMOG, SAPALDIA and Schindler et al. (2009) studies. We applied 3.9 incident cases annually per 1000 adults at risk who reported symptoms at the second survey but not at baseline. For RADs we also have no relevant Swedish data and rely on the commonly used 0.092 RADs for a change of 1 µg/m³*person and year $PM_{2.5}$ (for PM_{10} corresponding to 0.065 RADs for a change of 1 µg/m³*person and year).

3.6.4 Health impact scenarios and applied Exposure-response functions (ERF)

1. <u>NO_2 and all-cause mortality</u>

a) NO_2 (possibly as an indicator also of pollutants) long-term effect on all-cause mortality in ages 30+ as distributed 2010 with 2010 baseline mortality. We apply the Risk Ratio (RR) from Denmark (Raaschou-Nielsen et al., 2012) of 1.08 per 10 µg/m³ (95% CI 1.01–1.14%) for all-cause mortality. Since the smooth risk slope is steep at low concentrations we use both a low 5 µg/m³ threshold and a no threshold scenario.

b) As a comparison over time for all ages we use the relative concentration distribution estimated for 2005 but with the 2010 population and baseline mortality.

2. $PM_{2.5}$ and all-cause mortality

a) $PM_{2.5}$ long-term effect on all-cause mortality in ages 30+ as distributed 2010 with 2010 baseline mortality. We assume the same effect (same RR) for all sources and use a low 2 µg/m³ threshold. We apply the RR from HRAPIE (WHO, 2013b) 1.062 (95% CI 1.040-1.083) per 10 µg/m³.

b) As a comparison over time for all ages we use the relative concentration distribution estimated for 2005 but with the 2010 population and baseline mortality.

3. Impacts of particulate matter from specific sources

Mortality
For a more detailed assessment of PM_{10}, including the fine fraction ($PM_{2.5}$), we assume that the effects on all-cause mortality are different for different types of particles.

For combustion generated particles, modelled as vehicle exhaust and wood smoke, we assume the total mass to be in the fine fraction, and apply the RR 1.17 (95% CI = 1.05–1.30) per 10 µg/m³ from the intraurban Los Angeles analysis of ACS data (Jerrett et al, 2005) in the age group 30+ years. This RR is for $PM_{exhaust}$ also supported by our calculations using results for EC. We assume no threshold.

For road dust PM_{10}, dominated by the coarse fraction, we assume a short-term effect on daily number of deaths in all ages, and apply the RR for the coarse fraction 1.017 per 10 µg/m³ increase (95% CI 1.002-1.032) observed in Stockholm (Meister et al., 2012). We assume no threshold, and as the study included all ages we assume the association to occur for all ages.

For the rest of $PM_{2.5}$ (not originating from vehicle exhaust, wood smoke or road dust) we apply the RR from HRAPIE (WHO, 2013b) 1.062 (95% CI 1.040-1.083) per 10 µg/m³ in the age group 30+ years. For these non-local sources we assume a low threshold of 2 µg/m³ of $PM_{2.5}$.

Morbidity
We assume $PM_{2.5}$ to have an impact on RADs both in the age group 15-64 years, and in the rest of the population, but have to assume the same effect from all sources. The effect is assumed to be 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.

We assume PM_{10}, $PM_{2.5}$ and NO_2 to have an effect on hospital admissions and chronic bronchitis 30+, but have to assume the same effect from all sources. From HRAPIE (WHO, 2013b) we applied the following RRs: for all respiratory hospital admissions the RRs were 1.0190 per 10 µg/m³ $PM_{2.5}$ (95% CI 0.9982–1.0402) and 1.0180 per 10 µg/m³ NO_2 (95% CI 1.0115–1.0245), and for cardiovascular hospital admissions the RR was 1.0091 per 10 µg/m³ $PM_{2.5}$ (95% CI 1.0017–1.0166). RR for chronic bronchitis of 1.117 (95% CI = 1.040, 1.189) per 10 µg/m³ PM_{10}. For chronic bronchitis and PM_{10}, we assume a low threshold of 3 µg/m³.

3.7 Socio-economic valuation

The method used for socio-economic valuation in this study is identical to the method used in Sjöberg et al. (2007). In this report, relevant updates on health end points are presented. There are two additions to the valuation performed in Sjöberg et al. (2007), valuation of Chronic Bronchitis (CB) and Restricted Activity Days (RAD).

In Sjöberg et al. (2007), most central estimates are based on the central values from the 2005 update of the ExternE project. Estimates in this report are from New Energy Externalities Developments for Sustainability (NEEDS, 2007), which is a follow up to ExternE. However, in NEEDS (2007) most central estimates are the same as in ExternE (2005) apart from the update of life lost (VOLY) and Chronic bronchitis.

3.7.1 Morbidity

In the ExternE update from 2005 (www.externe.info) it is recognised that the suitable valuation of health related effects consists of three components; Resource costs (costs for medical aid), Opportunity costs (loss in productivity) and Disutility (costs for discomfort etc).

These costs correspond to welfare parameters of relevance for valuation of health effects related to air pollution. They allow for consideration of all economic decision makers in society; individuals (households), firms and government. Ideally, all these costs should be taken under consideration during the valuation of health effects, but it is sometimes difficult to measure and calculate reliable estimates of these cost parameters. It can also be that some methods of valuation aggregate the above mentioned parameters thereby making it difficult to distinguish between the different types of costs.

The health effects related to high concentrations of NO_2 that is of interest for the economic valuation performed in this study is Chronic Bronchitis (CB).

The health effects related to high concentrations of $PM_{2.5}$ that are of interest for the economic valuation are: Chronic Bronchitis, Cardiovascular Hospital Admissions (CVA), Respiratory Hospital Admissions (RHA) and Restricted Activity Days (RAD). The methods for valuation of CVA and RHA are discussed in Sjöberg et al. (2007) and will not be further covered in this report.

3.7.2 Quantified results from the literature

OECD (2006) summarises the recent developments in the area of Cost Benefit Analysis (CBA) and valuation and presents the results from several studies including NEEDS (2007) within the ExternE project and European Commission (2014). Results on valuation of mortality expressed as the measure Value of Statistical Life (VSL) of relevance for our study are summarised in Table 5 below.

Table 5 VSL estimates of mortality from previous studies (OECD, 2006).

	VSL [$ million]	**Currency year**
Hammit, 2000	3 – 7	1990
Alberini et al., 2004	1.5 – 4.8 (small risk reduction) 0.9 – 3.7 (large risk reduction)	2000
Krupnick et al., 1999	0.2 – 0.4	1998
Markandya et al., 2004	1.2 – 2.8 0.7 – 0.8 0.9 – 1.9	2002
Chilton et al., 2004	0.3 – 1.5	2002

The values in the table are used mainly for illustration of the most common ranges of VSL in the literature on the subject. Valuations that are based on risk contexts such as occupational risks (accidents when at work), road traffic and fires are excluded from this table. The column indicating the currency year is necessary for a potential transfer of the results to other valuation studies. The results from Alberini et al. (2004), Krupnick et al. (1999) and Markandya et al. (2004) are all results from studies on risk reductions for persons in the age class 70-80 years.

Other values of interest for our study are the valuation of mortality expressed as the measure (Value Of Life Year) VOLY estimates comparing Chilton et al (2004) with Markandya et al. (2004), Table 6.

Table 6 VOLY estimates on mortality from previous studies (OECD, 2006).

	VOLY [£]	**Currency Year**
Chilton et al., 2004	27630	2002
Markandya et al., 2004	41975	2002

The VOLY given by Markandya et al. (2004) is an indirect estimate derived from the VSL estimate in the study while the VOLY from Chilton et al. 2004 is a direct estimate.

Furthermore, OECD (2006) also indicates morbidity valuations for several different health effects given in the available literature. The values of interest for our study are given in Table 7.

Table 7 Morbidity valuation estimates (OECD, 2006).

	Study quoted		
Type of Illness (morbidity)	**Ready et al., 2004**	**ExternE, 1999**	**Maddison, 2000**
Hospital admission for treatment of respiratory disease	€ 490	€ 7870	n.a.
3 days spent in bed with respiratory illness (3 RAD)	€ 155	€ 75	€ 195

In OECD there is no suggestion as to why the ExternE values for hospital admissions is so much higher than in Ready et al. (2004). On the other hand, these ExternE values are then updated in the following update of the ExternE project, see Table 9.

The following Table 8 lists the central estimates of monetary values for health effects that are of relevance to our study as they are valued in the latest update of the ExternE project, NEEDS.

Table 8 Economic values of health effects (NEEDS, 2007).

Mortality	**Value**	**Unit**
Value of a statistical life	1052000	€$_{2000}$ /case
Value of Life year lost	40 000*	€$_{2000}$ / year
Morbidity		
Chronic Bronchitis	200 000*	€$_{2000}$ /case
Cardiovascular Hospital admission, Health Care resource costs	620**	€$_{2000}$ / day in Hospital
Respiratory Hospital admission, Health care resource costs	323	€$_{2000}$ / day in Hospital
Hospital admission, cost for absenteeism from work	82	€$_{2000}$ / day
Hospital admission, WTP for avoided hospitalisation	437***	€$_{2000}$ / occurrence
Restricted Activity Day (RAD)	46	€$_{2000}$ / day

**) Updated estimates from NEEDS, (2007)*

***) The value for Cardiovascular health care resource costs is derived through multiplying the RHA with 1.92, a multiplier provided by ExternE (2005).*

****) Hospital treatment for respiratory disease lasting three days, followed by five days at home in bed. The value is based on Ready et al. (2004) but differs from the same value given by OECD (2006) due to exchange rates and currency year used.*

The values in Table 9 are from the European Commission (2014) and serve as best available data and are used to enable a comparison with the cost estimates from NEEDS (2007).

Table 9 Economic values of health effects - European Commission valuation data, 2014.

Mortality	**Value**	**Unit**
Value of a statistical life	2 220 000	€$_{2005}$ /case
Value of Life year lost	57 700*	€$_{2005}$ / year
Morbidity		
Chronic Bronchitis 27+	53 600*	€$_{2005}$ /case
Cardiovascular Hospital admission, Health Care resource costs	2 220	€$_{2005}$ / day in Hospital
Respiratory Hospital admission, Health care resource costs	2 220	€$_{2005}$ / day in Hospital
Restricted Activity Day (RAD)	92	€$_{2005}$ / day

The valuation estimates used in our project to calculate socio-economic costs of high levels of $PM_{2.5}$ are presented in Chapter 4.5 and Table 24.

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 2010, calculated with the URBAN model, is presented in Figure 12. The result is based on calculated two-month means in order to capture the seasonal variation, where higher concentrations usually occur during winter.

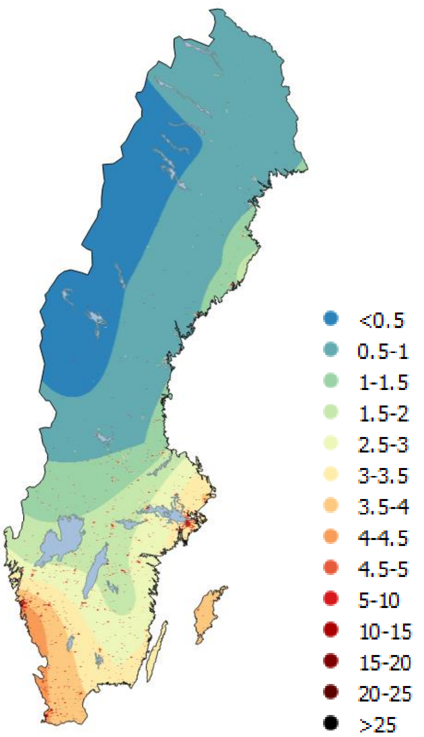

Figure 12 NO_2 concentration as annual mean for 2010 in Sweden, unit µg/m^3.

In Figure 12 it can be seen that most of the country have relatively low NO_2 urban background concentrations calculated for 2010, compared to the environmental standard for the annual mean value (40 µg/m^3). Most of the small to medium sized cities have NO_2 concentrations of less than 15 µg/m^3 in the city center. In the large cities and along the southern part of the west coast the concentration levels are higher, reaching up to 20-25 µg/m^3 in the major cities, which is of the same magnitude as the long-term environmental objective (20 µg/m^3 as an annual mean). The highest calculated urban background concentration of NO_2 occurred in the central parts of Gothenburg with concentrations approaching 40 µg/m^3. The highest calculated annual concentrations in Malmö and Stockholm were in the region of 30 µg/m^3.

There is an observable partition of Sweden north of Stockholm, possibly due to both a decreasing density of cities and meteorological factors. The location of cities determines the pattern of the concentrations, while the city size and the meteorological mixing parameters determine the concentrations levels. Thus, even in a small town high NO_2 concentrations can occur due to poor ventilation in the area.

According to the calculated results, there is only one 1 x 1 km grid (central Gothenburg) cell in all of Sweden where the annual air quality standard for NO_2 is exceeded. However, since the standards are also valid for concentrations in street canyons which normally are 1.5 times higher than the urban background (Persson and Haeger-Eugensson, 2006), there are likely additional exceedances at some "hot spots" reflected in the Air Quality Plans submitted by municipalities with violation of the limit values.

4.1.2 National distribution of PM_{10} concentrations

The annual mean concentrations of PM_{10} for 2010, calculated with the URBAN model, are presented in Figure 13. The result is based on calculated two months 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.

In Figure 13 it can be seen that the PM_{10} concentrations on a yearly basis to a large extent are governed by the regional background concentrations. There is a large latitudinal decrease to the north of the regional background concentration, due to the strong influence from the long range transport. The urban background concentrations in the largest towns in the southern and western parts of Sweden are calculated to be about 24-28 μg/m³, while the concentration in Stockholm is estimated to about 19 μg/m³. Compared to the environmental standard for the annual mean value (40 μg/m³) there are no exceedances in urban background in Swedish towns.

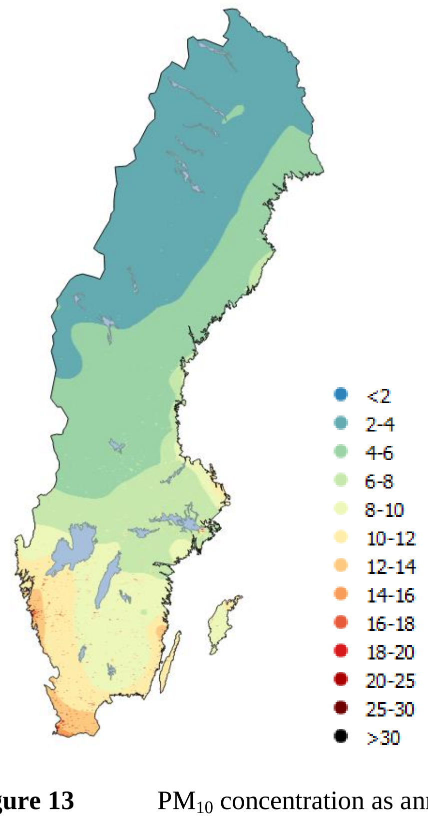

Figure 13 PM_{10} concentration as annual mean for 2010 in Sweden, unit µg/m³.

4.1.3 National distribution of $PM_{2.5}$ concentrations

The annual mean concentrations of $PM_{2.5}$ for 2010 are presented in Figure 14. The result is based on the earlier calculated PM_{10} concentrations in combination with calculated ratios based on empirical relationships of $PM_{10}/PM_{2.5}$.

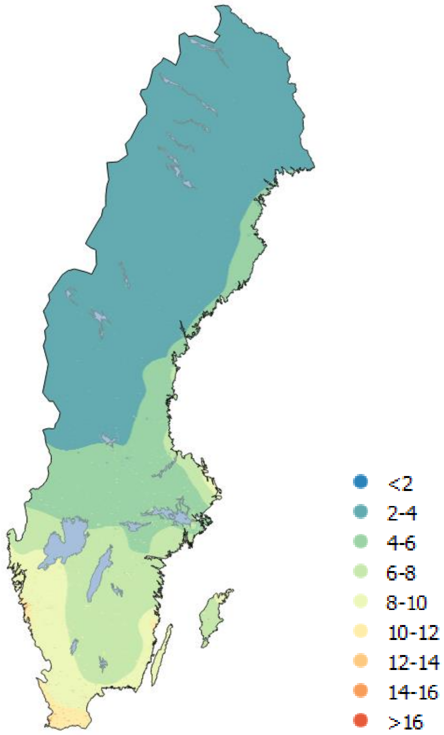

Figure 14 $PM_{2.5}$ concentration as annual mean for 2010 in Sweden, unit µg/m³.

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

As mentioned above, the highest NO_2 concentration levels will normally be found in street canyons. However, for studies of population exposure to air pollution it is custom to use the urban background air concentration levels or a relatively low geographic resolution, since this type of air quality data is used in the studies providing dose-response relationship for health impact calculations.

The number of people in Sweden exposed to certain levels of NO_2 in 2010 is shown in Table 12. The largest group of people, 45%, was exposed to annual mean concentrations of NO_2 less than 5 μg/m³. Another 32% were exposed to concentration levels between 5-10 μg NO_2/m³, and about 6% of the Swedish inhabitants experienced exposure levels of NO_2 above 15 μg/m³, se Figure 15.

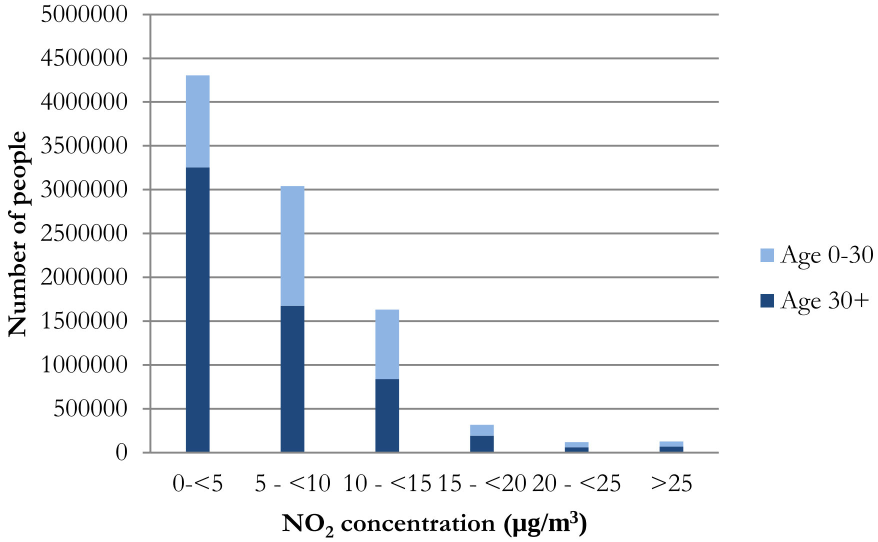

Figure 15 Number of inhabitants exposed to different NO_2 annual mean concentrations in Sweden in 2010, divided into two age categories 0 – 30 (light blue) and 30+ years of age (dark blue).

In 2010 37% of the population was of the age 0-30 years leaving 63% in the 30+ year category. However, Figure 15 show that at low concentrations 0-5 μg/m³ over 75% of the population belonged to the 30+ age class, whereas in the higher concentration almost half of the population were under 30 years of age.

4.2.2 Exposure to PM_{10}

The estimated number of people in Sweden exposed to different intervals of the total PM_{10} annual mean concentrations in 2010 is shown in Figure 16. Only 3.5% of the population was exposed to concentrations below 5 μg/m³. Approximately 70% of the population was exposed to PM_{10} concentrations between 5 and 15 μg/m³. About 10% of Swedish inhabitants were exposed to PM_{10} levels higher than 20 μg/m³. As for NO_2 the 30+ age class made up a larger percentage of the population included in low PM_{10} concentration intervals indicating that younger people are more often exposed to high concentration than the old.

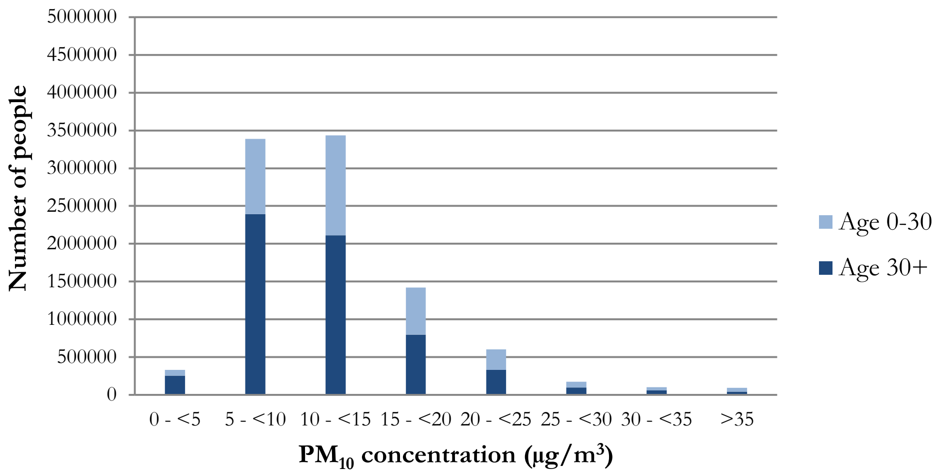

Figure 16 Number of inhabitants exposed to different PM_{10} annual mean concentrations in Sweden in 2010, divided into two age categories 0 – 30 (light blue) and 30+ years of age (dark blue).

The number of people exposed to the PM_{10} concentration contribution from the separate sources; regional background, road dust and other sources are presented in Figure 17-19. As mentioned before particles from traffic exhaust and domestic heating were assumed to all belong to the $PM_{2.5}$ fraction and is therefore presented in section 4.3.2.

Almost 40% of the Swedish population were exposed to regional background concentration of 4-6 µg/m³, in contrast to NO_2 and the total PM_{10} exposure there were no difference between age groups. Both road dust and other PM_{10} sources resulted in approximately 90% of the population being exposed to less than 4 µg/m³, and almost 29% of the total population was according to our model not exposed to negligible concentrations of road dust or other sources (Figure 18-19).

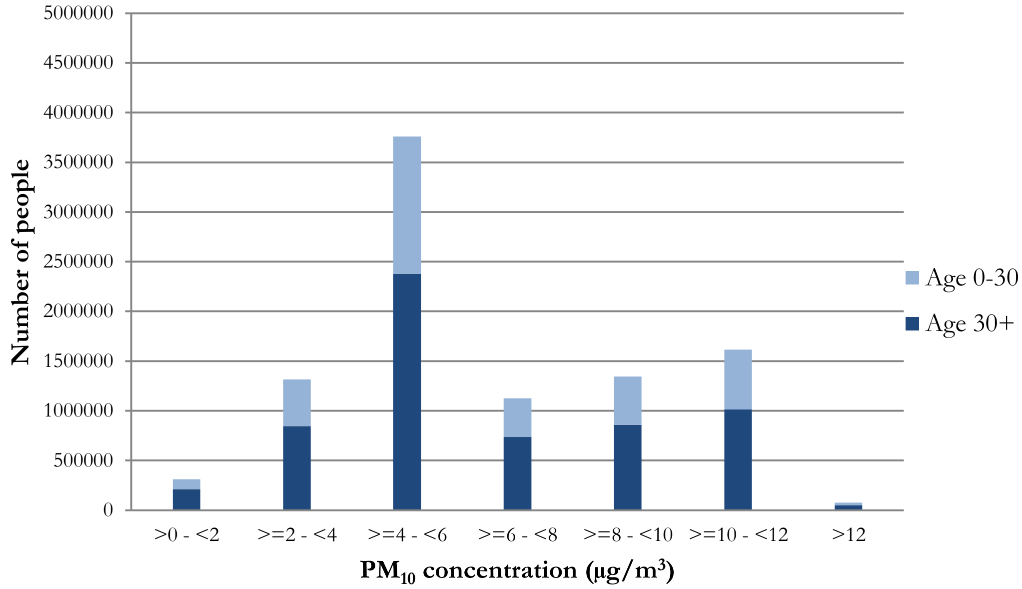

Figure 17 Number of inhabitants exposed to different PM_{10} annual mean concentrations from *regional background* in Sweden in 2010, divided into two age categories 0 – 30 (light blue) and 30+ years of age (dark blue).

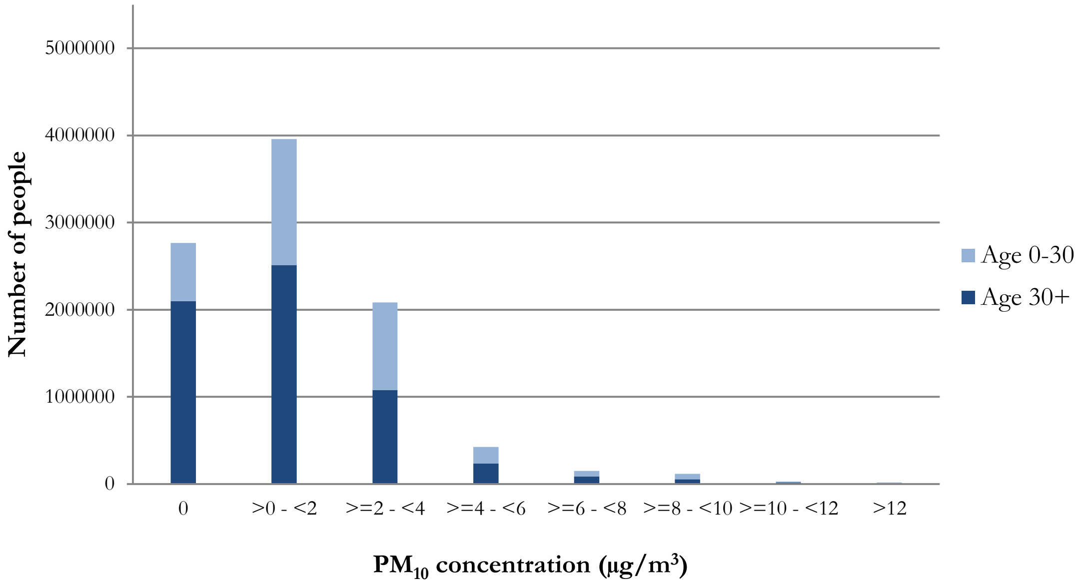

Figure 18 Number of inhabitants exposed to different PM_{10} annual mean concentrations from *road dust* in Sweden in 2010, divided into two age categories 0 – 30 (light blue) and 30+ years of age (dark blue).

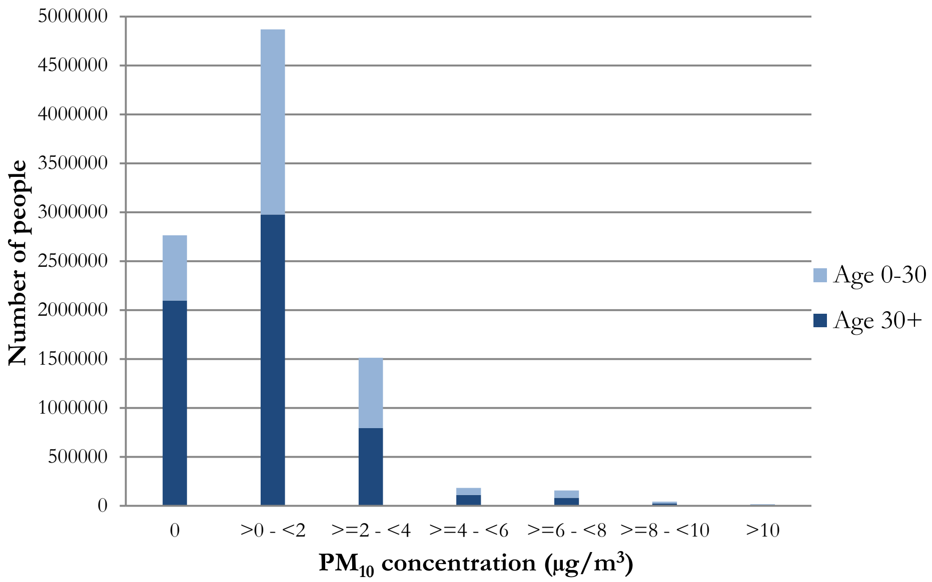

Figure 19 Number of inhabitants exposed to different PM_{10} annual mean concentrations from *other sources* in Sweden in 2010, divided into two age categories 0 – 30 (light blue) and 30+ years of age (dark blue).

4.2.3 Exposure to $PM_{2.5}$

The estimated exposure of the total annual mean concentrations of $PM_{2.5}$ is shown in Figure 20. The major part of the population, almost 72%, was exposed to $PM_{2.5}$ annual mean concentrations between 4 and 10 µg/m³. About 26% of the people in Sweden were exposed to levels between 10 and 20 µg/m³ and less than 1.5% was exposed to $PM_{2.5}$ concentrations above 20 µg/m³.

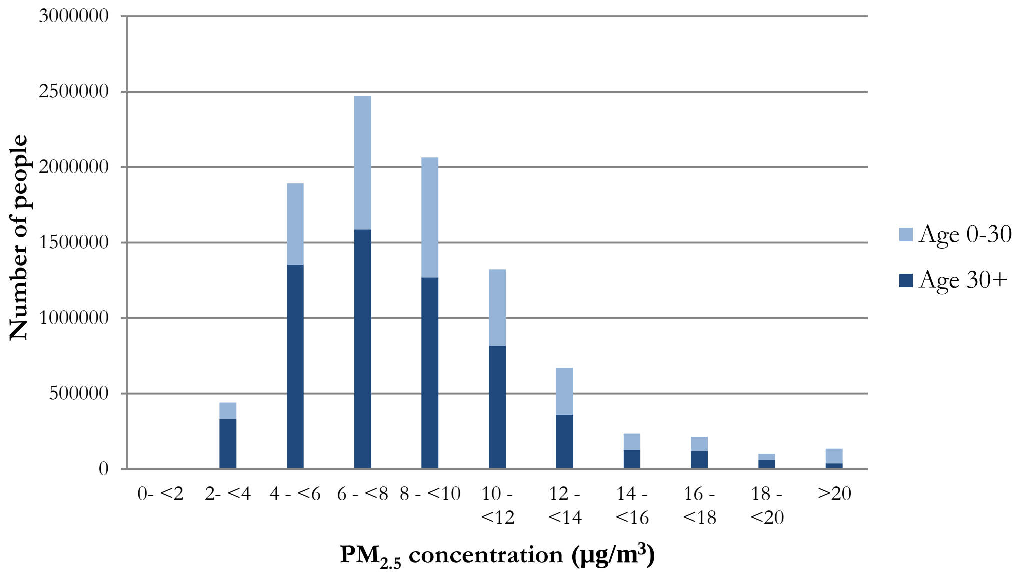

Figure 20 Number of inhabitants exposed to different $PM_{2.5}$ annual mean concentrations in Sweden in 2010, divided into two age categories 0 – 30 (light blue) and 30+ years of age (dark blue).

Figure 21 and 22 show the population exposure to $PM_{2.5}$ from traffic exhaust and domestic heating. According to the calculated contribution from traffic exhaust over 90% of the population was exposed to less than 0.5 µg/m³. As this does seem unrealistically low NO_2 concentration may be a better indicator of traffic exhaust pollution.

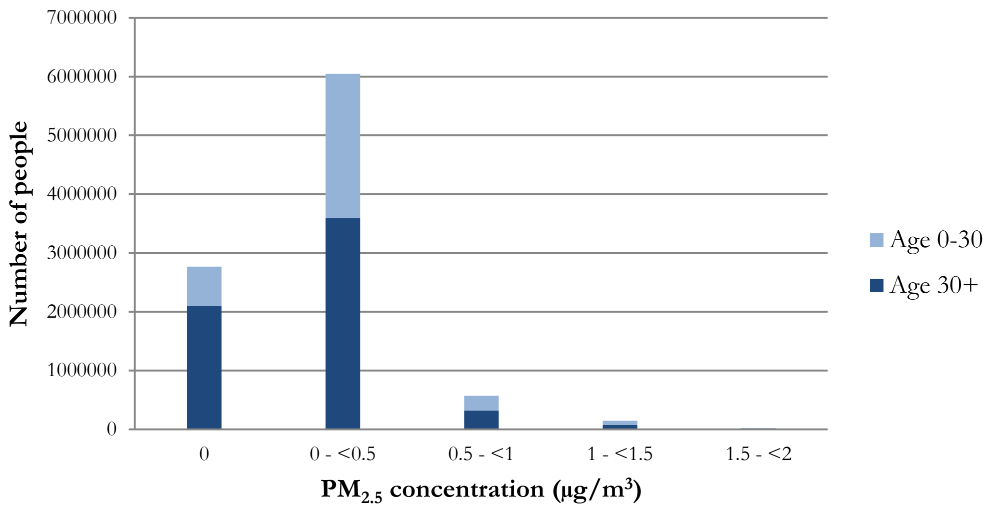

Figure 21 Number of inhabitants exposed to different $PM_{2.5}$ annual mean concentrations from *traffic exhaust* in Sweden in 2010, divided into two age categories 0 – 30 (light blue) and 30+ years of age (dark blue).

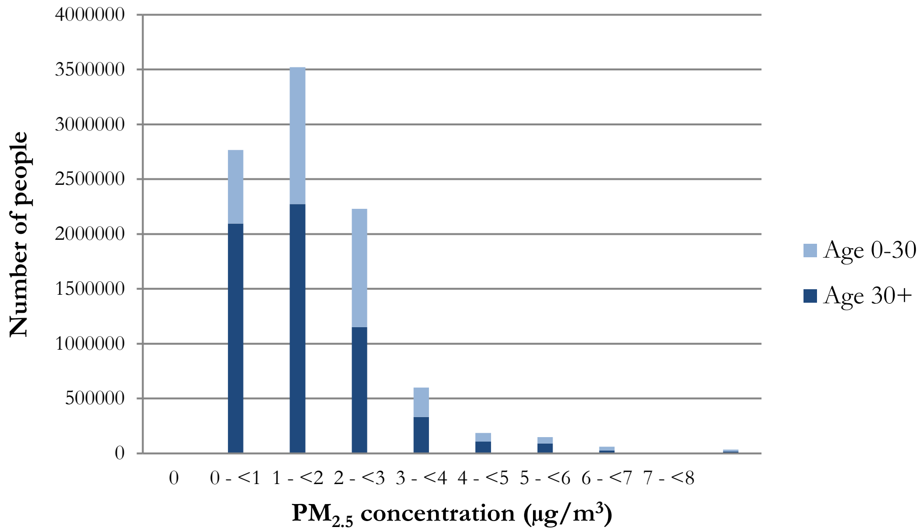

Figure 22 Distribution of exposure levels to $PM_{2.5}$ annual mean concentrations from wood related *domestic heating* in the Swedish population in 2010.

4.3 Trends in population exposure

4.3.1 NO_2

The NO_2 concentrations and the corresponding exposure situation have also been calculated for the calendar years 1990, 1995 and 1999. The numbers of monitoring sites available for the different years are given in Table 10. Since there were only a few sites of regional background measurements for 1990 new estimated concentrations, based on the geographical distribution in 1999, were developed. The relation between 1990 and 1999 at the five sites differs somewhat between the years, and therefore two-month means of the relation have been used. The calculated regional background concentrations have been evaluated with the two existing months of measurements (November and December), and the result shows that the differences in concentrations were ± 10%. However, the calculated concentrations were over-estimated more often than underestimated.

Table 10 The number of monitoring sites available for the different years.

Year	Regional background sites	Urban background sites
1990	5	62
1995	20	40
1999	73	45
2005	73	41
2010	17	41 (18)*

*18 stations with data for a complete year.

For calculation of exposure the same population density was used for all years. Figure 23 presents the extent of population exposure in different exposure classes for 1990, 1995, 1999 and 2005.

Figure 23 illustrates the percentage of the population exposed to NO_2, divided into concentration classes of 5 µg/m³, in the four studied years. An obvious trend of an increasing part of the population exposed to lower concentration levels can be observed. Compared to the situation in 1990 in 2005 about 15% less people were exposed to annual mean NO_2 levels above 15 µg/m³, while almost 20% more people were exposed to annual mean NO_2 levels in the lowest concentration class, 0-5 µg/m³.

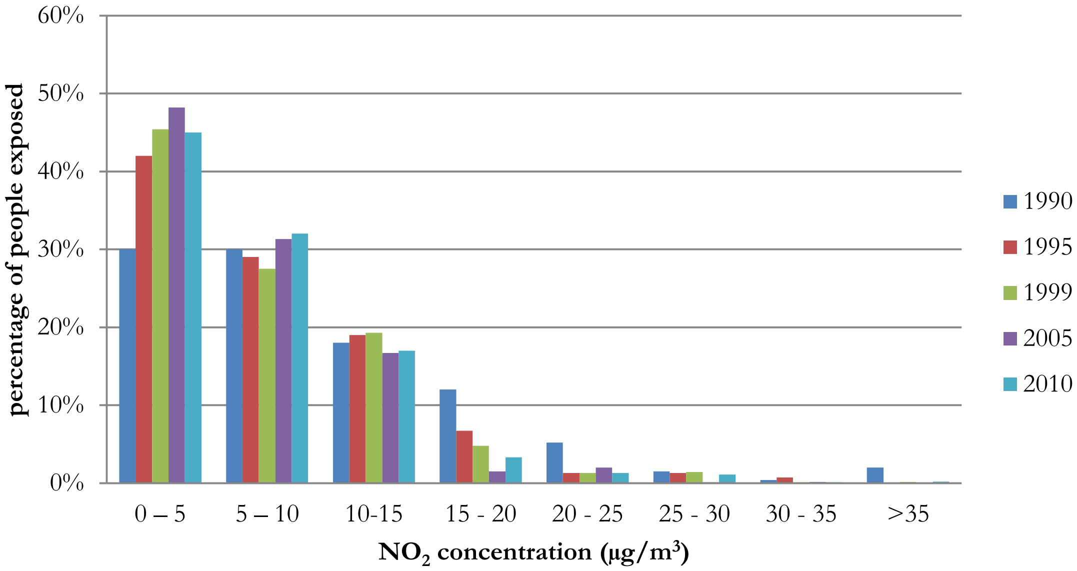

Figure 23 Percentage of the population exposed to NO_2 (µg/m³) in different annual mean concentration groups in 1990, 1995, 1999, 2005 and 2010.

4.3.2 Particles

Comparing the result from this study with the exposure data from the previous 2005 report (Sjöberg et al., 2009) a slight change in distribution can be seen, with more people exposed to concentrations of PM_{10} above 25 µg/m³ in 2010 than 2005 (Figure 24). There was almost twice as high a percentage of the population exposed to 15-20 µg/m³ in 2005 than 2010. The comparison for $PM_{2.5}$ yielded similar results to PM_{10}, see Figure 25.

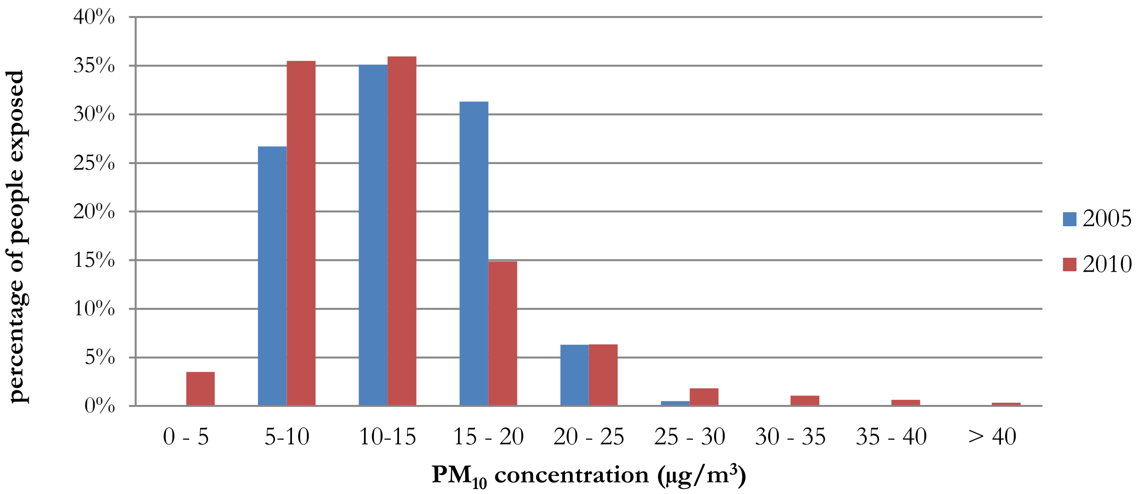

Figure 24 Percentage of the population exposed to PM_{10} (µg/m^3) in different annual mean concentration groups in 2005 and 2010.

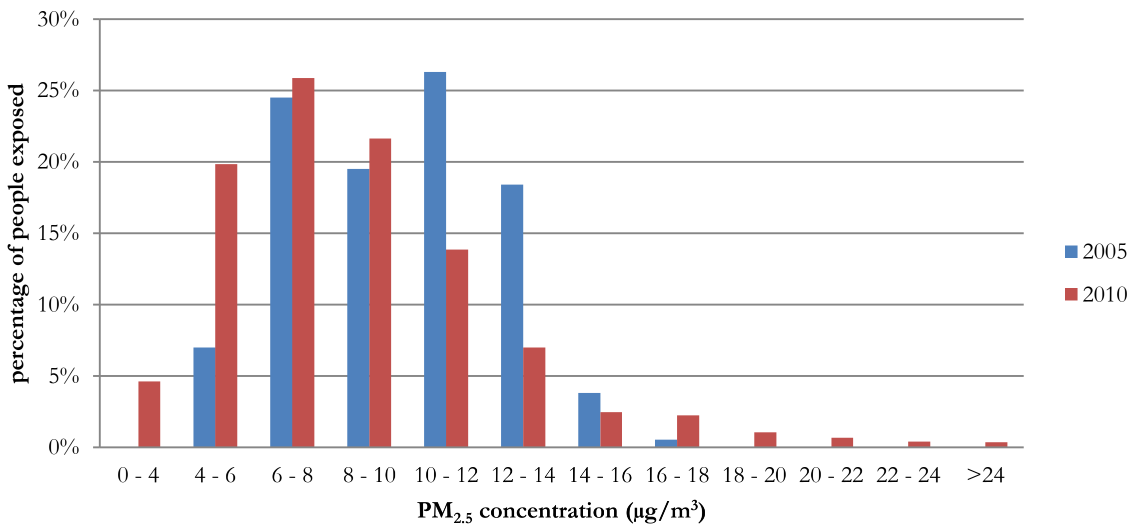

Figure 25 Percentage of the population exposed to $PM_{2.5}$ (µg/m^3) in different annual mean concentration groups in 2005 and 2010.

4.4 Estimated health impacts

4.4.1 Mortality

4.4.1.1 Effects associated with exposure to NO_2

We have estimated the excess mortality associated with exposure to NO_2 with two different cut-off values: effects starting above 5 μg/m³ and without any threshold. The more conservative scenario is assuming that all NO_2 is coming from combustion sources and is representing also other combustion products with no threshold effects. Without and with the threshold, we estimated that NO_2 exposure would be associated with 4 491 and 1 299 preterm deaths in the Swedish adult population (30+) in 2010, respectively (Table 11).

Altogether (assuming no threshold) NO_2 would cause annually 41 759 Years of Life Lost (YoLL) that would decrease on average life expectancy (LE) by 0.41 years among the adults at age 30+. Among more susceptible people who die because of air pollution, the average loss per excess death case would be 9.3 years.

Table 11 Estimated excess mortality due to exposure to total NO_2 in the Swedish adult population (age 30+) in 2010.

NO_2 concentration [μg/m³]	**Mean NO_2 [μg/m³]**	**Number of people**	**Percentage of population**	**Excess number of deaths**	
				Effects above 5 μg/m³	***Effects from 0 μg/m³***
0 - <5	2.7	3 253 108	53.4		1018
5 - <10	7.2	1 675 225	27.5	432	1429
10 - <15	11.8	839 468	13.8	677	1194
15 - <20	17.2	193 595	3.2	190	408
20 - <25	22.6	62 483	1.0	0	177
25 - <30	27.9	54 706	0.9	0	195
30 - <35	31.9	7 435	0.1	0	31
>35	40.7	0	0.0	0	40
Total	**6.2**	**6 086 020**	**100**	**1 299**	**4 491**

4.4.1.2 Effects due to exposure to particles (PM_{10} and $PM_{2.5}$)

We have estimated the excess mortality due to exposure to total $PM_{2.5}$ above 2 μg/m³ among adults at ages 30 years and older. It appeared to total $PM_{2.5}$ would cause 3 534 excess deaths in the Swedish population 30+ in 2010, respectively (Table 12). As we assume effects starting at very low levels, due to relatively low exposure, most of the effects also appear at low levels. Altogether fine particles would cause annually 34 523 Years of Life Lost (YoLL) that would decrease on average life expectancy (LE) by 0.34 years among the Swedish population.

Table 12 Estimated excess mortality due to exposure to total $PM_{2.5}$ in the Swedish adult population (age 30+) in 2010.

$PM_{2.5}$ concentration [µg/m³]	Mean $PM_{2.5}$ [µg/m³]	Number of people	Percentage of population	Excess number of deaths
0- <2		0	0	0
2- <4	3.4	329 191	5.4	40
4 - <6	5.1	1 353 155	22.2	384
6 - <8	6.9	1 587 504	26.1	714
8 - <10	9.0	1 269 257	20.8	811
10 - <12	10.9	817 579	13.4	667
12 - <14	12.9	360 094	5.9	363
14 - <16	15.0	128 159	2.1	155
16 - <18	16.7	119 037	2.0	164
18 - <20	19.2	58 719	1.0	95
20 - <22	20.6	36 553	0.6	65
22 - <24	22.7	22 017	0.4	34
>24	27.0	17 380	0.3	42
Total	**8.3**	**6 093 373**	**100**	**3 534**

Regarding the different sources, the largest proportion from specified sources would according to the model result be associated with domestic heating (1 047 excess deaths), followed by road dust (218 cases) and traffic combustion (180 cases) (Tables 13-15). These estimates are not in line with the results for NO_2, see comments in the Discussion. The category "other sources" (largely secondary PM) resulted in 3 015 excess deaths (Table 16). Regarding the combustion (originating from traffic and domestic heating) and road dust, the effects were assumed at any concentration and regarding other sources effects starting above 2 µg/m³. As we have assumed higher exposure-response function for combustion particles than for total $PM_{2.5}$, the total effect comes considerably larger: altogether 4 460 excess deaths.

Table 13 Estimated excess mortality due to exposure to PM_{10} from *road dust* in the Swedish population in 2010 (all ages).

PM_{10} concentration [µg/m³]	Mean PM_{10} [µg/m³]	Number of people	Percentage of population	Excess number of deaths
0	0.0	2 765 281	29.0	0
0 - <1	0.5	2 167 907	22.7	18
1 - <2	1.5	1 788 399	18.7	40
2 - <3	2.5	1 396 215	14.6	53
3 - <4	3.4	687 938	7.2	36
4 - <5	4.4	325 286	3.4	22
5 - <6	5.4	100 042	1.0	8
6 - <7	6.4	89 022	0.9	9
7 - <8	7.5	61 277	0.6	7
8 - <9	8.2	92 440	1.0	12
9 - <10	9.5	25 357	0.3	4
10 - <11	10.0	13 804	0.1	2
11 - <12	11.6	15 509	0.2	3
12 - <13	12.5	0	0.0	0
13 - <14	13.2	18 069	0.2	4
Total	**1.3**	**9 546 546**	**100**	**218**

Table 14 Estimated excess mortality due to exposure to $PM_{2.5}$ from *traffic combustion* in the Swedish adults (30+) population in 2010.

$PM_{2.5}$ concentration [µg/m³]	Mean $PM_{2.5}$ [µg/m³]	Number of people	Percentage of population	Excess number of deaths
0	0.0	2 096 798	34.4	0
0 - <0.5	0.1	3 595 347	59.0	111
0.5 - <1	0.6	320 144	5.3	46
1 - <1.5	1.1	73 731	1.2	20
1.5 - <2	1.7	7 353	0.1	3
Total	**0.1**	**6 093 373**	**100**	**180**

Table 15 Estimated excess mortality due to exposure to $PM_{2.5}$ from wood related *domestic heating* in the Swedish population (age 30+) in 2010.

$PM_{2.5}$ concentration [µg/m³]	Mean $PM_{2.5}$ [µg/m³]	Number of people	Percentage of population	Excess number of deaths
0	0.0	2 096 798	34.4	0
0 - <1	0.4	2 273 593	37.3	218
1 - <2	1.4	1 150 677	18.9	386
2 - <3	2.4	331 692	5.4	187
3 - <4	3.4	107 678	1.8	89
4 - <5	4.6	88 890	1.5	99
5 - <6	5.6	26 665	0.4	36
6 - <7	0.0	0	0.0	0
7 - <8	7.4	17 380	0.3	32
Total	**0.7**	**6 093 373**	**100**	**1 047**

Table 16 Estimated excess mortality due to exposure to $PM_{2.5}$ from other sources than those specified above in the Swedish population (age 30+) in 2010.

$PM_{2.5}$ concentration [µg/m³]	Mean $PM_{2.5}$ [µg/m³]	Number of people	Percentage of population	Excess number of deaths
0- <2		0	0	0
2- <4	3.4	386 210	6.3	48
4 - <6	5.2	1 769 465	29.0	508
6 - <8	6.9	1 700 991	27.9	755
8 - <10	9.0	1 190 795	19.5	761
10 - <12	10.8	707 088	11.6	570
12 - <14	12.9	205 608	3.4	208
14 - <16	14.8	106 896	1.8	128
16 - <18	17.0	18 967	0.3	27
>24	18.2	7 353	0.1	11
Total	**7.4**	**6 093 373**	**100**	**3 015**

4.4.2 Morbidity effects

4.4.2.1 Effects due to exposure to NO_2

We have estimated that the total exposure to NO_2, without assuming any threshold, would cause 1 168 respiratory hospitalisations in the Swedish population in 2010 (Table 17). Around 80% of these effects occurs at very low levels <15 µg/m³. If we compare this estimate with the results for the 2005 exposure levels (keeping other factors as population and mortality constant to 2010) there is a 3.7% decrease in respiratory hospitalizations in 2010.

Table 17 Estimated respiratory hospitalizations due to exposure to total NO_2 in the Swedish population in 2010.

NO_2 concentration [µg/m³]	Mean NO_2 [µg/m³]	Number of people	Percentage of population	Respiratory hospitalizations
0 - <5	1.5	4 306 557	45.0	211
5 - <10	7.4	3 042 142	32.0	397
10 - <15	11	1 632 355	17.0	343
15 - <20	17	318 205	3.3	99
20 - <25	23	118 828	1.3	49
25 - <30	27	96 586	1.1	49
30 - <35	32	13 804	0.1	8
>35	41	18 069	0.2	13
Total	**6.2**	**9 546 546**	**100**	**1 168**

4.4.2.2 Effects due to exposure to particles (PM_{10} and $PM_{2.5}$)

Assuming no threshold, fine particles would cause altogether 3 044 hospitalizations (Table 18). Little more than half of them (1 540 cases) appear due to respiratory causes and the other half due to cardiovascular causes (1 504 cases). If we compare this estimate with the results for 2005 exposure levels (keeping other factors as population and mortality constant to 2010) there is a 12.5% decrease in hospitalizations.

Table 18 Estimated respiratory and cardiovascular hospitalizations due to exposure to total $PM_{2.5}$ in the Swedish population in 2010.

$PM_{2.5}$ concentration [µg/m³]	Mean $PM_{2.5}$ [µg/m³]	Number of people	Percentage of population	Hospitalizations	
				Respiratory	Cardiovascular
0- <2		0	0.0	0	0
2- <4	3.4	440 902	4.6	28	27
4 - <6	5.2	1 893 640	19.8	182	178
6 - <8	7.0	2 469 830	25.9	322	315
8 - <10	9.0	2 064 335	21.6	347	339
10 - <12	10.9	1 322 759	13.9	270	263
12 - <14	12.9	668 704	7.0	162	159
14 - <16	14.9	235 331	2.5	66	65
16 - <18	16.7	214 313	2.2	67	66
18 - <20	19.3	101 134	1.1	37	36
>20	22.8	135 598	1.4	59	57
Total	**8.6**	**9 546 546**	**100**	**1 540**	**1 504**

Furthermore, particles would cause altogether 4 576 041 RADs (restricted activity days) among people in working age (16-64) (Table 19) using $PM_{2.5}$ and no threshold. The largest total effect can here be seen in levels around 10 µg/m³.

If we compare this total estimate with the results for 2005 exposure levels (keeping other factors as population and mortality constant to 2010) there is a little more than 10% decrease in the number of RADs.

Table 19 Estimated RADs due to exposure to total $PM_{2.5}$ in the Swedish population (16-64) in 2010.

$PM_{2.5}$ concentration [µg/m³]	Mean $PM_{2.5}$ [µg/m³]	Number of people	Percentage of population	RADs
0- <2		0	0.0	0
2- <4	3.4	292 773	4.9	90 488
4 - <6	5.1	1 309 525	22.0	620 292
6 - <8	6.9	1 557 601	26.1	995 076
8 - <10	9.0	1 236 564	20.7	1 021 588
10 - <12	10.9	813 448	13.6	812 750
12 - <14	12.9	371 598	6.2	440 838
14 - <16	15.0	135 245	2.3	186 560
16 - <18	16.7	133 504	2.2	205 082
18 - <20	19.2	71 060	1.2	125 599
>20	20.6	40 960	0.7	77 769
Total	**8.3**	**5 962 278**	**100**	**4 576 041**

We have also estimated that total PM_{10} at levels above 3 µg/m³ would cause 2 155 chronic bronchitis incidences in the Swedish adult population in 2010 (Table 20). Most of these effects appear at levels 5 - 20 µg/m³. If we compare this estimate with the results for the 2005 exposure levels (keeping other factors as population and mortality constant to 2010) there is a 12.2% decrease.

Table 20 Estimated chronic bronchitis incidence cases due to exposure to total PM_{10} in the Swedish adult population (30+) in 2010.

$PM_{2.5}$ concentration [µg/m³]	Mean $PM_{2.5}$ [µg/m³]	Number of people	Percentage of population	Chronic bronchitis incidence cases
0- <3		0	0.0	0
3 - <5	4.2	254 347	4.2	14
5 - <10	7.8	2 393 027	39.3	508
10 - <15	12.2	2 112 356	34.7	886
15 - <20	17.0	797 975	13.1	441
20 - <25	22.1	332 113	5.5	158
25 - <30	27.0	98 636	1.6	60
30 - <35	32.6	60 874	1.0	46
35 - <40	37.3	26 665	0.4	24
>40	44.8	17 380	0.3	19
Total	**12.0**	**6 093 373**	**100**	**2 155**

4.5 Socio-economic costs

4.5.1 Results of socio-economic valuation

Central Estimate, NO_2

The social costs in Sweden caused by health effects linked to air concentration levels of NO_2 are estimated by adapting the socio-economic values from available literature. These values are applied to the number of occurrences of the considered health effects as estimated in this study. We have estimated the socio-economic costs from exposure to NO_2 for effects starting above 5 µg/m³ and without any threshold (Table 21).

Table 21 Annual Socio-economic costs of high long term NO_2 levels without cut-off levels in Sweden, 2010 - Central Estimate.

	Socio-economic cost of health Effect [million SEK_{2010} / case]	**Health effects from *0 µg/m³***	**Socio-economic cost [million SEK_{2010}]**
Total Sweden			**24 637**
Out of which:			
Value of prevented fatality (VSL/VPF) (11 years of prolonged life)	5.48	4 491	24 607
Hospitalization, generic (respiration)	0.03	1 168	30

As shown the total annual socio-economic costs related to NO_2 without a threshold amount to approximately 24.5 billion SEK_{2010} and an absolute majority of these costs relate to loss of life years.

In these calculations all the estimates from the literature are recalculated into SEK at the value in 2010. This is done by adjusting the currencies with respect to Consumer Price Indices (CPI) and Purchase Power Parity (PPP). CPI is used to adjust the values from 2005 to 2010 values while PPP is used to adjust for national differences.

The updated VSL value (NEEDS, 2007) is lower than the former (ExternE, 2005), which is due to different valuation techniques. NEEDS based the valuation on loss of life expectancy (LE) whereas ExternE on the number of premature deaths. A VSL value of 5 500 000 SEK_{2010} is used for the valuation in this report; which is lower than other common estimates of VSL. This is mainly an effect of the adjustment of the VSL value from the expected life loss of 11 years, as is the estimate in our study. The normal number of years lost when estimating a VSL value is around 40.

The VSL estimate in our central estimate (i.e. 5 500 000) is not corrected for the respondents' time preferences (i.e. a present bias where future costs are valued less than costs taken today) since the origin of the value specifically indicates that annual payments should be made over 10 years, thereby inducing discounted values given by the respondents.

Table 22 presents the socio-economic costs of health effects from the European Commission (2014). These values serve as best available European Commission estimates and are used to enable a comparison with the costs from NEEDS 2007.

The total annual socio-economic costs related to NO_2, without a threshold, amount to approximately 35.5 billion SEK_{2010}, when valued with the economic values used by the European Commission. The difference in estimated socio-economic costs of health effects between the socio-economic costs from NEEDS and the European Commission data can to a high extent be explained by the use of different data.

Table 22 European Commission Valuation data (2014) of high long term NO_2 levels without cut-off, 2010 – Central Estimate.

	Socio-economic cost of health Effect [million SEK_{2010} / case]	**Health effects from** *$0\ \mu g/m^3$*	**Socio-economic cost [million SEK_{2010}]**
Total Sweden			**35 528**
Out of which:			
Value of prevented fatality (VSL/VPF) (11 years of prolonged life)	7.90	**4 491**	35 496
Hospitalisation, generic (respiration)	0.03	**1 168**	32

When we assume that there are no effects below 5 µg/m3, the socio-economic costs related to NO_2 is approximately 7 billion SEK_{2010} (Table 23). This is less than a third of the socio-economic cost compared to NO_2 without a threshold. This value corresponds to the health effects from local vehicle exhaust emissions in Sweden.

Table 23 Annual Socioeconomic costs of high long term NO_2 levels with cut-off values above 5 µg/m^3 levels in Sweden, 2010 - Central Estimate. The values correspond to the contribution from locally generated vehicle exhaust emissions.

	Socio-economic cost of health Effect [million SEK$_{2010}$ / case]	**Health effects above *5 µg/m³***	**Socio-economic cost [million SEK$_{2010}$]**
Total Sweden			**7 149**
Out of which:			
Value of prevented fatality (VSL/VPF) (11 years of prolonged life)	5.48	**1 299**	7 118
Hospitalisation, generic (respiration)	0.03	**1 168**	32

As shown the socio-economic data from the European commission is slightly higher than the estimates from NEEDS and thus also lead to higher socio-economic costs. The socio-economic costs due to exposure to NO_2 for effects starting above 5 µg/m^3 are estimated to about 10.5 billion SEK$_{2010}$ (Table 24) when using the same valuation data as the European Commission.

Table 24 European Commission Valuation data (2014) of high long term NO_2 levels with cut-off values above 5 µg/m^3, 2010 - Central Estimate. The values correspond to the contribution from locally generated vehicle exhaust emissions.

	Socio-economic cost of health Effect [million SEK$_{2010}$ / case]	**Health effects above *5 µg/m³***	**Socio-economic cost [million SEK$_{2010}$]**
Total Sweden			**10 299**
Out of which:			
Value of prevented fatality (VSL/VPF) (11 years of prolonged life)	7.90	**1 299**	10 267
Hospitalisation, generic (respiration)	0.03	**1 168**	32

Central Estimate, PM-sources (Road dust, domestic combustion, other sources)

The social costs in Sweden caused by health effects linked to high annual ambient air concentration levels of $PM_{2.5}$ are estimated by adapting the socio-economic values from available literature. These are applied to the number of occurrences of the considered health effects as estimated in this study, see Table 25. In the central estimate on socio-economic costs calculated in this project, health effects are included for the sources: road dust; domestic combustion; and other sources. Health effects from locally generated vehicle exhaust emissions are best associated with the health effects from NO_2-exposure above 5µg/m^3, as presented above. The calculations are made for excess deaths in the Swedish population in 2010 among adults at age above 30 years, due to exposure to $PM_{2.5}$

The total annual socio-economic costs related to high emissions of PM from road dust, domestic combustion and other sources is some 34.5 billion SEK_{2010}, and the absolute majority of these costs relate to loss of life years. If we compare this estimate with the estimation from 2005 (~25.5 billion SEK_{2005}), there is an increase in total annual socio-economic costs. The reason for that is, in the 2005 report, a cut-off value of 4 µg/m^{-3} was used since there was less scientific support for effects below these levels, whereas in this study a cut-off of 2 µg/m^3 was used. This lower cut-off is primarily due to better classification of source-specific exposure in 2010.

Table 25 Annual Socioeconomic costs of PM emissions from road dust, domestic combustion and other sources in Sweden, 2010 - Central Estimate.

	Socio-economic cost of health Effect [million SEK_{2010}/ case]	**Health effects in 2010**	**Socio-economic cost [million SEK_{2010}]**
Total Sweden			**34 711**
Out of which:			
Value of prevented fatality (VSL/VPF) (11 years of prolonged life)	5.48	**4 280**	23 451
Chronic Bronchitis	2.13	**2 155**	4 586
Hospitalisation, cardiology	0.05	**1 504**	70
Hospitalisation, generic (respiration)	0.03	**1 540**	42
RAD (age group 16-64)	1.43E-03	**4 576 041**	6 562

The results from the European commission data show that the total annual socio-economic costs related to PM-emissions from road dust, domestic combustion and other sources is about 40.5 billion SEK_{2010} (see Table 26). This value is higher than our estimates. As mentioned above there are differences in estimates between NEEDS and the data from the European Commission. For instance, the lower VSL value in the European Commission data is 5.5 million SEK_{2010}, i.e. the same as the central estimate in NEEDS. This difference between VSLs can explain the difference in the total annual socio-economic cost. .

Further, the value for chronic bronchitis is 2.1 million SEK_{2010} in NEEDS whereas the corresponding cost by the European Commission is 0.7 million SEK_{2010}. In an earlier study by the European commission in 2005 the estimated value for chronic bronchitis was approximately 2.5 million SEK_{2010} (€$_{2005}$ 208 000), which is more in line with the NEEDS value. However, due to new recommended values for chronic bronchitis from the HEIMSTA study (2011) the European Commission in discussion with the HRAPIE team decided to reduce the valuation for chronic bronchitis.

Table 26 Annual Socioeconomic costs of PM emissions from road dust, domestic combustion and other sources in Sweden, 2010 - European Commission Valuation data (2014) - Central Estimate.

	Socio-economic cost of health Effect [million SEK_{2010} / case]	Health effects in 2010 [cases]	Socio-economic cost [million SEK_{2010}]
Total Sweden			**40 591**
Out of which:			
Value of prevented fatality (VSL/VPF) (11 years of prolonged life)	7.90	**4 280**	33 829
Chronic Bronchitis	0.67	**2 155**	1 438
Hospitalisation, cardiology	0.03	**1 504**	42
Hospitalisation, generic (respiration)	0.03	**1 540**	43
RAD (age group 16-64)	1.15E-03	**4 576 041**	5 240

4.5.2 Sensitivity Analysis

In order to estimate a plausible range of socio-economic costs related to high levels of $PM_{2.5}$, some simple sensitivity analyses are performed. Matters of interest are what the results would be if health effect values from other studies were used and what effect a discounting of the VSL value would have on the total socio-economic cost. First we estimate the effect of different values on VSL as shown in Table 27. These estimates are taken from NEEDS (2007).

Table 27 Low / High Estimates of VSL from NEEDS (2007).

	Socio-economic cost related to $PM_{2,5}$ [million SEK_{2010}]
Low estimate VSL	25 837
High estimate VSL	157 830

The analysis shows that our central estimate is on the lower bound of the NEEDS estimates. The lower estimates from the European commission are slightly higher than the lower value from NEEDS, whereas the high estimate is much higher in NEEDS than the estimates from the European commission 2014 (Table 28). Again, this can likely be explained by the use of different data and also that the values from NEEDS are older.

Table 28 Low / High Estimates of VSL European Commission valuation data 2014.

	Socio-economic cost related to $PM_{2.5}$ [million SEK_{2010}]
Low estimate VSL	30 212
Central estimate VSL	40 590
High estimate VSL	88 078

Furthermore, time preferences are of general interest when valuing health effects. In economic valuation estimates time preferences are considered by introducing a discount rate, thereby reducing the value of future events. For the sake of comparison we discount the VSL values previously used with a 4% discount rate, which is a common rate used in valuation of health effects related to air pollution, Table 29.

Table 29 Discounted Low / Central / High Estimates of NEEDS (2007).

	Socio-economic cost related to $PM_{2.5}$ [million SEK_{2010}]
Low estimate VSL	19 037
High estimate VSL	54 216

From the discounting it can be seen that the values are sensitive to the choice of discount rate. However, it is our opinion that the values we use from NEEDS (2007) should remain undiscounted in the central estimate since they are valued with a method that allows the survey respondents to discount the values themselves.

Table 30 Discounted Low / High Estimates of VSL European Commission valuation data.

	Socio-economic cost related to $PM_{2.5}$ [million SEK_{2010}]
Low estimate VSL	25 438
Central estimate VSL	33 702
High estimate VSL	71 523

For policy purposes, our central estimate of the annual socio-economic costs related to high $PM_{2.5}$ levels in Sweden seem to be fairly robust. Even if discounted with a 4% discount rate, the socioeconomic cost would almost stay at 20 billion SEK_{2010} annually.

4.6 Cost-benefit analysis case study - reduced exposure to vehicle exhaust emissions in the three largest cities in Sweden

Key message from the case study

The reduction of exposure to exhaust emissions from cars by a large scale introduction of modern electric cars in the larger city regions of Stockholm, Göteborg, and Malmö could have motivated some indicative 13 – 18% of the additional costs required for an electric vehicle in 2010. Using the higher range of economic value on avoided fatalities as presented in the sensitivity analysis above, more than 60% of the costs for electric vehicles could have been socio-economically motivated by the impacts on human health. This case study exemplifies the need to monetize air pollution impact on human health from greenhouse gas emission abatement measures. This chapter gives a brief presentation of the case study; more detailed information is presented in *Appendix A – Detailed description of the Cost-Benefit Analysis of Electric* cars.

Prevailing problems with air pollution in cities

As this report has shown and reconfirmed, air pollution in Sweden is still causing severe health problems, and exhaust gases are according to the best available knowledge one of the most important culprits. The current Swedish projection on the development of car traffic is mainly considering a rejuvenation of the car fleet implying improved emission control, and a shift from gasoline to diesel engines. These cars will deliver emission reductions of both NOx, PM_{10}, and $PM_{2.5}$, but given that many Swedish cities are projected to experience relatively high levels of air pollution in 2020 and 2030 (Omstedt et al., 2012; Holmin Fridell et al., 2013) it is worthwhile exploring further options.

Given the last years technical development in electric cars these were chosen as a vehicle technology to be analyzed in a counterfactual case study of what the air pollution situation in three selected cities in Sweden could have been like in 2010, and what the net benefit for these cities would have been from an air pollution perspective if this greenhouse gas abatement measure would have been introduced.

How did we perform this case study?
Based on regional maps over the city regions we selected 26, 7, and 10 municipalities in and around Stockholm, Göteborg, and Malmö respectively that in this hypothetical case would have had a larger share of electric vehicles in 2010. The geographical selection was made rather large in an effort to include most of the cars driving in the Stockholm, Göteborg, and Malmö municipalities. This expansion was needed in order to get a reasonable relation between changes in emissions and changes in concentration of pollutants in the municipalities. We then used municipality-specific data on the number of purchased cars in 2010 as the numbers of electric cars to be introduced in this case study (Trafikanalys, 2014). A comparison with data on the existing car fleet in 2010 gave us the region specific percentage value of how large share of the total car fleet that would have been electric if all cars purchased in 2010 would have been electric in these regions. This region-specific percentage was then used to calculate changes in emissions from road transport. Changes in emissions from road transport were calculated based on the calculations performed and data used for the National emission projections (Gustafsson, Sjödin et al. 2013). On these data for 2010, the national average car fleet was changed so that X% was electric cars, where X corresponded to the regional-specific data on purchased vehicles in 2010. This introduction was either having an impact on the number of other newer conventional cars (minimum impact case) or an impact on the use of older cars (maximum impact). The total number of cars, the total transport work (vehicle kilometers), and the number of cars and average kilometer driven in each original emission standard category was also kept constant (identical to data for 2010). The impact on emissions from cars was then complemented with the emissions from other road transport vehicles to provide a total percentage change in emissions from road transport. We used the average value of the minimum and maximum impact case as a central case for continued analysis. The value of the region-specific percentage change in road transport emissions was then used to describe how much the source contribution from vehicle exhaust emissions would have to change for each considered municipality. Given this change in air pollution concentration, a change in health impacts and monetary health benefits was calculated following the methods described earlier in this report.

Calculating the abatement costs
In order to calculate the abatement costs we used estimates from the Swedish Consumer Agency (2014). The abatement cost was estimated as the difference in annual purchase and operating costs for the vehicle owners (net of taxes and subsidies) buying and using an electric vehicle instead of a conventional car. Annual purchase cost was in this case represented by the annual depreciation cost of the vehicle. The abatement costs were calculated for a low, mid, and high level reflecting different comparative vehicles, ownership duration and miles driven per year.

Most important assumptions
This case study is a pure 'what-if' analysis of an alternative reality in 2010. We have therefore allowed ourselves to make a number of assumptions. Most importantly, we have assumed that the current price of an electric car would have been available also in 2010. Furthermore, we assumed that a Y% change in vehicle exhaust emissions from a region would corresponded 1:1 with a Y% change in source contribution to air pollution concentration in the municipalities included in the region. Another important assumption was that relative contribution of personal cars to total road transport emissions in the regions corresponded to the national average in 2010.

The setting of this case implied that there would have been a large amount of used cars being sold so that the right mileage need would be given to the right consumer. New cars are usually used more than old cars, but electric cars have an upper limit on mileage per full charge, for the vehicles considered in this case study an upper limit of ~130 kilometers (Electric Power Research Institute, 2013). So the car owners in need for a car with long transport capacity would have had to buy a used car.

What would the socio-economic impact have been of electric cars?
The results show that a quite substantial part of the costs of electric vehicles could have been motivated by air pollution related health concerns.

	No. ELE cars	**% reduced emissions from road transport (mid case)**		**Total cost Million SEK_{2010}/ year (mid case)**	**Total monetized health impact Million SEK_{2010} / year (mid case)**	**% of cost motivated for health reasons (mid case)**	**% change in car fuel use**
		NO_x	$PM_{2.5}$				
Stockholm	77 476	6.2	7.2	1 728	236	14	~11
Göteborg	31 377	6.9	8.1	700	124	18	~13
Malmö	21 493	5.1	6.0	479	63	13	~9

In the sensitivity analysis the impact on emissions and concentrations, as well as the cost and benefit estimates were varied. The sensitivity analysis showed that in the low impact/high cost case, 8, 10, or 7% of the total cost could be motivated by health concerns for the Stockholm, Göteborg, and Malmö regions respectively. For the high impact/low cost case, 137, 172, and 131% of the total cost could be motivated by health concerns.

The most important limitations of this case study
This case study was based on the assumed relationship between emission reductions and reduced source contributions from vehicle exhaust emissions. In order to use a 1:1 relationship between transport emission reductions and source contribution from vehicle exhaust emissions in the city municipalities we expanded the amount of municipalities considered. However, vehicles from other municipalities than the ones consider also travel in the cities of Stockholm, Göteborg, and Malmö. This would reduce the 1:1 relationship. However, the vehicles considered in this case study also travel outside the municipalities considered, thereby contributing to improved air quality outside our analysis. We therefore continued with the use of a 1:1 relationship between transport emission reductions and source contribution from vehicle exhaust in the considered municipalities.

The constitution of the vehicle fleet in the city regions is not identical to the Swedish average vehicle fleet. Comparisons with regional data show however that the national average traffic work is a reasonable indicator of the traffic work in the regions. For the city of Stockholm in 2008 the traffic work was similar to the national average traffic work in Sweden 2010 used in this analysis (Burman and Johansson, 2009).

The average mileage per vehicle in Sweden ranged between 33 – 60 kilometers per day as a fleet average in 2010, dependent on ownership and whether all days of the year or only work days were considered (Trafikanalys, 2014). The vintage vehicle with the highest number of kilometers driven in 2010 corresponded to some 46 – 97 kilometers per day. These averages are well below the considered daily battery capacity in this analysis.

The vehicle costs considered in this analysis is based on vehicles available in 2014. The prices for these electric vehicles have decreased sharply in the years after 2010. However, given that this case study is counterfactual one could consider an alternative reality where investments in electric vehicles would have been made earlier, thereby pushing down prices in 2010. Also, the cost methodology used in this case study was based on the assumed value depreciation of the vehicles. If costs would have been calculated based on socioeconomic annualisation of investment costs over the lifetime of the vehicle, costs would have been lower. This latter method is the method most often use when calculating emission abatement costs from stationary sources.

Can we draw any conclusions?
Sharp conclusions are not suitable due to the counter-factual nature and the necessary assumptions made in this case study. It is however worth repeating that the socio-economic benefits related to human health impacts from air pollution is quite often not considered when comparing different greenhouse gas emission abatement options. And as we have shown, in 2010, it could have been a relatively good idea to push for a faster introduction of electric vehicles. The indicative results from our analysis showed a range between 3 - ~300% of the costs being covered by the socioeconomic impacts on air pollution (13 – 18% in the central case). The most important factors determining the size of this co-benefit were the costs for electric vehicles, and which economic value that should be assigned to an avoided fatality. What is happening today is a very fast shift to diesel vehicles (Trafikanalys, 2011; BilSweden, 2014). These have relatively high emissions of NO_x and $PM_{2.5}$ compared to other types of vehicles, and might not be on par with other cars until 2017 at the earliest. This shift risk missing out on an opportunity to ensure co-benefits between greenhouse gas emission abatement and air pollution emission control.

5 Discussion

The methodology of using an empirical model on a national basis, combined with advanced meteorological parameters, but still generating the results in a good geographical resolution, have not been used before Sjöberg et al. (2004). However, a similar approach has been used focusing on greater city areas (Amann, 2007). The great advantage with the URBAN model, compared to ordinary dispersion models, is that an emission database is not needed, and thus eliminating its uncertainties and limited possibilities to capture unknown changes. The URBAN model reflects changes both in large and local scale concentrations, through monitoring data, as well as in meteorology, through the input from the TAPM model.

The method used to estimate PM_{10} concentrations in urban areas, based on the relation to the levels of NO_2, has earlier been applied by i.a. UK (Muri, 1998). The relationship was adjusted to Swedish conditions, reflecting both latitudinal and seasonal variations. A comparison between the calculated PM_{10} concentrations, based on the PM_{10}/NO_2 ratio, and monitoring data in urban background has shown good agreement (Sjöberg et al., 2009). However, sources of NO_2 and PM_{10} are not always the same, for instance long range transport is the dominating source to the actual particle levels observed in Sweden, whereas the main sources of NO_2 are traffic and energy production. Since there are only four sites in Sweden where PM is measured in regional background it is difficult to, with certainty, estimate the contribution from long range particle transport. This does impose some uncertainty into the particle modelling.

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, where PM emissions are significant. 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 in larger cities, and apply the dispersion pattern to the URBAN model.

Environmental standards as well as environmental objectives are to be met everywhere, also at the most exposed kerb sites. However, for exposure calculations it is more relevant to used urban background data, on which also available exposure-response functions are based. The results from the urban modelling show that in 2010 most of the country had rather low NO_2 urban background concentrations compared to the environmental standard for the annual mean (40 μg/m^3). Likewise the PM_{10} urban background concentrations compared to the environmental standard for the annual mean (40 μg/m^3) were also low in most parts of the country. However, in some parts, mainly in southern Sweden, the concentration levels were of the same magnitude as the environmental objective (20 μg/m^3 as an annual mean) for the year 2010. The majority of people, 90%, were exposed to annual mean concentrations of PM_{10} less than 20 μg/m^3. Less than 5% of the Swedish inhabitants experienced exposure levels of PM_{10} above 25 μg/m^3.

The modelling results regarding $PM_{2.5}$ show that the urban background concentration levels in 2010 were below the environmental objective (12 μg/m^3 as an annual mean for the year 2010) in a quite large part of the country. More than 75% of the population was exposed to $PM_{2.5}$ annual mean concentrations less than 12 μg/m^3, while less than 15% experienced levels above 12 μg/m^3.

Updates to the population data used in this study compared with the one used for the exposure calculations for 2005 (Sjöberg et al., 2007; Sjöberg et al., 2009) showed a population increase of nearly 647 000 individuals over a 9 year period as the study 2005 used population data from 2001. The average exposure has decreased slightly between 2005 and 2010, from 6.3 to 6.2 μg/m^3 for NO_2, 12.7 to 12.5 μg/m^3 for PM_{10} and 9.8 to 8.6 μg/m^3 for $PM_{2.5}$. However, the number of people living in the highly polluted areas has increased, which at least partially is associated with the increase in the total population size. The population in these polluted areas tends to be skewed towards young people, whereas areas with low concentration of NO_2 and PM tend to have an older population.

To assess the change in the number deaths associated with NO_2 in the Swedish total population (all ages) between 2005 and 2010, the number of preterm deaths was recalculated using the NO_2 concentration distribution for 2005 but for the 2010 population. Using the 2010 ERF there is a 2.0% decrease in preterm deaths between 2005 and 2010 for the no threshold scenario, but a 13.4% increase with a 5 μg/m^3 cut-off (Table 31). The differences between the two cut-off scenarios are due to a larger proportion of the population exposed to higher concentrations in 2010. The estimated effects are also relatively larger in 2010 because of increased population in 2010. Note that in this comparison we used the total population (all ages) as was done in the 2005 study.

Table 31 Comparison of estimated number of excess deaths due to exposure to total NO_2 in the Swedish total population (all ages) in 2005 and 2010 with corresponding mortality rate applying the different ERF and cut-offs that have been used in this and previous 2005 study.

	2005 cut-off (10 μg/m^3) and ERF (1.13)	**2010 cut-off (5 μg/m^3) and ERF (1.08)**	**2010 cut-off (0 μg/m^3) and ERF (1.08)**
2005 NO_2 and 2005 population	828	1736	4541
2010 NO_2 and 2010 population	1002	2018	4564
2005 NO_2 and 2010 population	849	1780	4655

If we compare the estimated number of death associated with $PM_{2.5}$ in the Swedish total population (all ages) in 2005 and 2010 with corresponding mortality rate, applying the two cut-offs that have been used, there is a 15.7% decrease in effects of total $PM_{2.5}$ for the whole population (Table 32). Note that in this comparison we used the total population as was done in the 2005 study. Nevertheless, the sensitivity analyses with different populations, cut-offs, ERFs shows that the calculation are highly sensitive to these factors and chose of those can crucially affect the total calculated effects.

Table 32 Comparison of estimated number of excess deaths due to exposure to total $PM_{2.5}$ in the Swedish total population (all ages) in 2005 and 2010 with corresponding mortality rate applying the different cut-offs that have been used in this and previous 2005 study.

	2005 cut-off (4 μg/m³) and ERF (1.062)	2010 cut-off (2 μg/m³) and ERF (1.062)
2005 $PM_{2.5}$ and 2005 population	3243	4379
2010 $PM_{2.5}$ and 2010 population	2624	3754
2005 $PM_{2.5}$ and 2010 population	3296	4451

In contrast to the 2005 evaluation of PM exposure (Sjöberg et al., 2009) the separation between PM sources were done somewhat differently due to new exposure-response functions. The new exposure-response functions for PM focus on $PM_{2.5}$ and therefore a separation between $PM_{2.5}$ sources were made with particle emissions from traffic exhaust and from domestic heating assumed to only contribute to the $PM_{2.5}$ fraction. Ideally other $PM_{2.5}$ sources should be included, but due to gaps in the current knowledge there were not enough sound information to define these sources. Additionally the calculated exposure to $PM_{2.5}$ from traffic exhaust seems unrealistically low and we therefore would recommend caution when interpreting this result. We suggest that NO_2 would be a better indicator of traffic exhaust pollution.

Assessment of health impacts of particle pollution is difficult since PM is a complex mixture where different components very likely have different toxicity. However, WHO in HRAPIE (WHO, 2013b) and other assessments, lacking evidence enough for differential quantification, still choose to assume the same relative risk per particle mass concentration regardless of source and composition. This may be a too conservative approach and unwise with respect to the implications for actions. For this reason we apply different exposure-response functions for primary combustion generated particles (from motor vehicles and residential wood burning), for road dust and for other particles (the regional background of mainly secondary particles).

The recent WHO review REVIHAAP (WHO, 2013a) states that recent long-term studies show associations between $PM_{2.5}$ and mortality at levels well below 10 μg/m³, and thus concludes that for Europe it is reasonable to use linear exposure-response functions and to assume that any reduction in exposure will have benefits. This conclusion from REVIHAAP is also incorporated as a basic assumption in HRAPIE (WHO, 2013b).

Regarding long-term exposure and mortality the REVIHAAP report also concludes that more studies have now been published showing associations between long-term exposure to NO_2 and mortality (WHO, 2013a). This observation makes the situation a bit more complicated when it comes to impact assessments for vehicle exhaust particles, where the close correlation between long-term concentrations of NO_2 and exhaust particles may cause confounding in epidemiological studies.

The potential confounding problem in studies of effects from NO_2 and $PM_{2.5}$ on mortality was dealt with in a recent review paper focusing on 19 epidemiological long-term studies of mortality using both pollutants as exposure variable. In the review, studies with two-pollutant models ($PM_{2.5}$ and NO_2 in the same model) showed decrease in the effect estimates of NO_2, however still suggesting partly independent effects. One problem with such analyses is that the associations with mortality could partly be caused by exhaust particles having a correlation with both NO_2 and $PM_{2.5}$. In such a situation it would cause a problem of double counting if mortality effects of both $PM_{2.5}$ and NO_2 are added.

We believe that the effect on mortality of vehicle exhaust exposure in this national study is better described by NO_2 levels than by $PM_{2.5}$, and have in our calculations decided to use a large Danish cohort study of NO_2 and mortality not included in the meta-analysis mortality (Raaschou-Nielsen et al., 2012).

Which, if any, cut off level to use in a study like this is rather arbitrary, since we do not exactly know the natural background levels nor the shape of the exposure-response association in the lowest concentration intervals. There is no evidence of a specific toxicological threshold level shown to support a specific cut off level. For the assessment of different policies, especially in countries with a high background level of air pollutants, this question is not so relevant. When the total burden is estimated, the assumption of a lower threshold or not becomes relevant and makes a difference. Thus, we believe that for the total particle exposure, as for $PM_{2.5}$ from other sources than traffic and wood burning, a low threshold of 2 $\mu g/m^3$ $PM_{2.5}$, corresponding to the natural background in northern Sweden, is motivated. For PM_{10} the corresponding threshold is 3 $\mu g/m^3$. For the long-term effect of NO_2 on mortality, as a general traffic pollution indicator, we use 5 $\mu g/m^3$ as threshold, in addition to having no threshold at all. Without the threshold the regional background concentration is fully included, which likely results in double counting if added to the estimated impact of total $PM_{2.5}$.

The assessment of health impacts using $PM_{2.5}$ as exposure indicator is most valid for the regional background particle pollution. At first, urban background PM is largely built up by secondary particles, where a large part originates from remote sources. Secondly, the most often applied exposure-response relations for long-term effects on mortality come from studies where such particles were important for the contrasts in exposure. Recent research has shown that within-city gradients in air pollution seem to be very important for health effects (Jerret et al, 2005; WHO, 2006a). However, as suggested in this study particle mass concentration (as PM_{10} or $PM_{2.5}$) is not a good indicator of vehicle exhaust levels. Street levels of PM_{10} may be a good indicator for traffic when there is a lot of road dust, in particular during winter and spring where studded tires are used. NO_2 is on the other hand in most areas a good indicator of air pollution from the transport sector (cars, trucks, shipping). This does not mean that NO_2 is the definite main causal agent behind the health effects related to air pollution. Even if the exhaust particles are thought to contribute most to the health impacts, the health effects from local-regional gradients in vehicle exhaust are likely much better studied using NO_2 or NO_X as an indicator, rather than using particle mass as PM_{10}. Thus, the calculation using NO_2 is therefore a better indication of the magnitude of the mortality effects from traffic in Sweden than the estimates for exhaust PM and road dust PM in this assessment.

The estimation of respiratory and cardiovascular hospital admissions due to the short-term effect of $PM_{2.5}$ and NO_2 may seem to give a low number of admissions in comparison with the estimated number of deaths, new chronic bronchitis cases and restricted activity days. However, for hospital admissions only the short-term effect on admissions can be estimated, and thus not the whole effect on hospital admissions following morbidity due to PM. The total yearly number of hospital admissions in persons that developed their disease due to air pollution exposure may well be 10-20 times higher. It would be valuable to have morbidity indicators also for other long-term effects of air pollution exposure than chronic bronchitis. Most of the excess cases are related to large numbers exposed to low-moderate urban background levels, why current EU targets will have a minor effect.

All in all, a total of approximately 5 500 premature fatalities and a number of other health effects were estimated for 2010 assuming division between PM sources and a NO_2 cut-off of 5 μg/m³. Our comparison with the corresponding previous health impact calculations shows that the changes in calculated total PM2.5 population exposure results in a bit smaller health impacts all else (including population) being the same. Ozone has not been included in this study, but has also an impact on preterm deaths and causes also other adverse health effects.

It is important to put the socioeconomic costs into perspective and to discuss solutions that would ensure cost efficient measures to abate health effects from exhaust emissions and $PM_{2.5}$. All in all, socio-economic costs of ~35 billion SEK_{2010} can be associated with high levels of $PM_{2.5}$ in 2010. Furthermore, 7 and 25 billion SEK_{2010} are related to high NO_2 levels. Using the division between PM sources and NO_2 with a 5 μg/m³ as an indicator of traffic combustion the total socio-economic cost would be approximately 42 billion SEK_{2010}.

The high levels of PM will put some 0.3% of the Swedish working force out of their daily occupation. As a comparison, the Swedish Gross Domestic Product (GDP) was 3 338 billion SEK_{2010} in 2010 and the number of lost employment equals approximately 12 500 full time employments. However, which type of abatement measures that will provide most net benefit for society needs further analysis. As presented earlier in this report, the health impact of $PM_{2.5}$ seems to be source-dependent. So even though the total socio-economic cost associated with $PM_{2.5}$ is higher than the socio-economic cost associated with NO_2, it might still be the case that abatement measures focusing on NO_2 will provide a preferable net benefit. The issues of double-counting will however be of key concern when exploring suitable abatement measures.

The number of RADs affecting the Swedish 'working force' (age group 16 - 64 years) linked to $PM_{2.5}$ concentrations are about 4.5 million, which equals approximately 12 500 full time employments (given 251 working days per year and an unemployment rate of 8.7%). As a comparison, Volvo AB had approximately 23 000 employees in Sweden during 2010, the number of lost employment equals more than half of the amount of employees at the Volvo AB in Sweden. The number of full time occupancies in Sweden during 2010 amounts 3.33 million full time employments (www.scb.se, 2014). A comparison of these numbers gives that over 0.3% of the Swedish working force with an occupation in the age group 16 - 64 are impeded from participating in their occupation due to high levels of $PM_{2.5}$ concentrations.

The counter-factual analysis indicated some 13-18% of the electric vehicles to be motivated just by monetized health impacts in the three largest Swedish city regions. These percentages were especially sensitive to the costs of electric vehicles as well as the economic Value of a Statistical Life (VSL). In this study, a comparatively low VSL was used, indicating that the percentage could be even higher. However, given the future decrease in vehicle emissions due to more stringent Euro standards the relative impacts of electric vehicles will decrease. This reduction in relative impacts might however be counteracted by the reduction in costs for electric vehicles. So it may still be reasonable that a 13-18% share of the costs for electric vehicles could be motivated by air pollution health impacts in the future.

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Appendix A – Detailed description of the Cost-Benefit Analysis of Electric cars

Introduction

As this report has shown and reconfirmed, air pollution in Sweden is still causing severe health problems, and exhaust gases from road transport are according to the best available knowledge one of the most important culprit. The current Swedish projection on the development of car traffic is mainly considering a rejuvenation of the car fleet implying improved emission control, and a shift from gasoline to diesel engines. These cars will deliver emission reductions of both NO_x, PM_{10}, and $PM_{2.5}$, but given that many Swedish cities are projected to experience relatively high levels of air pollution in 2020 and 2030 (Omstedt, Andersson et al., 2012; Holmin Fridell, Jones et al., 2013) it is worthwhile exploring further options.

Given the last years technical development in electric cars these were chosen as a vehicle technology to be analysed in a counterfactual analysis of what the air pollution situation in three selected cities in Sweden could have been like in 2010, and what the net benefit for these cities would have been from an air pollution perspective if this greenhouse gas abatement measure would have been introduced.

Method and data

Data on road transport emissions of NO_x and $PM_{2.5}$ were taken from the analysis performed by the Swedish Road Administration and used in the Swedish projection of emissions as reported in March 2013 (Gustafsson, Sjödin et al., 2013). Data on 2010 vehicle usage and vehicle purchases in the regions were taken from Trafikanalys (2014).

Table A 1: National NO_x and $PM_{2.5}$ emissions from road transport in 2010.

Vehicle type	NO_x (ton)	Share	$PM_{2.5}$ (ton)	Share
All vehicles	64657	100%	1567	100%
Gasoline cars	15034	23%	115	7%
Ethanol/gasoline cars	351	1%	3	0%
Gas/gasoline cars	16	0%	0	0%
Diesel cars	6926	11%	384	24%
Diesel heavy duty trucks	30355	47%	556	35%
Gasoline light duty trucks	725	1%	7	0%
Diesel light duty trucks	5713	9%	380	24%
Diesel bus	5536	9%	122	8%

As is seen in the table above, passenger cars constituted some 35% of the total road transport NO_x emissions in 2010 (31% for $PM_{2.5}$).

In order to derive the number of cars that should be considered for analysis we selected a number of municipalities surrounding the city regions of Stockholm, Göteborg and Malmö. At this stage it was important to select a sufficiently large amount of municipalities so that it would be fairly reasonable to assume a 1:1 correlation between changes in road transport emissions in the city municipalities and changes in air pollution concentration from exhaust gases in the city municipalities. The key issue was that the source contribution of exhaust gases in the city municipalities origin not only from cars bought and owned within that municipality. Other cars are also contributing significantly. Table A 2 indicates the municipalities considered for health impacts and vehicle purchase in this analysis.

Table A 2: Swedish municipalities considered in the CBA case.

Stockholm region	Göteborg region	Malmö region
UPPLANDS VÄSBY	HÄRRYDA	STAFFANSTORP
VALLENTUNA	PARTILLE	BURLÖV
ÖSTERÅKER	ALE	VELLINGE
VÄRMDÖ	LERUM	KÄVLINGE
JÄRFÄLLA	GÖTEBORG	LOMMA
EKERÖ	MÖLNDAL	SVEDALA
HUDDINGE	KUNGÄLV	MALMÖ
BOTKYRKA		LUND
SALEM		LANDSKRONA
HANINGE		ESLÖV
TYRESÖ		
UPPLANDS-BRO		
NYKVARN		
TÄBY		
DANDERYD		
SOLLENTUNA		
STOCKHOLM		
SÖDERTÄLJE		
NACKA		
SUNDBYBERG		
SOLNA		
LIDINGÖ		
VAXHOLM		
NORRTÄLJE		
SIGTUNA		
NYNÄSHAMN		

In these municipalities, Trafikanalys (2014) provides information on the number of cars used in these municipalities and the number of newly purchased vehicles in 2010. This information was used as an indicator on how much emissions should change in the CBA case. Table A 3 shows the number of vehicles and the potential number of electric vehicles to be considered in the analysis.

Table A 3: Number of vehicles considered in the regions.

	Stockholm region	Göteborg region	Malmö region
Cars in traffic 2010	800 534	285 290	263 936
New ele cars / Corresponding to new purchase of cars in 2010	77 476	31 377	21 493
% ele vehicles	9.7%	11%	8.1%

The impact on emissions in the regions was analysed based on the Swedish average vehicle fleet for 2010. In this case, the region-specific percentage of electric vehicles was introduced into the vehicle fleet, where they either replaced old cars or new cars. The total number of vehicles and vehicle kilometres were kept constant as in the original data for 2010. This analysis then produced max and min impacts on emissions, which were then averaged in the central case analysis.

In order to calculate the abatement costs we used estimates from the Swedish Consumer Agency (2014). The abatement cost was estimated as the difference in annual purchase and operating costs for the vehicle owners (net of taxes and subsidies) buying and using an electric vehicle instead of a conventional car. Annual purchase cost was in this case represented by the annual depreciation cost of the vehicle. The abatement costs were calculated for a low, mid, and high level reflecting different comparative vehicles, ownership duration and miles driven per year.

Assumptions

- We assumed that the vintage of the national average car fleet was identical to the vintage in the city regions.
- We assumed that the relative contribution of different vehicle types to total road transport emissions in the city regions corresponded to the national average. The two assumptions above can at least partly be justified by the similarity in average traffic work in Sweden 2010, and traffic work in Stockholm 2008, see Table A 4.
- We assumed that electric vehicles would replace gasoline, diesel, gas, and ethanol cars in proportion to their share of the total passenger car fleet (maximum impact case), or in proportion to their share of total newly purchased cars in 2010 (minimum impact case).
- The setting of this case implied that there would have been a large amount of used cars being sold so that the right mileage need would be given to the right consumer. New cars are usually used more than old cars, but electric cars have an upper limit on daily mileage. So the car owners in need for a car with long transport capacity would have had to buy a used car.
- We assumed that a % change in vehicle exhaust emissions from a municipality would correspond 1:1 with a % change in source contribution to air pollution concentration in these and other included municipalities.
- We assumed that X% change in NO_x emissions would correspond in an X% change in NO_2 concentrations.
- We assumed that electric vehicles have zero tailpipe emissions.
- We assume that the electricity used for these cars would origin from renewable non-combustion sources.
- We assume that the price difference between average and electric cars observed in 2014 could have been observed in 2010 (given earlier investments in technology development).

Comparison with 2008 data for Stockholm shows a decent fit between the national average transport work in 2010 and the Stockholm region transport work in 2008, see Table A 4.

Table A 4: Transport work share per vehicle category in Sweden 2010 (Gustafsson, Sjödin et al. 2013) and Stockholm 2008 (Burman and Johansson 2009).

	Sweden 2010	Stockholm 2008
All vehicles	100%	100%
gasoline cars	58%	58%
E85/gasoline car	4%	9%
Gas/gasoline car	0%	1%
Diesel car	20%	12%
Diesel heavy duty truck	6%	3%
Gasoline light duty truck	1%	5%
Diesel light duty truck	9%	11%
Diesel bus	1%	1%

Results

The region-specific increase in number of electric vehicles would in this analysis have an impact on vehicle usage and vehicle mileage. The impact shown in Table A 5 and Table A 6 represent a 10 % introduction of electric vehicles in 2010 on the national average passenger car fleet. The impact per fuel category depends on the share of the fuel categories of the total cars sold in 2010 (Trafikanalys, 2011).

Table A 5: The number of vehicles, average km, and vehicle km in 2010 considered in the CBA (minimum impact on emissions since electric vehicles replace new vehicles).

		National estimate 2010			CBA case 2010		
Fuel	Emission class	Number vehicles	average km/yr	vehicle km/yr	Number vehicles	average km/yr	vehicle km/yr
E85/gasoline	PreEuro						
E85/gasoline	Euro 1						
E85/gasoline	Euro 2						
E85/gasoline	Euro 3						
E85/gasoline	Euro 4	199 729	14 340	2 864 026 040	162 526	14 340	2 330 562 046
E85/gasoline	Euro 5	4 763	8 553	40 736 696	0	0	0
E85/gasoline	Electric	0	0	0	41 965	13 683	574 200 689
Sub-total E85		**204 492**	**14 205**	**2 904 762 736**	**204 492**	**14 205**	**2 904 762 736**
Gas/gasoline	PreEuro						
Gas/gasoline	Euro 1						
Gas/gasoline	Euro 2						
Gas/gasoline	Euro 3	1 060	11 053	11 712 717	1 060	11 053	11 712 717
Gas/gasoline	Euro 4	23 089	14 534	335 586 390	2 968	14 534	43 144 019
Gas/gasoline	Euro 5	862	9 254	7 973 831	0	0	0
Gas/gasoline	Electric	0	0	0	20 983	14 317	300 416 202
Sub-total Gas		**25 011**	**14 205**	**355 272 938**	**25 011**	**14 205**	**355 272 938**
Gasoline	PreEuro	358 407	6 715	2 406 777 788	358 407	6 715	2 406 777 788
Gasoline	Euro 1	336 807	10 279	3 462 092 435	336 807	10 279	3 462 092 435
Gasoline	Euro 2	917 821	12 237	11 231 479 812	917 821	12 237	11 231 479 812
Gasoline	Euro 3	400 587	13 717	5 495 011 643	400 587	13 717	5 495 011 643
Gasoline	Euro 4	1 456 082	15 294	22 268 996 981	1 338 960	15 294	20 477 772 222
Gasoline	Euro 5	29 757	8 920	265 437 974	0	0	0
Gasoline	Electric	0	0	0	146 878	14 003	2 056 662 732
Sub-total gasoline		**3 499 461**	**12 896**	**45 129 796 634**	**3 499 461**	**12 896**	**45 129 796 634**
Diesel	PreEuro	6 035	4 776	28 824 818	6 035	4 776	28 824 818
Diesel	Euro 1	13 460	11 733	157 931 371	13 460	11 733	157 931 371
Diesel	Euro 2	49 324	16 797	828 497 241	49 324	16 797	828 497 241
Diesel	Euro 3	72 718	23 682	1 722 112 169	72 718	23 682	1 722 112 169
Diesel	Euro 4	443 639	27 759	12 314 866 130	255 798	27 759	7 100 638 771
Diesel	Euro 5	21 985	17 938	394 362 875	0	0	0
Diesel	Electric	0	0	0	209 826	26 730	5 608 590 234
Subtotal diesel		**607 162**	**25 441**	**15 446 594 604**	**607 162**	**25 441**	**15 446 594 604**

As is seen in Table A 5, in the minimum impact, the electric vehicles replace the newest cars available in each fuel category.

Table A 6: The number of vehicles, average km, and vehicle km in 2010 considered in the CBA (maximum impact on emissions since electric vehicles replace old vehicles).

		National estimate 2010			CBA case 2010		
Fuel	**Emission class**	**Number vehicles**	**average km/yr**	**vehicle km/yr**	**Number vehicles**	**average km/yr**	**vehicle km/yr**
E85/gasoline	PreEuro						
E85/gasoline	Euro 1						
E85/gasoline	Euro 2						
E85/gasoline	Euro 3						
E85/gasoline	Euro 4	199 729	14 340	2 864 026 040	179 938	14 340	2 580 234 418
E85/gasoline	Euro 5	4 763	8 553	40 736 696	4 763	14 340	68 299 352
E85/gasoline	Electric	0	0	0	19 791	12 947	256 228 965
Sub-total E85		**204 492**	**14 205**	**2 904 762 736**	**204 492**	**14 205**	**2 904 762 736**
Gas/gasoline	PreEuro						
Gas/gasoline	Euro 1						
Gas/gasoline	Euro 2						
Gas/gasoline	Euro 3	1 060	11 053	11 712 717	0	0	0
Gas/gasoline	Euro 4	23 089	14 534	335 586 390	21 729	14 364	312 118 304
Gas/gasoline	Euro 5	862	9 254	7 973 831	862	14 534	12 522 949
Gas/gasoline	Electric	0	0	0	2 421	12 655	30 631 686
Sub-total Gas		**25 011**	**14 205**	**355 272 938**	**25 011**	**14 205**	**355 272 938**
Gasoline	PreEuro	358 407	6 715	2 406 777 788	19 728	6 715	132 476 906
Gasoline	Euro 1	336 807	10 279	3 462 092 435	336 807	6 715	2 261 726 305
Gasoline	Euro 2	917 821	12 237	11 231 479 812	917 821	11 507	10 561 691 794
Gasoline	Euro 3	400 587	13 717	5 495 011 643	400 587	12 466	4 993 670 062
Gasoline	Euro 4	1 456 082	15 294	22 268 996 981	1 456 082	14 927	21 735 107 170
Gasoline	Euro 5	29 757	8 920	265 437 974	29 757	15 294	455 098 001
Gasoline	Electric	0	0	0	338 679	14 734	4 990 026 395
Sub-total gasoline		**3 499 461**	**12 896**	**45 129 796 634**	**3 499 461**	**12 896**	**45 129 796 634**
Diesel	PreEuro	6 035	4 776	28 824 818	0	0	0
Diesel	Euro 1	13 460	11 733	157 931 371	0	0	0
Diesel	Euro 2	49 324	16 797	828 497 241	10 058	7 559	76 027 691
Diesel	Euro 3	72 718	23 682	1 722 112 169	72 718	17 461	1 269 755 880
Diesel	Euro 4	443 639	27 759	12 314 866 130	443 639	27 219	12 075 306 078
Diesel	Euro 5	21 985	17 938	394 362 875	21 985	27 759	610 281 015
Diesel	Electric	0	0	0	58 761	24 084	1 415 223 940
Subtotal diesel		**607 162**	**25 441**	**15 446 594 604**	**607 162**	**25 441**	**15 446 594 604**

NO_x and $PM_{2.5}$ emission changes were then calculated based on the differences between the national estimate for 2010 and the CBA case vehicles and vehicle usage. The sharp increase in purchase and usage of electric vehicles, followed by a decline in purchase and usage of other conventional fuel cars would in this analysis imply changes in emissions and air pollution concentration as presented in Table A 7 and **Error! Reference source not found.**. The resulting percentage impact on emissions in 2010 is shown in Table A 7, with the following 1:1 impact on NO_2 and $PM_{2.5}$ concentration from vehicle exhaust emissions in **Error! Reference source not ound.**.

Table A 7: Potential reduction in regional transport emissions if all newly purchased passenger cars in 2010 were electric. Values in paranthesis represent the values for the minimum and maximum impact calculations.

Reduced emissions	Stockholm region	Göteborg region	Malmö region
NO_x (%)	6.2 (4.1–8.2)	6.9 (4.7–9.2)	5.1 (3.4–6.9)
$PM_{2.5}$ (%)	7.2 (6.1–8.2)	8.1 (7.1–9.1)	6.0 (5.0–7.0)
Fuel use (~ % CO_2)	~11-12	~13 – 14	~9 – 10

Again, this 1:1 ratio between changes in road transport emissions and contribution to NO_2 and $PM_{2.5}$ concentrations from road transport is a case specific relationship that we consider as reasonable given the number of municipalities included in the analysis for each region. There are of course passenger cars from other municipalities and regions that enter the city municipalities (thereby reducing the 1:1 ratio assumed in this analysis), but the cars in the studied municipalities would likewise affect municipalities outside the regions considered in this analysis (thereby implying health impacts not considered in this analysis).

The health impacts and economic benefits calculated from this reduction in air pollution concentrations followed the methodology presented in the main report and will not be further presented here. These emission reductions would come with an average annual cost per vehicle, as presented in Table A 8.

Table A 8: Electric car abatement costs, based on three electric vehicles.

Car Make	Years of ownership			Mileage cost per year			Total annual cost			Annual abatement cost per ELE car (SEK_{2010})		
	Low	Mid	High	Low	Mid	High	Low	Mid	High	Low	Mid	High
Nissan LEAF EL Visia 2014	5	3	1	4 500	3 000	1 500	66 300	69 300	85 000	**10 000**	**22 300**	**38 700**
Chevrolet Volt 1.4	5	3	1	5 100	3 400	1 700	80 500	82 300	80 000	25 500	35 300	33 700
Opel Ampera	5	3	1	5 100	3 400	1 700	72 300	72 000	68 800	16 000	25 000	22 500
Volkswagen Golf 2 1.4 TSI	5	3	1	21 200	14 100	7 100	55 000	47 000	44 000	-	-	-
Volvo V40	5	3	1	22 200	15 000	7 500	56 300	46 300	46 300	-	-	-

The annual abatement costs highlighted in bold were used in the min, max, and average CBA estimates. The other vehicles were included for comparisons of possible price ranges.

The impact on human health that would be associated with a large scale introduction of electric vehicles is presented in Table A 9, as well as the final Cost-Benefit analysis of the introduction.

Table A 9: Avoided health impacts and cost benefit analysis of a potential use of electric vehicles in 2010.

	Stockholm region	Göteborg region	Malmö region
Mortality (cases)			
Mid case	42	22	11
Min case	28	15	7
Max case	57	29	15
Chronic Bronchitis (cases)			
Mid case	1	1	1
Min case	1	1	1
Max case	2	1	1
CHA - Hospitalisation, cardiology (cases)			
Mid case	1	0	0
Min case	1	0	0
Max case	1	0	0
RHA hospitalisation (cases)			
Mid case	17	9	4
Min case	11	6	3
Max case	22	12	6
RAD (cases)			
Mid case	1819	1077	1013
Min case	1238	964	871
Max case	2333	1203	1362
Abatement Cost (million SEK_{2010})			
Mid case	1 728	700	479
Min case	2998	1214	832
Max case	775	314	215
Economic Benefits (million SEK_{2010})			
Mid case	265	138	71
Min case	152	85	40
Max case	1060	538	282
Economic values from European Commission	305	159	81
% of cost motivated for health reasons			
Mid case	15	20	15
Min case	8	10	7
Max case	137	172	131
Economic values from European Commission	18	23	17

As is seen in the table above, the CBA results show a large variation in the net benefit associated with human health benefits in 2010. The CBA results are mostly dependent on the costs associated with electric cars and the economic value assigned to avoided fatalities.

Discussion

Sharp conclusions from this CBA are not suitable due to the counter-factual nature and the necessary assumptions made. It is however worth repeating that the socio-economic benefits related to human health impacts from air pollution is quite often not considered when comparing different greenhouse gas emission abatement options. And as we have shown, in 2010, it could have been a relatively good idea to push for a faster introduction of electric vehicles. The indicative results from our analysis showed a range between 3 - ~300% of the costs being covered by the socioeconomic impacts on air pollution (13 – 18% in the central case). The most important factors determining the size of this co-benefit were the costs for electric vehicles, and which economic value that should be assigned to an avoided fatality. What is happening today is a very fast shift to diesel vehicles (Trafikanalys, 2011; BilSweden, 2014). These have relatively high emissions of NO_x and $PM_{2.5}$ compared to other types of vehicles, and will not be on par with other cars until 2017 at the earliest. This shift risk missing out on an opportunity to ensure co-benefits between greenhouse gas emission abatement and air pollution emission control. Considering the impact of electric cars, a final note is that while emissions of newer cars is expected to go down in the future, thereby reducing the benefits on human health from introduction of electric vehicles, so will probably the costs of electric vehicles, which would increase the cost-benefit of an introduction of electric vehicles.

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