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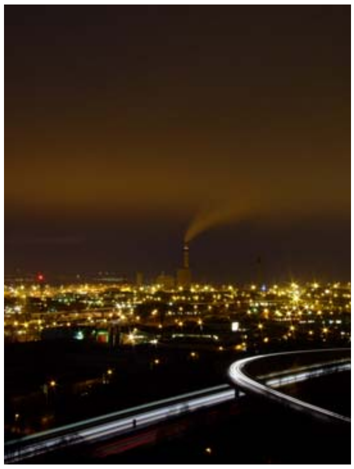

The mortality cost of particulate matter due to emissions in the Stockholm area – an investigation into harmfulness, sources and the geographical dimension of their impact

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The mortality cost of particulate matter due to emissions in the Stockholm area– an investigation into harmfulness, sources and the geographical dimension of their impact

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It has long been recognized that emissions from traffic have a negative impact on human health. In recent years there has been emerging consensus that the main influence is due to particulate matter. From an economic point of view these negative effects are external costs caused by traffic that, if not accounted for in decision making regarding transport, will result in a non-optimal allocation of resources leading to welfare losses. In the Impact pathway approach (IPA), that has been developed in the ExternE projects, the external cost is calculated as the product of exposure, effect and value. In this study the effect we focus on is health impacts (mortality). In TESS project the purpose has been to investigate how important the external health cost of road traffic generated PM is in relation to the cost of other sources of PM. To do this we have both investigated how the exposure varies between sources but also assessed if it is reasonable to assume that the impact differs between PM from different sources. Whether or not to assume that PM of different origin is equally harmful is of particular interest in Sweden where non-exhaust PM makes a large contribution to the concentrations of PM in urban areas. In the project we have used Stockholm as a case study and we have focused on mortality since this is the health impact that has been found to have the largest impact on health cost in other studies.

The findings in this report are that there is not an one-to-one correspondence between emissions and costs. The reason for this is that the cost is based on health impacts which in turn are related to population exposure. Combustion PM from road traffic and residential heating gives rise to a higher cost than for example power plants, although the emissions are smaller, since their emissions are released in close proximity to people's place of residence. Regarding non-exhaust PM (mainly road wear) the estimated cost is lower than that for combustion PM (exhaust emissions) from road traffic. This depends on the use of different exposure-response functions where non-exhaust PM is considered to have less impact on mortality. The results also show that the emissions from the sources in Stockholm also have an influence on the population exposure in the rest of Sweden and in Europe. This is due to the formation of secondary PM from gaseous pollutants, mainly NOx. For power plants this estimated cost is similar to that for the health impacts that occur locally in Stockholm for this source.

Keywords:
External cost, Air pollution, Particulate matter, Road traffic

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Författare:
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Kostnaden för dödlighet som partiklar från emissioner i Stockholm orsakar – en studie i hur skadliga olika partiklar är, vilka källorna är och den geografiska dimensionen

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Det resultaten visar är att förhållandet mellan emissioner och kostnader inte är ett till ett. Detta beror på att kostnaderna är baserade på de beräknade hälsoeffekterna vilka i sin tur beror på den beräknade befolkningsexponeringen. Förbränningspartiklar från vägtrafik och bostadsuppvärmning ger exempelvis upphov till högre kostnader än energianläggningar, trots att deras emissioner är lägre, eftersom deras utsläpp sker i närheten av bostäder. När det gäller partiklar från vägslitage så är kostnaden lägre än den som orsakas av avgaspartiklar från vägtrafiken. Detta beror framförallt på olika antaganden när det gäller hälsoeffektsambanden där slitagepartiklar antas ha en lägre påverkan på dödligheten. Resultaten visar också att emissionerna i Stockholm även bidrar till att människor i övriga Sverige och i Europa exponeras för partiklar. Orsaken till detta är att det bildas sekundära partiklar av emissioner såsom NOx. För energianläggningar är den kostnad som dessa emissioner orsakar ungefär densamma som de kostnader som denna källa orsakar lokalt i Stockholm.

Nyckelord:
Externa effekter, luftföroreningar, partiklar, vägtrafik

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Preface

This is a report resulting from the research project TESS – Traffic emissions, socioeconomic valuation and socioeconomic measures – that has been financed by EMFO (The Emissions Research Programme). In 2002 an agreement about the EMFO programme was reached between the partners from the Swedish Vehicle Research Council, PFF. EMFO was a sector-wide research competence to develop vehicles and vehicle components with emission levels that are sustainable in the long term. The aim of EMFO was to offer academia, industry and authorities access to necessary knowledge and pioneering solutions that are necessary if vehicle technology is to develop in a sustainable direction. One important task has been to coordinate activities within the programme with both national and international research in the field.

EMFO comprised subsidiary programmes and two of these were: "Socio-economic evaluation of the health and environmental impact of different emissions" and "Optimal range of socio-economic measures". TESS undertook research in these two areas but it was also related to the subsidiary programme: "Health and environmental impact". The application was approved in 2005 and the project took place 2005–2008.

The basis for the research in TESS is the valuation methods developed in the EU funded ExternE projects where the external cost of emissions is calculated by tracing the effects that the emissions have on human health and then valuing these effects. The aim of TESS was to calculate the external costs related to particulate matter that local emissions (from road traffic and other sources) generate on a local and regional scale using Stockholm as a case study. Based on this information an analysis was also made on what reduction measures are likely to be efficient from an economic point of view.

The analysis undertaken in TESS requires collaboration between researchers from different research disciplines and therefore there were four parties involved in this project. Coordinator for the project has been VTI, where Lena Nerhagen was project leader as well as responsible for the economic analysis. Christer Johansson and Kristina Eneroth at SLB analys (Environment and Health Administration, Stockholm) contributed with information about local emissions and performed exposure calculations for the Stockholm area. Robert Bergström at SMHI performed the regional scale dispersion and exposure modelling. Finally, Bertil Forsberg at Umeå Universiy undertook the health impact assessment.

This report presents the results of the valuation part of the TESS project. It is based on the results from the sub-projects undertaken (see references in text) and summaries of these results are presented in the report. In Nerhagen and Li (2008) the results of the cost-effectiveness analysis that has been undertaken within the TESS project are presented.

Borlänge December 2008

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Quality review

Review seminar was carried on 5th December 2008 in Stockholm where Mats Gustafsson, researcher at VTI, reviewed and commented on the report. Based on these comments and comments from Gunnar Lindberg alterations have been made to the final manuscript of the report. Participants at the seminar were also Gunnar Lindberg, Göran Friberg, Jan-Eric Swärdh, Maud Göthe-Lundgren, Svante Mandell and Anna Mellin. The research director Mats Andersson then examined and approved the report for publication.

Kvalitetsgranskning

Rapporten presenterades och granskades på ett seminarium i Stockholm den 5 december 2008 då Mats Gustafsson, forskare på VTI, lämnade synpunkter på rapporten. Vid seminariet deltog även Gunnar Lindberg, Göran Friberg, Jan-Eric Swärdh, Maud Göthe-Lundgren, Svante Mandell och Anna Mellin. Utifrån de kommentarer som Mats Gustafsson lämnade samt synpunkter från Gunnar Lindberg har rapporten justerats och slutligt rapportmanus tagits fram. Forskningschef Mats Andersson har därefter granskat och godkänt publikationen för publicering.

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The mortality cost of particulate matter due to emissions in the Stockholm area – an investigation into harmfulness, sources and the geographical dimension of their impact

by Lena Nerhagen, Robert Bergström[*], Bertil Forsberg[**], Christer Johansson[***] and Kristina Eneroth[***]
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SE-581 95 Linköping Sweden

Summary

It has long been recognized that emissions from traffic have a negative impact on human health. In recent years there has been emerging consensus that the main influence is due to particulate matter. From an economic point of view these negative effects are external costs caused by traffic that, if not accounted for in decision making regarding transport, will result in a non-optimal allocation of resources leading to welfare losses. To be able to implement road pricing measures, but also for the evaluation of other control measures through benefit-cost analysis, information on the external cost of traffic emissions is needed. In the Impact pathway approach (IPA), that has been developed in the ExternE projects, the external cost is calculated as the product of exposure, effect and value. In this study the effect we focus on is health impacts (mortality).

Regarding particulate matter (PM) there is recognition among the research community that there are different types of PM and that it is likely that their impact on human health differs. Still the current practice is to treat fine PM (which are considered to be most detrimental to health) as equally harmful irrespective of origin. In TESS the purpose has been to investigate how important the external health cost of road traffic generated PM is in relation to the cost of other sources of PM. To do this we have both investigated how the exposure varies between sources but also assessed if it is reasonable to assume that the impact differs between PM from different sources. Whether or not to assume that PM of different origin is equally harmful is of particular interest in Sweden where non-exhaust PM makes a large contribution to the concentrations of PM in urban areas. In the project we have used Stockholm as a case study and we have focused on mortality since this is the health impact that has been found to have the largest impact on health cost in other studies.

The findings in this report are that there is not an one-to-one correspondence between emissions and costs. The reason for this is that the cost is based on health impacts which in turn are related to population exposure. Combustion PM from road traffic and residential heating gives rise to a higher cost than for example power plants, although the emissions are smaller, since their emissions are released in close proximity to people's place of residence. Regarding non-exhaust PM (mainly road wear) the estimated cost is lower than that for combustion PM (exhaust emissions) from road traffic. This depends on the use of different exposure-response functions where non-exhaust PM is considered to have less impact on mortality. The results also show that

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the emissions from the sources in Stockholm also have an influence on the population exposure in the rest of Sweden and in Europe. This is due to the formation of secondary PM from gaseous pollutants, mainly NO_x. For power plants this estimated cost is similar to that for the health impacts that occur locally in Stockholm for this source.

These results also have implications for policy. It is concluded that if the purpose is to reduce human mortality more focus should be placed on reducing combustion emissions that contribute to the more dangerous, fine PM. To evaluate policies regarding non-exhaust emissions, more research is needed to clarify their impact on health. The regional impact and the impact on other emissions should also be more in focus in policy evaluations. It is also concluded that for a full picture on health cost of particulate matter there is a need for more research regarding morbidity; what exposure response functions as well as economic values should be used. Finally, all inputs are the result of ongoing empirical research and they are all related to uncertainties, hence the mortality cost that is calculated represents the results of the current state of the art.

Kostnaden för dödlighet som partiklar från emissioner i Stockholm orsakar – en studie i hur skadliga olika partiklar är, vilka källorna är och den geografiska dimensionen

av Lena Nerhagen, Robert Bergström*, Bertil Forsberg**, Christer Johansson*** och Kristina Eneroth***
VTI
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Sammanfattning

Det är sedan länge väl känt att emissioner från vägtrafik har en negativ påverkan på människors hälsa. Den forskning som skett under senare år visar på att den främsta orsaken till detta är de partiklar som emissionerna orsakar. Inom nationalekonomin är denna negativa påverkan på människors hälsa en extern effekt som kommer att leda till beslut som inte är optimala och därmed välfärdsförluster. För att utvärdera vilka åtgärder som behövs för att reducera välfärdsförlusterna krävs kunskap om storleken på de externa kostnaderna. I Impact Pathway ansatsen (IPA) som utvecklats i de EU-finansierade ExternE-projekten beräknas den externa kostnaden som produkten av exponering, effekten och den ekonomiska värderingen av effekten. I denna rapport är den effekt som vi studerar påverkan på människors hälsa (dödlighet).

När det gäller partiklar så har dessa olika ursprung och olika kemiska sammansättning och de förväntas därför ge upphov till olika påverkan på människors hälsa. Trots det är det antagande som för närvarande används vid hälsoeffektberäkningar att alla partiklar har samma påverkan. Syftet med TESS har varit att studera hur stora vägtrafikens hälsokostnader är i förhållande till dem som orsakas av partikelemissioner från andra källor. För att göra detta har vi undersökt hur befolkningsexponeringen skiljer sig åt mellan olika källor men också om det är rimligt att anta att hälsopåverkan av olika partiklar är densamma. Den senare frågan är av speciellt intresse för situationen i Sverige eftersom partiklar från vägslitage är en viktig orsak till höga partikelhalter i större tätorter. I studien har vi gjort beräkningar för Stockholm och den hälsoeffekt vi studerat är dödsfall eftersom denna effekt i andra studier har visat sig ha störst betydelse för den beräknade kostnaden.

Det resultaten visar är att förhållandet mellan emissioner och kostnader inte är ett till ett. Detta beror på att kostnaderna är baserade på de beräknade hälsoeffekterna vilka i sin tur beror på den beräknade befolkningsexponeringen. Förbränningspartiklar från vägtrafik och bostadsuppvärmning ger exempelvis upphov till högre kostnader än energianläggningar, trots att deras emissioner är lägre, eftersom deras utsläpp sker i närheten av bostäder. När det gäller partiklar från vägslitage så är kostnaden lägre än den som orsakas av avgaspartiklar. Detta beror framförallt på olika antaganden när det gäller hälsoeffektsambanden där slitagepartiklar antas ha en lägre påverkan på dödligheten. Resultaten visar också att emissionerna i Stockholm även bidrar till att människor i övriga Sverige och i Europa exponeras för partiklar. Orsaken till detta är att det bildas sekundära partiklar av emissioner såsom NO_x. För energianläggningar är den

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kostnad som dessa emissioner orsakar ungefär densamma som de kostnader som denna källa orsakar lokalt i Stockholm.

Dessa resultat har även policy-implikationer. En slutsats är att om man önskar minska den effekt på dödlighet i befolkningen som dessa källor orsakar bör man fokusera på emissionerna från förbränning snarare än slitagepartiklar. När det gäller slitagepartiklar behövs det mer forskning kring hur dessa påverkar dödlighet och sjukdom. Vid utvärdering av olika åtgärder är det också viktigt att ta hänsyn till den regionala påverkan som emissioner bidrar till samt hur olika åtgärder påverkar andra emissioner, t.ex. buller. Dessutom är det önskvärt med mer forskning när det gäller den påverkan på sjukdomar som partiklar ger upphov till – både vilka hälsoeffekterna är och hur dessa ska värderas. Slutligen, alla de delar som dessa beräkningar bygger på är resultaten av empirisk forskning där det fortfarande finns osäkerheter. Detta innebär att den kostnad vi beräknat reflekterar nuvarande kunskapsläge och den kan ändras över tid.

1 Introduction

1.1 Project description

1.1.1 Background and purpose

It has long been recognized that emissions from traffic have a negative impact on human health. In latter years there has been emerging consensus that the main influence is due to particulate matter (WHO, 2006). From an economic point of view these negative effects are external costs caused by traffic that, if not accounted for in decision making regarding transport, will result in a non-optimal allocation of resources leading to welfare losses. There are however various measures in place aimed at reducing the negative health impact (i.e. the external costs) of the emissions from traffic. The measures include emission control legislation but also air quality objectives for local concentration levels in urban areas that if exceeded compels the local authorities to take action. Also road pricing measures are increasingly considered as an option since the new information technology has opened up for new technical solutions. One such example is the Stockholm trial where rush hour road pricing was implemented, resulting in reduced traffic to and within the city area and thereby reductions in emissions and concentration levels (Johansson et al., 2008).

To be able to implement road pricing measures, but also for the evaluation of other control measures through benefit-cost analysis, information on the external cost of traffic emissions is needed. In the Impact pathway approach (IPA), that has been developed in the ExternE projects, the external cost is calculated as the product of exposure, effect and value (Friedrich and Bickel, 2001). In this study the effect we focus on is health impacts (mortality) but the same approach is also used to calculate the cost for other effects such as crop yield. All inputs are the result of ongoing empirical research and they are all related to uncertainties, hence the external cost that is calculated is not “the” cost. Regarding particulate matter (PM) there is for example recognition among the research community that there are different types of PM and that it is likely that their impact on human health differ. Still the current practice is to treat fine PM (which are considered to be most detrimental to health) as equally harmful irrespective of origin. Hence, there is only one function used for the health impact of fine PM (so called $PM_{2.5}$).

What is mostly measured in urban areas however is the concentration of PM_{10}, which contain both fine and coarse PM, since the air quality guidelines that have been in use up to June 2008 is based on these[1]. The most important local source of PM_{10} in many urban areas in Sweden is coarse PM from road wear (Johansson et al., 2007; Norman and Johansson, 2006; Omstedt et al., 2005). In spring, when the roadways are dry, the contribution from road wear PM may be 30 times the direct emissions from the exhaust pipe. These mechanically generated road dust PM are however not considered in calculations of the external cost that is based on the original ExternE-methodology (Friedrich and Bickel, 2001; Bickel and Friedrich, 2005).

In TESS the purpose has been to investigate how important the external health cost of traffic generated PM are in relation to the cost of other sources of PM. To do this we have both investigated how the exposure varies between sources but also assessed if it is

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[1] The new Directive 2008/50/EC of the European Parliament and of the Council of 21 May 2008 on ambient air quality and cleaner air for Europe, which includes standards for PM2.5, entered into force on 11 June 2008 (http://ec.europa.eu/environment/air/quality/legislation/existing_leg.htm).

reasonable to assume that the impact differs between PM from different sources. Recent research studies, which in various ways have tried to estimate the separate impact of traffic exhaust emissions on health, have found larger effects than studies using $PM_{2.5}$ (Forsberg, 2008). One reason for this could be that although the mass concentration of exhaust PM is small the exhaust emissions largely contribute to the number of PM in urban air (Johansson et al., 2007) or that they are much more toxic than the particle fraction that dominate the $PM_{2.5}$ levels (Schlesinger et al. 2006). Nitrogen oxide concentrations are highly correlated with the number of exhaust PM (Gidhagen et al., 2004; Olivares et al., 2007), therefore NO_x or NO_2 can be a good indicator for the exposure to particle exhaust emissions. In the present study we have also investigated the influence of local traffic and other sources on a regional scale, for example by studying secondary PM formation due to local emissions of NO_x, SO_2 and NH_3. In the project we have used Stockholm as a case study and we have focused on mortality since this is the health impact that has been found to have the largest impact on health cost in other studies (Bickel et al., 2006; Nerhagen et al., 2005)[2].

1.1.2 Particulate matter concentrations in Stockholm – an overview

The measured concentrations of fine PM and PM_{10} in an urban area is composed of several types of PM. Primary PM such as combustion PM from different sources and non-exhaust PM from road wear are mainly due to local sources while secondary PM, which are formed through chemical transformation of gaseous emissions, to a large extent are transported in from other areas. Therefore it is not possible to assess the actual impact on health from local traffic emissions using measurement data of the total concentrations. The impact of different contributions to the total PM_{10} concentrations at street canyon and urban background in central Stockholm is illustrated in Figure 1.

[2] As discussed in Nerhagen et al., (2005) there is also less research done on morbidity endpoints and hence less is know both about what exposure-response functions and about what economic values that are relevant to use .

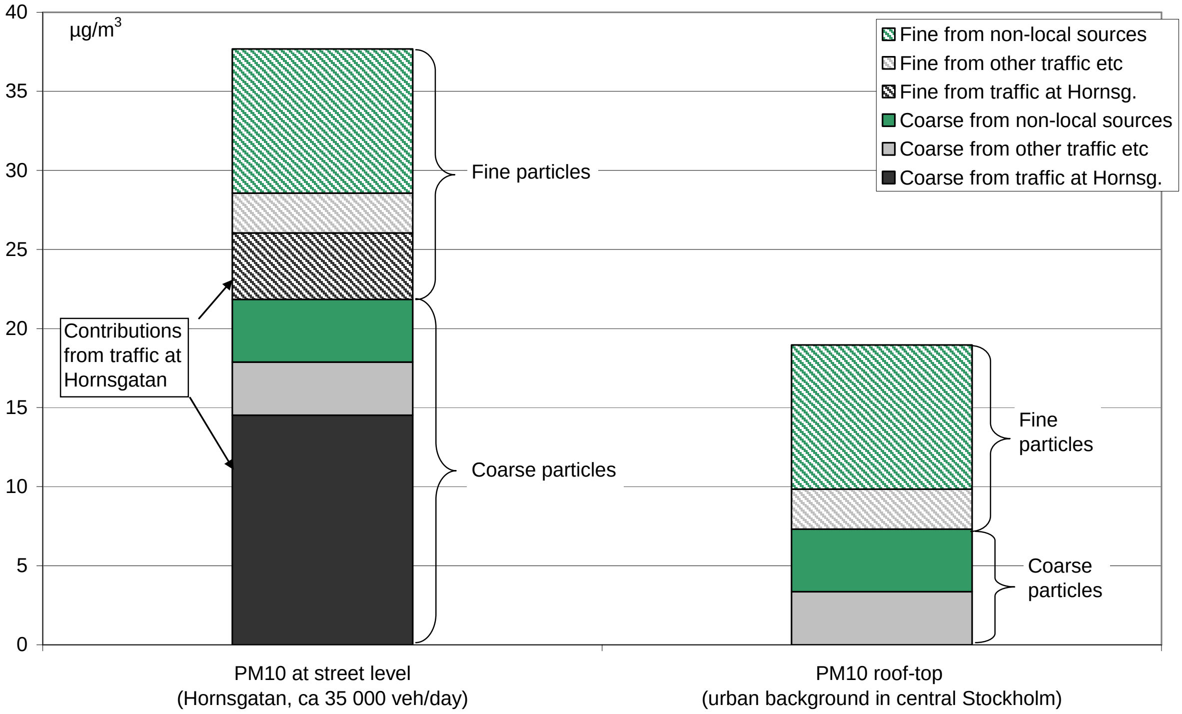

Figure 1 The relationship between the contribution from traffic and other sources to the PM_{10} concentration at a densely trafficked site (Hornsgatan) and at urban background (a roof-top site) in central Stockholm (annual mean contributions). "Fine" PM refers to $PM_{2.5}$ and "Coarse" PM to PM_{10}- $PM_{2.5}$. Source: Johansson and Eneroth, 2007.

Hence, in order to undertake analysis of the influence of traffic emissions on human health dispersion models are needed. There is however an additional problem with the current measurement on which Figure 1 is based. If we are only interested in exhaust PM from local traffic, measurements or modelling of $PM_{2.5}$ are not relevant. This is because exhaust PM consist mainly of ultrafine PM (with diameters <0.1 µm) and hence their contribution to the concentration of $PM_{2.5}$ is small (Johansson et al., 2007). Therefore, in TESS we will not base the analysis on $PM_{2.5}$ or PM_{10} but we will model the contribution of *combustion (*exhaust*)* and *non-exhaust* (road-wear) and *secondary* PM.

1.2 Content of report

The Impact pathway approach is a bottom-up approach where the emissions are traced from source to those exposed (Friedrich and Bickel, 2001; Bickel et al., 2006; Muller and Mendelsohn, 2007). Since emission sources in Stockholm contribute to PM_{10} concentrations both locally and on a regional scale, we have modelled emissions of the combustion process, the contribution from road wear (non-exhaust PM) and also the formation of secondary PM from NO_x, SO_2 and NH_3. Previous research has shown the importance of accounting for the location of the source in relation to those exposed, particularly for traffic and residential heating, since the concentrations decrease rapidly with distance to the source (Nerhagen et al., 2005; Bickel et al., 2006). In this paper we have therefore used high resolution emission- and population data to model the exposure on a local scale. For the regional scale, we have used a relatively high spatial resolution for the area surrounding Stockholm, and a somewhat coarser resolution for other parts of Europe. The emissions sources included in the paper are the energy sector, road traffic, residential heating, and the maritime transport that takes place within Stockholm[3].

In this paper we deduct the contribution from one emission source at a time to see what impact this has on total concentrations locally and regionally. However, opposite to what is done in for example Muller and Mendelsohn (2007) we reduce the yearly emissions from each source and not the marginal contribution such as one kg or one ton. The main reason for this is that both effect estimates (i.e. the exposure-response functions) and values normally used in the cost calculations are linear relationships and hence do not change with the size of the change in concentration levels (see Bickel et al 2006 for a detailed discussion). In theory the impact and value functions could be dependent on the magnitude of the emission changes and/or the concentration levels, but in practice we do not have such detailed information. The exposure-response functions, for example, are not derived from the impact of marginal changes in concentration levels over a short period of time. Moreover, the meteorology used represents the conditions for a particular period and these conditions vary from year to year. Furthermore, the exposure calculations are based on the concentration levels where people live (official residence addresses) and this will only be an approximation of their true exposure. Finally, although economists are usually interested in the costs resulting from a marginal change in quantity, what is to be considered as marginal in the air pollution context is not strictly defined. The traffic in Stockholm for example only makes a minor contribution to the total NO_x emissions in Sweden and Europe and hence reducing them could be considered a marginal change[4].

The outline of the paper is as follows. In the next chapter we briefly describe the methods and inputs on which the calculations are based and the resulting data that will be used in the cost calculation. In part three we present results of the calculations including sensitivity analysis for some inputs followed by a discussion on what the implications are for policy and further research.

[3] The emission data for shipping only includes part of the shipping that takes place in the Stockholm area (ferries and merchant ships).

[4] As long as the assumptions of linearity are correct whether or not we explore the impact of a small change in emissions (a kilo) or a large (several tons) is not of importance. However, for emissions that contribute to the formation of secondary pollutants such as nitrates or ozone the size in emissions may be of importance since these are non-linear relationships. Hence, reducing only one with a small amount can influence the amount of other pollutants in a different way than if the change is large. We however expect this effect to be minor since the amounts we have accounted for in these calculations are relatively small.

2 Methods

2.1 The Impact Pathway Approach

IPA is a bottom-up approach where the calculated external cost is a function of what influence a certain emission has on human health, and the value of this health impact. The following equation, (which is a modification of an equation in Ostro and Chestnut, 1998), describes the yearly external health cost due to PM from a change in emissions from a specific source:

Health cost = Δyearly exposure · effect · value

$$= (\Delta PM_{a;i} \cdot POP) \cdot (B_{a;j} \cdot P_{i;j}) \cdot V_j \qquad (1)$$

where

$\Delta PM_{a;i}$ =	change in annual average exposure for pollutant i (µg/m3)
POP =	population exposed to $\Delta PM_{a;i}$
$B_{a;j}$ =	baseline annual health impact rate in population for health impact j (number of cases per inhabitant)
$P_{i;j}$ =	effect on health impact j per µg/m^3 of pollutant i (relative risk)
V_j =	value of health impact j.

This calculation has to be done separately for each type of PM since the health impacts are likely to differ. The cost calculated for each pollutant and each health endpoint can then be added up to arrive at the total external cost for each source. The determinants of the final cost are the outcome of the exposure calculation, the assumptions in the effect estimation (especially the exposure-response function P) and the value used. In the following, we will discuss each of these inputs and how they have been derived in this paper.

2.2 Dispersion and exposure modeling

2.2.1 Emissions and local exposure

This part of the project has been undertaken by SLB analys at the Stockholm Environment and Health Protection Administration. In this part we give an overview of the work done and of the results; details regarding the data and the modelling are given in a separate report (Johansson and Eneroth, 2007). The modelling is based on the Airviro Air Quality Management system (http://airviro.smhi.se)[5]. SLB analys has been involved in air quality modelling research for many years and the models used in this

[5] If the interest is in the contribution of local sources to air pollution measurement data cannot be used since such measurements only capture the total concentration level within a city. Measurements will be influenced by a number of local and regional sources, see discussion in section 1.1.2.

paper have been continually validated against measurements at air quality monitoring stations in Stockholm and Uppsala (a neighbouring city) (Johansson et al., 1999; Eneroth et al., 2006).

To be able to calculate the local exposure of air pollutants information about the emissions from each source and the population within Stockholm is needed. Temporally and spatially resolved emissions are available from the emission inventory of the Stockholm and Uppsala Air Quality Management Association (http://www.slb.nu/lvf). Information in this data base has been updated yearly since 1993. All data used in the dispersion modelling is for the year 2003 which was a year with quite normal meteorological conditions.

Emissions from road traffic are based on traffic data for separate road links. The road links are classified into 45 different road types depending on signed speed limit, the percentage heavy duty traffic and the temporal variation of the traffic. Emission factors for exhaust emissions from road traffic are obtained from the EVA model of the Swedish Road Administration. Emission factors for non-exhaust PM (mainly road wear but including some contributions from brake wear and tyre wear) were obtained from measurements in a street canyon using NO_x as tracer (Ketzel et al., 2007). Emissions from sea traffic as well as energy production are describes as point sources. The emission data for sea traffic includes data on merchant ships and ferries that call at Stockholm ports[6]. Emissions from energy production plants are updated yearly by the supervisory authorities.

The emissions from residential heating are divided into oil-heating and wood combustion, and are described as grid sources (1 x 1 km). The emissions are based on regional fuel statistics from Statistics Sweden[7]. Information about the distribution of environmental certified and not environmental certified wood boilers – with or without accumulator tanks – is based on the annual report of chimney sweeping 2002. The grid sources are distributed according to population statistics in areas where district heating is missing. The emissions from wood combustion are associated with uncertainties due to limited knowledge about the heating appliances used and their emission factors.

In the emission inventory we have also included the emissions from off-road vehicles although they are not included in the final analysis. They have not been included in the final analysis since these are diffuse emission sources and therefore their contribution to exposure is unclear. However, they could be of importance since they make a relatively large contribution to total emissions of combustion PM.

Table 1 presents total emissions of NO_x and PM in the Greater Stockholm area. Even though we are not interested in NO_x as a pollutant, we have included information about these emissions because it is a proxy for the number of PM emitted from different sources, but also because it makes an important contribution to secondary PM[8]. Road

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[6] The data on sea traffic has been obtained from the County Administrative Board in Stockholm which in turn have based their estimates on a survey by MariTerm AB and the Institute of Shipping Analysis, ”Kartläggning av Östra Mellansveriges hamnkapacitet – Kartläggning och analys”, 2001 (in Swedish).

[7] New values have been included in the emission database in 2006. Regarding PM, these are 1/5 of the previous emission factors while for NOx the new emission factors are slightly higher than those used previously.

[8] For the regional scale modeling information on emissions of other pollutants from these sources were also used. The main inputs in the calculations were SO_x (40 tonnes from LDV, 2 tonnes from HDV, 182 tonnes from sea traffic, 1642 from power plants and 299 tonnes from residential heating),

traffic dominates the emissions of both PM and NO_x. Regarding PM, the largest contribution comes from non-exhaust PM. For NO_x and combustion PM we have made separate estimations for light duty vehicles[9] (LDV) and heavy duty vehicles (HDV). The energy production sector and off-road vehicles are important NO_x sources.

Table 1 Total emissions (tonnes/year) of NO_x and PM from road traffic and other sources in Greater Stockholm area during 2003.

	Road traffic	Sea traffic [a]	Power plants	Residential heating	Off-road vehicles	Other	Sum
NO_x (LDV)	3029						3029
NO_x (HDV)	2645						2645
NO_x (other sources)		885	2002	487	1913	231	5518
NO_x Total	**5674**	**885**	**2002**	**487**	**1913**	**231**	**11192**
Combustion PM (exhaust LDV)	*82*						**82**
Combustion PM (exhaust HDV)	*40*						**40**
Combustion PM (other sources)		33	249	98	110	46	**536**
Non-exhaust PM (road, brake and tyre wear[b])	1859						**1859**
PM_{10} Total	**1981**	**33**	**249**	**98**	**110**[c]	**46**	**2517**

[a] Only emissions from merchant ships and ferries that call at ports are included.

[b] Only non-combustion PM with a diameter <10 μm. The size is mainly larger than 1 μm.

[c] Only combustion PM due to the use of diesel fuel.

The outcome of the exposure calculation will have an important influence on the calculated cost. We have previously investigated what impact the assumptions regarding the spatial resolution and the population density close to the emissions source will have on the estimated exposure (Nerhagen et al., 2005). Using changes in average concentration levels for larger areas with no account of peoples' location would underestimate the exposure. Hence, in this study population weighted concentrations were obtained by combining the dispersion calculations with gridded population data.

The dispersion calculations were performed with a Gaussian dispersion model, part of the Airviro Air Quality Management system. The model calculation height is 2 m above ground level in open areas, 2 m above roof height over cities. That is, the exposure calculations in this study correspond to roof level concentrations since the exposure-response functions from epidemiological studies are based on such concentrations. The calculation area covers the Greater Stockholm area (35km x 35 km) with a spatial resolution of 100m x 100 m (122 500 receptor points). The gridded (100 x 100 m) population data was obtained from Statistics Sweden. In 2003, 1 405 600 people lived within the 35km x 35km calculation grid.

NMVOC (6720 tonnes from LDV, 227 tonnes from HDV, 1101 tonnes from sea traffic, 265 tonnes from power plants and 415 tonnes from residential heating) and finally NH_3 (371 tonnes from LDV).

[9] Included in this category are all vehicles (including cars) smaller than 3.5 tonnes.

Table 2 shows calculated yearly average concentrations of NO_x and PM. These results tell us what impact reductions of different emission sectors would have on the yearly average concentration levels in Stockholm[10]. Road traffic makes the largest contribution both to the NO_x and the PM concentrations. Non-exhaust PM dominates the contribution to the total PM concentration. There is however not a one to one correspondence between changes in emissions and changes in concentrations as the dispersion pattern will depend on where the emissions occur and at what height. For example, changing the emissions from power plants have less influence on local concentration levels in Greater Stockholm since they are emitted at higher altitudes. Therefore, residential heating makes a larger contribution to the NO_x concentrations than power plants, although the latter have higher total emissions. The contribution of each source to the total concentrations in the Stockholm area is however relatively small. These concentrations can for example be compared with the rural background concentrations[11] for these pollutants which are around 10 $\mu g/m^3$ for PM_{10} and 4 $\mu g/m^3$ for NO_x. (e.g. SLB rapport 1:2004; LFV 2008:4).

Table 2 Annual mean contributions to the concentrations of NO_x and PM for all grid cells from different source sectors in the Greater Stockholm area ($\mu g/m^3$).

Substance	Road traffic	Road traffic, LDV[1]	Road traffic, HDV[2]	Sea traffic	Power plants	Residential heating
NO_x	2.46	1.32	1.14	0.16	0.26	0.37
Combustion PM	0.053	0.036	0.017	0.0052	0.037	0.07
Non-exhaust PM	0.79					

[1] LDV = light duty vehicles

[2] HDV = heavy duty vehicles

The population weighted exposure estimates that are the basis for the health impact assessment are presented in Table 3. The simulated concentration in each grid square (100m x 100m) has been multiplied with the grid square population. Hence, sources that cause high concentrations in densely populated areas will get higher estimates in relation to their emissions. For traffic, the population weighted estimates for NO_x and PM in Table 3 are about twice as high as the estimates in Table 2, while for sea traffic it is only 20% higher. The latter is due to the fact that sea traffic emissions are located to the east of the most populated areas and that prevailing winds are westerly.

[10] It should be noted that the annual mean contributions to concentrations of NO_x and PM include all grid cells in the Greater Stockholm area, i.e. also many grid cells where the contribution from local sources are very small, making the mean of all cells quite small. The maximum contributions are due to traffic emissions and occur in central Stockholm. Here the calculated contributions (at roof level, not street canyons) are around 20 $\mu g/m^3$ for NO_x and 3-5 $\mu g/m^3$ for PM_{10}.

[11] Rural background concentrations are those that are valid for a large area outside the cities and they are due to both natural and man-made emissions.

Table 3 Population weighted annual mean concentrations of NO_x and PM from different source sectors in the Greater Stockholm area ($\mu g/m^3$).

Substance	Road traffic	Road traffic, LDV[1]	Road traffic, HDV[2]	Sea traffic	Power plants	Residential heating
NO_x	5.68	3.14	2.44	0.18	0.36	0.68
Combustion PM	0.14	0.10	0.036	0.0063	0.051	0.12
Non-exhaust PM	1.70					

[1] LDV = light duty vehicles

[2] HDV = heavy duty vehicles

Road traffic is the most important source for human exposure to PM and to NO_x. Non-exhaust PM clearly dominates regarding exposure to PM. It can also be concluded that residential heating is an important source for exposure to combustion PM. The reason for the importance of these two sources is that these are emissions that occur in close proximity to where people live. However, the emission data for residential heating is uncertain. In this study we have used new estimates that give 1/5 of the PM-emissions from earlier studies.

2.2.2 Regional impact

The second part of this study, the estimation of the dispersion and exposure on a regional scale, has been undertaken by the Swedish Meterological and Hydrological Institute (SMHI). Details regarding the modelling can be found in Bergström (2008). The MATCH (Multi-scale Atmospheric Transport and Chemistry) model was used for the regional modelling. MATCH has participated in many international studies where different models have been evaluated against observational data and each other (see, e.g., van Loon et al., 2007; Hass et al., 2003). Detailed information about the model can be found in Andersson et al, 2007.

Primary PM emitted in Stockholm will, to some extent, be transported out of the city and contribute to the PM levels in the region surrounding the city but also other parts of Europe. In addition, the gaseous NO_x, SO_x and NH_3 emissions also contribute to PM exposure since they are chemically transformed into secondary inorganic PM, that is, particulate nitrate, sulphate and ammonium[12]. Secondary PM may be more important outside the emission area since it takes some time for it to form. Since primary PM will have a larger influence on the neighbouring region, the model calculations to determine the impact of emissions in Stockholm were performed on two different scales/regions. For impacts closer to Stockholm, a smaller domain was used, with a horizontal resolution of 5 km; this high resolution domain is called the Mälardalen domain. A larger model domain covering most of Europe, with a horizontal resolution of about 44km, was used to study the impacts on a continental scale. The model areas used are illustrated in Figure 2. All calculations were performed for one complete year (meteorology from 2003 was used).

[12] Secondary *organic* particles were neglected in this study.

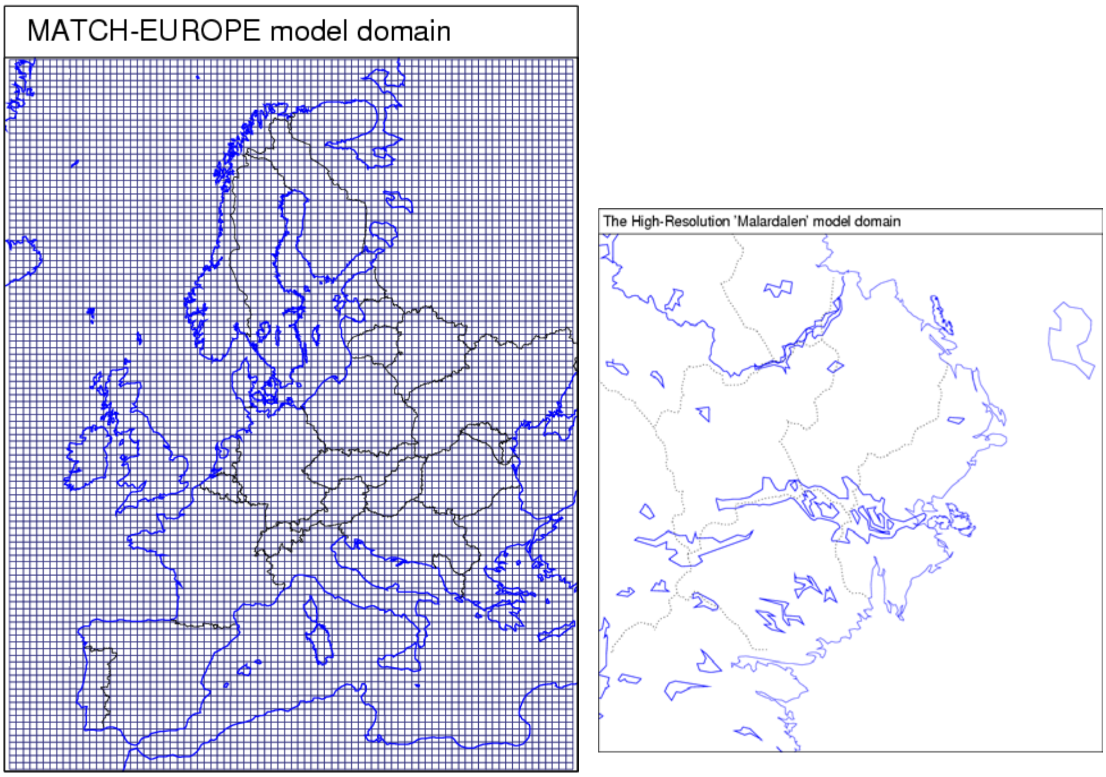

Figure 2 The modelling domains. Left: The European scale model domain with ca 44km horizontal resolution (each square represents one model grid cell). Right: The High-resolution, "Mälardalen" model domain; the domain is split into 59 × 60 grid cells, with 5km horizontal resolution in the Stockholm region. (Source: Bergström, 2008).

Emission data for Stockholm for the regional scale modelling were provided by SLB analys. To describe the chemical evolution of the emissions from Stockholm, and the resulting production of secondary PM, accurate emissions are also needed for the change in other emissions and the total emissions in the rest of Sweden and Europe. For Sweden NO_x, SO_x, VOC, CO and NH_3 emission data from the SMED (Swedish Methodology for Environmental Data, www.smed.se) project, with 1km resolution, were used. For emissions outside Sweden data from EMEP (www.emep.int) were used. EMEP data are provided with a horizontal resolution of 50km. All emission data were regridded to the two different calculation grids used in MATCH. Population data with approximately 1km resolution from the European Environmental Agency (EEA) was used for the EU countries and Croatia (EEA/JRC, 2006). For countries outside EU population data were taken from the Columbia University data base (CIESIN, 2005).

Examples of the resulting PM exposure from emissions in Stockholm on the regional scale are shown in Figure 3. The exposures have been calculated as the product of the number of persons in each grid cell and the average concentration resulting from emissions in Stockholm and are expressed in the unit # persons × $\mu g/m^3$. In the figures the importance of the population density within each grid cell is revealed. The high exposure areas are coloured red in the figure and they coincide with the larger cities within Europe. The emissions from Stockholm contribute somewhat to PM-exposure in most of Europe[13].

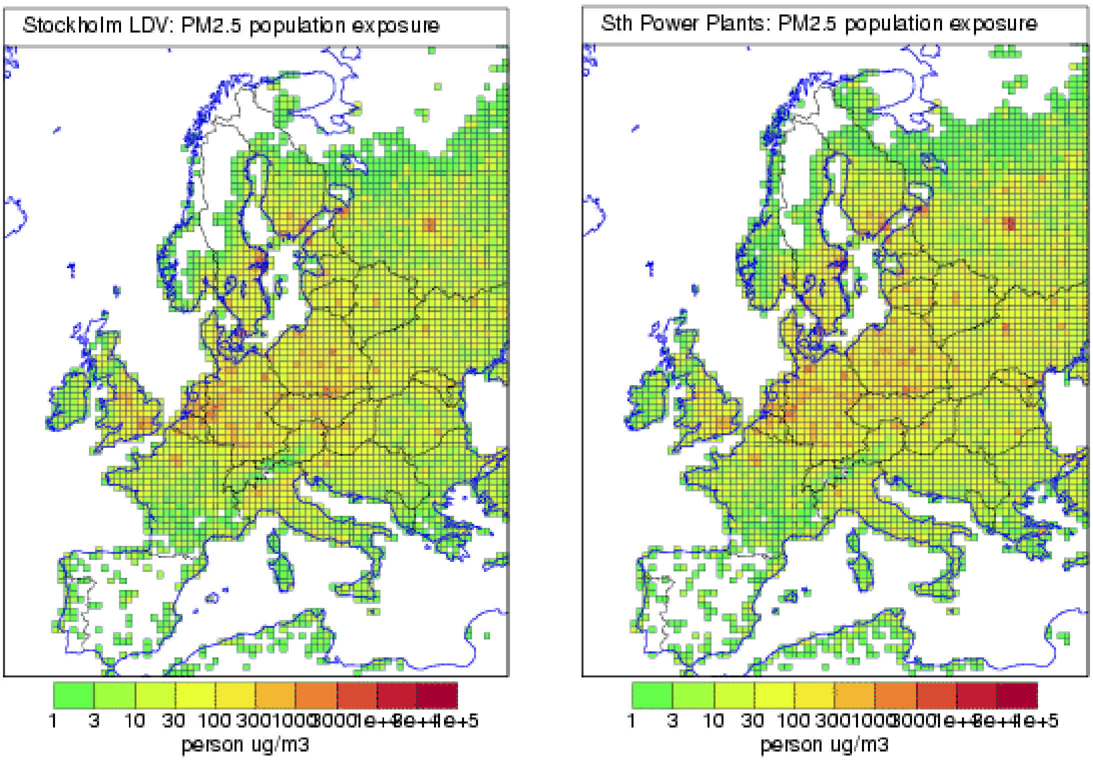

Figure 3 Yearly average population exposure to $PM_{2.5}$ (primary PM+secondary inorganic PM) from Light Duty Vehicle (left figure) and Power plants (right figure) in Stockholm. White areas are due to no or low population densities. (Unit:# persons × $\mu g/m^3$). Source: Bergström, (2008).

The final results of the calculations done by SMHI regarding PM are presented in Tables 4–6.

Table 4 gives exposure estimates for population within the Stockholm area (that is, the emission area). This also includes results from the local scale modelling by SLB analys. In addition to the primary PM that was accounted for by SLB analys, SMHI has also modelled the contribution from secondary PM (particulate nitrate, sulphate and ammonium) due to the emissions from NO_x, SO_x and NH_3. We find that non-exhaust PM has the largest impact on human exposure within the Stockholm area. However, we find that secondary PM due to the local $NO_x/SO_x/NH_3$ emissions also has a non-negligible impact on the local exposure in Stockholm. The relatively high contribution from LDV, compared to HDV, to particulate nitrate is due to fairly large ammonia (NH_3) emissions from LDVs with catalytic converters. The NH_3 reacts with nitric acid (HNO_3) in the city air to form particulate ammonium nitrate. However, we are uncertain

[13] It is to be noted that the information presented in Figure 3 is the population exposure in Europe due to emissions in Stockholm. Hence, the results reflect the combined effect of contribution in concentration and population density. Although westerly winds results in somewhat higher concentrations in for example Finland than the rest of Europe (although this effect is not as large as often assumed) this difference is not seen in Figure 3 due to differences in population density.

about this result and think it needs to be verified by more measurements and modelling.[14]. Residential heating seems to be an important source for exposure to combustion PM but it does not make a large contribution to the formation of secondary PM.

Table 4 Population exposure in Greater Stockholm from different PM components (person $\mu g/m^3$).

PM component	Road traffic	Road traffic, LDV[a]	Road traffic, HDV[b]	Sea traffic	Power plants	Residential heating [c]
Combustion PM	191 500	141 000	50 500	9 000	71 500	167 000
Non-exhaust PM	2 190 000					
Secondary PM[d]	117 000	115 500	1 500	12 000	11 500	28 500
Total pop. exposure	2 190 000	256 500	52 000	21 000	83 000	195 500

[a] LDV = light duty vehicles

[b] HDV = heavy duty vehicles

[c] The combustion PM and NOx emissions from residential heating are very uncertain which means that the population exposure values for this sector are uncertain as well.

[d] This is the exposure due to particulate nitrate, particulate sulphate and particulate ammonium. Of these, particulate nitrate in in most cases makes the largest contribution to population exposure. For details see Bergström (2008).

Table 5 contains the modelling results for the Mälardalen region (excluding the studied emission area of Stockholm). The population in this region was about 2 160 000 people. As seen from the table, the influence from the emissions in Stockholm on exposure is much lower in this area. This is due to two main factors, lower concentration levels and lower population densities. As for the Greater Stockholm area, non-exhaust PM gives rise to the largest exposure.

Table 5 Population exposure in Mälardalen from different PM components (person $\mu g/m^3$).

PM component	Road traffic	Road traffic, LDV[a]	Road traffic, HDV[b]	Sea traffic	Power plants	Residential heating [c]
Combustion PM	7 500	5 000	2 500	4 500	2 000	6 000
Non-exhaust PM	196 000					
Secondary PM[d]	15 000	12 500	2 500	2 000	5 500	2 500
Total pop. exposure	196 000	17 500	5 000	6 500	7 500	8 500

[a] LDV = light duty vehicles

[b] HDV = heavy duty vehicles

[c] The combustion PM and NO_x emissions from residential heating are uncertain which means that the population exposure values for this sector are uncertain as well.

[d] This is the exposure due to particulate nitrate, particulate sulphate and particulate ammonium. Of these, particulate nitrate in most cases makes the largest contribution to population exposure. For details see Bergström, 2008.

[14] This is a somewhat unexpected result and we have tried to verify these results with measurement data. The latter indicates that local traffic makes a small contribution to particulate ammonium nitrate. However, since this is a pollutant taking part in chemical processes it is more problematic to capture the contribution through measurements. Hence, further research is needed in order to clarify what the actual contribution from traffic is likely to be.

Table 6, finally, contains the estimated exposure for the rest of Europe. Although the contribution to the concentration levels on the continent of emissions from the Stockholm area is small, the estimated exposure is large when compared to population exposure in Greater Stockholm and Mälardalen, due to the number of people exposed. In this case however, the most important contribution to exposure is due to secondary PM (where particulate nitrate makes the largest contribution). For many of the sources, the total exposure on this scale is even higher than the exposure within Greater Stockholm (see Table 4). For residential heating however the impact on this scale is relatively small which is due to relatively small NO_x -emissions. We also find that non-exhaust PM also has an impact on this scale. Hence policy measures, such as road charging, that reduce emissions locally in the Stockholm area will have an important impact on human health in other parts of Europe.

Table 6 Population exposure in the rest of Europe from different PM components (person $\mu g/m^3$).

PM component	Road traffic	Road traffic, LDV[a]	Road traffic, HDV[b]	Sea traffic	Power plants	Residential heating [c]
Combustion PM	6 000	4 000	2 000	3 500	13 000	4 700
Non-exhaust PM	133 000					
Secondary PM[d]	300 000	187 000	113 000	56 500	176 000	39 000
Total pop. exposure	133 000	191 000	115 000	60 000	189 000	43 000

[a] LDV = light duty vehicles

[b] HDV = heavy duty vehicles

[c] The combustion PM and NOx emissions from residential heating are uncertain which means that the population exposure values for this sector are uncertain as well.

[d] This is the exposure due to particulate nitrate, particulate sulphate and particulate ammonium. Of these, particulate nitrate in most cases makes the largest contribution to population exposure. For details see Bergström, 2008.

A summary of these results is presented in Figure 4. As seen in the figure non-exhaust PM from road traffic (mainly road wear and dust) have the largest influence on human exposure. This is due to the large amount of emissions. The PM exposure of the other sources is due the burning of fuels. In the figure it is seen how much exposure is due to combustion PM (primary emitted PM) and how much is due to secondary PM. The latter mainly has an influence on populations outside of Stockholm. While road traffic LDV and residential heating are important sources for exposure to combustion PM locally, the main influence of the other sources is in the rest of Europe. (According to this figure road traffic LDV also make a large contribution to exposure to secondary PM but as mentioned previously, this is a result that needs to be investigated further.) This figure however does not reflect the relative harm done by each pollutant since they are expected to have different health impacts. Hence, as will be shown in the next section, non-exhaust PM is not likely to have such a large impact on mortality as this figure might suggest.

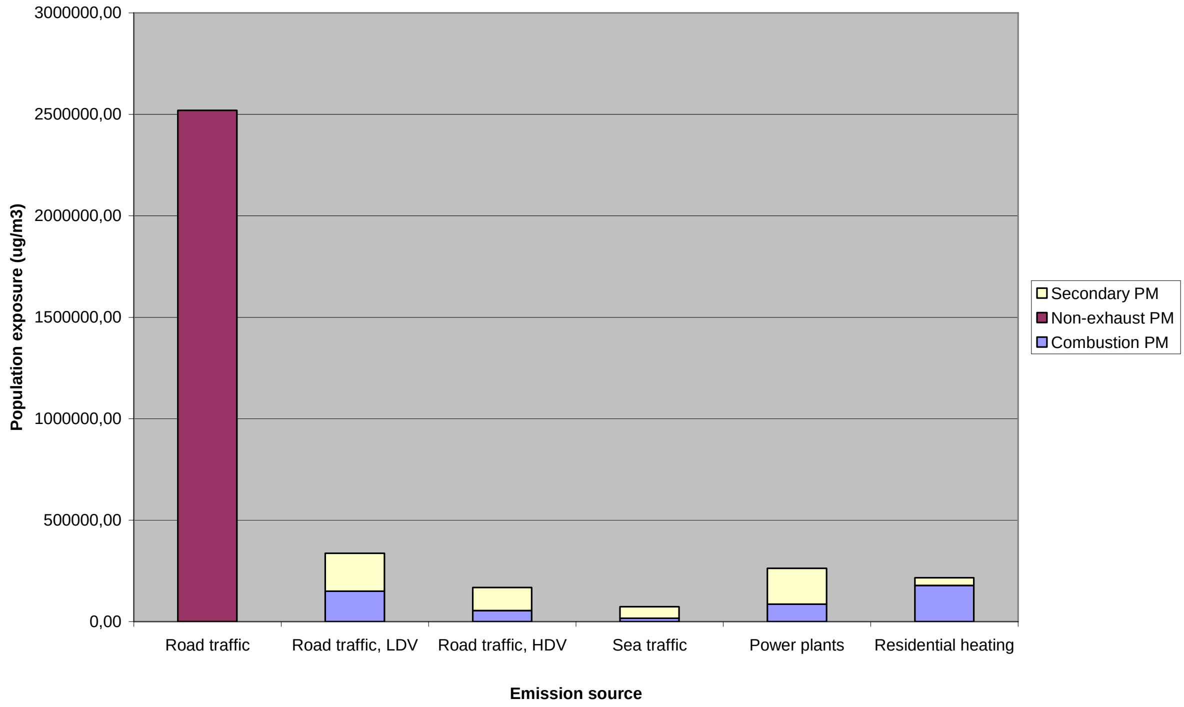

Figure 4 Total population exposure in Europe due to PM from emission sources in Stockholm (person $\mu g/m^3$).

2.3 Estimation of health impacts

2.3.1 Methodology

The second step in the IPA is the health impact estimation that, according to equation 1, is composed of two parts; a baseline estimate and an exposure-response estimate. The inputs used in these calculations are mainly obtained from epidemiological studies. They have, to some extent, been validated by toxicological studies. Both mortality and morbidity effects have been assessed in different studies. The World Health Organization (WHO) repeatedly reviews the current level of knowledge and issues Air Quality Guidelines for different pollutants and also relevant exposure-response (ER) functions. These are general estimates (not country-specific). For mortality there is a distinction made between deaths that will occur due to long-term exposure (chronic mortality) and those that occur directly after exposure (acute mortality). If both of these relationships have been found for the same pollutant, there is a risk of double counting and hence it is often assumed that the acute cases are accounted for in the calculation of chronic mortality (Friedrich and Bickel, 2001; Bickel et al., 2006; Muller and Mendelsohn, 2007). The health effect baseline incidences are the number of health events per year per unit of population. These estimates are impact- and country-specific.

To arrive at relevant baseline and ER-functions to be used in this study, Forsberg (2008) has undertaken a literature survey of peer-reviewed studies. Since most external cost studies have found that mortality effects give rise to the highest costs, we have focused on such studies. The current WHO guidelines contain an exposure-response function for $PM_{2.5}$ assuming a long-term effect on mortality of 1.06 (6%) for a 10 $\mu g/m^3$ increment of average concentration levels at roof level. No threshold effect is assumed. It is also stated that the same ER-function should be used for the finer fraction irrespective of origin; hence it is assumed that different PM components are equally harmful. This is also the ER-function used in most assessments, such as the Clean Air for Europe (CAFE) calculations. Regarding the coarser fraction ($PM_{10-2.5}$), these have not been included in the ExternE Methodology Update 2005 (Bickel and Friedrich, 2005). There is however an ongoing discussion among the research community of whether or not it is correct to assume that all PM have the same impact irrespective of origin or size. Hence, there is research undertaken that tries to assess the health impact of different PM components.

The epidemiological studies from which ER-functions are obtained have mainly focused on short-term effects of exposure. Up to now there have only been a few cohort studies (with panel data) from which long-term effects can be assessed. Therefore, although the long-term effects may be dominant, the results from short-term effect studies can give useful indications of differences in harmfulness between different PM components. Another problem with epidemiological studies is that they build on measured mass concentration of PM_{10} or $PM_{2.5}$, both influenced by several sources. Therefore, so-called indicator variables have been used in recent studies to assess the influence of local sources such as traffic. In the traffic environment NO_x has been found to be highly correlated with the number of exhaust PM and have therefore been used as and indicator of combustion PM from traffic in some studies. Moreover, studies based on within-city gradients of PM have been used to try to exclude the impact of interurban differences in PM composition.

Another issue concerning the health impact assessment, which is of importance for external cost calculations of mortality impacts, is in what ”unit” the calculation is done. In epidemiological studies it is common to assess the number of cases or the number of deaths in a population. However, for the economic assessment the number of years lost due to the health impact is of importance. For results from cohort studies this information can be obtained by studying the mortality or survival in different ages. Calculations of potential years of life lost (YOLL) are done with known life table, usually assuming that the relative risk per unit increase in exposure affects the mortality in every age class in a population in the same way. This way the potential years of life lost due to excess mortality from the population exposure can be given per calculated excess death or divided by the number of persons in the population as a mean value. The same approach can however not be used for the results from time-series studies. Here the estimates used are rather based on the condition of people who have died from the health conditions that are caused by air pollution.

2.3.2 Baseline and ER-functions used[15]

In this project we do not apply different baseline rates for different cities, regions or countries. The reasons for this are that the exposure calculation does not make a separation between different regions or countries and nor do we want the mortality effects to be modified by recent mortality rates. Therefore, the baseline mortality rate used in the study is the one relevant for Sweden of 1010 deaths per year from non-external causes per 100 000 persons. This estimate is obtained from The Swedish Board of Health and Welfare (register unit EpC). Moreover, although it is assumed in some studies that no deaths occur among those younger than a certain age (often 30 years), we

[15] For a detailed discussion and references, see Forsberg (2008).

have not made that assumption. This is because there are some studies pointing at an impact on infant mortality and also because inclusion of the younger age groups only has marginal impact on the estimated results.

The studied literature indicates indeed that there are differences in the harmfulness between different PM components. Several studies suggest that vehicle exhaust and other combustion PM are strongly related to mortality. Also concentrations of secondary PM, especially sulphate, are statistically important for mortality according to American cohort studies. Moreover, the studies based on within-city gradients of PM, seem to find a higher relative risk per concentration unit of combustion PM (or the indicator variable). A study that used data from the American Cancer Society (ACS) gives a relative risk of 17% per 10 $\mu g/m^3$ $PM_{2.5}$ (Jerret et al., 2005). Our conclusion is that this is, especially for combustion PM, a realistic alternative in a sensitivity analysis to the 6% per 10 $\mu g/m^3$ suggested by WHO. Regarding nitrate there are few studies done on this pollutant. Since it is unknown what exactly the harmful component in secondary PM is, we have chosen to treat nitrate as equally harmful as sulphate, and hence use the ER-function suggested by WHO for $PM_{2.5}$. An alternative assumption is to assume half this effect, as suggested for example in the most recent ExternE recommendation (2005). In our calculations we have also included ammonium nitrate since this pollutant is always found in measurements of secondary inorganic aerosols.

Regarding non-exhaust PM, there is little support from cohort studies of an effect on mortality. In time-series studies a short-term effect on daily mortality has been found but it is usually more strongly related to finer PM. In line with these findings preliminary results from another research-project undertaken in Stockholm (TRAPART) indicate non-significant short-term effect on mortality from road wear[16]. Thus non-exhaust PM may be seen as having no effect on mortality. We have therefore decided to assume, as a higher estimate, that non-exhaust PM have the same short-term effect on mortality as PM_{10} in general. Since studies such as APHEA-2 (Zanobetti et al., 2002) have shown that the short-term effect in fact lasts over several weeks, we chose as upper estimate to assume a cumulative effect of 1% increase in all cause non-external mortality per 10 $\mu g/m^3$ and as a lower bound no impact on mortality. The latter however is an underestimate of the effect of this pollutant since it is known that they worsen the conditions for persons with respiratory illnesses and other morbidity endpoints. Still we have chosen not to include morbidity endpoints because both exposure-response functions and values are uncertain.

The relative increase in mortality per unit increase in exposure from cohort studies can be assumed to effect mortality in every age class in the population in the same way with a known life table. The potential years of life lost (YOLL) due to excess mortality can then be calculated per excess death or divided by the number of persons in the population to have a mean value for the whole population. We found for Stockholm that every excess death due to long-term effects of PM corresponds to 11.2 YOLL. Time-series studies do not give any information on how much each excess death is brought forward. Therefore we have used the same assumption as in the CAFE calculations for short-term effects on mortality of PM_{10}, 1 YOLL (a reasonable range for this estimate is from 6 to 18 months).

[16] However, chance cannot be excluded as causing this result. If the time-series had been larger and the effect stayed the same, it would become statistically significant.

2.4 Economic valuation of health impacts

2.4.1 Methodology

The final input into the external cost calculation in equation 1 is the economic values placed on the estimated health impacts. Values used in these types of calculations usually have two components; the costs related to illness (medical treatment and lost production) and the value placed on the loss of welfare of being ill. While the former information can be obtained from register data, so called valuation studies are used for the latter. These are of two kinds, revealed preference studies that rely on observed behaviour and stated preference studies that use responses to hypothetical questions.

There are a number of issues discussed in relation to this type of valuation. It is for example acknowledged that the valuation of small changes in risk is one of the most challenging. Up to now there has also been a focus on the valuation of accident risk. Hence, more research is needed on what are the relevant estimates to be used in this context. More detailed discussions on the valuation of changes in risk can be found in Carson et al., (2001) and Andersson and Treich, (2008).

The theoretical basis for this type of analysis is that people are willing to trade money for reductions of risks and that this "willingness to pay" will be an approximation for the utility that is lost when in a state of bad health. The research undertaken in this area has mainly focused on obtaining values for the reduction of mortality risks. By dividing the willingness to pay for a risk reduction by the change in the probability of death, the value of a statistical life (VSL) is obtained. Hence, if the change in the risk of dying is 5/100 000 and the stated willingness to pay is 50 euro then the VSL is equal to 1 million euro (Boardman et al., 2006).

A particular problem for the cost calculations discussed in this paper is that most studies concerning VSL come from studies of working age people and hence apply to cases where several years of life are assumed to be lost. Since air pollution mainly have an impact on elderly, it has been questioned if the VSL should be applied uniformly to people of all ages or whether the VSL should be differentiated by age. In the ExternE Methodology report (Bickel and Friedrich, 2005) this issue is discussed at length. Previously the values used were obtained from available literature, such as estimates from the valuation of fatal transport accidents. These estimates were then adapted to account for the length of lifetime lost by changing the metric from VSL to the VOLY (value of life year lost) by annuity calculation. However, since this transfer rests on a number of questionable assumptions, a study was undertaken that account for both the air pollution context and the influence of years of life lost. It is a study that uses a survey instrument that has been used in previous American and Canadian studies (Alberini et al., 2004a) and that is based on the contingent valuation method[17].

2.4.2 Economic values used

In this study we have decided to apply the estimates suggested by Bickel and Friedrich (2005) since they are similar to those used in the cost-benefit analysis in CAFE (AEA

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[17] The conclusion to rely on VOLY is however not without critiques. Krupnick (2004) in a peer review of the methodology to be used in the cost-benefit analysis in CAFE questioned this and therefore the final work included both VSL and VOLY estimates. Hammit (2007) on the other hand states that a change in survival probabilities can be estimated by either estimate but that due care needs to be taken to other factors that influence these estimates (state of health, income etc.).

Technology, 2005). We use these estimates since no studies in Sweden have tried to estimate VOLY. The result from the valuation study undertaken within ExternE was a mean VSL of 2.258 million euro and a median estimate of 1.052 million euro (Alberini et al., 2004b). The change in risk on which this estimate is based was a 5/1000 reduction in the risk of dying during the next ten years (hence 5/10 000 reduction in risk of dying). The study was undertaken in three countries and the estimates differ between countries. The estimates are also associated with income, sex, education, old age and previous health record.

A VOLY estimate was obtained by conversion where each respondent's WTP was divided by that respondent's life expectancy extension that would result from the 5/10000 change in risk. The calculation of life expectancy extension was based on life table methods as suggested by Rabl (2002). The result of this recalculation was a mean estimate of 125 250 euro and a median estimate of 55 800 euro (price year 2003). The final recommendation is, contrary to previous Externe studies and other valuation studies, to use the median estimate, adjusted down to 50 000 euro, because this was the estimate was unaffected by distributional assumptions. This is assumed to be a correct estimate for chronic mortality and the undiscounted estimate to be used for acute mortality is 75 000 euro, using a discount rate of 3% (Bickel and Friedrich, 2005)[18].

In Sweden only VSL-values have been used in policy evaluations. It is the Swedish Institute for Communication Analysis (SIKA) that suggests values to be used in infrastructure planning. In the latest revision (SIKA, 2008), a mean value of SEK 22.3 million was suggested for VSL based on recent Swedish research (price level 2006). This is approximately equal to 2.4 million euro, hence about the same size as the mean VSL estimate obtained in the ExternE study. However, this is an estimate for traffic safety where on average approximately 38 years are lost (SIKA). Based on this information we can estimate an approximate VOLY for Sweden using annuity calculation:

$$VOLY=(VSL*(1+r)^t*r) / ((1+r)^t-1). \qquad (2)$$

The result is about 100 000 euro (using a discount rate of 4%) which is a higher VOLY than the median estimate recommended in ExternE (Bickel and Friedrich, 2005). Hence, the values we use in this study are conservative estimates and can be seen as a lower bound. However, we think more research is needed in order to determine what a reasonable value in this health context is. VSL studies in Sweden looking at private traffic safety devices have for example found much higher VSL estimates[19].

[18] These median estimates are lower than those previously used in ExternE-projects (that were based on mean VSL estimates from other studies). In the UNITE project VOLY for acute mortality 131 400 euro and VOLY chronic was 76 400 euro (Nellthorpe et al., 2001).

[19] According to economic theory VSL estimates for public projects should be higher than for private safety since the former encompass many more individuals. Swedish studies have however found the opposite result. One possible explanation for this is that since most health care in Sweden is paid for through taxes, people think that government should pay for important safety enhancing projects using funds collected by the existing taxes. This and other aspects regarding the valuation of health risks are investigated in an ongoing research project VASSLA at Örebro University.

3 Results and discussion

3.1 Health impact and external cost calculation

Based on the information presented in section 2.2 and 2.3 we are able to calculate the health impacts due to emissions in Stockholm. Since different assumptions can be used regarding the ER-functions, we will perform calculations for a high case and a low case. In the low case we used the ER function suggested by WHO for combustion and secondary PM and we assumed that non-exhaust PM does not have an impact on acute mortality. In the high case estimation we used the ER function for combustion PM based on Jerret et al. (2005), which implies a higher impact than the one suggested by WHO, and included acute mortality estimates for non-exhaust PM. We believe the ER-functions of Jerret et al. (2005) for combustion PM is more representative for the current level of knowledge while the estimate for non-exhaust PM, although uncertain, can be used as a high estimate for the total external health cost of non-exhaust PM (since mortality in many studies results in much higher costs than the morbidity health endpoints). We base the calculation for mortality on years of life lost (YOLL). For combustion PM (chronic mortality) we assume 11.2 years lost per death while for acute mortality we assume 1 year. The estimated health impacts are presented in Table 7, where we have separated the health impacts that occur within Stockholm, within the Mälardalen region and in the rest of Europe.

Table 7 Estimates of YOLL due to emissions in Stockholm in 2003 – low case and high case for combustion and non-exhaust PM.

PM component	Road traffic	Road traffic, LDV[b]	Road traffic, HDV[c]	Sea traffic	Power plants	Residential heating[d]
Stockholm						
Combustion PM		93.5–266.2	34.1–95.7	5.5–16.5	47.3–135.3	110.4–313.1
Non-exhaust PM[a]	0–22.1					
Secondary PM[d]		75.8	1	7.8	7.5	18.8
Total YOLL	*0–22.1*	*169.3–342*	*35.1– 96.7*	*13.3–24.3*	*54.8–142.8*	*129.2–331.9*
Mälardalen						
Combustion PM		3.3–9.9	2.2–4.4	3.3–8.8	1.1–4.4	4.2–12.1
Non-exhaust PM [a]	0–2.0					
Secondary PM[d]		8.1	1.5	1.2	3.6	1.7
Total YOLL	*0–2.0*	*11.4–18*	*3.7–5.9*	*4.5–10*	*4.7–8*	*5.9–13.8*
Rest of Europe						
Combustion PM		2.2–7.7	1.1–3.3	2.2–6.6	8.8–24.2	3.3–9.5
Non-exhaust PM[a]	1.3					
Secondary PM[e]		122.5	74.2	37	115.4	25.6
Total YOLL	*1.3*	*124.7–130.2*	*75.3–77.5*	*39.2–43.6*	*124.2–139.6*	*28.9–35.1*

[a] The health impact for non-exhaust PM is acute mortality.

[b] LDV = light duty vehicles

[c] HDV = heavy duty vehicles

[d] The estimates of combustion PM and NO_x emissions from residential heating are uncertain.

[e] This is the exposure due to particulate nitrate, particulate sulphate and particulate ammonium. Of these, particulate nitrate in most cases makes the largest contribution to population exposure. For details see Bergström, 2008.

The first thing to notice is the importance of LDV and residential heating for YOLL in the Stockholm area. It is also somewhat unexpected that for LDV secondary PM is also an important source for premature deaths locally within Stockholm. As stated above the reason is the fairly large ammonia (NH_3) emissions from LDVs with catalytic converters. Another thing to notice is the influence of emissions in Stockholm on YOLL in the rest of Europe, which is mainly due to the emissions of NO_x. Here LDV and power plants are the most important sources. For non-exhaust PM it can be noticed that although non-exhaust PM is a dominating source for measured PM_{10} concentrations and PM exposure, the number of YOLL is small compared to that for the other PM components. For all pollutants the influence on Mälardalen is relatively small which can be explained by the dispersion pattern of emissions from local sources, that not much nitrate has been formed within this distance and also the relatively low population density in the area.

For the economic analysis we multiply the above estimates with the values presented in section 2.4. For chronic mortality due to combustion PM and secondary PM, the value used is 50 000 euro. The values used for acute mortality due to non-exhaust PM is 75 000 euro. This is because these deaths occur directly and hence should not be estimated based on a discounted present value as the chronic mortality estimate is. The results are presented in Table 8.

Table 8 Estimate of external health costs of mortality due to PM from different sources in Stockholm for 2003, with a low and a high estimate for combustion and non-exhaust PM (million euro).

PM component	Road traffic	Sea traffic	Power plants	Residential heating [b]	Total external cost (million euro)
Stockholm					
Combustion PM	6.4–18.1	0.3–0.8	2.4–6.8	5.5–15.6	14.6–41.3
Non-exhaust PM[a]	0–1.1				0–1.1
Secondary PM[c]	3.8	0.4	0.4	0.9	5.5
Total external cost	*10.2–23*	*0.7–1.2*	*2.8–7.2*	*6.4–16.5*	*20.1–47.9*
Mälardalen					
Combustion PM	0.3–0.7	0.2–0.4	0.1–0.2	0.2–0.6	0.8–1.9
Non-exhaust PM[c]	0–0.2				0–0.2
Secondary PM[c]	0.5	0.1	0.2	0.1	0.9
Total external cost	*0.8–1.4*	*0.3–0.5*	*0.3*	*0.3*	*4.4*
Rest of Europe					
Combustion PM	0.2–0.5	0.1–0.3	0.4–1.2	0.2–0.5	0.9–2.5
Non-exhaust PM[c]	0–0.1				0–0.1
Secondary PM[c]	9.8	1.8	5.8	1.3	18.7
Total external cost	*10.0–10.4*	*1.9–2.1*	*6.2–7*	*1.5–1.8*	*19.6–21.3*

[a] Acute mortality due to exposure to non-exhaust PM from LDV and HDV in Stockholm.

[b] The estimates of combustion PM and NO_x emissions from residential heating are uncertain.

[c] This is the exposure due to particulate nitrate, particulate sulphate and particulate ammonium. Of these, particulate nitrate in most cases makes the largest contribution to population exposure. For details see Bergström, 2008.

According to these calculations, road traffic gives rise to large external health costs both locally, due to combustion PM, and regionally, due to the contribution to secondary PM. It dominates because of its health impact, resulting in several YOLL, the high value for this health end point (premature deaths) and that the emissions occur near ground, relatively close to people's homes. Residential heating contributes substantially to local external costs while power plants seem to have a relatively large impact on a regional scale.

Non-exhaust PM, with very high emissions and exposure of PM_{10} (see table 3), only give rise to a minor total cost on all geographical scales investigated. The reason for the final low estimate is the low estimated health impact from the, mostly coarse, non-exhaust PM.

As expected, the estimated external cost for the Mälardalen region is relatively small. Still, emissions in Stockholm have a negative influence on its closest neighbouring regions (and vice versa).

3.2 Discussion and suggestions for further research

Air quality is high on the environmental policy agenda in the European Union and one of the policy measure implemented is air quality limit values for PM_{10}. In addition, a new Directive on ambient air quality has recently been implemented that imposes standards also for $PM_{2.5}$. This is done although the knowledge about the harmfulness of different PM components is still very uncertain. In order to design policy measure to reduce the emissions in an efficient way such information in addition to information about the sources of air pollution is crucial. In this paper we have therefore investigated the PM exposure in Europe due to the emissions in Stockholm. Moreover, we have used this information to calculate the external health cost of different emission sources. The study implements the Impact Pathway approach (IPA) that has been developed in the EU funded ExternE-projects but using Swedish data and models.

In the IPA, the external cost is calculated based on a chain of events. Since the functions and values used are constants, the cost is the product of exposure, health impact and value. The estimated exposure is the result of dispersion modelling in combination with population data (Johansson and Eneroth, 2007; Bergström 2008). The health impact assessment is based on the results from epidemiological studies (Forsberg, 2008), while the economic values used are obtained from valuation studies (see discussion in section 2.4). As mentioned in the introduction to this report all these estimates are related to uncertainties, some of which we have discussed in the report and return to below. In this study we have chosen to focus on the uncertainties in one part of the chain, the effect of using different assumptions regarding the ER-functions. The reason for this choice is that this has not been done before for the Swedish situation where non-exhaust PM makes a large contribution to the concentrations of PM_{10}. So far the focus in most studies has been on combustion PM. Still, the exposure modelling has been performed using accepted methods and models so these represent the state of the art. As for the economic values used, these are lower bound estimates. Hence, we consider the calculated costs to be reasonable and conservative estimates that represent the current state of knowledge.

One important point to note is that there is not a one to one correspondence between emissions and costs. This is illustrated in Table 11 where we have compiled a summary of the results for primary PM (combustion and non-exhaust PM). The impact of these emissions is mainly local so the larger share of the cost is due to exposure within Greater Stockholm. We have included the results from the external costs that were calculated in the low case and in the high case. The main difference between these two is the ER-functions used (see section 2.3.2).

Table 11 Total PM emissions (tonnes/year) and total external cost of mortality (million euro), due to primary combustion and non-exhaust PM, from emissions in the Stockholm area in 2003.

	Road traffic non-exhaust	Road traffic combustion	Sea traffic [a]	Power plants	Residential heating [b]	Sum
PM emissions	1859	122	33	249	98	2361
Cost PM low	0	6.9	0.6	2.9	5.9	16.3
Cost PM high	1.4	19.3	1.7	8.2	16.7	47.1

[a] Only emissions from merchant ships and ferries that call on ports are included.

[b] The combustion PM emissions from residential heating are very uncertain.

The first thing to notice in Table 11 is that non-exhaust PM, that makes the larges contribution to PM_{10} emissions in Stockholm, results in the lowest cost. The reason for this is that the health impacts that we have based the calculations on are acute mortality that results in fewer years of life lost and therefore are given a lower value. In this study we have not included any morbidity effects. Emissions of combustion PM from power plants are higher than those from road traffic but their costs are lower. In this case the reason is that power plant emissions are released in less densely populated areas and from high chimneys.

A somewhat unexpected result is that residential heating seems to create relatively large health impacts also in the Stockholm region. This is unexpected since district heating makes the largest contribution to residential heating in Stockholm. Still, it confirms what have been concluded in EU-reports that small scale residential heating is a potential problem regarding ambient air quality in Europe in the future. Although our results for residential heating are uncertain, the estimated large impact underlines the importance of improving emission inventories for this source to use as basis for health cost calculations like the one in this report.

Another important result from this study is that the external costs that results from exposure outside Greater Stockholm (that is in the rest of Sweden and Europe) in many cases outweigh the cost that results from exposure within the city. Hence, when designing policy measures the *regional* dimension needs to be taken into account. Furthermore, since both local and regional sources contribute to local concentrations, the exact composition of PM will differ between cities and regions. If the effects of different PM components vary, the same total PM_{10} concentration may be more harmful in one area than in another.

These results are also important in a policy context. In Sweden, the current air quality limit values for PM_{10} have resulted in lengthy discussions and policy proposals regarding emissions from road wear (non-exhaust PM). Larger cities in Sweden do not reach the stated air quality limit values, mainly due to the use of studded tires in wintertime. Since the external cost of non-exhaust emissions have not been addressed in previous calculations of the external cost of PM-emissions (for example in the CAFE-programme), the importance of this source for the health cost has not been known. Our results indicate that at least regarding the mortality impact this source is of less importance. Hence, if the purpose is to reduce human mortality more focus should be placed on reducing exhaust emissions that contribute to the more dangerous, fine PM. To evaluate policies regarding non-exhaust emissions, more research is needed to clarify their impact on health.

The calculations presented here are based on input data from different disciplines. It is important that research continues within each separate field. For example, we have found that there are large uncertainties related to emission data for residential heating, a source that is potentially harmful since the emissions mostly occur in proximity to where people live. The importance of this source may increase over time due to the concern over climate change and the conversion from oil to bio fuels. Our results for example suggest that it may be a poor solution to increase small scale residential heating in densely populated areas.

The health impact of non-exhaust PM also needs further exploration, especially regarding the ER-functions and values to be used for morbidity endpoints. There is also need for further research regarding the valuation of these types of health risks, especially the valuation of morbidity.

Moreover, there is a need for more policy evaluations. Since air pollutants come from several sources, evaluating local policy measures for one source at a time can result in policy mixes that are not optimal. Hence we think that more integrated assessments such as the one we have undertaken in another part of this project (Nerhagen and Li, 2008) are needed. These should also include the impact on more than one outcome. Our results suggest that road traffic is one of the most important sources for health impacts from both PM and NO_x. Hence, policy evaluations should include impacts on several harmful emissions and not focus on one at the time, which is currently often the case. For example, one of the advantages with the reduction in road traffic that may result from road pricing measures is that it has an impact on several harmful pollutants and also CO_2. For the transport sector it may also be relevant to include other health impacts such as noise in these kinds of evaluations since noise is also expected to have similar health impacts.

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