

# Measuring the external health cost of particulate matter from road traffic and other sources in Stockholm, Sweden.

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## Abstract

*This paper measures the external health cost due to emissions from different sources in the Stockholm area using the Impact pathway approach. The estimated health impact is the result of detailed dispersion modelling with high spatial resolution. We make separate calculations for the impact that occur within the Stockholm area, the surrounding region and the rest of Europe. The pollutants considered are combustion and secondary particulate matter (PM) from the burning of fuels and also road wear (non-exhaust PM) that makes a large contribution to measured concentrations of PM locally in Stockholm. We also investigate the influence of assumptions made regarding the exposure-response functions used in these calculations since PM of different origin are expected to have different health impacts.*

*According to the results road traffic makes important contributions to the external health cost both on a local and a regional scale compared to other sources. This is in part due to emissions being released in close proximity to where people live but also because of the amount of pollutants emitted. Although non-exhaust PM makes a large contribution to local population exposure within Stockholm the external health cost is relatively small which is due to other health impact being relevant for this emission source. Residential heating also makes an important contribution to exposure and external health cost on a local scale while power plants have a large influence regionally.*

**Key-words:** Health cost, particulate matter, dispersion modelling, exposure-response functions.

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## Introduction

It has long been recognized that emissions from traffic have a negative impact on human health (Small and Kazimi, 1995; Delucchi, 2000). In latter years there has been emerging consensus that the main influence is due to particulate matter (WHO, 2006; Muller and Mendelsohn, 2007; Jensen et al., 2008). Air quality limit values for particulate matter (PM) have therefore been implemented in the EU and other countries in order to achieve a healthier living environment, especially in cities (Olsthoorn et al., 1999; Monzon and Guerrero, 2004; Sugiyama et al., 2009). Up to 2008, the air quality limit values in the European Union have been based on measurement of  $PM_{10}$ , which is PM that is not larger than 10 microns in diameter<sup>1</sup>. These are PM that are small enough to be inhalable. There is however various components from different emission sources which contribute to concentration levels of PM in cities. In order to design abatement measures that are welfare improving and economically efficient, it is important to assess the damage done by each component and each emission source. In the project reported in this paper we have therefore chosen to investigate the influence from both road traffic and other emission sources in Stockholm, Sweden.

There are several reasons for why we chose to focus on PM. First of all this is the component that makes the largest contribution to total cost in most calculations due to its expected negative impact on human health, especially mortality (Small and Kazimi, 1995; Delucchi, 2000; Nerhagen et al., 2005; Bickel et al., 2006; Muller and Mendelsohn, 2007; Jensen et al., 2008). So far the focus in these calculations have been on the contribution from different combustion processes where the resulting air pollution belongs to the group called fine PM (or  $PM_{2.5}$ ), which are those with a diameter of less than 2.5 microns. It is believed that they are more harmful since they are more likely to penetrate further down into the airways (Small and Kazimin, 1995; Forsberg, 2008). In Stockholm however the most important local source of  $PM_{10}$  is non-exhaust PM that originates from road wear (Johansson et al., 2007; Norman and Johansson, 2006; Omstedt et al., 2005). This is mainly coarse PM ( $PM_{10-2.5}$ ). In spring, when the roadways are dry, the contribution from non-exhaust PM may be 30 times the direct emissions from the exhaust pipe. These mechanically generated dust PM have not been considered in calculations of the external cost that is based on the original ExternE-methodology (Friedrich and Bickel, 2001; Bickel and Friedrich, 2005) or other similar studies (Small and Kazimi, 1995; Jensen et al., 2008). Hence, one purpose with this study is to include them in these types of calculations to get a fuller picture of the PM problem.

In addition we have reason to believe that the current focus on measurement and modelling of  $PM_{2.5}$  in these calculations are somewhat misleading. The main contribution to local concentrations of  $PM_{2.5}$  comes from secondary PM (nitrates and sulphates) which are transported in from other regions. These are aerosols formed by chemical transformation of gaseous emissions and the formation of these PM species is relatively slow. Another contribution to measured  $PM_{2.5}$  however comes from combustion  $PM^2$ . These are ultrafine PM ( $PM_{0.1}$ ), with a diameter of 0.1 microns, and hence their contribution to measured concentrations is small. According to the recommendation by WHO (2006) secondary and combustion PM should be considered to be equally harmful. However, there is a growing recognition among the research community that these are PM of different origins and hence that it is likely that their impact on human health differ (WHO, 2006; WHO, 2007)<sup>3</sup>. Another purpose with this study is therefore to assess the scientific evidence regarding the health impact of different PM components and to compare the outcome of the cost calculations when different assumptions are used.

The method used in the calculation is the Impact pathway approach (IPA), where the external cost is calculated as the product of exposure, effect and value, which has been formalized in the ExternE projects (Friedrich and Bickel, 2001; Jensen et al., 2008). This is similar to the approaches used in other studies (Delucchi, 2000; Muller and Mendelsohn, 2007). In this study we have based the calculation on a detailed assessment of how the exposure varies between sources since previous research has shown the importance of accounting for the location of the source in relation to those exposed (Nerhagen et al., 2005; Bickel et al., 2006; Muller and Mendelsohn, 2007; Jensen et al., 2008). Moreover, since part of the cost for each source is

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<sup>1</sup> In the US air quality limit values for  $PM_{2.5}$  have been in place since 1997. A new EU Directive, which includes standards for  $PM_{2.5}$ , entered into force on 11 June 2008 ([http://ec.europa.eu/environment/air/quality/legislation/existing\\_leg.htm](http://ec.europa.eu/environment/air/quality/legislation/existing_leg.htm)).

<sup>2</sup> In this paper we will use combustion PM to describe emissions of PM that are released directly into the air, without taking part in any transformation, after the burning of different fuels. These types of PM are also referred to as primary PM. The term commonly used in relation to road transport is exhaust PM but for consistency we will use the term combustion PM also for these emissions.

<sup>3</sup> One reason for why the smaller PM from combustion could be more harmful is that they make a large contribution to the total particulate number in city air (Johansson et al., 2007), something that is not accounted for in current measurements.

due to a regional impact (European scale), we have also investigated this aspect by studying the secondary PM formation and dispersion that is due to local emissions of NO<sub>x</sub>, SO<sub>2</sub> and NH<sub>3</sub>. The emissions sources included in the paper are power plants, road traffic, residential heating, and part of the maritime transport that takes place within Stockholm<sup>4</sup>.

The economic values used in the study are similar to those used in the cost-benefit analysis (CBA) that was undertaken in the programme Clean Air for Europe (CAFE) (AEA Technology, 2005a). For mortality the values used are in line with estimates used in Swedish policy analysis. Exactly what estimate to use in this context however is a matter of discussion (Friedrich and Bickel, 2001; Muller and Mendelsohn, 2007). Our decision is to make the calculations based on years of life lost (YOLL) and the value of a life year (VOLY) since this is what is commonly used in ExternE projects. We chose to use this since we both consider chronic mortality and acute mortality as health endpoints and the estimated YOLL for these two endpoints differs by a factor 10. Hence, these assumptions have a considerable influence on the final cost estimates that would not be evident if we used a single VSL estimate for each health outcome. For emissions from road traffic we have also included a morbidity endpoint that is relevant for non-exhaust PM, restricted activity days (RAD). The value used for this endpoint is the same as in the CBA for CAFE (AEA Technology, 2005b)

The results of the calculations reveal that combustion PM and non-exhaust PM mainly have a local impact in Stockholm. As expected, for combustion PM the marginal cost is highest for road traffic and residential heating. This is because these emissions are released in close proximity to peoples place of residence. Non-exhaust PM makes the largest contributions of the local sources to average population exposure to PM in Stockholm but the estimated marginal cost is comparatively low. Although these emissions are also released in densely populated areas, the health impacts are of different kinds which have an important influence on the final cost estimates. The largest total cost due to exposure on a regional scale is estimated for secondary PM due to emissions of NO<sub>x</sub> SO<sub>2</sub> and NH<sub>3</sub> from road traffic and power plants. Somewhat unexpectedly we have also found that emissions of NO<sub>x</sub> also contribute to the formation of secondary PM locally in Stockholm. This is a result that has not been accounted for in other studies. Jensen et al., (2008) for example report that formation of nitrates and sulphates is very limited on the local scale.

The outline of the paper is as follows. In the next chapter we briefly describe the methodology. We then present the inputs on which the cost calculations are based. In part three we present the results including sensitivity analysis for some inputs. We start with a comparison between the different emission sources that contribute to combustion PM, followed by separate calculations for PM emissions from road traffic. The paper ends with conclusions.

## 1 - Methodology

The impact pathway approach (IPA) is a bottom-up approach where the calculated cost is a function of what influence the emission of a certain pollutant has on human health, and the value of this health impact. This approach has been formalized in the EU funded ExternE-projects but in principle is the same as the damage functions approach that has been used previously for these types of calculations (Small and Kazimi, 1995; Delucchi, 2000). This approach is based on a chain of causality linking emissions to costs and it could be expected that non-linear relationships would occur at several points in the chain. However, in most applications linear relations are assumed (Small and Kazimi, 1995; Olsthoorn et al., 1999; Bickel et al., 2006; Jensen et al., 2008). Why this is a reasonable assumption for most part of the chain is discussed at length in Small and Kazimi (1995) and Bickel et al., (2006).

There are however some aspects in the chain that will imply non-linearity (Muller and Mendelsohn, 2007; Jensen et al., 2008). One reason is that population exposure will vary depending on the location of the emission source. Hence the cost for a pollutant that increases concentrations of PM locally will be higher if it is released in urban areas where the population density is high. Since this aspect has been found to have an important influence on the cost estimates, detailed exposure modelling like the one we use in this paper for the local scale is called for.

Another reason for non-linearity would be if the formation of secondary PM (sulphates and nitrates) depended upon what pollutants are already in the air or on the amount of the pollutant that is released. Small and Kazimi (1995) argue that for small changes in emissions, these relationships can be assumed to be linear. This issue is also discussed in Muller and Mendelsohn (2007) and they conduct an experiment to test this assumption. The result of this experiment is that there is no interaction between pollutants at the margin.

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<sup>4</sup> The emission data for shipping only includes part of the shipping that takes place in the Stockholm area (ferries and merchant ships).

Jensen et al., (2008) undertakes a similar experiment and their result is that there is interaction between some pollutants such as NO<sub>x</sub> and NMVOC.

The approach we have chosen to use in this paper is to deduct the contribution from one emission source at a time to see what impact this has on total concentrations of PM 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 the effect estimates and the values used in the cost calculations are assumed to be linear relationships and hence do not change with the size of the change in concentration levels. Moreover, 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 yearly emissions from traffic in Stockholm for example only make a minor contribution to the total NO<sub>x</sub> emissions in Sweden and Europe and hence we believe that reducing these emissions can be considered a marginal change with minor interaction effects.

Hence, our cost calculations are based on the following equation, (which is a modification of an equation in Ostro and Chestnut, 1998). It describes the yearly external health cost due to PM from a change in emissions from a specific source:

$$\begin{aligned} \text{Health cost} &= \Delta \text{yearly exposure} \cdot \text{effect} \cdot \text{value} \\ &= (\Delta \text{PM}_{a,i} \cdot \text{POP}) \cdot (B_{a,j} \cdot P_{i,j}) \cdot V_j \end{aligned} \quad (1)$$

where

$\Delta \text{PM}_{a,i}$  = change in annual average exposure for pollutant  $i$  ( $\mu\text{g}/\text{m}^3$ )

POP = population exposed to  $\Delta \text{PM}_{a,i}$

$B_{a,j}$  = 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  $\mu\text{g}/\text{m}^3$  of pollutant  $i$  (relative risk)

$V_j$  = value of health impact  $j$ .

Details regarding the calculation of  $\Delta \text{PM}_{a,i}$  is to be found in descriptions of the dispersion models used. This calculation has to be done separately for each type of PM since the effect estimates  $P_{i,j}$  (also called the exposure-response functions) 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) and the value used. To arrive at the marginal cost (Euro/ton or Euro/kg) we divide this total cost with the total annual emissions for each pollutant and each source.

## 2 – Data

### 2.1 Emissions and exposure modeling

The modelling for the local scale (i.e. the influence within the Stockholm area) is based on the Airviro Air Quality Management system which has been used in many similar studies (Johansson et al., 1999; Eneroth et al., 2006). All data used in the dispersion modelling is for the year 2003 which was a year with quite normal meteorological conditions (for details, see Johansson and Eneroth, 2007).

Emission data for road traffic are based on traffic data for separate road links. 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<sub>x</sub> as tracer (Ketzel et al., 2007). Emissions from sea traffic as well as power plants are describes as point sources. The emission data for sea traffic includes data on merchant ships and ferries that call at Stockholm ports. Emissions from power plants are updated yearly by the supervisory authorities. The emissions from residential heating are probably the most uncertain. They are divided into oil-heating and wood combustion. The emissions are based on regional fuel statistics from Statistics Sweden. The grid sources are distributed according to population statistics in areas where district heating is not available.

Table 1 presents total emissions and the resulting average exposure of NO<sub>x</sub> and PM in the Greater Stockholm area. Even though we are not interested in NO<sub>x</sub> for its direct health impact on the local scale, we have included information about these emissions because we are interested in its contribution to secondary

PM<sup>5</sup>. The population weighted concentrations that are the basis for the local health impact assessment were obtained by combining the modelled changes in concentrations on the local scale with gridded population data<sup>6</sup>. Road traffic dominates the emissions of both PM and NO<sub>x</sub>. Regarding directly emitted PM, the largest contribution comes from non-exhaust PM. The power plants are an important NO<sub>x</sub> sources. To be noticed is that since the exposure estimate depends on the population density in proximity to the emission source there is not a one to one correspondence between total amount of emissions and average exposure. Concerning combustion PM for example the emissions from power plants is about twice of those from road traffic (249 vs 122 ton) but the average exposure for road traffic is about three times as high (0.14 vs 0.051). To put these figure in relation to total exposure they can be compared to the contribution made to local air quality from regional sources. To obtain an estimate of this contribution it is common to measure the concentrations at measurement stations in the countryside. For Stockholm these rural background concentrations are about 10 µg/m<sup>3</sup> for PM<sub>10</sub> and 4 µg/m<sup>3</sup> for NO<sub>x</sub> (e.g. SLB Analys, 2004; LFV, 2008).

**Table 1: Total emission and exposure (population weighted annual mean concentrations) of NO<sub>x</sub> and PM from different source sectors in the Greater Stockholm area.**

Substance	Road traffic		Sea traffic <sup>a</sup>		Power plants		Residential heating	
	Emissions (ton)	Exposure (µg/m <sup>3</sup> )	Emissions (ton)	Exposure (µg/m <sup>3</sup> )	Emissions (ton)	Exposure (µg/m <sup>3</sup> )	Emissions (ton)	Exposure (µg/m <sup>3</sup> )
NO <sub>x</sub>	5674	5.68	885	0.18	2002	0.36	487	0.68
Combustion PM	122	0.14	33	0.0063	249	0.051	98	0.12
Non-exhaust PM <sup>b</sup>	1859	1.70	-	-	-	-	-	-

<sup>a</sup> Only emissions from merchant ships and ferries that call at ports are included.

<sup>b</sup> Only non-combustion PM (road, brake and tyre wear) with a diameter <10 microns. The size is mainly larger than 1 micron.

Based on the data provided by SLB Analys, the Swedish Meteorological and Hydrological Institute (SMHI) did dispersion modelling and exposure calculation on a regional scale (Bergström, 2008). The MATCH (Multi-scale Atmospheric Transport and Chemistry) model was used (see Andersson et al, 2007; Hass et al., 2003)<sup>7</sup>. For impacts closer to Stockholm, a smaller calculation area was used with a horizontal resolution of 5 km (the Mälardalen region). A larger calculation area covering most of Europe, with a horizontal resolution of about 44km, was used to study the impacts on the rest of Europe. The exposure calculation is done using the same approach as for the local scale i.e. the average concentration resulting from emissions in Stockholm in each grid cell, multiplied with the number of persons in each grid cell<sup>8</sup>.

A summary of exposure calculations for the local and regional scale are presented in Table 3 where exposure is measured as the number of people during a year that are exposed to 1 µg/m<sup>3</sup> PM. This is the first input in the cost calculation in equation (1). The first four rows present the results for the Greater Stockholm area. We find that non-exhaust PM has the largest impact on human exposure on this scale. However, somewhat unexpectedly, we find that secondary PM due to the local NO<sub>x</sub>/SO<sub>x</sub>/NH<sub>3</sub> emissions also has a non-negligible impact on the local exposure in Stockholm. This is due to the formation of particulate

<sup>5</sup> The gaseous pollutant NO<sub>x</sub> is not expected to have a major impact on human health directly, although it can worsen the symptoms for those with respiratory problems, but it is often used as a traces for road traffic emissions in epidemiological studies and is therefore of interest.

<sup>6</sup> The calculation area covers the Greater Stockholm area (35km x 35 km) with a spatial resolution of 100m x 100 m. The gridded population data was obtained from Statistics Sweden. In 2003, 1 405 600 people lived within the 35km x 35km calculation grid.

<sup>7</sup> To describe the chemical evolution of the emissions from Stockholm, and the resulting production of secondary PM, accurate emissions are needed for the change in other emissions and the total emissions in the rest of Sweden and Europe<sup>7</sup>. For Sweden NO<sub>x</sub>, SO<sub>x</sub>, VOC, CO and NH<sub>3</sub> emission data from the SMED (Swedish Methodology for Environmental Data, [www.smed.se](http://www.smed.se)) project, with 1km resolution, were used. For emissions outside Sweden data from EMEP ([www.emep.int](http://www.emep.int)) were used.

<sup>8</sup> The population data for Sweden are obtained from Statistics Sweden. The population for the Mälardalen region was about 2.2 million people. For the rest of Europe population data was obtained from two sources. 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).

nitrate<sup>9</sup>. 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.

Row 5 to 8 contains the modelling results for the Mälardalen region (excluding the studied emission area of Stockholm). As seen from the table, the influence on exposure from the emissions in Stockholm 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.

Finally, in row 9 to 12 the results for the influence on the rest of Europe are presented. Although the contribution to the concentration levels of emissions from the Stockholm area on the rest of Sweden and the continent is small, the estimated exposure is large due to the number of people exposed. In this case the most important contribution to exposure is due to secondary PM. For many of the sources, the estimated exposure on this scale is even higher than the exposure within Greater Stockholm. For residential heating however the impact on this scale is relatively small which is due to relatively small NO<sub>x</sub>-emissions from this emission source.

**Table 2: Population exposure in the different calculation areas from different PM components (person µg/m<sup>3</sup>).**

PM component	Road traffic <sup>c</sup>	Sea traffic	Power plants	Residential heating <sup>a</sup>
Combustion PM	191 500	9 000	71 500	167 000
Secondary PM <sup>b</sup>	117 000	12 000	11 500	28 500
Non-exhaust PM	2 190 000	-	-	-
<i>Total Greater Stockholm</i>	<i>2 498 500</i>	<i>21 000</i>	<i>83 000</i>	<i>195 500</i>
Combustion PM	7 500	4 500	2 000	6 000
Secondary PM <sup>b</sup>	15 000	2 000	5 500	2 500
Non-exhaust PM	196 000	-	-	-
<i>Total Mälardalen</i>	<i>205000</i>	<i>6 500</i>	<i>7 500</i>	<i>8 500</i>
Combustion PM	6 000	3 500	13 000	4 700
Secondary PM <sup>b</sup>	300 000	56 500	176 000	39 000
Non-exhaust PM	133 000	-	-	-
<i>Total Other Europe</i>	<i>439 000</i>	<i>60 000</i>	<i>189 000</i>	<i>43 000</i>
<i>Total exposure from Stockholm</i>	<i>3 142 500</i>	<i>87 500</i>	<i>279 500</i>	<i>247 000</i>

<sup>a</sup> The Combustion PM and NO<sub>x</sub> emissions from residential heating are very uncertain which means that the population exposure values for this sector are uncertain as well.

<sup>b</sup> 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).

<sup>c</sup> For combustion and secondary PM we have made separate calculations for light duty vehicles (LDV) and heavy duty vehicles (HDV) that we have used in the analysis for road traffic emissions. For results, see Johansson and Eneroth (2008) and Bergström (2008).

## 2.2 Data for health impact assessment

The second step is the health impact assessment that, according to equation (1), is composed of two parts; a baseline estimate and an exposure-response (ER) function. 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. We have focused on mortality since this is the health endpoint that gives rise to the largest cost in most studies, but will in our calculations for emissions from road traffic also include a morbidity endpoint, Restricted Activity Days (RAD).

For mortality there is a distinction made between deaths that will occur due to long-term exposure

<sup>9</sup> The relatively high contribution from LDV, compared to HDV, to particulate nitrate is due to fairly large ammonia (NH<sub>3</sub>) emissions from LDVs with catalytic converters. The NH<sub>3</sub> reacts with nitric acid (HNO<sub>3</sub>) in the city air to form particulate ammonium nitrate. 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.

(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 World Health Organization (WHO) repeatedly reviews the current level of knowledge and issues Air Quality Guidelines for different pollutants and also relevant E-R functions. The current WHO guidelines contain an ER function for PM<sub>2.5</sub> assuming a long-term effect on mortality of 1.06 (6%) for a 10 µg/m<sup>3</sup> increment of average concentration levels at roof level (WHO, 2006). No threshold effect is assumed. It is also stated that the same ER-function should be used for the finer fraction of PM irrespective of origin; hence it is assumed that different PM components are equally harmful. There is however an ongoing discussion within the research community of whether or not it is correct to assume that all PM have the same impact irrespective of origin or size (WHO, 2006; WHO 2007).

Based on the results from the literature review by Forsberg (2008) we have concluded that, especially for combustion PM, a realistic alternative in a sensitivity analysis is a relative risk of 17% per 10 µg/m<sup>3</sup> PM<sub>2.5</sub> (Jerret et al., 2005). 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 for secondary PM we use the ER-function suggested by WHO for PM<sub>2.5</sub>. 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 of an effect on mortality (Brunekreef and Forsberg, 2005). In time-series studies a short-term effect on daily mortality has been found but it is usually more strongly related to finer PM. We have therefore decided to assume, as a higher estimate, that non-exhaust PM have the same short-term effect on mortality as PM<sub>10</sub> 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 µg/m<sup>3</sup> and as a lower bound no impact on mortality.

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<sup>10</sup>. 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 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 potential years of life lost (YOLL) due to excess mortality can be calculated per excess death or divided by the number of persons in the population to have a mean value for the whole population. This is because 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. 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 premature death is brought forward. Therefore we have used the same assumption as in the CAFE calculations (AEA Technology, 2005b) for short-term effects on mortality of PM<sub>10</sub>, 1 YOLL (a reasonable range for this estimate is from 6 to 18 months).

Other applications where the impact pathway approach has been used, such as in CAFE (AEA Technology, 2005a), also include morbidity impacts<sup>11</sup>. In these studies however they only consider and compare the cost due to PM emissions from the burning of fuel. In our study we have to compare for road traffic the cost resulting from combustion and non-exhaust PM that are PM of different origin. Even though morbidity impacts have made a small contribution to total cost in most studies, it may be of importance regarding non-exhaust PM since the estimated exposure to this pollutant is comparatively large. Recent research has therefore tried to separate the influence that PM of different origin has on different health impacts but the result is still unclear. It appears that non-exhaust PM can be expected to have a larger influence on respiratory problems.

Hence, in order to get some indication on the possible influence of morbidity on the marginal cost for road traffic, we have added an estimate for restricted activity days (RAD). This is the second most important morbidity endpoint for the total health cost result in CAFE (AEA Technology, 2005). The impact estimate

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<sup>10</sup> We use this estimate since we do not want the mortality effects to be modified by recent mortality rates.

<sup>11</sup> In CAFE CBA morbidity impacts accounted for 30% of the total health cost (15% when a higher value was used for VOLY) and RAD accounted for 10% (5% with a higher value for VOLY).

used in the CAFE, and suggested in the ExternE Methodology update (Bickel and Friedrich, 2005), is from a study by Ostro from 1997 using PM<sub>2.5</sub> measurements. For Europe this implies an impact of 90.2 RAD per 1000 adults that are exposed to 1 µg/m<sup>3</sup> PM<sub>2.5</sub> during a year. To apply this estimate to non-exhaust PM we convert this estimate using the same relation used in APHEIS (Medina et al., 2004) which results in an estimate of 65 RAD per 1000 adults for a change of 1 µg/m<sup>3</sup> non-exhaust PM during a year. We will use this for the total population not separating out adults. Hence we may overestimate the cost for this morbidity endpoint but we expect this effect will be relatively small. No other morbidity endpoints were included due to a lack of ER functions for non-exhaust PM.

## 2.2 Economic values for health impacts

The final input into the cost calculation in equation (1) is the economic values placed on the estimated health impacts. For mortality the value commonly used is the value of a statistical life (VSL). There are several aspects that are raised in the literature on the methods used to obtain these estimates and how this value can vary depending on age, context etc, see Anderson and Treich (2008) for an overview. However, a particular problem for cost calculations in our context is that most studies concerning VSL come from studies of working age people. Hence they 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 this study we have decided to follow the approach commonly used in ExternE-projects and apply the estimates of the value of a life year (VOLY). We have used the values suggested by Bickel and Friedrich (2005) since they are similar to those used in the CBA in CAFE (AEA Technology, 2005b)<sup>12</sup>. They are the result of a valuation study undertaken within an ExternE project (Alberini et al., 2006). Contrary to previous Externe studies and other valuation studies they recommend the use of 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)<sup>13</sup>.

While the major component of the value of mortality impacts is the welfare loss, this is only one of the aspects considered when valuing morbidity impacts. Included in these values is also, if relevant, the value of lost production and the cost related to the illness. Hence, for restricted activity days the value to be used depends upon the kind of restriction imposed on the individual. In CAFE (AEA Technology, 2005b) they separated the value for welfare loss, 49 euro, and the production loss, 88 euro. They however also assumed that some individuals are affected but still working, a health endpoint called Minor restricted activity day. Based on assumptions on the share of people affected in different ways they arrive at an average value for a working adult of 83 euro/RAD which we have used in this study.

## 3 - Results

### 3.1 External costs for combustion PM per emission source

Based on the information presented above we are able to calculate the external health cost due to emissions in Stockholm. In this part we have calculated the cost for PM resulting from the burning of fuels. Since different assumptions can be used regarding the ER-functions, we have performed 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. In the high case estimation we used the ER function for combustion PM based on Jerret et al. (2005). The total cost per region for each emission source is presented in Table 3.

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. Road traffic dominates on the local scale because the emissions occur near ground, relatively close to people's homes. For the same reason residential heating contributes substantially to external health costs locally in the Stockholm area. Power

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<sup>12</sup> 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.).

<sup>13</sup> 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). They are however similar to those used by Jensen et al., (2008); 48 000 euro for chronic mortality and 81 000 euro for acute mortality in 2002 prices.



plants have a relatively large impact on a regional scale which is due to the formation of secondary PM from the gaseous emissions including NO<sub>x</sub>. The contribution from sea traffic on all scales is small which is likely to be due to the maritime traffic that we have accounted for in this study. 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). There is also a relatively large difference between the low and high estimate which is due to the ER-functions used. Hence, the effect estimates used have an important impact on the final results.

**Table 3: Estimate of the mortality cost due to PM from combustion from different sources in Stockholm for 2003, with a low and a high estimate for combustion PM (million euro).**

PM component	Road traffic	Sea traffic	Power plants	Residential heating
Combustion PM	6.4 – 18.1	0.3 – 0.8	2.4 – 6.8	5.5 – 15.6
Secondary PM	3.8	0.4	0.4	0.9
<b>Total Stockholm</b>	<b>10.2 – 21.9</b>	<b>0.7 – 1.2</b>	<b>2.8 – 7.2</b>	<b>6.4 – 16.5</b>
Combustion PM	0.3 – 0.7	0.2 – 0.4	0.1 – 0.2	0.2 – 0.6
Secondary PM	0.4	0.1	0.2	0.1
<b>Total Mälardalen</b>	<b>0.8 – 1.2</b>	<b>0.3 – 0.5</b>	<b>0.3</b>	<b>0.3</b>
Combustion PM	0.20 – 0.5	0.1–0.3	0.4 – 1.2	0.2 – 0.5
Secondary PM	6.1	1.8	5.8	1.3
<b>Total Other Europe</b>	<b>10 – 10.3</b>	<b>1.9 – 2.1</b>	<b>6.2 – 7</b>	<b>1.5 – 1.8</b>

To arrive at the average annual marginal cost<sup>14</sup> we divide these total cost estimates by the total emissions since the calculations are based on linear relationships, hence marginal cost is equal to average cost. The results of these calculations are presented in Table 4 for PM and Table 5 for NO<sub>x</sub>. Emissions of combustion PM from power plants are higher than those from road traffic and residential heating but their marginal costs are lower since the power plant emissions are emitted in less densely populated areas and from high chimneys. The same reasoning applies for sea traffic. Maybe somewhat unexpectedly the marginal cost for residential heating is higher than the cost for road traffic. These figures however have to be treated with caution since the exposure modelling for this emission source is based on more uncertain assumptions regarding the emissions and the locations of emission sources. Still, they highlight that residential heating in closely populated areas is a potential threat to human health and we believe that their influence should be explored in further research.

**Table 4: Marginal mortality cost due to PM from combustion from different emission sources in Stockholm for 2003, with a low and a high estimate for combustion PM.**

PM component	Road traffic	Sea traffic	Power plants	Residential heating
Combustion PM (million euro)	6.9 – 19.3	0.6 – 1.5	2.9 – 8.2	5.9 – 16.7
<i>Total emissions (ton)</i>	122	33	249	98
<i>Marginal cost (euro/kg)</i>	<i>56.6 - 158.2</i>	<i>18.2 - 45.5</i>	<i>11.6 - 32.9</i>	<i>60.2 - 170.4</i>

In Table 5 we have performed the same calculation for the NO<sub>x</sub> emissions. However, these estimates are only an approximation of the marginal cost of these emissions since the formation of secondary PM also depends on other gaseous emissions. These estimates are more similar in size which is because they are the result of changes in concentrations over a larger area with varying population densities. The high estimate for road traffic and residential heating is in part due to the modelled formation of secondary PM locally in Stockholm. The impact of secondary PM that is formed locally has not been fully explored in previous research, but these results indicate that their influence on cost is non-negligible. Hence, we believe further research on this issue is needed. Still, the contribution to the total marginal cost for each emission source is small.

<sup>14</sup> This is an average estimate for the marginal cost since exposure in reality will vary depending on season which will influence the amount of emissions for example, meteorology etc.

**Table 5: Marginal mortality cost due to secondary PM from NO<sub>x</sub> from combustion from sources in Stockholm in 2003.**

PM component	Road traffic	Sea traffic	Power plants	Residential heating
Secondary PM ( <i>million euro</i> )	14.1	2.3	6.4	2.3
<i>Total emissions (ton)</i>	<i>5674</i>	<i>885</i>	<i>2002</i>	<i>487</i>
<i>Marginal cost (euro/kg)</i>	<i>4.83</i>	<i>2.6</i>	<i>3.2</i>	<i>4.7</i>

### 3.2 External costs for road traffic emissions of different origin

In Table 6 we have instead compared the cost for the different PM emissions resulting from road traffic for the same endpoint as before, mortality. We have presented the results for the different regional scales for which the analysis has been done. We have also made separate calculations for combustion PM and secondary PM resulting from gaseous emissions from light duty vehicles (LDV) and heavy duty vehicles (HDV).

The first thing to notice in Table 6 is that non-exhaust PM, that makes the largest contribution to PM exposure in Stockholm, results in the lowest marginal cost for mortality. The reason for this is that the health impacts that we have based these calculations on are acute mortality with a different ER-function and fewer years of life lost per death than for combustion PM. The lower estimate for mortality for non-exhaust PM is zero, hence making the assumption of no impact from these emissions on mortality. Another thing to notice is that the cost is higher for LDV than for HDV which in part is due to more emissions being released in proximity to where people live. But the difference is also due to differences in the size of emissions where HDV contribute to 1/3 of the total combustion PM for road traffic but 47% of the total emissions of NO<sub>x</sub>.

**Table 6: Estimate of the mortality cost due to PM from road traffic in Stockholm for 2003, with a low and a high estimate for combustion and non-exhaust PM (million euro).**

Pollutant	Road traffic non-exhaust PM	Road traffic combustion PM <sup>a</sup>	
	<i>LDV+HDV</i>	<i>LDV</i>	<i>HDV</i>
PM (non-exhaust or combustion)	0 – 1.1	4.7 – 13.3	1.7 – 4.8
Secondary PM	-	3.8	0
<b>Total Stockholm</b>	<b>0 – 1.1</b>	<b>8.5 - 17.1</b>	<b>1.7 - 4.8</b>
PM (non-exhaust or combustion)	0 – 0.2	0.2 – 0.5	0.1 – 0.2
Secondary PM	-	0.4	0.1
<b>Total Mälardalen</b>	<b>0 – 0.2</b>	<b>0.6 - 0.9</b>	<b>0.2 - 0.3</b>
PM (non-exhaust or combustion)	0 – 0.1	0.15 – 0.4	0.1 – 0.3
Secondary PM	-	6.1	3.7
<b>Total Other Europe</b>	<b>0 – 0.1</b>	<b>6.25 - 6.5</b>	<b>3.75 - 3.8</b>
<b>Total cost for mortality</b>	<b>0 – 1.4</b>	<b>15.35 – 24.5</b>	<b>5.65 – 8.9</b>

<sup>c</sup> For details regarding the underlying emission and exposure data used in these calculations, see Eneroth and Johansson (2008) and Bergström (2008).

One of the reasons for the interest in the external cost of PM emissions is that these estimates can be used to evaluate different measures that are implemented in order to reach air quality limit values. Regarding choice of policy measures, in Stockholm there has been a focus on non-exhaust PM, and reductions in the use of studded tires, since these are the local emissions that are the main reason for the exceedances of the air quality limit values for PM<sub>10</sub>. In order to get some indication on the possible importance for human health of this emission source in relation to combustion PM we have, as discussed in section 2, added an estimate for the second most important morbidity endpoint for the final results in CAFE (AEA Technology, 2005a), restricted activity days (RAD). This calculation is based on the exposure to combustion PM and non-exhaust PM that occurs within the Stockholm area resulting from local emissions. The results are presented in Table 7. The final results presented in the table are marginal cost (euro per kg) and marginal cost (euro per vkm) since the latter is more relevant when comparing the cost for different vehicles travelling on a certain road. Since this table only reflects the cost on a local scale, this is only part of the cost for LDV and HDV. To these the cost for secondary PM needs to be added.

**Table 7: Total and marginal cost for mortality and morbidity due to PM emissions from road traffic in Stockholm for 2003, with a low and a high estimate for combustion PM and non-exhaust PM.**

Health endpoints	Road traffic non-exhaust PM	Road traffic combustion PM	
	LDV+HDV	LDV	HDV
Mortality (million euro)	0 – 1.1	4.7 – 13.3	1.7 – 4.8
Morbidity (million euro)	0 – 11.8	1	0.4
<b>Total cost (million euro)</b>	<b>0 – 12.9</b>	<b>5.7 – 14.3</b>	<b>2.1 – 5.2</b>
Total emissions (ton)	1859	82	40
<b>Marginal cost (euro/kg)</b>	<b>0 - 6.9</b>	<b>69.5 – 174</b>	<b>52.5 – 130</b>
Emissions per vkm (g/vkm)	0 - 0.29	0.014	0.10
<b>Marginal cost (euro/vkm)</b>	<b>0 - 0.0019</b>	<b>0.0009 – 0.0024</b>	<b>0.005 – 0.013</b>

To give a more complete account of the marginal cost of emissions from road traffic will not be possible without further clarifications regarding the health impact of pollutants of different origin. Still, this analysis illustrates the importance of accounting for different health impacts when evaluating policy measures that aim at improvements in local air quality. According to these estimates the marginal cost per vehicle km for combustion PM from LDV and non-exhaust PM may be of a similar size, although the exposure to non-exhaust PM is about ten times larger than exposure to combustion PM. Hence, reducing exposure to combustion PM by reducing traffic may be as beneficial to health as reducing the emissions of non-exhaust PM by reduced use of studded tires. There are also measures such as congestion charging that will contribute to less exposure from both emission sources since they reduce vehicle km driven in densely populated areas<sup>15</sup>. Moreover, this policy measure also contributes to a reduction of secondary PM. Hence, for a welfare assessment, policy analysis concerning improvements in air quality should account for the impact of the different sources on a local and a regional scale but it should also consider the contribution from different sources that contribute to PM concentrations.

## Conclusions

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<sub>10</sub>. In addition, a new Directive on ambient air quality has recently been implemented that imposes standards also for PM<sub>2.5</sub>. 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 locally in Stockholm and regionally 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 this study we have included a pollutant, non-exhaust PM, that is important for measured PM<sub>10</sub> concentrations in Sweden, and the exceedances of the air quality limit values. Since this is a pollutant of a different origin (mainly road wear) it can be expected to have different health impacts than PM emissions from the burning of fuel that has been accounted for in previous studies. There is also an ongoing discussion among the research community if directly emitted PM (combustion PM) has different health impacts than secondary PM from gaseous pollutants. While combustion PM and non-exhaust PM mainly have a local impact, and therefore can be reduced by local policy measures, secondary PM is transported in from other areas.

The results of the calculations reveal that on emissions that occur in close proximity to where people live, mainly emissions from light duty vehicles and residential heating, are most important for the health impact and cost on a local scale. Non-exhaust PM makes large contributions to total emissions of PM in Stockholm but the estimated cost is relatively low. This is because they are expected to have other health impacts.

<sup>15</sup> A evaluation undertaken by the Environmental Division of Stockholm (SLB Analys report 8:2009) finds that the congestions charges contributed to lower emissions and concentrations of several pollutants.

Somewhat unexpectedly we have however found that emissions of NO<sub>x</sub> also contribute to the formation of secondary PM locally in Stockholm. This is a result that has not been accounted for in other studies. The largest cost due to exposure on a regional scale is estimated for emissions from road traffic and power plants mainly resulting from their emissions of NO<sub>x</sub>.

Another important result from this study is that the total costs that result from exposure outside Greater Stockholm 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<sub>10</sub> concentration may be more harmful in one area than in another. This geographical dimension should be accounted for when designing EU wide policies such as environmental air quality limit values.

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