



HEALTH EFFECTS OF TRANSPORT EMISSIONS

A review of the state of the art of methods and data used for external costs calculations

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Preface

In Sweden cost-benefit analysis is used in infrastructure investment analysis. Until April 2010, before the creation of the new Swedish Transport Administration, there were four government agencies handling these issues, one for each transportation mode. In order to have a coherent framework for how cost-benefit analysis was done a joint working group (*ASEK*¹) undertook research and set standards.

The background for this project is a request from ASEK in September 2009. They needed an update on the state of the art regarding health impacts and external cost calculations of emissions from traffic. The work within this project has been carried out by Anna Mellin and Lena Nerhagen between March and June 2010. The project has been funded by CTS. An external peer review was performed by Elisabet Idar Angelov (WSP) before the result was published.

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¹ *Arbetsgruppen för Samhällsekonomiska Kalkyler* which can be translated to the Working group for CBA. The aim of the group is that the same principles and values should be used in socio-economic analyses in the transport sector in Sweden.

Definitions and Abbreviations

In the literature of health effects from traffic emissions there are several terms that are frequently used. Here follow explanations of some of the terms we will use in this report to facilitate the reading:

Acute mortality – is defined as the change in the number of mortalities caused by a short period of increased exposure to a pollutant (Nerhagen et al., 2005). In other words, the impact occurring within a few days of the exposure (Bickel and Friedrich, 2005). The same terminology applies to morbidity (i.e. illness).

Association –the correlation between a pollutant and an end-point (Bickel and Friedrich, 2005).

Chronic mortality – is defined as the increased mortality caused by exposure over a longer period of time (Nerhagen et al., 2005). In other words, the total mortality increase due to chronic exposure (Bickel and Friedrich, 2005). The same terminology applies to morbidity.

Dose-response function – describes the impact (e.g. cases of asthma) a quantity of a pollutant (dose) has on a receptor (e.g. population). The term is usually used in a wider sense than the actual dose absorbed by a receptor especially for air pollution, e.g., the concentration of a pollutant in the air. Therefore the terms exposure-response function (ERF) and concentration-response functions (CRF) are commonly used (Bickel and Friedrich, 2005).

End-point – another word for health impact (Bickel and Friedrich, 2005).

Epidemiologic studies – aim to define the human health impact of different substances based on e.g. statistical analyses and measurements. It is often difficult in epidemiologic studies to define the health effects of specific pollutants (Sehlstedt et al., 2007). One example of a common method in epidemiology is to use time series. The researcher looks for the correlations between a daily concentration of a (mix of) pollutant(s) which a population is exposed to and, e.g., the number of hospital admissions for asthma attacks.

Sensitivity analysis – to test the robustness of a cost-benefit analysis some of the input parameters can be changed and tested in a sensitivity analysis to see how much the output values changes (how sensitive they are). The sensitivity analysis reveals the degree of uncertainty in the cost-benefit analysis.

Toxicology studies – are tests in experimental cells, animals and humans performed in laboratories to describe a pollutant's toxicity (poisonousness) (Sehlstedt et al., 2007).

Abbreviations

CBA – Cost Benefit Analysis

COI – Cost of illness

CVM – Contingent valuation method

EPA – Environmental protection agency

IPA – Impact pathway approach

RP – Revealed preference

SP – Stated preference

VKM – Vehicle-kilometer

VOC – Volatile organic compounds

VOLY – Value of a life year

VSL – Value of statistical life

WTA – Willingness to accept

WTP- Willingness to pay

Summary

The purpose is to provide a background for a discussion concerning the methods and values used in cost-benefit analysis in Sweden for air pollutions', from traffic, impact on human health and the research needs in this area. We provide an overview of the current state of the art of models used for and input needed for external cost calculations of the health impacts. The calculations are not straightforward and depend on the collaboration between several research disciplines. In the ExternE projects, which have been used as a reference point in this study, there are still uncertainties concerning which pollutants to take into consideration.

Regarding the health impacts, we have recapitulated some of the main conclusions in a review by the American Heart Association (2010). They state that e.g. the following issues need further research: the importance of ultrafine particles, what constituent parts make traffic related air pollution more harmful than $PM_{2.5}$ in general and the importance of coarse particles.

Concerning external cost calculations these can be of help to reveal important health aspects to consider in further research, if done in a transparent way. Some pollutants which are very harmful are released in such small concentrations that the overall effect is still relative limited. Hence, undertaking external cost calculations gives an indication of which pollutants to cover in the models and analyses to make them relevant but at the same time manageable.

Further, there are the questions of how to handle the relationship between Value of a Statistical Life and age, and of which values that should be used for children. This is an area where little research has been carried out. One important area is how to use discounting to account for the time dimension since current air pollution may influence children's health in the future. More research is also needed regarding the valuation of morbidity. Here there are two issues to consider, the value of the welfare loss from being ill and the cost of illness. We have not found reliable estimates of these components for Sweden.

1 Introduction

Background and purpose

The overall objective of Swedish transport policy is to ensure the economically efficient and sustainable provision of transport services for people and businesses throughout the country. The Swedish Transport Administration² is from the first of April 2010 responsible for long-term planning of the transport system for road, rail, maritime and air traffic, as well as for building, operating and maintaining public roads and railways. In order to achieve this in an efficient manner, investments in infrastructure are preceded by different types of analyses including cost-benefit analysis (CBA). For this purpose a number of analytical methods and models have been developed³.

Since there were previously different authorities responsible for the different transportation modes, general guidelines for how to perform these analyses and values to be used were commissioned by the SIKa institute⁴. Their recommendations were preceded by discussions in a working group called ASEK where representatives from different transport authorities in Sweden participated. There were also representatives from the Swedish Environmental Protection Agency in the working group.

This report is produced in response to a request from this working group. The purpose is to provide a background for a discussion concerning the methods and values used in cost-benefit analysis in Sweden for air pollutions' impact on human health and the research needs in this area. In this report we will present the current state of the art on how these external costs are calculated and the inputs used in these calculations. The main focus is on the current knowledge regarding the health impacts of transport emissions. We use the latest version that has been presented of the ExternE methodology (Bickel and Friedrich, 2005) as our reference point. We also provide information on more recent research regarding pollutants and health impacts as well as emission- and exposure modelling⁵.

² This a newly formed public authority. Previously the same issues for road were administered by the National road administration and for Rail by the National Rail Administration.

³ A summary of different models used are presented on the webpage <http://www.trafikverket.se/Foretag/Planera-och-utreda/Planerings--och-analysmetoder/Prognos--och-analysverktyg/> (In Swedish. Accessed April 20, 2010).

⁴ Swedish Institute for Transport and Communications Analysis. Since the first of April its work is performed by the newly formed agency Transport Analysis (Trafikanalys).

⁵ For a broader overview of the development of methods and inputs in this type of analyses see Forslund, Marklund and Samakovlis (2007).

There are several reasons for the focus on the ExternE methodology. Since, as discussed in WHO (2008a), the literature regarding all the inputs in these calculations is vast much more resources (in time, budget and competence) would be needed if a full literature overview were to be undertaken. Hence there is a need to rely on other sources that has made summaries of the current evidence. Furthermore, as described in Nerhagen et al. (1995), the ExternE methodology is the basis for how the current calculations are done in Sweden. This is also the methodology underlying most of the external cost estimates used for policy evaluation within the EU and also the method used in EU studies such as HEATCO, CAFE and *The Handbook on estimation of external cost in the transport sector* as well as in the WHO health reports from 2008 (Bickel et al., 2006; Hurley et al, 2005; Maibach et al., 2007; WHO, 2008a; WHO, 2008b). This approach is also used in the model BenMAP developed by US EPA (see next section).

The most recent research, however, has not influenced the values and models currently used in these calculations in Sweden. As discussed in Nerhagen et al. (2005) there are several arguments in favour for an update. The current calculations are the result of work done by Leksell (2000) based on earlier versions of the ExternE methodology. In Nerhagen et al. (2005) the results from the Swedish calculation approach was compared to the calculations done with the EU version of ExternE. Two major problems were then discovered with the current calculations used in Sweden. First of all the exposure calculation is crude and its accuracy has not been assessed. Secondly the value placed on exhaust emissions of particulate matter is too high. Another problem found with the current method of calculation is that it is not as transparent in all its parts as the original ExternE method. This in turn makes it almost impossible to revise the model in the light of new empirical evidence regarding health impacts or economic values. For more discussion on this issue, see Nerhagen et al. (2005).

Before presenting our results, however, we provide a brief description of why economists use information on external costs. We think this is important since many research disciplines are involved in these calculations and for many the concept of external costs may be unfamiliar. We will also give a short description of the ExternE projects and the Impact pathway approach that has been developed in these projects. After this introduction we provide a brief summary in *chapter 2* on aspects that need to be considered and assumptions made when making these calculations. In *chapter 3* we present an overview of recent knowledge and research about airborne pollutants and their health impacts. This is followed by an overview in *chapter 4* of

economic valuation of health impacts and the estimates used in different applications. In *chapter 5* we discuss the information that is needed in practice to do these calculations, especially regarding exposure quantification and the development that has taken place regarding modelling in this area. We also present results from some recent studies and discuss how different components in the calculation influence the final external cost estimate. Finally, in *chapter 6*, we discuss if there are aspects missing in the current ExternE approach and suggest areas for future multidisciplinary research.

Why external cost calculations

Societal decision making implies making trade-offs between different objectives. Approaching these objectives often results in direct or indirect costs to society (national or local governments) but also for individuals. In some cases these decisions are straightforward with few and clearly defined costs and benefits, in economic terms, but this is most often not the case in decisions regarding environmental improvements. Air pollution, which is the problem in focus in this study⁶, comes from various sources and has several different impacts on the natural environment and/or on human health. Moreover, the effects can occur instantly but also some time into the future. In order not to neglect negative impacts of pollution in decision making, economic valuation methods have been developed with the purpose of “placing a price” (referred to as internalising) on these impacts.

The basis for internalising is the impact of pollution (measured as costs) on third parties; the term external cost is commonly used in economics. There are two main reasons for obtaining this information. One is that if we have an estimate on the cost of pollution, the external cost, a pollution tax can be placed on the production of a good. As discussed in economic theory, internalising external costs through pollution taxes will correct for the market failure caused by pollution. This is because prices are bearers of information that sends signals in a market system. Pollution taxes give actors (consumers and producers as well as policy makers) economic incentives to act and to change behaviour since they raise the cost of shirking. However, for various reasons, a pollution tax is not always possible to impose on the production of a good. Therefore, the other reason for having information on the external cost of pollution related to a good is because it is useful for the design of other policy measures, for example standards or limit values (see for example Hanley et al., 1997

⁶ This report focuses on the problems related to air pollution but the underlying theory and the methods described are the same as those used to calculate the external cost for noise, see Andersson and Ögren (2007) for a Swedish example. Air pollution also has a negative influence on the natural environment but in this report we only focus on the calculation of external health costs.

for a description of environmental economics, i.e. the theory behind the use of prices in environmental policy)⁷.

External cost estimates are also important for investment analysis for example to provide values for resources that are not traded in a market, such as clean air. Their influence on the outcome however may be minor. This is because the underlying data on which the calculations are made, the effects estimate, can be small. In transportation analysis the evaluation is based on two traffic forecasts, one for the reference alternative (i.e. the current situation and infrastructure) and one for the investment alternative. The effect is the difference between the two. If the change in traffic is small between the two alternatives, then the change in emissions will be small and hence also the total external cost estimate for the health impacts. When evaluating the results of a CBA, or other cost or benefit calculations, it is important to analyse the drivers of the results, is it the external cost estimates used or is it the calculated impact⁸. We will return to this issue in chapter 5 when we discuss how to interpret results from recent studies on external cost calculations.

For emissions the external cost calculations are often based on an assessment of the impact that the emissions have on the environment and the economic value placed on these impacts. Early examples of these type of calculations for the transport sector are found in Small and Kazimi (1995) and Delucchi (2000). They referred to these calculations as the damage cost approach. In the environmental economics literature it has instead been called the dose-response method and can be applied to different types of pollution. In recent years, in order to obtain consistent estimates for the negative impacts of air pollution, efforts have been made both within EU and the US to develop more standardised approaches for this type of assessment. In EU the methodology developed in the ExternE projects is commonly used. A brief description is given in the next section (for detailed descriptions see Friedrich and Bickel, 2001; Nerhagen et al., 2005; Nerhagen et al., 2009). The US Environmental Protection Agency (EPA) has developed a similar tool referred to as BenMAP⁹.

⁷ The evaluation criteria used in economics is economic efficiency which is achieved if the benefit of production or consumption outweighs the cost (based on the assumption that all benefits and cost are accounted for). A central underpinning of economic theory is that there are scarce resources and that trade-offs are needed. Hence, for the sake of accomplishing other welfare enhancing actions, society in most cases has to accept a certain level of "bads" such as some impact on the environment.

⁸ For a more detailed discussions on CBA and what the drivers of the results are, see Hultkrantz, Li and Nerhagen (2003) as well as Nilsson and Pyddoke (2009) where the latter discusses this issue in relation to estimates of carbon dioxide emissions.

⁹ Abbreviation for Environmental Benefits Mapping and Analysis Program. See <http://www.epa.gov/air/benmap/>.

The ExternE projects and the Impact Pathway Approach

ExternE (External costs of Energy) is a project that started in 1991 with several European partners involved, financed by the European Commission. The aim of the project initially was to make an assessment of the externalities associated with electricity generation (ExternE, 1995) while the methodology used for transportation was developed in the end of the 1990s. It is presented in great detail in Friedrich and Bickel (2001). The model has also been used in several other EU funded research projects such as UNITE, HEATCO, and CAFE¹⁰. A handbook for external cost calculations for transports has also been produced, where the estimates presented for different countries are based on the results from HEATCO and CAFE (Maibach et al., 2007). As discussed in Lindberg (2009) it is difficult to assess how relevant these values are in a Swedish context without detailed calculations (case studies)¹¹.

Within the ExternE projects the Impact – Pathway Approach (IPA) has been developed, which is a method for quantification and monetisation of impacts, i.e. external cost calculations (European Commission, 2003; Krewitt, 1998). This method is divided into four different steps, see Figure 1 (European Commission, 2003). The first step is to identify the source and the quantity of the emissions. The second step is to calculate the dispersion of these emissions throughout the area of interest for the study. In the third step the application of exposure-response function is used to yield loss, i.e. for example the health effects of the emission in focus. Finally, a monetary valuation is made using world market prices if possible. However, since human health does not have a world market price other methods are used for the valuation. Mainly willingness-to-pay (WTP) or willingness-to-accept (WTA) values are used derived from stated or revealed preferences studies (Hurley et al., 2005). The Impact pathway approach can indicate the relevance of different emissions in comparison to each other and the total impact of traffic emissions. The external cost calculations are done with EcoSense, a tool which has been developed within ExternE. See more about the model in chapter 5.

¹⁰ Unification of Marginal Costs and Transport Accounts for Transport Efficiency (UNITE). Harmonised European Approaches for Transport Costing (HEATCO). Clean Air For Europe (CAFE).

¹¹ For more information on the partners participating in the ExternE project series and the different projects undertaken, see the project home page, www.externe.info.

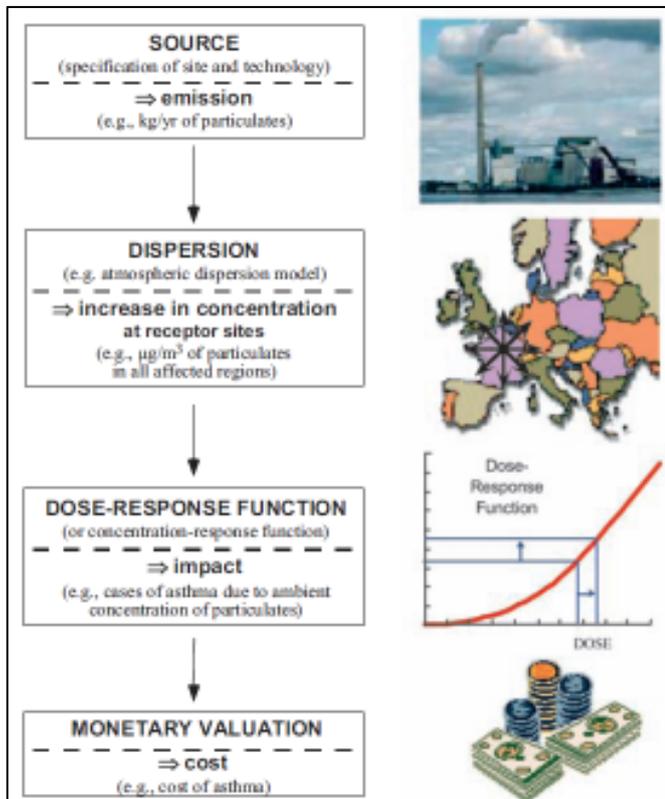


Figure 1 The four principal steps of the impact pathway approach
 Reference: Bickel and Friedrich (2005)

2 The Impact Pathway Approach - theoretical issues and assumptions

The Impact Pathway Approach is a bottom-up approach where the calculated external cost is a function of what influence the emission of a certain pollutant has on human health, and the value of this health impact. Irrespective of context, these calculations are multidisciplinary tasks and involve a number of steps as was illustrated in Figure 1.

A more formal description of how the external health cost calculations are done in the Impact Pathway Approach is given by the following equation (1), (which is a modification of an equation in Ostro and Chestnut (1998)¹². It describes the yearly cost (benefit) due to a increase (reduction) in concentration C from a change in emissions of a certain pollutant from a specific source:

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

where

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

POP = population exposed to $\Delta C_{a,i}$

$B_{a,j}$ = 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 .

This calculation has to be done separately for each pollutant since the effect estimates $P_{i,j}$ (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 yearly health cost for the change in emissions from each source¹³. What this expression reveals is that this calculation requires data from several different sources. What is done in one step will also influence the information needed and assumptions made in the following steps. We will therefore give an overview of some important issues that have been discussed in the literature regarding these calculations.

Non-linearities

The calculation presented in equation 1 rests on an assumption of linear relationships and in most applications linear relations are assumed (Small

¹² In the study by Ostro and Chestnut (1998) they only included the first two components in the expression (1). This is commonly referred to as a health impact assessment (HIA).

¹³ In theory this is quite straightforward but in practice it is more difficult since it is difficult to determine with certainty what the health impacts of a certain pollutant are. Hence, when choosing pollutants and health endpoints to include in the calculation the analyst has to consider how to avoid double-counting.

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). However, according to economic theory, assuming linearity implies that only minor changes from the current state can be evaluated. The reason is that both economic values and exposure-response relationships are likely to change the further we move away from the current situation. The exposure-response functions for example are derived from studies with marginal changes in air pollution concentrations. As for economic values it is common to assume marginal diminishing utility which implies that the more we have of a good the smaller is the value an individual places on an additional unit of the good. For more details on this issue see Viscusi and Gayer (2005). They conclude that extrapolating results beyond the actual underlying empirical data, and making incorrect assumptions, will have a large impact on the final cost estimate.

There are however some nonlinear relationships in the exposure calculation that may be important to account for in these calculations (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 locally will be higher if it is released in urban areas where the population density is high. Another reason for non-linearity is if the formation of secondary pollutants (resulting from chemical transformations) 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 while other studies have show mixed results (Muller and Mendelsohn, 2007; Jensen et al., 2008).

Latency and discounting

Another aspect to account for in these calculations is that the health effects caused by air pollution can occur instantly but also some time into the future, i.e. with latency. Hence, when calculating the external cost for a certain health impact there needs to be a correspondence between the estimated health impact and the economic value used. For chronic diseases such a chronic bronchitis the value used should reflect that this disease will affect an individual's quality of life several years into the future. The approach economists use to account for the time dimension is discounting which implies that the value placed on outcomes that occur in the future are given a lower weight than if they occur today. There are several reasons for the use of discounting. One is that investments today will generate greater amount of resources in the future. Another is that people prefer consumption today (or health improvements today) instead of in the future, a proposition that has also been verified by empirical analysis (Cropper, Aydede and Portney, 1994).

One important issue discussed in relation to discounting is the size of the discount rate. This choice is of importance since using a higher discount rate

implies giving less weight to benefits that occur in the future. For consistency it is common that governmental policies are evaluated using the same discount rate. In Swedish transport policy it used to be the SIKA¹⁴ institute that gave recommendations on the discount rate (based on discussions in the ASEK group) to be used in CBA.

Uncertainties

These types of calculations are complex and based on assumptions in every part of the chain. Hence, for the user of the result it is important to be able to assess the reliability of the results and how they are influenced by the assumption made. One way to validate the results is by comparing them with similar calculations where other models have been used. For this reason transparency has been one of the objectives in the ExternE projects (Friedrich and Bickel, 2001).

To achieve this objective the descriptions of the ExternE methodology contains thorough discussions on the underlying assumptions, the inputs into the calculations and the reasons for the choice of inputs (Friedrich and Bickel, 2001; Bickel and Friedrich 2005). One issue they discuss is the risk for double counting for example due to the close correlation between emissions of different pollutants. To reduce the risk of double counting they make a careful choice regarding the pollutants and the health impacts to include in the calculations. They also discuss the possibility of using “conventions for evaluating external costs” that involves reaching agreement on key values to be used in these analyses, for example the discount rate. One of the advantages they see with using such an approach is that it helps narrowing down the range of estimates to be used in a sensitivity analysis, which is used for testing the robustness of a study (Bickel and Friedrich, 2005 page 6). In the documentation they also provide upper and lower bound estimates that they recommend for use in sensitivity analysis¹⁵.

¹⁴ The latest recommendations are presented in SIKA rapport 2009:3.

¹⁵ In the model BenMAP developed by US EPA they have used a different, more statistical, approach based on calculations where exposure-response functions and valuation functions are inputs in the calculations. Hence, the results of these calculations are statistical distributions for the estimated costs.

3 Air pollution and health impacts

Introduction

The main information used to calculate dose-response functions are results from epidemiological studies. In these studies the aim is to define the human health impact of different substances based on e.g. statistical analyses and measurements. One example of a common method in epidemiology is to use time series data. The researcher looks for the correlations between a daily concentration of a (mix of) pollutant(s) which a population is exposed to and for example the number of hospital admissions for asthma attacks (Bickel and Friedrich, 2005).

In general, the scientific research strongly indicates that traffic emissions do generate a negative impact on human health (Sehlstedt et al., 2007; American Heart Association, 2010). For economic valuation the contribution of each specific pollutant is important to know. However, in epidemiological studies it is difficult to separate the contribution from individual pollutants (Sehlstedt et al., 2007; American Heart Association, 2010). One complication is that many emissions are correlated and hence it may be difficult to state exactly which pollutant causes which effect. Conducted epidemiologic studies are often based on measurement data and measurements are only done for a few pollutants. Therefore NO_x is often used as an indicator for other pollutants from traffic since there are long measurement series for NO_x .

It is in particular problematic to separate sulphur dioxides (SO_2), nitrogen oxides (NO_x) and particulate matter (PM_{10}), since these tend to vary together and it is not clear if PM is reflecting the effects of SO_2 and NO_x or the other way around (Bickel and Friedrich, 2005). Further, it could also be the case that other pollutants absorbed on PM affect the toxicity of PM (Bickel and Friedrich, 2005). Due to this some of the dose-response functions found in the literature has not been used in ExternE, since the specific effects has not strong enough evidence. If the dose-response functions for a specific pollutant reflect the effects of other pollutants as well, this can generate double counting¹⁶.

Here follows an overview of the specific known pollutants from traffic and their associated health effect. We start with the pollutants and the assumptions made in the most recent version of the ExternE methodology (Bickel and Friedrich, 2005). We then describe issues that have been discussed in more recent literature on what seems to be relevant assumptions

¹⁶ This problem is discussed in greater detail in for example the overview presented by the American Heart Associations (2010). The extent of correlation between pollutants will depend on situation-specific circumstances. In Sweden there is less correlation between measured PM_{10} concentrations and secondary pollutants since the PM_{10} concentrations are dependent upon the emissions of road wear that have a seasonal pattern with high concentrations in spring.

to make. The surveyed literature mainly focuses on health impacts related to adults or the general population. Recently, the impact of air pollution on children's health has received increasing attention and projects have been undertaken both internationally and nationally. For details about current knowledge and ongoing research in this area, see WHO (2008a and 2008b), Naturvårdsverket (2010a), and Naturvårdsverket (2010b).

Directly emitted pollutants

*Nitrogen oxides (NO_x)*¹⁷

According to Bickel and Friedrich (2005) there is no convincing evidence of a direct impact of nitrogen oxides (NO_x) on human health. Therefore it is suggested not to be included in the cost calculations in ExternE (Bickel and Friedrich, 2005; Nerhagen et al., 2005). NO_x has been measured during a long period of time, since it has been considered as a harmful pollutant for human health (Socialstyrelsen and Karolinska Institutet, 2009). Health effects which have been indicated to have associations with NO_x are e.g. pulmonary effects in asthmatics and reduced lung-growth in children (WHO, 2005). This knowledge has lately been questioned and there has been evidence in support of lowering the valuation of NO_x. However the evidence has not been assessed as strong enough and the risk of correlation with other pollutants are strong (WHO, 2005) However, NO_x is still of interest for environmental medicine purposes (Pershagen, 2009), and is often used as an indicator for other pollutants (WHO, 2005). In the EU there are laws regulating the emission levels of NO_x and they are also valued in the recommendations made by the ASEK-group (Nerhagen et al., 2005; Naturvårdsverket, 2004).

Sulphur dioxides (SO₂)

In ExternE there are neither economic valuations nor dose- response functions (health effects) used for SO₂. This is due to the risk of double counting, since SO₂ also generates secondary pollutants and since the evidence of SO₂ specifically causing adverse health effects is low. However, Bickel and Friedrich (2005)¹⁸ refer to a study conducted in Hong Kong which indicates that there could be significant direct health effects of SO₂. SO₂ could generate mortality and cardio-pulmonary (heart and lung) morbidity, such as hospitalisation, consultation of doctor, asthma, sick leave and restricted activity.

Particulate matter (PM)

Particulate matter is usually measured as PM_{2.5} and PM₁₀. The numbers indicate the size of the particles, i.e. less than 2.5 micrometer (µm) and less than 10 µm in diameter (Sehlstedt et al., 2007). In general, smaller particles are causing cardiovascular effects while larger particles are more likely to

¹⁷ NO_x includes NO + NO₂

¹⁸ Hedley et al. (2002), however, a complete reference is missing in Bickel and Friedrich (2005).

cause respiratory diseases (Sehlstedt et al., 2007; American Heart Association, 2010). The reason is that particles above 10 μm are filtered out in the upper airways while smaller particles are not. PM from combustion processes are $\text{PM}_{2.5}$ while PM from road wear is in the coarse ($\text{PM}_{10-2.5}$) range (American Heart Association, 2010). Hence, when *measuring* PM_{10} the estimate will be influenced both by emissions from combustion sources and from road wear. In ExternE they originally referred to PM_{10} in their documentation since this used to be the most common measure although they in reality only did calculations based on measurement of PM from combustion.

Earlier, ExternE was treating the emissions from power plants as equally toxic as PM_{10} and vehicle (exhaust) emissions as $\text{PM}_{2.5}$ (Bickel and Friedrich, 2005, p. 84). In the update in 2005 these assumptions have been revised and will be presented below.

Vehicle exhaust, $\text{PM}_{2.5}$

Vehicle exhaust particles are usually grouped into $\text{PM}_{2.5}$, which embraces several different particles with a diameter less than 2.5 μm as mentioned above. Most of the exhaust particles are however mainly equal to or smaller than $\text{PM}_{0.1}$ (Nerhagen et al., 2005). $\text{PM}_{2.5}$ has been proved to cause mortality and morbidity, both acute (short-term) and chronic (long-term). Associated health effects are both respiratory and cardiovascular (Bickel and Friedrich, 2005).

The smallest particles, nanoparticles or ultrafine particles ($< 0.1 \mu\text{m}$), seem to be very toxic due to their size and the very large number of particles. Even though the mass is small the total surface area may be extremely large. When we inhale particles it is primarily the surfaces of the particles that interact with our cells. This makes the surface area critical for toxic effects (Sehlstedt et al., 2007). According to WHO (2005) there are not enough epidemiologic evidences for the health effect associated with ultrafine particles to conclude on a dose-response relationship. The report from the American Heart Association (2010) concludes that it appears that traffic-related pollution appears to increase cardiovascular risk but concludes that it is unclear if this is due to the ultrafine particles.

Wear particles, $\text{PM}_{10-2.5}$

In ExternE wear particles are not considered¹⁹. However, they do use the measurement PM_{10} but for vehicle exhaust only. The main source for wear particles are wear of vehicle components (e.g. brakes and studded tyres) and resuspension of road dust by traffic. The levels vary between seasons, and in Sweden the highest concentrations are measured during winter/spring due to the use of studded tyres and winter operation, i.e. sanding and salting (Sehlstedt et al., 2007; Gustafsson et al., 2010). The wear particles are coarser than exhaust particles and are usually grouped under the definition $\text{PM}_{10-2.5}$. They make a large contribution to measured PM_{10} concentrations in

¹⁹ Neither in HEATCO nor in CAFE.

Swedish cities, especially at street level. Measured PM₁₀ is associated with acute and chronic morbidity as well as acute mortality. The evidence for chronic mortality is however insufficient (WHO, 2005). However, it is unclear to what extent this is caused by the coarser fractions of PM. In the report by the American Heart Association (2010) it is stated regarding coarse PM that recent findings do not support an association between cardiovascular risk and coarse PM.

Secondary pollutants

Nitrates (NO₃)

Nitrates (NO₃) are formed by oxidation of NO_x and are hence considered as secondary particles. Nitrates can be present on PM_{2.5} (Sehlstedt et al. 2007). There have not been many studies on nitrates' effect on human health. This is due to the lack of time series data since nitrates have not been measured until rather recently (Bickel and Friedrich, 2005). The current view is that primary particles emitted locally have the same impact as secondary particles dispersed on a regional level (Nerhagen et al., 2005). However, in ExternE the impacts from nitrates are assumed to be 0.5 times the toxicity of PM₁₀ (Bickel and Friedrich, 2005).

The view on the impact on human health of nitrates is somewhat divided. According to Sehlstedt et al. (2007) the toxicity of nitrates seems low and only few epidemiological studies have been carried out.

Sulphates (SO₄)

Sulphates (SO₄) are secondary particles formed by oxidation of SO₂. SO₄ is assumed in ExternE to have the same health impact as PM₁₀. For SO₄ there are few studies indicating health association (Bickel and Friedrich, 2005). According to the newer study by Sehlstedt et al. (2007) SO₄ is associated with less health endpoints than PM_{2.5}. However, there are only few studies on SO₄ and these are showing inconsistent results. SO₄ is also correlated with metals from combustion processes (Sehlstedt et al., 2007).

Ozone (O₃)

Ground level ozone (O₃) is not emitted directly through vehicle exhaust, it is a secondary pollutant formed when NO_x and volatile organic compounds (VOC) react with sunlight. O₃ causes adverse health effects, such as acute mortality and morbidity (Bickel and Friedrich, 2005; Hurley et al., 2005). There is no strong scientific evidence that long-term exposure of O₃ would cause any additional health effects to those aggregated over time of short-term exposure (Bickel and Friedrich, 2005). The health effects associated with O₃ are respiratory hospital admission, minor restricted activity days, increased need of respiratory medication used by people with respiratory diseases and cough and lower respiratory symptoms (LRS) among children in the general population (Hurley et al., 2005). Ozone seems to affect the

lung function (respiratory system) in the same way for both people with asthma and without. However, new reports indicate that asthmatic people might be more sensitive (Karolinska Institutet, 2010a).

Hydrocarbons

The ExternE Methodology assesses the importance of different pollutants that belong to the chemical group hydrocarbons. The substances they consider are discussed below. This is in contrast to the current external cost calculations used in Sweden where the estimation is based on assessment of the emissions of VOC as a group (Nerhagen et al., 2005; SIKa, 2009)

Polycyclic Aromatic Hydrocarbons

Polycyclic Aromatic Hydrocarbons (PAHs) include a large number of different hydrocarbons consisting of two or more condensed aromatic rings (Sehlstedt et al., 2007). These are usually bond to particles (Sehlstedt et al., 2007). The most studied and so far the only PAH which is regulated within the EU is Benze-[a]-pyrene (BaP). According to Bickel and Friedrich (2005) there is strong evidence from epidemiological studies that some PAHs, especially BaP, are carcinogenic. Flouranthene is a volatile (gaseous) PAH which is also mentioned in ExternE. From modern diesel cars volatile PAH such as Flouranthene seems to be a greater health risk than the lower emissions of BaP (Naturvårdsverket, 2004). On the other hand Karolinska Institutet (2010c) states that the carcinogenic effect of Flouranthene is estimated to be about twenty times lower than BaP, and the concentration on the ambient air in Sweden only ten times the one of BaP.

The literature review by Sehlstedt et al. (2007) suggests that quinones are particularly reactive and affects the respiratory system as well as vascular system (e.g. increased blood pressure). These together with nitro-PAHs are suggested by Sehlstedt et al. (2007) to be studied further.

1,3 butadiene and Benzene

1,3 butadiene and Benzene are both associated with cancer. However, while Benzene has been classified by the International Agency for Research on Cancer (IARC) as carcinogenic, 1,3 butadiene is only classified as potentially carcinogenic (Bickel and Friedrich, 2005). Lately the effect of 1,3 butadiene has also been reduced. The dose-response functions for these pollutants are rather uncertain and needs to be investigated further (Nerhagen et al., 2005). These constitute a small part of the total health cost caused by traffic emissions (Nerhagen et al., 2005).

Formaldehyde

Formaldehyde has been classified by IARC to be a probable human carcinogen (Bickel and Friedrich, 2005). The cancer effect at low levels of exposure, in absence of damage to respiratory tract tissue, seems to be negligible (Bickel and Friedrich, 2005).

Other pollutants considered in ExternE

Diesel particles

There have been weak associations between diesel particles, or soot, and lung cancer, but these particles are classified as probably carcinogenic to humans by IARC (Friedrich and Bickel, 2001). Diesel particles are probably correlated with other carcinogenic pollutants, e.g. BaP, and should not be valued due to risk for double counting, both in terms of health risk and economic values (Friedrich and Bickel, 2001; Nerhagen et al., 2005).

Carbon oxide (CO)

Carbon oxide has been identified to have adverse health effects for elderly people above 65 years of age. The associated health effect is cardiovascular (Bickel and Friedrich, 2005). It is however difficult to do specific dose-response functions for CO and it only constitutes a small part of the total health cost from traffic emissions (Nerhagen et al., 2005).

Dioxins

Dioxins are one of the most studied pollutants and there have been several epidemiologic studies conducted. The main health effect associated with dioxins is cancer (Bickel and Friedrich, 2005). Earlier one of the main sources of dioxin in Sweden was traffic exhaust, now the main source is via food consumption (Karolinska institutet, 2010b; Socialstyrelsen and Karolinska Institutet, 2009).

Metals

Arsenic (As), Cadmium (Cd), Chromium (Cr-VI), Nickel (Ni), Mercury (Hg) and Lead (Pb) are the metals discussed in ExternE. These are all associated with cancer, except for Hg and Pb which are associated with neurotoxic diseases and for Pb IQ decrement seems to be the major health impact (Bickel and Friedrich, 2005).

Summary

The following Table 1 shows an overview of the different emissions associated with adverse health effects in ExternE. We have summarised the emissions which we have interpreted as being used in cost calculations in ExternE. For several emissions the scientific evidences for the health impacts are uncertain as stated above and the literature is not all that clear on whether the calculations are being used or not today.

One example is black smoke or soot which has been discussed in Friedrich and Bickel (2001) to be used as an alternative to PM. However, they concluded not to use black smoke. In the updated version Bickel and Friedrich (2005) black smoke is stated in the summary tables, but not further discussed in the text. The same goes for diesel particles which are also discussed in Friedrich and Bickel (2001) but not in the Bickel and Friedrich (2005).

The classic pollutants PM₁₀, SO₂, NO₂, and O₃ are considered in ExternE but SO₂, NO₂ only when it comes to their contribution to secondary PM (Bickel and Friedrich, 2005; Hurley et al., 2005). SO₂ and NO₂ are closely correlated to primary PM and are not included due to the risk of double counting. Hence, when an external cost is set for the exhaust emissions of SO₂ and NO₂, these estimates are based on the health associations (dose-response functions) of the secondary particles nitrates and sulphates.

Further, in ExternE dose-response functions for hydrocarbons and the other pollutants are derived from the US EPA for consistency. Concerning formaldehyde the recommendation is only to use the function in sensitivity analyses.

For quantifications of relative risks and dose-response functions for the different pollutants see Bickel and Friedrich (2005).

Table 1 Traffic emissions health effects considered in ExternE

Impact Category	Receptor	Pollutant	Morbidity/Mortality
Respiratory	Asthmatics Adults Children	PM ₁₀ O ₃	Morbidity & Mortality Morbidity & Mortality
	All	PM _{2.5} PM ₁₀ O ₃ NO ₃ SO ₄	Morbidity & Mortality
Cardiovascular	Elderly 65+	PM ₁₀ CO	Morbidity & Mortality Morbidity
	All	PM ₁₀	Morbidity & Mortality
		PM _{2.5}	
		NO ₃ SO ₄	
Cancer	All	Benzene Benzo[a]Pyrene 1,3 butadiene Dioxins Formaldehyde ²⁰ As Cd Cr-VI Ni	Morbidity & Mortality
Neurotoxic	All	Pb	-

Reference: Friedrich and Bickel, 2001; Bickel and Friedrich, 2005

²⁰ Only in sensitivity analysis.

4 Economic valuation of health impacts

Introduction

There is an extensive literature that deals with questions on economic valuation in general, but also on the valuation of health risk reductions in particular. Initially, the value used in this context was related to the financial costs lost or paid due to a health outcome. In the case of premature mortality the present value of lost income, the so called human capital approach, was used. Similarly, the valuation of morbidity endpoints was based on the cost of illness approach where the benefits were assumed to be equal to the savings from medical expenditure plus forgone opportunity cost for being ill.

Both of these approaches however underestimate the welfare loss of a health risk reduction since they do not account for the disutility that individuals experience if the outcome occurs. Hence, current valuation methods seek to estimate individuals' WTP for risk reductions. These methods rest on the assumption that an individual's WTP is an approximation of a change in utility that the risk reduction entails²¹. A brief formal treatment of the difference between the production functions approaches described above and the WTP approach is given in Viscusi and Gayer (2005)²².

The first attempts to obtain WTP estimates relied on the use of market data using revealed preference methods. These methods derive economic values from individuals' choice behaviour in real markets. An early example in the case of mortality risk reductions was the hedonic wage model. In this case the estimate rests on the compensating wage differential that workers receive for riskier jobs. However, a major drawback with revealed preference methods is that there are a limited number of risk contexts that can be explored using actual choices. There are limitations since the choices are generally not representative for a larger population but also because not all risk contexts can be controlled by actions made in markets. Therefore, so called stated preference methods are increasingly used.

In stated preference methods information is obtained from survey data exploring individuals' choice behaviour. The analyst designs a choice context that resembles a market situation or a referendum. The earliest approach used in environmental economics was the contingent valuation method where the respondent was asked to state their WTP (open-ended format) or accept or reject a certain bid (closed-ended format) for a certain

²¹ For more information see the special issue in the journal *Environmental & Resource Economics* (2006) 33.

²² There are numerous books written on issues related to economic valuation. Overviews of this method (fast vilken?) and other economic valuation methods and the theory behind them are given in introductory texts in environmental economics such as Brännlund and Kriström (1998) or Tietenberg (2007). Viscusi and Gayer (2005) provides a summary of issues related to the quantification and valuation of environmental health risks in particular.

improvement. Another more recent format is to ask the individuals to respond to several alternatives in a row where the alternatives are described in terms of a number of attributes. This is often called a choice experiment which has its origin in valuation studies in transport economics. The main objection raised against stated preference methods is that it is difficult to validate that answers to these questions represents actual choice behaviour (a problem often referred to as hypothetical bias).

One problem in finding economic values for this type of assessment is that few valuation studies have been designed for the type of health impacts that are relevant in the air pollution context. Hence, in addition to the uncertainties related to the economic values themselves there are uncertainties related to the transfer of one value from one context to another. This is referred to as benefit transfer by economists and as discussed by Viscusi and Gayer (2005) this type of extrapolations to other groups or contexts is based on strong assumptions.

Valuation of mortality risk reductions

Mortality is the benefit that often comes out as the most important aspect when making assessments of health risk reductions in the general population. The value used in this context is commonly called the value of a statistical life (VSL) because the estimate is derived from an individual's WTP for his/her own marginal (statistically determined) change in the risk of dying prematurely. The question of how large this value is has generated an extensive literature where many different aspects on the problem of valuation have been raised. One of the latter issues considered is if the value should vary with respect to age. This issue has been raised in relation to air pollution since this is a risk that mainly affects a smaller group, the sick and elderly, in the population. However, so far no firm conclusions on this issue have been reached.

The first estimates on the value of premature mortality relied on estimation of production losses. This is the so called human capital approach. However, as discussed in the introduction of this section, this is only one part of the total value of a statistical life. To this should be added an estimate of the welfare loss. An example of components that are included in calculations of the total value of a statistical life is found in the value used in the transport sector. The value of a statistical life in this context is composed of two parts, the welfare estimate equals to 21 million SEK and the so called material costs (hospitalization, property losses, administration and production loss) amounts to 1.32 million SEK (SIKA, 2009).

One way to arrive at an estimate that accounts for length of life in the case of premature mortality was formalised in early work in the ExternE projects. They used the estimate of VSL to obtain an estimate of the value of a life year (VOLY) using annuity calculations (Friedrich and Bickel, 2001):

$$VSL = \sum_{t=0}^T \frac{VOLY_{acute}}{(1+r)^t}$$

where

VSL the value of a statistical life
VOLY_{acute} the value of a life year lost
t year
T average expected length of life
r discount rate.

This however rests on the assumption that every year is given the same value which is not a finding supported by empirical research. Many studies have found that WTP first increases and then decreases with age.

Latency is an additional complication. For adults it is expected that some pollutants will contribute to the development of heart conditions which in turn will result in premature deaths in the future. Hence, when calculating benefits resulting from exposure reductions today we need to discount the value for health gains that will occur in the future.

The calculation of the discounted value is straightforward if we know the value of a life year and the discount rate. In ExternE they use the following formula to arrive at a present value for a life year expected to be lost in the future (Friedrich and Bickel, 2001):

$$VOLY_{chronic} = \frac{VOLY_{acute}}{(1+r)^t}$$

where

VOLY_{chronic} the value of a discounted life year
VOLY_{acute} the value of a life year
t latency period, the time between exposure and death
r the discount rate.

However, as discussed earlier, the problem is to determine VOLY. Moreover, the length of the latency period is also an estimate that is difficult to establish empirically. The present value estimate will also be influenced by size of the discount rate; a higher discount rate implies a lower present value. In Sweden the recommended value for the discount rate from ASEK is four percent (SIKA, 2009)²³.

Since how to value premature deaths for young and old is an important issue, especially in a policy situation where the policy maker often has to rely on benefit estimates from other studies and other contexts, many studies have explored these aspects in recent years. Examples include a study within the

²³ There is a vast literature on discounting and on ways to derive discount rates. Since this is an influential parameter in cost-benefit analysis it is common to use discount rates set by some government agency but also to use a high and a low value in a sensitivity analysis.

NewExt project (Alberini et al., 2004), a study commissioned by DEFRA²⁴ in the UK (Chilton et al., 2004) and a recently published American study that include morbidity and mortality risks in the same questionnaire (Bosworth et al., 2009). Unfortunately, none of these answers the basic question that is if people value risk reductions for children higher than for adults and if so, by how much. They rather confirm the general finding that understanding individual choice behaviour and determinants of choice behaviour regarding changes in risk is a challenging task. For a more thorough discussion on the valuation of saving lives see for example Nerhagen et al., (2005), Bickel and Friedrich (2005) and Andersson and Treich (2008).

Regarding values to be used for premature mortality the ExternE Methodology Update (Bickel and Friedrich, 2005) gives recommendations for median and mean estimates for both VSL and VOLY based on a study by Alberini et al. (2004). The median estimates are 50 000 euro for VOLY and 1 million euro for VSL. The mean estimates are 125 250 euro for VOLY and 2.3 million euro for VSL. All these estimates are for chronic mortality. For acute mortality they use a VOLY estimate of 75 000 euro. All estimates are in price year 2000. They recommend the use of the median estimates since they are unaffected by the distributional assumptions made in the statistical analysis. However, according to economic theory the mean estimate is a better reflection of the true WTP in a population and should therefore be used (Mitchell and Carson, 1989). The mean estimate in different studies is also the basis for the value of 21 million SEK (about 2 million euro) recommended by the SIKa institute for Swedish analyses. This recommendation is based on a recent research review done by Hultkrantz and Svensson (2008).

The point estimate recommended for VSL in the US EPA's model BenMAP is 6.3 million USD in price year 2000 (around 5 million euro). The basis for this estimate is 26 valuation studies. In the description of the model they do not discuss nor provide an estimate for VOLY but conclude that their VSL can be used for deaths that occur at different ages (US EPA, 2008). The question of VSL and age however is also discussed in the research done in the US. For an overview see Viscusi and Aldy (2007).

Valuation of morbidity risk reductions

This is not an area that is as researched as the valuation of mortality risks so there are fewer studies to obtain values from. The aspects to include in a benefit calculation however are quite clear. First of all it is common to account for the cost of illness which is the production loss from being away from the regular occupation in addition to the medical expenses related to being ill, ranging from cost of drugs to cost of hospitalization. Secondly, it is the welfare loss that should be valued using WTP estimates.

²⁴ DEFRA = Department for Environment Food and Rural Affairs

Since valuation of morbidity has been less researched compared to mortality, Ready et al. (2004) undertook a valuation study in several European countries (see also Eftec, 2004). This study is the basis for the economic values recommended in the ExternE Methodology update. The original values have been adapted to adhere with the calculation framework in ExternE (see Bickel and Friedrich, 2005 for details). In their recommendation they have summed up the willingness to pay estimates from Ready et al (2004) with cost of illness (COI) estimates. The components and the final estimates are presented in Table 2.

Table 2 Recommended values in ExternE (Euro). Price level 2000.

Health endpoint	COI	WTP	Recommended value
Respiratory Hospital admission	1600	400	2000/admission
General practitioner visit	38	15	53/consultation
Restricted activity day (1 day)	84	46	130/day
Asthma attack	450	220	670/visit
Respiratory medication use	1	-	1/day
Cough	-	38	38/day
Symptom day	-	38	38/day

Reference: Bickel and Friedrich (2005)

As can be seen from these estimates, although the COI values account for the larger share of the total value, if the WTP estimate is omitted a relatively large share of the disutility of being ill would not be accounted for. Some of the final values also need to be added up in a benefit calculation. It can be expected for example that the total cost for a day when a visit is made to a general practitioner is both the cost for the visit (53 euro) and the cost for a symptom day (38 euro) and maybe also the cost of a restricted activity day (130 euro), hence 221 euro in total.

In order to determine how relevant these estimates are for Sweden they would need to be compared to results from Swedish studies. This is difficult however since, as discussed in Nerhagen et al. (2005), almost no research has been done in Sweden on the value of morbidity risk reductions. Regarding the willingness to pay estimates there are, according to our search, only one study that can be used for comparison and that is Samakovlis och Svensson (2004) who did a study using the same questions as in Ready et al. (2004). The results from these two studies are presented in Table 3.

As seen in the table, the estimated values in Sweden are much lower than those for Norway (where those in Norway are at a similar level as values in

other countries in the Ready et al study). Unfortunately, Samakovlis and Svensson (2004) do not discuss the reason for this difference and hence it is not possible to assess which of the values that are the most relevant for Sweden.

Table 3 Willingness to pay estimates for morbidity (Euro). Price level 1998.

Health endpoint	Ready et al. (2004)	Samakovlis and Svensson (2004)
Respiratory Hospital admission	482	205
Restricted activity day (3 days in bed)	190	64
Symptom day	50	14

Reference: Nerhagen et al. (2005)

We have also searched for COI data for Sweden and found a couple of studies that provide such estimates. Most of the studies however do not provide information that can be used for the kind of calculation discussed in this report since they do not provide cost for a particular case of illness²⁵. The previously mentioned study by Samakovlis and Svensson (2004) added some estimates for the cost of illness in their calculations of total health cost from a study done by Huthala and Samakovlis (2003). For production loss they use an average estimate for daily income in Sweden of 647 SEK in 1999 (about 60 euro using an approximate exchange rate of 10 SEK/euro). They also used an estimate for medical expenses of 8 SEK per day for asthmatics. This was derived from information on the total cost for medical services in Sweden which in turn was based on the cost for medicine, general practitioner visits and hospital admission.

A more recent study by Vredin-Johansson and Forslund (2009) uses the same kind of underlying data and provide an estimate of 6117 SEK/day for each case of respiratory illness (in 2007 price level). Unfortunately, none of these estimates gives information with the level of detail needed for comparison with the estimates proposed by ExternE. In the case of Vredin-Johansson and Forslund (2009) the problem is that it is unclear what is included in the cost for medical treatment²⁶.

²⁵ The health cost of air pollution in Sweden has also been estimated in another recent study (Vägverket, 2009). Unfortunately, the calculation in this report is based on disability adjusted life year (DALY) estimates and only total costs are presented, hence they are not comparable to the case based measures that we have presented in this report. We have also searched for other studies using COI-estimates in Sweden. We have found some but they all present total cost estimates. See for example Bolin and Lindgren (2004) and Olofsson (2008).

²⁶ In a study done regarding the health cost of smoking by Bolin and Lindgren (2004), estimates from Roberts (2001) are reported that gives an average estimate of 4740 SEK per hospital treatment and 981 SEK for a visit to a doctor. Hence, the estimate for medical treatment presented by Vredin-Johansson and Forslund (2009) could be a high estimate more likely to represent the cost for hospital treatment.

Another problem with the valuation of morbidity is that most of the research has focused on the valuation of short term impacts that relate to persons already suffering from symptoms or having a disease. Long-term exposure to air pollution may however also result in new cases of a disease. Few studies have looked into the problem of valuing this type of outcome. There are some studies on the valuation of new cases of chronic bronchitis among adults. In Bickel and Friedrich (2005) they discuss this issue and use an estimate from Health and Safety Executive (HSE, a national regulatory body responsible for promoting better safety and health in workplaces in England). HSE has in their calculations added up the case specific costs relating to asthma in order to arrive at an estimate for a new case. The value they arrived at for this outcome was about 60 000 euro and it included: loss of income, medical treatments and the value of pain and suffering (HSE, 2010). In BenMAP developed by US EPA they use an estimate for avoiding chronic asthma of 38 947 USD/case, price year 2000 US EPA (2008). This estimate is based on WTP studies.

5 Factors that influence the external cost estimates

Introduction

So far in this report we have given a general description of the approach used for external cost calculations, some of the aspects that need to be considered when performing them and the current knowledge regarding health impacts from transport's emissions and the economic values placed on them. The outcome of these calculations will however to a large extent be influenced by other inputs on which the calculations are based, especially the emission and exposure modelling that are the first and second step in Figure 1 (where the result in turn depends on the dispersion modelling). We will therefore in this chapter give an update on the research done in this area. Another aspect we think is important to consider when interpreting these types of estimates and the influence they have in different types of analyses, is that they are the result of a combination of different inputs. We will therefore present some results from recent studies and describe the factors that influence the final estimate.

Emission and exposure modelling

Within the ExternE projects a tool has also been developed that can be used for external cost calculations. The model is called EcoSense and it is described as an integrated environmental impact assessment model²⁷. Given that emissions from different sources are provided this model performs the second, third and fourth step in Figure 1. This model can be used for detailed bottom-up-calculation for emissions from single sources in different geographical areas. There is an online version of this model called EcoSenseLE that is free to use. This model however only allows for cost calculations of emissions from different sources on a national scale. Hence, due to the large scale, it is not relevant for the purpose of evaluation of single policies or investments. More detailed calculations for specific areas require that the model is operated by the developer since accounting for local conditions is more complicated. In this case the user has to provide the emission data. This is for example how the calculations for Sweden were done that is presented in Bickel, Schmid and Friedrich (2003).

Hence, the emissions that are the basic input into these calculations is provided by other sources. For transport there has also been emission models developed in EU funded research projects. The most recent is called ARTEMIS (Assessment and Reliability of Transport Emission Models and Inventory Systems)²⁸. This was a research project that started in 2000 and where about 40 research institutes were involved. The project resulted in an emission model that is now being implemented and used in Sweden.

²⁷ See www.externe.info and Tools.

²⁸ See <http://www.trl.co.uk/artemis/>.

There is also ongoing research regarding exposure modelling and an integrated tool for external cost calculation in Sweden. It started with the development of the dispersion modelling tool SIMAIR undertaken by SMHI (the Swedish Meteorological and Hydrological Research Institute). The tool was introduced to help local communities to assess local air quality. Further development has now adapted the model so that it can also be used for exposure assessment. The most recent development is to create an integrated assessment tool for the purpose of assessment of air pollution trends in Sweden and their related external costs. This is ongoing work in a project called Scenarioverktyg²⁹. This tool is based on the Impact Pathway Approach, see figure 1 and equation 1. The input is emissions calculated using the ARTEMIS model and data on Swedish traffic activity.

As for the current external cost calculations used in transport CBA:s in Sweden they are based on a very simplified assessment of exposure and it is not possible to follow the different steps in the calculations. Therefore, as discussed in Nerhagen et al. (2005), how well this assessment represents the change in exposure due to a reduction in emissions is not known.

One thing to note about these integrated assessment models is that some of them can be used both for point and line sources. Road traffic is considered to be a line source since emissions are released along a longer distance. Hence, SIMAIR also has a model that can be used to evaluate changes in emissions from wood combustion, i.e. from point sources. This possibility also allows for evaluation of the cost of emissions from other highly local point sources such as harbours or marshalling yards.

The performance of these models is evaluated in different ways. Regarding the performance of the dispersion models their results are compared to measured concentrations of different pollutants. Regarding cost calculations the outcome of different models for the same calculation area can be compared. Jensen et al. (2008) for example compared the outcome of the EcoSense model with a similar model in Denmark. According to their results the EcoSense model gave higher regional exposure factors. Nerhagen et al. (2005) also compared the result from the EcoSense model for Stockholm with results when more detailed Swedish data was used. Again it was found that the assumptions made about the emissions from traffic and the geographical resolution used when modelling exposure had an important influence on the final cost estimate. Hence, when interpreting the results from these calculations the assumptions used in the exposure modelling need to be evaluated.

Interpretation of results from recent studies

As described above the exposure modelling will have an influence on the final cost estimate but the latter will also be influenced by the combined

²⁹See <http://www.smhi.se/forskning/forskningsomraden/luftmiljo/scenarieverktyg-visualisering-och-utvardering-av-utslappsscenarier-1.363>

effects of other inputs. The dose-response functions for some pollutants are for example stronger and more certain compared to others as addressed in chapter 3. This may also be reflected in the economic valuation, since a pollutant with more severe health impacts will be valued higher. However, when it comes to the total impact it is important to also address the actual concentrations of the pollutant being emitted. Because even if a pollutant may be very toxic the concentrations in the ambient air might be small, indicating a small impact on human health. Meanwhile another pollutant which is indicated to be less toxic can be emitted in much higher concentrations and therefore might cause a higher impact.

To illustrate the influence of these two forces we have included the results from two recent studies. In Table 4 results from Nerhagen et al. (2005) on external cost calculations for Stockholm are presented. The value (price) that is used in CBA is presented in the last column (SEK/vkm). According to these results exposures to vehicle exhausts (chronic mortality) and wear particles (morbidity) result in the highest cost per vehicle-kilometre (vkm). The reason for the former is the high value placed on each kg emitted while the latter to a greater extent is due to the amount emitted per vkm. The highest cost per kg emitted however is given to BaP (due to their impact on cancer) but since the amount emitted per vkm is very low the final cost estimate is also very low.

Table 4 Cost per vehicle-kilometre in Stockholm. 1998 price level

Pollutant and health endpoint	External cost (SEK/kg)	Emissions (g/vkm)	Cost (SEK/vkm)*
Vehicle exhaust (chronic mortality)	552 (1454)	0.027	0.015 (0.039)
Vehicle exhaust (morbidity)	265	0.027	0.007
Wear particles (acute mortality)	4.25 (23.3)	0.343	0.0015 (0.008)
Wear particles (morbidity)	104 (0)	0.343	0.036 (0)
CO	0.005	9	0.000045
Benzene	3.37	0.062	0.0002
1-3 butadien	12.66	0.024	0.00003
BaP	35 606	0.00000042	0.000014
Total			0.060 (0.054)

* The cost in brackets are calculated from assumptions recommended for sensitivity analysis.

Reference: Nerhagen et al. (2005)

The same pattern is found in the results presented in Table 5. In this case the calculations were done with the EcoSense model for the airport in Frankfurt. The external cost calculations are based on exposure assessment for the change in emissions the results from the LTO (landing and takeoff) cycle. Again we find that BaP has the highest cost per kg emitted. The emissions of this pollutant per LTO, however, are very small and hence the final cost estimate for this pollutant is among the smallest. Instead NO_x has the highest final cost estimate due to the large amounts of emissions emitted.

Table 5 Emissions and external costs per LTO for two airplanes at Frankfurt's airport

Pollutant *	External cost (Euro/Ton)	A340-211		B737800	
		Emissions (Ton/LTO)	Cost Euro/LTO	Emissions (Ton/LTO)	Cost (Euro/LTO)
PM _{2.5}	86 000	4.00E-05	3.44	1.90E-05	1.63
NO _x	1 440	4.32E-02	62.21	1.06E-02	15.26
SO ₂	4 300	2.16E-03	9.27	7.31E-04	3.14
NM VOC	1130	3,64E-03	4,11	6,20E-04	0,70
CO	1,4	2,33E-02	0,03	6,10E-03	0,01
Benzene	340	6,90E-05	0,02	1,20E-05	0,00
1-3 butadien	13400	6,50E-05	0,87	1,10E-05	0,15
BaP	4000000	8,00E-09	0,03	3,00E-09	0,01

* The external cost estimates for NO_x and SO₂ are calculated based on the health impact of the secondary pollutants nitrates and sulphates that are formed due to these emissions.

Reference: Hansen and Nerhagen (2008)

Other differences that will influence the final cost estimate are the health impacts that are considered regarding a certain pollutant. Chronic mortality and chronic morbidity appears to result in higher costs compared to acute impacts.

6 Discussion on research needs

The report provides an overview of the current state of the art of models used for and input needed for external cost calculations of the health impacts caused by traffic emissions. As can be seen in this report calculation of the external health costs of traffic emissions is not straightforward and depends on the collaboration between several research disciplines. In the ExternE projects, which have been used as a reference point in this study, there are still uncertainties concerning which pollutants to take into consideration. The emissions and the health endpoints that currently are considered has been summarised in Table 1 and most of the economic values are summarised in Table 2. In the following we highlight some issues where we think further research is needed.

Regarding the health impacts, a newly published review by the American Heart Association (2010) gives a very good overview of the difficulties connected to the determination of health endpoints related to different emissions. Instead of making our own assessment of important research issues we will recapitulate some of the main conclusions in this report. They state that the following issues need further research³⁰:

- the importance of ultrafine particles for health risks,
- what constituent parts that make traffic related air pollution more harmful than PM_{2,5} in general,
- the possible combined effects of particulates and other pollutants, both air borne pollutants and noise,
- the importance of coarse particles where current evidence is that they have no impact.

In practice, this implies further research regarding all aspects of the health impacts of air pollutants resulting from different emissions sources. However, given that this is a complex pattern to disentangle, we think it should be assessed whether a more “structured” research could be undertaken. The aim with such an approach should be to try to rule out the pollutants that are likely not to be health risks. Is it for example possible to make toxicological assessments that have a close connection to the studies undertaken by epidemiologists? So far much research has had to rely on available data on concentration measurements but maybe the increased use of dispersion and exposure modelling can help to design studies in different ways than before.

Moreover, we believe that the external cost calculations also can be of help to reveal important health aspects to consider in further research if done in a transparent way. As shown in chapter 5, some pollutants which are very

³⁰ They are also discussing other issues, e.g. if obesity increases the health risk. This is probably more of an issue in the US than in Sweden. Still, this is an area where more research may be needed since these types of differences between populations may influence the estimated dose-response functions and their transferability to another context.

harmful are released in such small concentrations that the overall effect is still relative limited. Hence, undertaking external cost calculations gives an indication of which pollutants to cover in the models and analyses to make them relevant but at the same time manageable.

On the valuation side, the questions of how to handle the relationship between VSL and age and of which values to be used for children are unresolved. One particularly important area here, where more research is needed, is how to use discounting to account for the time dimension. Children for example may develop a disease when adults due to the accumulated exposure to pollutants over time. Moreover, while a number of projects have estimated the value of a statistical life, less is done regarding morbidity. Here there are two issues to consider, the value of the welfare loss from being ill and the cost of illness. We have not found reliable estimates of these components for Sweden.

As discussed in chapter 5, a calculation tool based on Swedish models and data is being developed in the project Scenarioverktyg. This is an important step forward towards a unified framework for this type of calculation in a Swedish context. Currently however the purpose of the project is to be able to assess changes in total external costs for different air pollution scenarios on a national scale. It is likely that this tool could also be used for policy evaluation in the transportation context. If and how this could be accomplished is also an area where more research is needed. In such a project we believe the multidisciplinary collaboration is very important.

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