Air pollution and children’s health in Sweden

An enquiry into how the economic benefit of improvements in children’s health resulting from reductions in air pollution can be assessed

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SWEDISH ENVIRONMENTAL PROTECTION AGENCY
Preface

The Swedish parliament has adopted 16 Environmental Objectives, which are to be met within a generation, to guide action towards a sustainable environment. One goal is Clean air which states that the air must be clean enough not to represent a risk to human health or to animals, plants or cultural assets. This project is part of the work undertaken by the Swedish Environment Protection Agency (Swedish EPA) to bring about the fulfilment of this goal.

Much research has looked into the valuation of health risks in general and mortality risks in particular. Few studies however have assessed how to value the health risks that children are exposed to and fewer still has assessed the impact of environmental health risks. This is an issue that has received increasing interest since the turn of the century, an example being the Fourth Ministerial Conference on Environment and Health held in Budapest in 2004 that emphasized the connection between a degraded environment and children’s health. However, as is discussed in this report, these are calculations involving methodological complexities so there is not a straight road forward. One of the reasons for the complexity is that several research disciplines provide input to these calculations and there are interconnections between the data used in each step. The assumption and outcome of one step will influence the data needed in the second step.

An OECD-report from 2006, that provided valuable information to this study, ended suggesting that more multi-disciplinary research (gathering economists, epidemiologists, sociologists, psychologists etc) may be necessary in order to obtain sound estimates. We are happy to say that this study is a step in this direction. In this project two epidemiologists and one economist has worked jointly to try to give an update on the current state of knowledge. We have given an example of how benefit calculations of the impact of environmental improvements on children’s health can be assessed and what aspects to consider in such calculations. The main conclusion however, is that still many details need further clarification before the full pattern is understood. Hopefully there will be more multi-disciplinary research undertaken in the future that will help sort out of some of the unresolved issues and uncertainties.

The authors alone are responsible for the contents of this report, which should not necessarily be regarded as reflecting the views of the Swedish Environmental Protection Agency. Contact person at the Swedish EPA was Maria Ullerstam.
Contents

PREFACE 3

SUMMARY 6

SAMMANFATTNING 9

INTRODUCTION 13
Purpose and content of the study 13
Benefit calculation – the dose-response method 14

AIR POLLUTION AND CHILDREN’S HEALTH 18
Impacts due to long-term exposure 18
  Lung function development 18
  Sensitisation to allergens 19
  Development of air-way disease (including asthma) 19
  Reproductive disorders 20
  Cancer 20
Impacts of short-term exposure 20
  Panel studies of children 21
  Register studies of ERVs and admissions 22
  School absence 22
Summary and conclusion 23

ECONOMIC VALUATION OF HEALTH RISK REDUCTIONS 25
Children’s willingness to pay – methodological issues 26
Valuation of morbidity risk reductions 28
  Results from recent valuation studies on adults and morbidity 29
Valuation of mortality risk reductions 31
Other methods used to weigh and compare different health outcomes 34
Summary of evidence and usable values 35

CHILDREN’S HEALTH AND THE BENEFIT OF REDUCING AIR POLLUTION – CALCULATIONS FOR TWO CITIES 38
Exposure quantification 39
Health impact and benefit of reductions in long-term exposure 41
Health impact and benefit of reductions in short-term exposure 42
Discussion of results and uncertainties 43

FINAL REMARKS AND FUTURE RESEARCH 46
Summary

Clean air is one of 16 Environmental Objectives adopted by the Swedish parliament to guide action towards a sustainable environment. This project is part of the work undertaken by the Swedish Environment Protection Agency (Swedish EPA) to bring about the fulfilment of this goal. Much research has been undertaken regarding air pollution and health impacts in the adult population but much less is known about how pollutants influence children’s health. The overriding purpose of this study has therefore been to see how and to what extent the economic benefit from reducing these impacts can be calculated.

To answer this question we provide a brief introduction on the method commonly used to do these kinds of benefit calculations. Two crucial inputs into these calculations are estimates of the health impacts and estimates of the economic values for the health impacts. We therefore start by providing a summary of the current state-of-art regarding these inputs which is based on a survey of the literature in each area. We then perform two case studies that describe how these economic benefits can be calculated and what influences the results. The calculation is based on the findings in the literature reviews and we also describe the exposure assessment that is another crucial input into these calculations. The report ends with suggestions for future research.

Regarding air pollution and health impacts, the finding is that air pollution exposure has been associated with a number of health outcomes in children, many of these partly overlapping and related to respiratory effects. Both long-term exposure and short-term fluctuations have been correlated with adverse effects. However, the involved exposure variables are often not source specific, but may in some cases act as acceptable indicators of traffic related air pollution. Only for a limited number of health effects we have found exposure-response functions that may be used to quantify health effects in children. Most of these have been described also in a previous report (Naturvårdsverket, 2010). New for this report is an estimated exposure-response function for the development of airway disease in the 5-18 age group. For the short-term effects such as hospital admissions, it is possible to calculate baseline frequencies needed for the impact assessments from register data. It is more complicated to estimate the baseline in terms of prevalence (occurrence of disease) or onset of disease, but some types of impacts can be estimated using combinations of data and assumptions. On the relationship between traffic pollution and restricted activity days (for example school absences), effects on pregnancy outcome and in infancy as well as effects of early exposure later in life there is limited information.

As for the economic valuation of health impacts, the conclusion in the literature is that the valuation of children’s health risks is more challenging than that of adults. There are several reasons for this where children not being able to assess and value risk reductions by themselves is the most important one. There is however also the
difference in age between children and adults which is likely to make a difference for the values. As in the case of the quantification of health impacts, little research has been done on the valuation of children’s health risks. Therefore, so far mainly proxies have been used such as willingness to pay estimates derived from parents’ choices and behaviour. The general conclusion is that economic values used for adults in general underestimate the benefits to children and that as high as two times these estimates can be relevant. Since almost no economic valuation studies of this kind have been undertaken in Sweden the estimates we propose are those used in other, mainly European, studies.

Based on the findings in the literature surveys we have, as an example, calculated the benefit of a reduction in children’s exposure of 1 µg/m³ of NO₂ in Stockholm and Umeå. The difference between the cities that we could account for was the number of children that are exposed. The calculation was done for two endpoints; that children having wheeze develop asthma and that asthmatic children are admitted to hospital due to respiratory symptoms. According to our calculations this reduction in exposure in Greater Stockholm would generate a benefit to society of 168 million SEK per year because of fewer cases of asthma, and 47 000 SEK due to fewer hospital admissions (for the price levels in 2000). For Umeå the benefits are smaller, 8 million SEK and 2000 SEK per year.

These benefit estimates however are based on a quite large reduction in air pollution. 1 µg/m³ NO₂ is approximately the reduction in population exposure that resulted in the inner city of Stockholm from the trial with congestion charges where traffic in this area decreased by 15%. To achieve the same reduction in Greater Stockholm or Umeå would require measures that result in quite important emission reductions from transport. To determine if such measures are beneficial from a socioeconomic point of view would require a comparison of benefits and costs on the local scale of the chosen measures. In general it is found in the literature that the benefits are larger when emissions are reduced in densely populated areas.

We also discuss how different assumptions influence the results and the uncertainties related to these types of calculations. There are uncertainties in every part of the calculation chain; exposure, impact assessment and economic valuation. One way to account for these uncertainties is by doing a sensitivity analysis where alternative assumptions are used for important inputs. In our calculations an influential assumption is for example the probability that children with wheeze develop asthma later in life. The largest uncertainty however is probably the cause and effect of single pollutants. In this study NO₂ is used since it is a good indicator of emissions from traffic but if this is the true cause of the effects is still a matter of research and discussion.

This is the first attempt to calculate the benefits for children in Sweden of reducing air pollution. Due to lack of data we have only been able to give an indication of the size of the benefits and only for endpoints related to respiratory diseases.
Therefore, further research is needed in order to determine the accuracy of these estimates, the size of the benefit for other endpoints and all children in Sweden and how the benefits vary between different geographical areas. However we consider such research to be warranted since our estimates suggests that reducing children’s exposure to air pollution result in important economic benefits and there is a need for policy makers to know if and when this is the case.
Sammanfattning

*Frisk luft* är ett av de 16 miljömål som antagits av Sveriges riksdag för att styra samhället mot en hållbar utveckling. Detta projekt har genomförts som en del i det arbete som Naturvårdsverket bedriver för att genomföra detta miljömål. Mycket forskning har genomförts när det gäller luftföroreningar och deras påverkan på vuxna men mindre är känt när det gäller påverkan på barns hälsa. Syftet med detta projekt har därför varit att fastställa hur och i vilken omfattning som dessa hälsövinster kan kvantifieras i ekonomiska termer.


När det gäller ekonomisk värdering av hälsöeffekter är slutsatsen i litteraturen att det är svårare att härleda sådana värden för barn än för vuxna. Orsakerna till detta är flera. De viktigaste är att barn inte förstå eller värdera risker på samma sätt som vuxna. Det finns också en åldersskillnad mellan barn och vuxna som förväntas medföra en skillnad när det gäller det ekonomiska värdet. Det är dock få studier som genomförts när det gäller ekonomisk värdering av barns hälsöeffekter så många av de värden som anges i litteraturen baseras på olika indirekta mått såsom föräldrarnas val och beteende. En generell slutsats från de studier som genomförts
är dock att ekonomiska värden för vuxna är en underskattnning och att ekonomiska värden när det gäller barn kan vara upp till två gånger så höga. Eftersom få svenska ekonomiska värderingsstudier har genomförts kring miljörelaterade hälsorisker rekommenderar vi användning av ekonomiska värden från framförallt europeiska studier.

Baserat på slutsatserna i litteraturöversikterna har vi, som ett exempel, beräknat hälsovinsten av att reducera barns exponering för NO\textsubscript{2} med 1 µg/m\textsuperscript{3} i Stockholm and Umeå. En skillnad mellan dessa städer, som påverkar resultatet av beräkningen, är antalet barn som exponeras i respektive stad. Beräkningen är genomförd för två hälsoeffekter, att barn med pipande andning utvecklar sjukdomen astma samt att barn som har astma läggs in på sjukhus till följd av andningsproblem. Enligt våra beräkningar skulle hälsovinsten för samhället för denna minskning i exponering i Storstockholm vara 168 miljoner SEK per år p.g.a färre fall av sjukdomen astma samt 47 000 SEK p.g.a färre inläggningar på sjukhus (i prisnivån för år 2000). Hälsovinsten i Umeå är betydligt lägre, 8 miljoner SEK och 2000 SEK per år.

Dessa beräkningar är dock baserade på en ganska stor förändring i exponeringen. En reduktion i exponering med 1 µg/m\textsuperscript{3} NO\textsubscript{2} är ungefär den effekt som uppnåddes i Stockholms innerstad när försöket med trängselavgifter genomfördes. Detta blev följden av en minskning av trafikarbetet i innerstaden med 15%. Att åstadkomma en sådan reduktion i exponeringen även för Storstockholm eller för Umeå skulle alltså kräva åtgärder som kraftigt minskar emissionerna. Huruvida detta är samhällsekonomiskt lönsamt kräver en analys på lokal skala både av samhällets kostnader för att genomföra åtgärderna och av dess hälsovinsters. När det gäller hälsovinstersna så är en generell slutsats att dessa är större om reduktionen sker i tätbefolkade områden.


Detta är det första försök att i ekonomiska termer beräkna hälsovinsterna för barn av minskade halter av luftföroreningar som genomförts i Sverige. Slutsatsen av vår genomgång är att det fortfarande finns mycket kunskapsluckor när det gäller luftföroreningar och barns hälsa vilket får konsekvenser för möjligheten att genomföra beräkningar av detta slag. Våra beräkningar ger därför endast en indikation på hur
stora hälsovinsterna kan vara och då endast till följd av minskad förekomst av sjukdom i luftvägarna. Fortsatt forskning behövs för att validera våra resultat, för att klärlägga hälsovinsterna till följd av andra hälsoeffekter samt för att visa hur hälsovinsterna varierar mellan olika geografiska områden. Våra resultat pekar på att hälsovinsterna för barn kan vara betydande vilket vi menar berättigar till att fortsatt forskning genomförs inom detta område eftersom beslutsfattare behöver känna till om och när detta är fallet.
Introduction

Purpose and content of the study

The Swedish parliament has adopted 16 Environmental Objectives, which are to be met within a generation, to guide action towards a sustainable environment. One objective is Clean air which states that the air must be clean enough not to represent a risk to human health or to animals, plants or cultural assets. This project is part of the work undertaken by the Swedish Environment Protection Agency (Swedish EPA) to bring about the fulfilment of this goal. Much research has been undertaken regarding air pollution and health impacts in the adult population but much less is known about how pollutants influence children’s health.

To obtain information on the current state of art regarding air pollution and children’s health the Swedish EPA commissioned the Institute of Environmental Medicine at Karolinska Institutet (KI) to undertake a review of the scientific literature. This study summarised and evaluated results from recent epidemiological studies concerning effects of short- and long-term exposure to traffic-related air pollution on respiratory health and allergic sensitization in children (Naturvårdsverket, 2010). The KI study did not however quantify the possible gains in children’s health from reducing air pollution in Sweden. This is information that is necessary in order to make policy proposals that can lead to improvements in children’s health. The overriding purpose of this study is therefore to see how and to what extent the benefits from reducing these impacts can be estimated.

The approach we use in this study is to assess the benefits in economic terms. The method we apply is referred to as the dose-response method in the environmental economics literature. It rests on establishing a relationship between environmental quality changes and health changes. The final benefit estimate is the product of the change in environmental quality (i.e. air pollution), the estimated health impact and the economic value placed on the health impact. Hence, the severity of a health impact is measured by the economic value placed on it\(^1\). We will in principle follow the approach that has been developed in the EU-funded ExternE research projects\(^2\). This is an approach that has been used in a number of EU funded research projects on external cost calculations: UNITE (Unification of Marginal Cost and Transport Accounts for Transport Efficiency), HEATCO (Harmonised European Approaches for Transport Costing and the cost-benefit analysis undertaken in the CAFE (Clean Air For Europe) programme. It is also the basis for the approach for

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\(^1\) This method is not so common in health care analysis. In such applications other methods have been developed that tries to account for the fact that the quality of life differs between different health outcomes by giving different weights to different symptoms. The most common of these are QALY (quality adjusted life years) and DALY (disability adjusted life year).

\(^2\) ExternE – the External cost of Energy – is research funded by the EU on the assessment of environmental damage cost. The research stared in the beginning of 1990 and involves research teams in several countries, see http://www.externe.info/ for more information. Within this project they have developed the impact pathway approach which is the same as the dose-response method. In early publications in transport such as Small and Kazimi (1995) and Delucchi (2000) it was called the damage cost approach. In ExternE they have also developed an integrated calculation tool called EcoSense.
external cost calculation for transport presented in WHO (2008). For readers who are unfamiliar to this method, a brief description is presented in the next section in this chapter.

This type of assessment is a multidisciplinary task that requires information from different research disciplines. In this study we have therefore used the information collected in the KI study, as well as other sources, to assess what the health impacts related to children are likely to be and which of these that are currently quantifiable. These results will be presented in chapter 2. In chapter 3, an overview is given about the approaches used to value changes in risk to children’s health and suggestions are given regarding values that can be used in these types of calculations.

In chapter 4, we use two case studies to illustrate how the benefit of reducing children’s exposure to air pollution can be calculated and the data that is required for this type of calculation. Since recent studies have highlighted the importance of traffic emissions, as they are often released in close proximity to where people live, we focus on this emission source. In this chapter we also discuss the uncertainties related to these calculations. In chapter 5 some final remarks are given on how different aspects will influence the final benefit estimates and what we consider to be the most important areas for future research regarding air pollution and children’s health.

Benefit calculation – the dose-response method

The dose-response method is one of many ways that has been developed to be able to assess the benefit of environmental or health changes in economic terms. The idea behind economic benefit calculation is to make it possible for the decision maker to compare obscure non-priced benefits with direct costs or benefits that result from different policy measures. This estimation is based on economic welfare and public finance principles that the evaluation of benefits and costs should rest on individuals’ willingness to pay. The idea of using willingness to pay is because this reflects the trade-off that individuals are willing to make in order to achieve environmental or health improvements.

For air pollution this type of calculation is quite complex since air pollution 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. Therefore, in order to obtain consistent estimates for these negative impacts, efforts have been made within the EU and the USA to develop more standardized approaches for this type of assessment, see Viscusi and Gayer (2005) for a description and discussion. In EU this approach has been formalised in

\[3\] There are numerous books written on issues related to economic valuation. Overviews of this 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).
the ExternE projects. The so called Impact Pathway Approach (IPA) that has been developed in these projects is commonly used and widely accepted and therefore our calculations in this study will be based on the same principles and reasoning (for detailed descriptions see Friedrich and Bickel, 2001; Nerhagen et al, 2005; Nerhagen et al., 2009; WHO, 2008).

This is a bottom-up approach where the calculated benefit (or cost in the case of increased emissions) 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 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).

Still, there are some aspects in the chain that will imply non-linearity that needs to be considered 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) depends 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).

A more formal description of the approach we have used is given by the following equation, (which is a modification of an equation in Ostro and Chestnut, 1998). It describes the yearly benefit (cost) due to a reduction (increase) in concentration $C$ from a change in emissions of a certain pollutant from a specific source:

Health benefit = $\Delta$yearly exposure $\cdot$ effect $\cdot$ value

$$= (\Delta C_{a;i} \cdot \text{POP}) \cdot (B_{a;j} \cdot P_{i;j}) \cdot V_{j}$$

where

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

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

$B_{a;j} = \text{baseline annual health impact rate in population for health impact } j \ (\text{number of cases per inhabitant})$

In their study they only included the first two components in the expression (1). This is commonly referred to as a health impact assessment (HIA).
\[ P_{i;j} = \text{exposure-response function, i.e. effect on health impact } j \text{ per } \mu g/m^3 \text{ of pollutant } i \text{ (relative risk or odds ratio)} \]
\[ V_j = \text{economic value of health impact } j. \]

Relative risk (RR) usually means the risk ratio (a ratio of probabilities), but sometimes the odds ratio (OR) is used instead, where the disease odds is the probability for disease P divided by (1-P).

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 benefit calculated for each pollutant and each health endpoint can then be added up to arrive at the total yearly health benefit for the change in emissions from each source.

What this expression reveals is that this calculation requires data from several different sources. We want to stress that this kind of benefit calculation is based on a number of assumptions and, as discussed in Viscusi and Gayer (2005), extrapolating results beyond the actual underlying empirical data, and making incorrect assumptions, will have a large impact on the final benefit estimate. Therefore input values needs to be carefully chosen and there is also a need to undertake a sensitivity analysis.

One issue here is the transfer of estimates from one context to another. Since studies on health impact assessments as well as economic valuation are quite demanding to undertake it is often not possible to derive situation specific estimates. Therefore it is assumed that the exposure-response functions as well as the economic values that are derived in one context are also relevant in another. In economics this is referred to as a benefit transfer. One way to test the relevance of such transfers is by comparing estimates between studies trying to account for differences in underlying circumstances (for example health status or income).

Another issue is that, according to economic theory, only minor changes from the current state can be evaluated. Hence, economists usually use so called marginal cost estimates as the basis for external cost calculations. The reason is that both economic values and health impacts 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 and this is also the case for the economic valuation studies.

In this study the exposure-response functions (the \( P_{i;j} \)) and the economic values (the \( V_j \)) that are relevant to include to assess benefits to children’s health are discussed.

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5 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.

6 Sensitivity analysis is common in economic analysis of different kinds. It is a way of investigating the robustness of the results. For a discussion of the use on sensitivity analysis in these kind of calculations see Bickel and Friedrich (2005 page 6).
in more detail in chapter 2 and 3. Another important input is $B_{nj}$, the baseline health impact. This is yearly prevalence of a health impact in a certain population. In Sweden such statistics is collected by The National Board of Health and Welfare. For the purpose of this study information is needed on the child population in a certain place and the amount of children affected by a certain symptom. Details on what estimates we have used in our case studies and how they were obtained will be given in the description in chapter 4.

In chapter 4 we will also give more information on the modelling behind the first part of the expression, the modelling of the change in yearly exposure due to a policy change. How this is done will have an important influence on the final benefit estimate. For our case study we need to consider that since we focus on traffic, that mainly influence $C$ on a local scale, the benefit estimate is likely to vary depending on the population density close to traffic. Hence, the population data should account for how the child population is located in relation to the emission source.
Air pollution and Children’s Health

The main sources of information regarding the effect of air pollutants on human health are epidemiological studies. In these studies the occurrence of different health outcomes in individuals from a specific population is related to air pollution concentrations. There are mainly two types of studies, using either spatial or temporal differences in concentration levels. In the first type, differences in health risks are evaluated in relation to spatial differences in air pollution concentrations, e.g. differences between neighbourhoods. This is typically the type of design that is mostly used for studying diseases that develop after a longer period of exposure. Incident (new) cases can be identified during a follow-up period, but also cross-sectional (prevalence) and retrospective data are used in some studies. In the latter type, health outcomes and exposures are compared on a temporal basis, typically on a day-by-day basis. This is a type of design that is used for health effects that appear after only a short period of exposure. Both study types can be used for both incident and recurrent diseases, i.e. both for the risk for a previously healthy person of getting a disease and for diseased persons of getting a relapse or an exacerbation. For long-term effects of air pollution, it is easier to assume causality in prospective settings where bias is less likely. For this reason more weight is often given to such studies (Bråbäck and Forsberg, 2009).

Since many of these studies are based on the general population, the results may be directly applied in estimating the health impact of a certain air pollution concentration. Since the air pollution mixture differs somewhat from place to place it is not however certain that the results from one population a priori can be transferred to another.

According to the review performed by KI (Naturvårdsverket, 2010) the effect of air pollution on children’s health can be divided into two main categories. First of all air pollution impairs normal development of the lung function during childhood and youth, and increases the risk of developing certain diseases. Secondly, air pollution worsens the health conditions for children that have already developed a disease, and increases the risk of healthy children to display different symptoms. In this chapter we discuss these two categories, and the endpoints related to each category, in turn and in the final section we make a summary of the important health outcomes and which of these that is currently possible to quantify.

Impacts due to long-term exposure

Lung function development

There is strong evidence that adverse long-term effects of air pollution occur on lung function growth in children, resulting in deficits of lung function at the end of adolescence. No study has, however, followed up adolescents until they reached the plateau phase of early adulthood. It therefore is not known whether growth deficits will be compensated by a prolonged growth phase, or whether these sub-
jects will enter the lung-function decline phase of later adulthood with a reduced lung function. Because of the diversity of the reviewed studies, formal quantitative comparisons of the findings are difficult. The KI review, as well as another recently published review (Götschi et al., 2008), refrained from summarising the findings in one exposure response functions.

**Sensitisation to allergens**

Hypersensitivity mediated by immunological mechanisms is called allergy. An objective sign of such hypersensitivity are increased levels of circulating antibodies to allergens. It should however be noted that not all with circulating antibodies experience symptoms, and that not all hypersensitivity reactions are allergic.

Exposure to NO$_2$ (or NO$_x$) is in some studies related to sensitisation to common allergens in children. In the KI review, the pooled estimate shows a 7% (95% confidence interval 0.8-14%) increased risk for sensitisation to outdoor allergens when living in areas with 10 µg/m$^3$ higher NO$_2$ levels. Just as for the association between environmental pollution and respiratory disease, a gene-environmental interaction is suggested; with the consequence that certain individuals may be especially prone to develop sensitisation to common allergens if exposed to traffic air pollution. Identification of individuals at higher risk in the population is however not yet feasible. The KI review concluded that the association between air pollution and sensitisation needs to be further studied before it can be used as a basis of preventive action.

**Development of air-way disease (including asthma)**

Respiratory disorders in children are of great concern, accounting for a substantial part of consultations with doctors for acute illness. Asthma is a major respiratory disease and the most common chronic disease in children, and may also be included in the wider definition allergy-related disorders, which besides asthma includes rhinitis, eczema and food allergy. In these disorders, symptoms are usually triggered by hypersensitivity to substances and environmental factors mostly tolerated in the normal population.

There is evidence that exposure to traffic-related air pollution including gases (CO, O$_3$, NO$_x$) and particles (PM$_{2.5}$, PM$_{10}$) early in life contributes to the induction of respiratory airway disease like asthma and allergic rhinitis during childhood, especially in children living within short distances of major roads. Furthermore, it is likely that traffic-related air pollution could interact with certain genetic factors making some individuals further susceptible to environmental agents. The KI review (Naturvårdsverket, 2010) as the review of cohort studies by Bråbäck and Forsberg (2009), concluded that due to the variation in methods and definitions among the studies on respiratory symptoms and disease reviewed, no combined analysis could be done on the long-term effects.
For the purpose of this paper, we have updated the literature search and identified combinations of design (follow-up, cross-sectional), spatial exposure assessment (hi-resolution, low-resolution), age group (0-2, 2-5, 5-18 years) and outcome (doctor’s diagnosis, wheeze) for which there are at least three reports with a meta-OR statistically different from unity. We found only two such combinations, both for the outcome wheeze in the 5-18 age group and with hi-resolution NO₂ as the exposure indicator. For follow-up studies there were four studies (Shima, 2000; Gauderman, 2005; Gehring, 2010; Ofteidal, 2007) and we estimated a meta-OR\(^7\) of 1.14 (95% confidence interval 1.05-1.25) for wheeze for an exposure difference of 10 µg/m\(^3\) NO₂. For cross-sectional studies there were five studies (Krämer, 2000; Nicolai, 2003; Janssen, 2003; Zhao, 2008; Sahsuvaroglu, 2009) and we estimated a meta-OR of 1.38 (95% confidence interval 1.16-1.64) for wheeze for the same exposure difference.

Reproductive disorders

A recent review reported how since the mid-1990s, the number of studies linking air pollutants to adverse effects such as low birth weight, small for gestational age, and preterm birth has grown steadily (Ritz and Wilhelm, 2008). Several pollutants have been associated with birth outcomes. Studies in the USA most consistently report associations for carbon monoxide and particles, with motor vehicle exhaust as possible causative agent. However, there still not easy to suggest the exposure-response assumptions for NO₂ or NOₓ as traffic pollution indicators to be used in health impact assessments.

Cancer

The risk of childhood cancer after increased exposure to air pollutants was evaluated by WHO (WHO, 2005). The conclusion was that “The weight of the epidemiological evidence to date indicates that no increased risk of childhood cancer is associated with traffic-related air pollution at home.” In view of the limited number of studies (15) it was cautioned: “Nevertheless, the low number of studies, the methodological limitations of epidemiological research and the absence of full consistency across the study results preclude a firm conclusion of no effect.”

Impacts of short-term exposure

These are impacts that occur shortly after exposure episodes. Studies focus on persons with disease such as a panel of asthmatics, or include all cases in the population such as hospitalizations for heart or lung problems. In the studies of short-term effects, urban background air pollution exposure is generally monitored on a

\(^7\) A weighted summary of risk (Odds Ratio) based on the result of several studies. The summary was performed in the logarithmic space using 1/variance as the weight for each result. The variance was estimated from the published confidence interval in each study.

\(^8\) We searched for more recent publications in PubMed by searching for studies that had referred to these 15 publications, and found another 11 studies. In seven out of these the authors report associations with air pollution in the abstract. This suggests that an update of the WHO review may be warrant-ed.
daily basis. In the panel data or time-series analysis, meteorological variables, season, time-trends and day of the week is usually controlled for. Some papers report results from single pollutant models only, some give results adjusted for other pollutants.

The KI review (Naturvårdsverket, 2010) presents two types of epidemiological studies, panel studies that monitor at the individual level the variation in respiratory symptoms in relation to air pollutants; and, time-series studies that examine the daily number of children requiring urgent medical attention because of asthma (studies of emergency visits and hospital admissions). It should be noted that overall there are few studies on acute effects only based on previously healthy children.

Panel studies of children
The KI review includes 17 panel studies, of which 7 were conducted in Europe, 9 in North America and one in Asia (Thailand). Some heterogeneity in the results of panel studies is expected, given differences in selection criteria, duration of follow-up and outcomes under investigation. Since panel studies are rather different, it is difficult to perform a meta-analysis. According to the review, the effects of air pollutants on asthma exacerbation do not appear fully consistent, which may be related to the complexity of the disease itself and the subsequent difficulties in estimating the impact of air pollution. Nevertheless, most of the reviewed studies found evidence of adverse effects of short-term exposure to air pollutants and respiratory symptoms in children. Thus, significant positive associations were shown for asthma symptoms in relation to exposure to PM$_{10}$, NO$_2$, SO$_2$ and ozone. Certain threshold levels (or safe levels below which there is no risk) could not be identified from the published studies.

Many of the panel studies recruited symptomatic children without detailed information on asthma status, for example the large European multicentre study PEACE$^9$ (Roemer, 2000). In such studies the proportion of children with asthma may be as low as 50% or even less.

A panel study of 138 children in Los Angeles with an asthma diagnosis (Ostro et al., 2001) has been used for impact assessments and is also included in the US software BenMAP (US EPA, 2008a)$^{10}$. For a 17 µg/m$^3$ increase in 24-h mean PM$_{10}$ (lagged 3 days) the odds ratio was 1.14 (95% confidence interval 1.04-1.24) for shortness of breath, 1.04 (95% confidence interval 0.98-1.10) for wheeze and 1.10 (95% confidence interval 1.04-1.16) for cough. The effect of NO$_2$ as daily 1-h maximum, was significant for wheeze and borderline significant for shortness of breath.

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$^9$ A EU financed multicentre study on acute pollution effects on asthmatic children.

$^{10}$ BenMAP (Environmental Benefits Mapping and Analysis Program) is a tool developed for the calculation of environmental health benefits by the US Environmental Protection Agency. For detailed information see http://www.epa.gov/air/benmap. It is based on an approach similar to that is used in the ExternE projects.
Register studies of ERVs and admissions
The KI review considered papers on emergency room visits and hospital admissions for asthma, upper and lower respiratory diseases in children as main outcomes of interest. Several studies included more specific symptoms (i.e., allergic rhinitis and wheezing). Of 12 reviewed studies two were carried out in Western Europe, two in North America, one in Australia, three in South America, and four in Asia. Overall, the reviewed studies found consistent associations between short-term exposure to air pollution and hospital admissions or emergency department visits for asthma or respiratory conditions in children. The reported associations with NO₂, PM₁₀ and CO indicate that traffic-related pollutants are the main pollutants associated with urgent medical help needs for respiratory conditions. According to the review, effects in the time-series studies appeared stronger for several day means or delayed day lags of exposure, although some studies have also shown significant increase in emergency room visits for the same day exposure. The combined estimates of the association with hospital admissions for asthma were 1.3% and 2% increase in such admissions, for a 10 µg/m³ increase of NO₂ (95% confidence interval 0.96-1.7, based on four studies), and PM₁₀ (95% confidence interval 1.7-2.3; based on five studies), respectively. No threshold levels below which there is no risk were suggested in these studies.

For the case studies in this report (see chapter 4) we choose as exposure-response relation for the association between the short-term level of traffic related pollution and respiratory effects in children, results from a high quality study of respiratory hospital admissions in Rome (Fusco et al., 2001) assumed to be representative for Europe. In this study the associations between daily mean levels of NO₂, CO, SO₂, particles and ozone and hospital admissions for respiratory conditions in the metropolitan area of Rome were analysed for a three year long period. Total respiratory admissions as admissions in children were significantly associated with same-day level of NO₂. The relative risk associated with a 10 µg/m³ increase in NO₂ was a 1.77% increase in respiratory admissions in the age group 0-14 years (95% confidence interval 0.27-3.30).

School absence
School absences due to illness have only been investigated in a few studies. In Seoul, Korea, several different lags were studied, but the same day mean concentration produced the best model fit (Park et al., 2002). An interquartile range increase in PM₁₀, 41µg/m³, resulted in a relative risk of 1.06 (95% confidence interval 1.04-1.09).

Illness-related school absences in the Southern California Children's Health Study were studied in relation to both short-term and long-term exposure (Rondeau et al., 2005). The analysis of short-term effects included a distributed lag model, where the cumulative effect of PM₁₀ over 30 days was 7.9% per 10 µg/m³.
Summary and conclusion

In this chapter we have discussed the knowledge regarding the health endpoints that are likely to be of most relevance for children’s health. However, the current level of knowledge about the relationship between exposure and outcome varies for the different endpoints. Hence, quantifications of health impacts are not always possible. In Table 1 and Table 2 we give an overview of the current level of knowledge regarding the health endpoints discussed in this chapter and the information needed for health quantification. If information is lacking in a cell in the table it means that no information is available. The rows shaded grey are those that we will use in our benefit calculation in chapter four.

<table>
<thead>
<tr>
<th>Health impacts of relevance for children</th>
<th>Inputs into quantification of health impacts</th>
<th>Pollutant</th>
<th>Concentrations</th>
<th>Exposure</th>
<th>Baseline</th>
<th>Exposure-Response</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lung function development</td>
<td>Emissions from traffic, NO₂ as indicator</td>
<td>Can be calculated based on measurement data for NO₂</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Not quantified</td>
</tr>
<tr>
<td>Sensitisation to allergens</td>
<td>Emissions from traffic, NO₂ as indicator</td>
<td>Can be calculated based on measurement data for NO₂</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Not quantified</td>
</tr>
<tr>
<td>Development of air-way disease</td>
<td>Emissions from traffic, NO₂ as indicator</td>
<td>Can be calculated based on measurement data for NO₂</td>
<td>Concentrations known for some cities but knowledge of influence from specific sources not available</td>
<td>Detailed data not available because subject for ethical judgement.</td>
<td>Exists and based on review of X published results.</td>
<td></td>
</tr>
<tr>
<td>Reproductive disorders</td>
<td>Emissions from traffic, CO, particles</td>
<td>Can be calculated based on measurement data for NO₂</td>
<td>Concentrations known for some cities but knowledge of influence from specific sources not available</td>
<td>From registers</td>
<td>Inconclusive</td>
<td></td>
</tr>
<tr>
<td>Cancer</td>
<td>Emissions from traffic, benzene,</td>
<td>Can be calculated based on measurement data for NO₂</td>
<td>Concentrations known for some cities but knowledge of influence from specific sources not available</td>
<td>From registers</td>
<td>Inconclusive</td>
<td></td>
</tr>
</tbody>
</table>
### Table 2  Overview of current knowledge of short-term (acute) health impacts

<table>
<thead>
<tr>
<th>Health impacts of relevance for children</th>
<th>Inputs into quantification of health impacts</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Pollutant</td>
</tr>
<tr>
<td>Children with asthma diagnosis</td>
<td>Emissions from traffic. NO₂ as indicator</td>
</tr>
<tr>
<td>Hospital admission</td>
<td>Emissions from traffic. NO₂ as indicator</td>
</tr>
<tr>
<td>Emergency care</td>
<td>Emissions from traffic. NO₂ as indicator</td>
</tr>
<tr>
<td>General practitioner visit</td>
<td>Emissions from traffic. NO₂ as indicator</td>
</tr>
<tr>
<td>Medication</td>
<td>Emissions from traffic. NO₂ as indicator</td>
</tr>
<tr>
<td>Restricted activity days</td>
<td>PM</td>
</tr>
</tbody>
</table>
Economic valuation of health risk reductions\textsuperscript{11}

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 sick.

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’ willingness to pay for risk reductions. These methods rest on the assumption that individuals’ willingness to pay is an approximation of a change in utility that the risk reduction entails. A brief formal treatment of the difference between the production functions approaches described above and the willingness to pay approach is given in Viscusi and Gayer (2005).

The first attempts to obtain willingness to pay estimates relied on the use of market data using so called 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 willingness to pay (open-ended format) or accept or reject a certain bid (closed-ended format) for a certain improvement. Another more recent format is to ask the individuals to respond to several alternatives in a row. This is often called a choice experiment which has been developed in valuation studies in transport economics and marketing. The main objection raised against stated preference meth-

\textsuperscript{11} This chapter is based on the overviews given in Viscusi och Gayer (2005) and OECD (2006). We will not state specific references in the text except for the case where we refer to a question or statement made by one specific author. For more detailed discussion on issues related to valuation, we refer to the original texts.
ods is that it is difficult to validate that answers to these questions represents actual choice behavior (a problem often referred to as hypothetical bias).

There are a multitude of aspects that has been discussed in relation to economic valuation, the methods used and the difference in the estimates produced by the different methods. However, the purpose of this study is to give an overview of how changes in children’s health risks can be valued. We have therefore chosen to focus on what the important differences are between economic valuations of the risk for adults compared to risks for children. The reason for this choice is that much has been done regarding the valuation of health risk reductions to adults and for policy purposes one way forward is to use accepted values for adults as the basis for deriving economic values for children.

This is a type of benefit transfer. As discussed by Viscusi and Gayer (2005) this type of extrapolations to other groups or contexts is based on strong assumptions. What we attempt to do in this overview is to sort out what assumptions that are reasonable to make according to findings in the literature. We start with an introduction on what aspects that has been raised in the literature regarding willingness to pay estimation regarding children’s health and we then discuss the valuation of morbidity and mortality, and the values that can be used in these contexts, in turn. We treat mortality after morbidity since this endpoint according to current epidemiological knowledge is of less importance in the case of environmentally related health risks to children. We will then briefly discuss other ways to weigh and compare different health outcomes, and how they relate to the willingness to pay measure, before ending the chapter with a summary of the main findings and their implications for policy and issues to be considered in future research.

**Children’s willingness to pay – methodological issues**

As stated in the introduction to this chapter, according to economic theory a correct measure of changes in an individual’s welfare (utility) is willingness to pay. An individual’s willingness to pay estimate can be positive or negative depending on the context. If a change is expected to improve the well-being of the individual (feeling more secure for example) the willingness should be positive and vice versa. Individuals will experience a certain change in different ways and hence we also expect the willingness to pay to differ between individuals according to socio-demographic differences or due to differences in attitude.

Economists use willingness to pay since it is a measure that allows comparison between different outcomes. The estimate is a reflection of the weight that an individual places on a certain change in their well-being. This weight is related to other choices that individual’s make that have an impact on their well-being. An additional reason for using the willingness to pay measure is that it is a measurement in
monetary terms that allows for comparison between benefits and costs for a certain policy change.

This line of reasoning is not directly applicable to the valuation of changes in children’s well-being. Children, especially in the younger ages, have not developed the cognitive capabilities to assess how a certain risk change will influence their own utility. Moreover, they are not in control over their financial situation and cannot make decisions on what money to spend on risk reductions. This however does not imply a willingness to pay that is zero. Both society at large and parents make arrangements to protect children from being harmed by accidents or by illness. Hence, willingness to pay to protect children could be based on such choices. The unresolved issue though is how well such estimates reflect the trade-offs that the children would make themselves.

Relatively few empirical studies have investigated the willingness to pay to reduce health risks to children and most of them are based on the choices made by parents. Most results however indicate that the willingness to pay is higher for children than for adults and this is irrespective of if the study is based on parental preferences or on other persons’ preferences. One explanation for this seems to be a general sense of responsibility to protect vulnerable individuals. It is acknowledged that children are not in the position to make their own choices regarding exposure to risk. An additional explanation is that the individual seem to account for the difference in outcome between adults and children where children have more to lose because they are expected to live longer. Yet another reason for parents and other persons to place a higher value on risk reductions for children is that this will be to their own benefit in the future. The reason is that it is expected that improvements for children will influence future economic and social development. It is however difficult to state exactly how much higher the willingness to pay should be in the case of children since it is difficult to disentangle altruistic and other components of the preferences.

When assessing what the expected difference between the willingness to pay for children and adults should be, it is also important to consider that these groups face different risks. According to Hoffman et al. (2006), a particular problem when obtaining values for children is that the uncertainties regarding the expected impact of a risk reduction is larger. This has to do with the difference in life expectancy. On the one hand this could increase the difference in willingness to pay since children will suffer for a longer time period but on the other hand there is also the possibility of improvements that will change the baseline risk in the future. This also

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12 An issue that has been raised in relation to these methods is what the stated willingness to pay encompasses and how it is related to the impact that is evaluated. First of all there is the question of if a certain answer is simply a reflection of “warm glow”, that is willingness to pay (make a donation) to a good cause irrespective of the expected impact. This is also referred to as impure altruism and this is not an estimate that should be included when calculating the willingness to pay. The question is however more complicated when we consider what is called altruism (see discussion in Hoffman et al., 2006 and in Andersson and Lindberg, 2009).
has to do with changes in children’s vulnerability over time which could be caused by overall changes in exposure to hazards but also to changes in medical technology. Improvements in neonatal care for example have reduced infant mortality but little is known about the susceptibility of early-borne. Hence, a major problem is the greater uncertainty in forecasting children’s future health states.

This problem is in the literature referred to as the influence of latency. For some health endpoints there will be a time lag between exposure to a pollutant and the onset of illness or death. Hence, the benefit of exposure reductions today will only materialise some time in the future. Since it is found that individuals prefer consumption today instead of in an uncertain future, economists use discounting to account for the time dimension.

### Valuation of morbidity risk reductions

This is the health outcome of greatest importance regarding environmental health risks to children. The aspects to include in a benefit calculation 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 willingness to pay estimates. We will discuss each of these values, and what should be accounted for when valuing changes for children, in turn. We will then give some examples of estimates from some recent studies that have assessed these values for adults. This however is not an area that is as researched as the valuation of mortality risks so there are fewer studies to obtain values from.

#### Cost of illness (COI)

The first component when calculating the cost of illness is the production loss. When valuing morbidity for adults this cost is based on income lost from not working. To this could be added the productivity loss due to hired less skilled or trained workers taking part in production. In the case of children’s illness, this is a relevant measure to the extent that parents or other employed relatives stay at home to care for the sick child. However, this cost is likely to accrue mainly for younger children since older children often can care for themselves if the symptoms are not too severe. There is also the question of what estimate to use in this context. If women are more likely to take care of sick children then the production loss estimate should be based on estimates for this working group.

There is however an additional component that should be added in the case of children, at least for those attending school. Being ill may have an impact on future income since it may influence educational choices and career paths. This component however is difficult to determine and is therefore likely to be left out of production loss calculations for children. Hence, what we can conclude from this is
that using production loss calculations for adults may be a downward biased estimate in the case of children.

The second component when calculating cost of illness is direct costs related to medical expenses and health care services. For this cost component there is no difference between the calculation for adults and children so these estimates can quite readily be collected using register data.

Welfare loss

This is the component that was discussed at length in the section on methodological issues. What is to be measured here is the reduction in quality of life from being ill. This is the result of not being able to take part in desired activities but also the pain and suffering related to being ill. Since children do not have the capabilities to make a judgement of these costs, it is common to base these estimates on studies of parents’ willingness to pay (WTP) to avoid illness. However, as discussed previously, the question is how well such estimates reflect the actual disutility of the child. To greater or lesser extent restrictions related to a child being ill also place restrictions on the activities that the parents can undertake. Hence, the difficulty is to separate out the various components that will influence the parents willingness to pay estimate.

Results from recent valuation studies on adults and morbidity

Since valuation of morbidity has been less researched compared to mortality, Ready et al. (2004) undertook a valuation study in several European countries (Eftec, 2004). One purpose with this study was to evaluate the possibility of benefit transfers of such estimates between countries. However, this study is also the basis for the economic values recommended in the ExternE Methodology update (Bickel and Friedrich, 2005). In their recommendation they have summed up the willingness to pay estimates from Ready et al. (2004) with cost of illness estimates. The components and the final estimates are presented in Table 3.

<table>
<thead>
<tr>
<th>Health endpoint</th>
<th>COI</th>
<th>WTP</th>
<th>Recommended central unit value in ExternE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Respiratory Hospital admission*</td>
<td>1600</td>
<td>400</td>
<td>2000/admission</td>
</tr>
<tr>
<td>General practitioner visit</td>
<td>42</td>
<td>15</td>
<td>53/consultation</td>
</tr>
<tr>
<td>Restricted activity (RAD)</td>
<td>84</td>
<td>46</td>
<td>130/day</td>
</tr>
<tr>
<td>Asthma attack</td>
<td>450</td>
<td>220</td>
<td>670/visit</td>
</tr>
<tr>
<td>Respiratory medication use</td>
<td>1</td>
<td>-</td>
<td>1/day</td>
</tr>
<tr>
<td>Cough</td>
<td>-</td>
<td>38</td>
<td>38/day</td>
</tr>
<tr>
<td>Symptom day</td>
<td>-</td>
<td>38</td>
<td>38/day</td>
</tr>
</tbody>
</table>

* Based on an assumption of 3 days in hospital and 5 days recovery at home

As can be seen from these estimates, if the willingness to pay estimate are 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) plus 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.

Since both willingness to pay and cost of illness estimates may differ between countries we have searched for Swedish studies to be able to assess the relevance of the estimates presented in Table 3 in a Swedish context. Regarding willingness to pay we only found one study by Samakovlis och Svensson (2004). This study is relevant since it used the same survey format as in Ready et al. (2004). The results from these studies are presented in Table 4. The value we have included from Ready et al. (2004) is the estimates for Norway since this is a country similar to Sweden in structure and culture. 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).

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Respiratory Hospital admission</td>
<td>482</td>
<td>205</td>
</tr>
<tr>
<td>Restricted activity day</td>
<td>190</td>
<td>64</td>
</tr>
<tr>
<td>Asthma attack</td>
<td>382</td>
<td>-</td>
</tr>
<tr>
<td>Cough</td>
<td>58</td>
<td>-</td>
</tr>
<tr>
<td>Symptom day</td>
<td>50</td>
<td>14</td>
</tr>
</tbody>
</table>

Unfortunately, Samakovlis and Svensson (2004) do not discuss the reason for this difference so which of these estimates that are the most reliable in a Swedish context cannot be determined. There are also only a few health endpoints that are included in the Swedish study. Hence, from this we conclude that there is a lack of reliable Swedish willingness-to-pay estimates.

We have also searched for cost-of-illness data for Sweden and found a couple of studies that provide such estimates\(^{13}\). The previously mentioned study by Samakovlis och Svensson (2004) used estimates for the cost of illness 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). This is somewhat lower than the comparable estimate 84 Euro used in ExternE (the COI for a restricted activity day, see Table 3). 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 ser-

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\(^{13}\) 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 illness. The health cost of air pollution in Sweden has been estimated in a recent study (Vägverket, 2009). Unfortunately, the calculation in this report is based on DALY estimates and only total costs are presented. Total cost estimates are also presented in Bolin and Lindgren (2004) and Olofsson (2008).
vices in Sweden which in turn was based on the cost for medicine, general practitioner visits and hospital admission. However, we have not been able to make a comparison for this estimate with those used in ExternE since their estimate is per symptom day instead of per day.

A more recent study by Vredin-Johansson and Forslund (2009) uses the same kind of underlying data as in Huthala and Samakovlis (2003) and provide an estimate of 6117 SEK/day for each case of respiratory illness (in 2007 price level). Unfortunately, neither in this case is it straightforward to compare it with those proposed by ExternE. This time it is because we do not know what kind of medical treatment the estimate is based on, is it general practitioner visit, hospital admission, both or something else? 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, this comparison suggests that the estimate for medical treatment presented by Vredin-Johansson and Forslund (2009) is a high estimate more likely to represent the cost for hospital treatment.

However, all estimates stated above relate to persons already suffering from symptoms or having a disease. Exposure to air pollution may 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 the ExternE methodology update (Bickel and Friedrich, 2005) they discuss this issue and use an estimate from HSE\(^{14}\). HSE added up the cases specific costs relating to asthma in their calculations in order to arrive at an estimate for a new case. The estimate for this outcome was about 60 000 euro and it included: loss of income, medical treatments and the value of pain and suffering. In the US model BenMAP, they use an estimate for avoiding chronic asthma of 38 947 USD/case, price year 2000 (US EPA, 2008b). This estimate is based on willingness to pay studies. This is approximately equal to 31 000 euro.

### Valuation of mortality risk reductions

Reduction in 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 willingness to pay for his/her own marginal (statistically determined) change in the risk of dying. 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, that could be particularly relevant for the valuation of improvements in children’s health, is if the value should vary with respect to age.

\(^{14}\) Health and Safety Executive, a national regulatory body responsible for promoting better safety and health in workplaces in England More information on the purpose of the calculation and how it was done can be found on [http://www.hse.gov.uk/ria/chemical/asthma.htm](http://www.hse.gov.uk/ria/chemical/asthma.htm).
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.

Our conclusion however is that the valuation of mortality is not really of importance in our context. It is because premature mortality among children due to the air pollution levels found in Sweden is so far without empirical evidence. Still, we will briefly describe the methods used and the most important issues raised regarding valuation of this health endpoint since new findings may change this conclusion. As discussed in chapter 2 there are three health impacts, if empirical studies can verify a connection with air pollution, that entail using values for premature mortality for children. These are if reduced lung function in young ages due to air pollution is a chronic condition that can be related to shorter length of life, if air pollution is one of the causes of infant mortality and finally, if air pollution contributes to the development of cancer.

Similar to the development path of the valuation of morbidity, 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, this is only part of the total value. To this an estimate of the welfare loss should be added. Contrary to the total value for morbidity, it is the welfare loss that makes up the largest part of the total value for preventing premature mortality.

An example of aspects that are important in calculations of the total value is found in the VSL used in the transport sector. The value in this context is composed of two parts, the welfare estimate equal to 16,2 million and the so called material costs (hospitalization, property losses, administration and production loss) which amounts to 1,2 million. The major part of these material costs are the production loss equal to 0,9 million. These are the estimates for 2001. Based on more recent research evidence a higher welfare estimate of 21 million is now used (SIKA, 2008).

For both of the value components, the question of children having longer life expectancy is relevant. When calculating the production loss it is quite clear that this value will be higher if a person dies at young age provided that we can expect similar income levels in the future. It is however more difficult to make firm conclusions regarding the value of the welfare loss. This is partly due to the problem that we cannot obtain willingness to pay estimates from the children themselves. The overview of the literature done in OECD (2006) however suggests that VSL estimates for children should be higher than those used for adults.

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 value of statistical life (VSL) to obtain an estimate of the value of a life year (VOLY) using annuity calculations (Friedrich and Bickel, 2001):
\[ VSL = \sum_{t=0}^{T} \frac{VOLY_{\text{acute}}}{(1 + r)^t} \]

where
- \( VSL \): the value of a statistical life
- \( VOLY_{\text{acute}} \): the value of a life year lost
- \( t \): year
- \( T \): average expected length of life
- \( r \): discount rate.

This 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 willingness to pay first increases and then decreases with age. Since the debate on the use of \( VSL \) or \( VOLY \) is ongoing, in the ExternE methodology update (Bickel and Friedrich, 2005) they suggest values for both. For \( VSL \) they arrive at an estimate of 1 million euro and for \( VOLY \) 75 000 euro. Both these estimates refer to premature deaths occurring at the time of exposure (acute mortality). For deaths at a later point in time resulting from current exposure (chronic mortality), the \( VOLY \) estimate they use is 50 000 euro.

Latency is an additional complication, also in the case of mortality valuation. 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. There are several reasons for using 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 et al., 1994).

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 an life year expected to be lost in the future (Friedrich and Bickel, 2001):

\[ VOLY_{\text{chronic}} = \frac{VOLY_{\text{acute}}}{(1 + r)^t} \]

där
- \( VOLY_{\text{chronic}} \): the value of a discounted life year
- \( VOLY_{\text{acute}} \): the value of a life year
- \( t \): latency period, the time between exposure and death
- \( r \): the discount rate.

15 Discounting is supported by empirical research but the size of the discount rate to be used in different contexts is a matter of discussion.
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.

Since the value to be used for premature mortality for young versus old are important issues, 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 New-Ext project (Alberini et al., 2004), a study commissioned by DEFRA\textsuperscript{16} in England (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 if people values 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) and Andersson and Treich (2008).

Other methods used to weigh and compare different health outcomes

There are also other methods that have been developed to weight different health outcomes. Such measurements are more common in evaluations in the health sector and can be used for example in a cost-efficiency analysis. These methods are based on non-economic preference scales. There are several different methodologies but we will describe the two most common, QALY and DALY. We include a description of them in this report since they are often put forth as an alternative to benefit estimation using economic valuation.

QALY is an index that assigns numeric values to various health states so that morbidity effects can be combined with mortality effects to arrive at an aggregate measure of health outcomes. The extreme points on the scale are death (=0) and perfect health (=1). The QALY value for a health state is found by multiplying the duration of a health state by a score reflecting the quality of a health state. This quality estimate is derived in different ways, but often using the ex ante judgement of people in good health and asking questions regarding people in general, not the person himself. When using QALY’s the value of a life year is treated equally for all individuals and saving one year for ten persons is given the same value as saving ten years for one person. Income will not influence the values placed on a health outcome for an individual.

\textsuperscript{16} DEFRA = Department for Environment Food and Rural Affairs
Empirical evidence show that QALY values obtained from parents and children differ. It is also found that children older than 8 years are able to answer the kind of ratings underlying the quality estimates. Hence, according to research older children’s preferences should be sought when estimating these values.

DALY summarises the number of healthy years that are lost during a year in a specific population due to illness and death from a certain cause. It is similar to QALY since it is expressed on a severity scale ranging from 0 to 1 and that the rating depends on health states and life years. The quality index however is in this case obtained from medical expertise. Moreover, this measure incorporates an age-weighting factor but not one that gives a higher weight to young children. Instead, the weighing factor gives higher weight to persons in young adulthood and the middle ages representing the judgement that people in these years contribute more to society.

How these approaches relate to the valuation approach using willingness to pay has been discussed by different authors in different contexts, especially in the US. A thorough description of the difference between impact evaluation using willingness to pay or QALY’s is given by Hammit (2006). What aspects that are important to consider in such comparisons is also presented in a special issue of Environmental and Resource Economics in 2006.

The most important conclusion from such comparisons is that willingness to pay is the only method that is based on individuals’ preferences and the only method that can truly be said to reflect what is beneficial from society’s point of view (Krupnick, 2004; Dickie och List, 2006; Hammit, 2006). Moreover, these measures also rely on measuring the utility, or quality, difference between different health endpoints which in turn implies a need for understanding and interpreting individuals’ preferences. Hence, although based on somewhat different assumptions, these methods face similar problems in measurement as economic valuation studies based on willingness to pay.

**Summary of evidence and usable values**

In this overview on the economic valuation of environmental health risk reductions to children we have to a great extent relied on results from a report produced as a result of a workshop organized by OECD. We have relied on this report since it is our opinion that it gives a good overview of the current state of art. It refers to the same authors as we found references to when making a literature search and it raises issues that have also been discussed in the ExternE Methodology report (Bickel and Friedrich, 2005).

The general conclusion from OECD (2006) is that the benefit to children from improved environmental conditions is underestimated. Hence, using economic values for risk reductions derived from the general population also in the case of
children will give conservative (i.e. low) estimates. In Table 5 and 6 we have summarised the findings regarding usable benefit estimates for the health endpoints that were summarised in Table 1 and 2. The rows in grey are the estimates that will be used in the benefit calculation in chapter 4. As discussed in the text, these are estimates for adults. In lack of further evidence on the economic valuation of children’s health risks, the question to be considered in policy is if the values that are presented in here should be adjusted and if so, by how much.

The major concern in this respect is most likely the values used for new cases of illness as well as possible losses due to premature mortality. These are the two components where the economic value is due to accumulated impacts over time. Few studies have addressed this issue. A summary of the estimates found in the literature for the long term impacts is given in Table 5.

<table>
<thead>
<tr>
<th>Health impacts of relevance for children</th>
<th>Inputs into benefit estimate</th>
<th>COI (euro)</th>
<th>WTP (euro)</th>
<th>Discounting</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lung function development</td>
<td>Not quantified</td>
<td>Not quantified</td>
<td>Needed</td>
<td></td>
<td>-</td>
</tr>
<tr>
<td>Sensitisation to allergens</td>
<td>Not quantified</td>
<td>Not quantified</td>
<td>Needed</td>
<td></td>
<td>-</td>
</tr>
<tr>
<td>Development of air-way disease</td>
<td>-</td>
<td>31 000</td>
<td>Done</td>
<td>BenMap (2008b)</td>
<td></td>
</tr>
<tr>
<td>Reproductive disorders</td>
<td>Not quantified</td>
<td>Not quantified</td>
<td>Needed</td>
<td></td>
<td>-</td>
</tr>
<tr>
<td>Cancer</td>
<td>Not quantified</td>
<td>VSL = 1 million</td>
<td>Done</td>
<td>Bickel and Friedrich (2005)</td>
<td></td>
</tr>
</tbody>
</table>

The value for the development of air-way disease is from one American valuation study and should therefore be treated with caution. Concerning premature mortality an argument for using VSL is that it is likely to ascertain an equal treatment between premature mortality in different policy areas. The question however is which VSL to use in a Swedish context. The VSL proposed in the ExternE methodology update (Bickel and Friedrich, 2005) was 1 million euro to be compared with the value used in the Swedish transport sector of about 2 million euro. The main reason for the difference in these estimates is that the former is the median estimate while the latter is the mean. Sticking to the conventions normally used in cost-benefit analysis, we recommend using the Swedish mean estimate as a basic value for VSL. There are examples in the literature where children’s welfare loss is valued twice as high as adults. Hence, two times these estimates may be used as an upper bound in a benefit calculation.

We want to stress though that the importance of these two components for the total benefit for children’s health is an issue that is left for future research. It is still unclear how relevant these health endpoints are since the epidemiological evidence is
sparse and since there is no or little empirical evidence on the valuation of these endpoints for children.

For the short term (acute) health impacts, the values proposed in the ExternE methodology update (Bickel and Friedrich, 2005) are according to our opinion currently the most relevant estimates also in a Swedish context. The estimates found in the literature are summarised in Table 6.

<table>
<thead>
<tr>
<th>Health impacts of relevance for children</th>
<th>Inputs into benefit estimate</th>
<th>Discounting</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hospital admission</td>
<td>1600</td>
<td>400</td>
<td>No</td>
</tr>
<tr>
<td>Emergency care (=asthma attack)</td>
<td>450</td>
<td>220</td>
<td>No</td>
</tr>
<tr>
<td>General practitioner visit</td>
<td>42</td>
<td>15</td>
<td>No</td>
</tr>
<tr>
<td>Medication</td>
<td>1</td>
<td></td>
<td>No</td>
</tr>
<tr>
<td>Restricted activity days</td>
<td>84</td>
<td>46</td>
<td>No</td>
</tr>
</tbody>
</table>

No firm conclusion can be drawn on how these adult values should be adjusted when used for children. We however believe that it is more reasonable with a smaller markup than twice the estimate for these short term impacts. The reason is that approximately the same consequences are due to sick children as to sick adults. That some premium could be reasonable is because illness may affect future income for the child in a different way than an adult but also because there can be some disutility to account for on the behalf of parent or other concerned relatives when children are sick.

Few Swedish economic valuation studies on environmental health risks have been undertaken and hence it is not possible to state how relevant the estimates presented in Table 5 and 6 are in a Swedish context. Therefore, economic valuation of health risks in general, particularly for morbidity, is an area in need of further research. While willingness-to-pay studies are more demanding we believe it would be quite easy to derive Swedish estimates for cost of illness. There are databases that collect this type of cost information so what is needed is an adaptation of the data to this kind of benefit calculation (i.e. cost per doctors visit or symptom day etc).
Children’s health and the benefit of reducing air pollution – calculations for two cities

In this chapter we will apply the dose-response method that was described in chapter 1. We have done these calculations for two health endpoints where reasonable exposure-response functions and economic values could be derived from the literature. The health endpoints are development of asthma and hospital admission due to respiratory problems (for details see chapter 2 and 3, especially the rows shaded grey in Table 1, 5, 2 and 6). We make the calculations for two different cities, Umeå and Stockholm. They were chosen because information about the number of children and some underlying data on their health status that is needed for the calculations was readily available. The reason for making calculations for two different cities, using only a few endpoints, is to give an illustration of how different components used in the calculation influence the final estimate. Since traffic is possibly the most important source for population exposure in cities, the discussion is based on the influence of this source.

As seen from equation 1 on page 11, an assessment of the change in population exposure is needed in these types of calculations. For policy purposes, what is relevant is to assess the influence of a certain policy measure on air pollution concentrations and population exposure, in this case regarding children. The population exposure is an important component since the benefit estimate will be a function of the population density close to the emission sources. Data with this level of detail however is not available for the two cities. Therefore, what we calculate is the benefit that would result from a reduction in the children’s average exposure in each city of 1 µg/m³ of NO₂ per year. We use this estimate since it is common in economic analysis to consider marginal changes and this change has been evaluated in other studies such as the study on Swedish data by Samakovlis and Svensson (2004)\(^\text{17}\).

\(^{17}\)One reason why economists commonly use marginal estimates is due to the problem with non-linearities as discussed by Viscusi and Gayer (2005). Another reason is that marginal estimates are used as the basis for policy evaluations where the changes in air pollution concentration (or other changes) are small. The basis for transport policy in the EU is also “marginal cost pricing” (EU Commission, 2008).

No detailed exposure estimates resulting from traffic was available for the two cities. Therefore, we cannot assess what policy measure or emission reduction that is needed in each city to achieve a reduction in average exposure of 1 µg/m³ of NO₂ per year, not for the general population and hence not for children. Instead, to give some indication on what this would require, we use available data for the two cities to illustrate how the concentrations vary within a city and how they are influenced by traffic. This information is presented in the next section. Thereafter we present the benefit calculation for the two health endpoints. The chapter ends with
a discussion of the results, how they can be used and the uncertainties related to them.

Exposure quantification

The exposure-response functions we will use are based on concentrations of NO₂ but the available data is modelling and measurement of NOₓ. The concentrations of these two pollutants are quite closely correlated and it is possible to convert the concentration for NOₓ to NO₂. In a study for Stockholm (Naturvårdsverket, 2005) the following conversion formula was estimated:

\[ \text{NO}_2 = \text{NO}_x^{0.66+34/(\text{NO}_x + 100)}. \]

As will be shown in the example provided below, the concentration of NO₂ is generally somewhat lower than the concentration of NOₓ.

The concentrations in the urban background (roof level) of NOₓ within a city will vary depending on the emissions from traffic and other local sources. They are also influenced by the background levels (those found in the countryside without influence of local emission sources). In Sweden this concentration is about 4 µg/m³ and somewhat lower in the north than in the south.

To give an illustration of how the concentrations can vary in a city we have included information from modelling done in Umeå, see Figure 1. The data presented in the figure is modelled estimates of average yearly exposure to NOₓ for children between 5 and 18 years old, obtained with recent traffic data and the Airviro dispersion model, adjusted using monitoring data. Every dot in the figure represents a child and the modelled exposure for this child. As seen in the figure there are quite a few places, especially in the city centre, where the average yearly concentrations are above 20 µg/m³ (the highest estimate is 60.79 µg/m³).

The estimate that is the basis for the information in the figure has also been used to calculate the average population weighted exposure for children in this age group in Umeå. According to this calculation it was 13.5 µg/m³ per year. This information can be used to calculate an approximate estimate of the total external health cost for exposure to NOₓ in Umeå. If we deduct the concentrations found in the background, we can conclude that emissions from local sources in Umeå, mainly motor vehicles, result in an average exposure among children of about 10 µg/m³ per year. This is the concentration that can be reduced by local policy measures.

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18 The estimates are the results of modelled concentrations based on traffic emissions in Umeå using the Airviro model that has been developed by SMHI (Swedish Meteorological and Hydrological Institute). To these modelled concentrations are added the regional background. Traffic emissions are the main source to the NOₓ concentrations since district heating is used and there are no emissions from maritime transport in the area. There are a few point sources but these emissions are released at high altitude and therefore have limited influence on local concentration levels. Additional details about the data can be provided by Bertil Forsberg (bertil.forsberg@envmed.umu.se).

19 The yearly air quality limit value for NO₂ is 40 µg/m³.
Unfortunately, for Umeå we do not have data available to illustrate the influence on concentrations of specific policy measures. Instead, some information on this issue is given in the following using data from Stockholm.

In the inner city of Stockholm in 2006 the measured average yearly estimate of NOx in the urban background was 21 µg/m³ (SLB Analys, 2007)\textsuperscript{20}. This estimate however cannot be used as an estimate for the average exposure to traffic exhausts since it is influenced by many sources. An estimate of the influence from traffic on average population exposure for the whole population is instead given in a study done about the effect of the trial with congestion charges in Stockholm (SLB Analys, 2006). According to this report the contribution from traffic to average exposure in the inner city was 8.41 µg/m³ before the congestion charges and the reduction in traffic emissions (about 8.5%) reduced population weighted exposure in this area by 0.81 µg/m³ (10%). Using the conversion formula we arrive at a change in NO2 of about 0.71 µg/m³.

The influence of the congestion charges on air quality in the greater Stockholm region was more limited. The contribution from road traffic to NOx levels in greater Stockholm was estimated at 4.42 µg/m³ before the implementation of congestion charges, evaluated as a population-weighted urban background level (Johansson et al., 2009). The decrease in traffic due to the congestion charges was estimated to reduce the NOX level by 0.23 µg/m³ (5.2%) to 4.19 µg/m³. This equals a reduction

\textsuperscript{20} They also measure NO2. The average yearly concentration was 17 µg/m³. There are also measurements per hour and the highest estimate was 264 µg/m³ on the 13\textsuperscript{th} of October, 2006.
in NO\textsubscript{2} of 0.17 µg/m\textsuperscript{3}. Hence, the congestions charges mainly contributed to reduced population exposure in the inner city which is because this is the area were charging was used.

As illustrated in this section the exposure to NO\textsubscript{x} (and NO\textsubscript{2}), and the influence of policy measures that reduce these emissions, will be highly context dependent. The influence on average population exposure of a policy measure will be higher if the emissions are reduced in areas with high population density such as city centres. However, the influence such measures will have on children’s health depends on the number of children living in the city centre. If children to a greater extent live outside of city centres then exposure estimates for the general population will overestimate the exposure of children.

Another thing to notice from these estimates is that measured concentrations cannot be used to assess the influence from a specific source. According to the modelling in Stockholm the contribution of local traffic to average population exposure in the inner city is about half of the concentration level that was measured in the urban background in the same area. As for the influence of reductions in traffic it was found that an 8.5% reduction in the emissions in the inner city resulted in a reduction in average population exposure of less than 1 µg/m\textsuperscript{3} NO\textsubscript{2}. Hence what we have illustrated in this section is that in order achieve an average reduction in population exposure of 1 µg/m\textsuperscript{3} NO\textsubscript{2}, which is what we will assume in our calculations in the next section, quite important reductions in traffic emissions in a city are required.

**Health impact and benefit of reductions in long-term exposure**

The health endpoint where we found most usable exposure-response functions was wheeze. It is expected that some children with this diagnosis will also develop asthma in the future. The estimate we use in this calculation is a meta-OR of 1.38 (95% CI 1.16-1.64) for an exposure difference of 10 µg/m\textsuperscript{3} NO\textsubscript{2}, based on the result of five cross-sectional studies studies, see page 18. This estimate is relevant for children in the age from 5 to 18 years, hence our calculations may underestimate the total impact for this diagnosis. Of these new cases of wheeze we expect that at least 50%, but maybe as many as 75%, will develop asthma in the future.

Since this estimate is a change in the likelihood of observing a symptom in the population, we have to know the underlying prevalence of the diseases in the population. Unfortunately we could not obtain an up to date estimate for this baseline risk from the National Board of Health and Welfare within the time frame of the project so we make an assumption based on results from another study (Bjerg et al., 2010)\textsuperscript{21}. In both Stockholm and Umeå we make the assumption that 13000/100 000

\textsuperscript{21} An application was made in the autumn of 2009 but there were problems in the ethical judgement that stopped the delivery of data.
children currently have wheeze symptoms; hence the current individual risk is 13%. Under the assumption of a linear relation between exposure level and prevalence of wheeze, the estimated meta-OR above corresponds approximately to a decrease in prevalence by \((1.38-1)/10*13 = 0.49\) percent units for each 1 µg/m³ decrease in NO₂. In greater Stockholm there are 220 000 children age 5-18 and reducing their exposure with 1 µg/m³ would imply 1087 (=0.49/100*220 000) fewer cases per year in Stockholm. In Umeå there are 10 500 children in this age. Here the reduction in number of cases would be 52 per year.

To calculate the benefits over time for this reduction in exposure we make the assumption that 50% of these children will develop asthma in the future, hence 544 cases in Stockholm and 26 cases in Umeå. For these cases we use the estimate presented in BenMAP that was based on willingness to pay to avoid asthma in the future (US EPA, 2008b). The estimate per case was 38 947 USD in price level 2000. Assuming an exchange rate of 8SEK/USD, we arrive at an estimate for this outcome of about 310 000 SEK. The benefit from reducing the average exposure to NO₂ by 1 µg/m³ would therefore be 168 million SEK (=544*310 000) in Stockholm and 8 million SEK (=26*310 000) in Umeå.

Health impact and benefit of reductions in short-term exposure

For the short-term effects the endpoint most suitable to use in this study of traffic pollution is the influence on the number of respiratory hospital admissions during an average year. The study by Fusco et al. (2001) is here assumed to be representative for Europe, see page 20. The relative risk associated with a 10 µg/m³ increase in NO₂ is a 1.77 % increase in admissions in the age group 0-14 years (95 % confidence interval 0.27 - 3.30).

Also in this calculation we need an estimate of the occurrence. According to official statistics, approximately 800/100 000 children in the age 0-19 years in Stockholm County were admitted to hospitals for respiratory problems in 2008 (http://www.socialstyrelsen.se/statistik/statistikdatabas). This implies that the annual risk for an individual to be admitted for such problems is 0.8%. We assume this risk in our calculations.

In Greater Stockholm there are 330 000 children in this age group. According to the estimates above, a reduction in exposure of NO₂ of 1 µg/m³ results in a risk reduction of 0.177% which would imply 4.7 (=0.00177*0.008*330 000) fewer

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\(^{22}\) The preventive potential in Stockholm for decreasing the influence of NO₂ on wheeze is estimated for the following population: 220 000 in the 5-18 age group (whole population: 1 400 000) in Botkyrka, Danderyd, Huddinge, Järfalla, Lidingö, Nacka, Salem, Sollentuna, Solna, Stockholm, Sundbyberg and Täby, in 2006

\(^{23}\) Approximately equal to 31 000 euro, the estimate presented in Table 5.

\(^{24}\) In the 0-19 age group in Botkyrka, Danderyd, Huddinge, Järfalla, Lidingö, Nacka, Salem, Sollentuna, Solna, Stockholm, Sundbyberg and Täby, in 2006.
cases yearly in Greater Stockholm. In Umeå there are 14 500 children in this age living in the City. Here the reduction in number of cases would be 0.2.

To estimate the benefit from this reduction we have to determine the implications of hospitalization in terms of costs. This will be influenced by the average time each child spends in hospital. In this case we make the assumption that the average days in hospital are 2.5. The ExternE estimate of 2000 euro per admission is an estimate assuming 3 days in hospital and 5 days at home. We therefore use half this sum, 1000 euro or 10 000 SEK, as an estimate for the cost of hospitalization. Using this estimate we find that for this health endpoint a reduction of 1 µg/m³ NO₂ gives a benefit of 47 000 SEK per year in Stockholm and 2000 SEK per year in Umeå.

Discussion of results and uncertainties

In Table 7 we have summarised the inputs that were the basis for the calculation in the previous section. The final result is the product of the values in column b to f. This table gives a good illustration of the data required for these calculations. The final result is dependent on the accuracy of the input in every part of this calculation chain. We will therefore end this chapter with a discussion of the assumptions made in our calculations and on the uncertainties related to this type of calculation in general. As in other socioeconomic evaluations the robustness of the results should be clarified using sensitivity analysis when used in policy analysis.

<table>
<thead>
<tr>
<th>Pollutant</th>
<th>Δ Exposure</th>
<th>Population*</th>
<th>Baseline</th>
<th>Exposure-Response Function**</th>
<th>Economic value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Development of air-way disease</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Umeå</td>
<td>NO₂ as indicator</td>
<td>1 µg/m³</td>
<td>50% of 10 500</td>
<td>13%</td>
<td>0.038</td>
</tr>
<tr>
<td>Greater Stockholm</td>
<td>NO₂ as indicator</td>
<td>1 µg/m³</td>
<td>50% of 220 000</td>
<td>13%</td>
<td>0.038</td>
</tr>
<tr>
<td>Respiratory hospital admission</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Umeå</td>
<td>NO₂ as indicator</td>
<td>1 µg/m³</td>
<td>14 500</td>
<td>0.8%</td>
<td>0.177%</td>
</tr>
<tr>
<td>Greater Stockholm</td>
<td>NO₂ as indicator</td>
<td>1 µg/m³</td>
<td>330 000</td>
<td>0.8%</td>
<td>0.177%</td>
</tr>
</tbody>
</table>

* For air-ways disease we make the assumption that 50% of the children are likely to develop a chronic condition. The population is different for the two health outcomes because the dose-response relationships are relevant for different age groups.

** For air-ways disease the odds-ratio is used while the relative risk is stated for respiratory hospital admission.
Regarding our calculations, the first things to notice is that for the same exposure reduction we get much higher benefit estimates for the long term effects in both cities (for Stockholm 168 million SEK vs 47 000 SEK). Since we have omitted a number of short term health impacts affecting asthmatic children, the estimates for the two endpoints are not directly comparable. Still, it appears that it is the risk of developing chronic conditions that give rise to the highest external cost for children. Similar results are found in calculations for the adult population. Hence, as in other health care it may be better to prevent a disease from occurring instead of reducing the symptoms afterwards. This finding may have implications for the design of policy measures since it may be more important to reduce the average concentration levels rather than reducing short-term episodes in local hot spots.

A second thing to notice is that when using these calculations for policy purposes it is important to assess the size and the extent of the impact. To give an illustration we will assess the impact of the congestions charges on children’s health in Stockholm. The example we use is the probability of developing an air-way disease. To do this we have to modify our previous calculations. First of all we should only include the children living in the inner city in the calculation since this is where we had the main influence on the concentration levels. About half of the 220 000 children in the age 5-18 living in Greater Stockholm live in the inner city. Secondly we have to account for the actual reduction in this area which was estimated to be 0.71 μg/m³ for NO₂. Hence the estimated impact of the congestion charging trial in Stockholm is less than half of the estimated cases of wheeze that we used in the previous calculation, 386 (=1087*0.5*0.71) per year. If half of these develop asthma then the benefit for children of the congestion charges would be about 60 million SEK (386*0.5*310 000) per year instead of the 168 million that a general reduction of 1 μg/m³ for the children in Stockholm would result in.

A third thing to notice is that there are a number of uncertainties related to these results. For development of air-way disease we have assumed that 50% of those exposed will develop the disease. There is however not an exact relationship determined for the share of children with wheeze symptoms that may develop asthma. Hence the expected health impact may be higher or lower. The same applies to our assumption on the impact of hospitalization due to respiratory problems. Our baseline data are also rather crude. The baseline may for example vary geographically. More accurate data is available from registers collected by the National Board of Health and Welfare, but we were not able to use it in this project because it was subject to ethical judgement.

As for the economic values we have used, they are taken from the models developed for external cost calculations in the EU and the US. Although these values have been based on careful evaluation of the state-of-the art research, there is the question of how relevant they are in a Swedish context. As discussed by Viscusi and Gayer (2005), using benefit transfers is this type of extrapolations to other groups or contexts is based on strong assumptions. Moreover, given that we have
used economic values for adults it may be that we have underestimated the benefit, at least for asthma attacks. Finally, as can be seen from Table 5, little is known about how to value long term health impacts.

It is also to be remembered that we in this study have used NO$_2$ as an indicator for emissions from traffic. This does not imply that reductions of emissions in NO$_x$ will bring about these benefits. The cause and effect between single pollutants and health impacts is maybe the most important uncertainty. This however is not only a problem for the calculation of external costs but for health impact assessment as such. For policy purposes this is problematic since efforts may be made that mainly reduce pollutants that are not causing the health impacts.

A final comment concerns the use of these estimates for policy evaluation. When deciding whether or not a reduction in emissions is beneficial from society’s point of view, the cost needed to achieve the reduction in the emissions also needs to be considered. In our case reducing emissions can be achieved by reductions in traffic in the two cities. Our benefit estimates are based on an average reduction in exposure of 1 µg/m$^3$ of NO$_2$. As discussed above, this change in exposure is quite large. According to the findings from the results of the congestions charges trial, a reduction of traffic of 15% in the inner city resulted in a exposure reduction of this size.
Final remarks and future research

This is a first attempt to calculate the benefits for children in Sweden of reducing air pollution. Due to lack of data we have only been able to give an indication of the size of the benefits per µg/m³ decrease in the mean concentration of NO₂. Further research is needed in order to determine the accuracy of these estimates, the size of the benefit for other endpoints and all children in Sweden and how the benefits vary between different geographical areas. We however consider such research to be warranted since our estimates suggest that reducing air pollution is important particularly because it reduces the risk of the development of chronic conditions such as asthma.

In the study, we have surveyed the literature both regarding air pollution and health impacts on children and economic valuation of children’s environmental health risks. Based on the findings we have calculated the benefit of a reduction in children’s exposure of 1 µg/m³ of NO₂ in Stockholm and Umeå. The difference between the cities that we could account for is the number of children that are exposed. As expected, the estimated cost was higher in Stockholm because more children live there. There will be other inputs that differ and that will influence the results, such as the share of asthmatics in the child population and the change in exposure due to a certain change in emissions, but this information is not currently available.

The calculation was done for two endpoints, developing an air-way disease (asthma) and hospital admissions due to respiratory symptoms. According to our estimates this reduction in exposure in Greater Stockholm per year would generate a benefit to society of 168 million SEK because of fewer cases of asthma, and 47 000 SEK due to fewer hospital admissions (for the price levels in 2000). These figures can be compared to the results in Huhtala and Samakovlis (2003) that arrived at an estimate for acute respiratory problems in the adult population in Sweden, for the same reduction in pollution, of 745 million SEK per year (for the price level in 1999). These figures suggest that the health benefits from a reduction of this size are quite important.

We however want to caution that both these studies are the first of their kind in Sweden and therefore the results needs further validation. Regarding the calculations in this study the exposure-response functions used are based on the results of a few studies. Moreover we have made assumptions regarding the share of children that develop asthma which should be verified scientifically. Regarding the economic values used they are estimates used in models in other countries so their relevance in a Swedish context also needs to be studied. As for the study done by Huhtala and Samakovlis (2003) their results are based on a survey of self-reported symptoms in the general population. How well reported data correspond to other types of quantifications of health impacts should also be assessed.
In addition, the benefit estimates presented above cannot readily be used for policy analysis. As discussed in the end of the previous chapter, the effects of a policy measure needs to be assessed on a case to case basis. The benefit estimates we have presented are based on a quite large reduction in air pollution. 1 µg/m³ is about 10% of the contribution to NO₂ from local sources in Swedish cities. This is approximately the reduction that resulted in the inner city of Stockholm from the trial with congestion charges where traffic in this area decreased by 15%. Hence, in order to determine whether or not to take action to reduce traffic, the costs and benefits of such measures needs to be compared on a local scale. The result will for example be dependent upon the impact a certain measure has on exposure. A general conclusion from studies in this area shows that the benefits are expected to be high if emissions are reduced in densely populated areas.

We have in this report shown that it is possible to undertaken benefit calculations also for children. Being the first attempt of its kind in Sweden we however have been limited by the data available. To our knowledge no estimates are available on children’s exposure to air pollution resulting from traffic and other sources. Moreover, we ran into problems when trying to obtain relevant baselines on the prevalence of illness in the child population in the two cities from The National Board of Health and Welfare. Hence, in addition to the uncertainties related to the inputs in these calculations there are also other problems. The latter however we believe should be quite easy to overcome. There are models developed that should be possible to use for exposure quantification and regarding the denial for baseline data, it could well be due to a misunderstanding of the intended purpose.

As for future research we suggest that the following should be considered:

First and foremost, more studies of benefit calculations should be done since it is in the actual practice that what is known and not known about the different parts of the calculation chain are clarified. There are now models developed for exposure assessment but they need to be combined with other models such as those for transport to assess the influence of different policies.

There is also a need for more research on health impacts and the economic value placed on reducing these health risks. Of particular importance is to try to determine which of the different emissions from traffic that gives rise to these and other health impacts. Furthermore, the same main sources and a similar geographical resolution should be used in the studies that produce the exposure-response functions as in the exposure model used for the health impact assessment.
References


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Air pollution and children’s health in Sweden

An enquiry into how the economic benefit of improvements in children’s health resulting from reductions in air pollution can be assessed

LENA NERHAGEN, TOM BELLANDER AND BERTIL FORSBERG

Frisk luft är ett av de 16 miljömål som antagits av Sveriges riksdag för att styra samhället mot en hållbar utveckling. Detta projekt bidrar till arbetet med att genomföra detta miljömål. Mindre är känt när det gäller luftföroreningars påverkan på barns hälsa än när det gäller vuxna. Syftet med projektet har varit att fastställa hur hälsocinster med minkad exponering för luftföroreningar kan kvantifieras i ekonomiska termer.

 För att svara på denna fråga ger vi inledningsvis en kort beskrivning av den metod som vanligtvis används för att beräkna hälsocinster i ekonomiska termer. Därefter illustrerar vi genom två fallstudier hur dessa beräkningar genomförs och vad som påverkar resultaten. Rapporten avslutas med en sammanfattning över områden där vi ser att fortsatt forskning behövs.

När det gäller ekonomisk värdering av hälsoeffekter på barn är det svårare att hitta uppgifter om kostnader för påverkan på hälsan av luftföroreningar på barn än när det gäller vuxna. Eftersom få svenska ekonomiska värderingsstudier har genomförts kring miljörelaterade hälsorisker rekommenderar vi användning av ekonomiska värden från framförallt europeiska studier.

Baserat på slutsatserna i litteraturöversikterna har vi, som ett exempel, beräknat hälsocinsten av att minska barns exponering för NO2 med 1 µg/m³ i Stockholm och i Umeå. Enligt våra beräkningar skulle hälsocinsten för samhället av denna minskning i exponering i Storstockholm vara 168 miljoner SEK per år p.g.a. färre fall av sjukdomen astma samt 47 000 SEK p.g.a. färre inläggningar på sjukhus (i prisnivån för år 2000). Hälsocinsten i Umeå är betydligt lägre, 8 miljoner SEK och 2000 SEK per år.

Detta är det första försök att i ekonomiska termer beräkna hälsocinstern för barn av minskade halter av luftföroreningar som genomförts i Sverige. Slutsatsen av vår genomgång är att det finns kunskapsluckor när det gäller luftföroreningar och barns hälsa vilket får konsekvenser för möjligheten att genomföra beräkningar av detta slag. Beräkningar ger därför endast en indikation på hur stora hälsocinstern kan vara.