1 Introduction

1.1 Background

1.1.1 Transport and health

Transport is an essential component of life. Effective transport networks bring health benefits by giving people access to work and essential services, shops, leisure facilities and enabling social contact (Killoran et al., 2006). However, different modes of transport have specific effects on society. The transport scene in Europe is dominated by road transport, which accounts for 85% of the motorized passenger-kilometres in the 25 countries that were members of the European Union (EU) before 1 January 2007 (European Commission, 2006). As road transport accounts for the largest share of transport activities, involves nearly the entire population, directly influences urban development and produces the greatest effects on emissions of pollutants and consumption of energy, this report addresses the health costs related to this mode of transport.

The range and extent of the health effects of road transport are increasingly being understood and clarified. Air and noise pollution, road crashes and deterrent effects on walking and cycling adversely affect health, as do less obvious factors such as social isolation and worsened quality of life in neighbourhoods affected by heavy road traffic.

During the past decades, road transport has been changing: more vehicles carrying fewer passengers per vehicle are making more and longer trips. These changes are affecting the health of more and more people.

1.1.2 Children

Children’s exposure to transport-related pollutants may differ from adults’ exposure, since children spend their time in other settings and behave differently. Further, children are suspected of being more susceptible to the effects of transport-related pollutants for several reasons: (1) their organs are not yet fully developed and their physiology differs in several ways (such as higher resting metabolic rate and rate of oxygen consumption per unit of body weight); (2) children are not always aware of the dangers; (3) children have different exposure profiles (such as being closer to the ground and thus to car exhaust pipes due to their size); and (4) children have not (fully) developed coping mechanisms and cannot change their situation, whereas adults may have the power and/or resources to do so (Tamburlini et al., 2002; van den Hazel & Zuurbier, 2005). These factors combine to generate or trigger a wide range of negative health effects. In addition, the results of observational studies have shown that some adult diseases may originate from childhood exposure. Road transport might play a role in this. Understanding how the environment affects children’s health and development is therefore important for preventing illness (Stansfeld et al., 2005).

1.2 Transport, Health and Environment Pan-European Programme

As part of implementing the joint UNECE (United Nations Economic Commission for Europe)/WHO Transport, Health and Environment Pan-European Programme (THE PEP), in 2003 six European countries launched a project on transport-related health effects with a particular focus on children. The aim was to make progress towards integrated assessment of major transport-related health effects. After the evidence was reviewed and various aspects of transport-related effects on environment and health were addressed, a set of key messages was developed. These were summarized in a synthesis report (THE PEP, 2004a) and an executive summary (THE PEP, 2004b), which were launched at the WHO Fourth Ministerial Conference on Environment and Health in Budapest in June 2004. One topic the project specifically addressed was the economic valuation of transport-related health effects (THE PEP, 2004c). Further research and work on this topic were recommended, and the following questions requiring priority attention were identified:

- how to select pertinent health effects in children and how to estimate the quantitative relationships between exposure and health effect (exposure–response function);
- how to accurately estimate the fraction of exposure coming from transport;
- how to quantify the health effects attributed to this exposure; and
- how to measure and express in monetary terms these health effects and how to foster comparability.

This project addressed these questions.

1.3 Aims of this project

The overall objective of this project was to develop practical approaches for carrying out economic valuation of road transport–related health effects, based on best available evidence and including a focus on children. The project was implemented through the following steps:

- Two reviews of relevant literature were carried out. The aim of the first review was to identify the health end-points related to road transport for which sufficient evidence and data (such as exposure–response functions) exists for including them in economic valuation. Literature was analysed for both adults and children. The second review examined the literature on methodological approaches to economic valuation of health costs due to road transport to derive guidance on the best way forward.
- Based on these two reviews, practical approaches for carrying out economic valuation of transport-related health effects were developed. In the development of these approaches, attention was given to orienting the reader on selecting the best approach, taking into account the specific conditions in different countries.
- The results of the literature reviews and an initial proposal of practical approaches were discussed at an international workshop (12–13 November 2007, Düsseldorf, Germany) with the project advisory group, a panel of experts with background in health, transport, environment and economics. The final draft version was again made available to the advisory group, three independent reviewers and the Steering Committee of THE PEP. Based on the feedback and comments received, the report was finalized.

The report also highlights present methodological limitations, especially related to valuating some health effects (such as psychosocial ones) and/or to some population groups (especially children). It also identifies where gaps in knowledge exist.

The approaches developed through this project are not meant to be prescriptive. They present a possible way forward based on best available evidence and clearly specifying the assumptions made and existing limitations. In addition, the rapid development of new scientific knowledge in both economics and epidemiology needs to be taken into account in further developing the proposed methods.

The presented approaches target non-health experts and practitioners in transport planning, ranging from experts operating at the national level to those dealing with subnational and local assessment. They are intended to facilitate the integration of health-related effects in the economic valuation of transport options. The geographical scope of application can be international, national and local. The approaches can also serve as general direction for very small-scale interventions paying, however, attention to the specificities of those situations.

Assessments developed by applying the approaches presented in this report can be used in a variety of types of economic valuation, including cost–benefit and cost–effectiveness analysis of transport options that aim at factoring in health-related costs. Ultimately, it is hoped that the result of this project will contribute to better integrating health concerns into transport policies, thereby supporting more informed decision making.

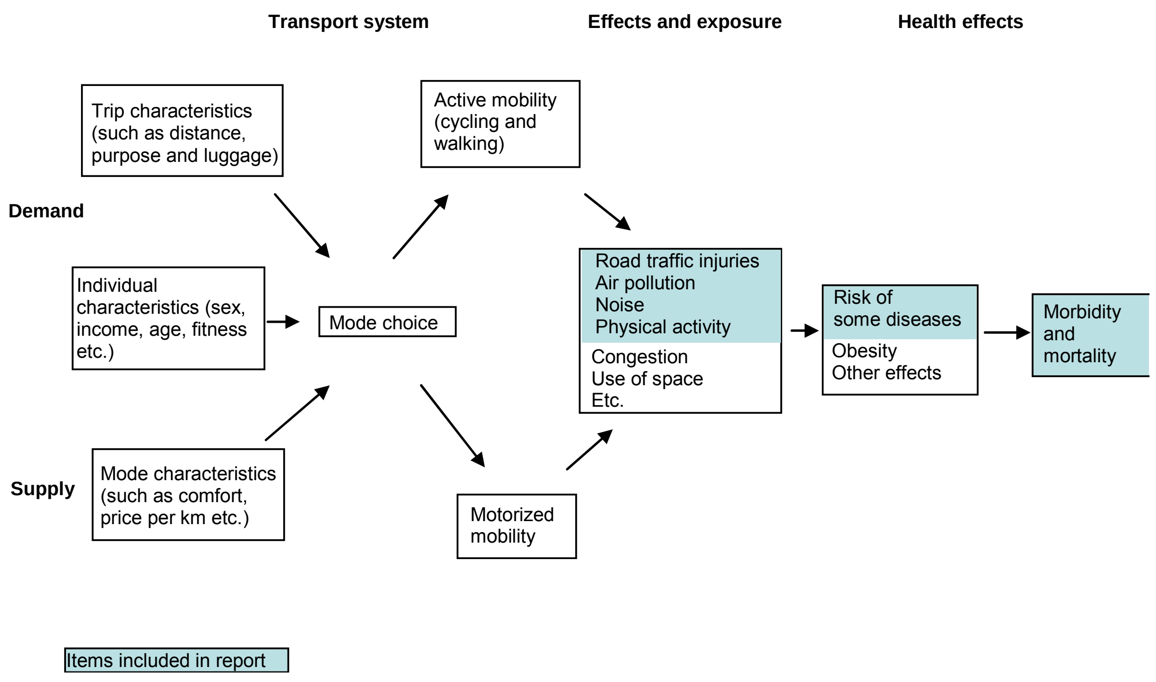
2 Methodological approach

2.1 Conceptual framework

This report is based on the conceptual framework shown in Fig 1. It shows the determinants of a choice of mode of transport, including both the demand and supply sides, that lead to a certain choice of mode. The mode of transport can either be motorized or non-motorized, which leads to different levels of effects and exposure, such as air pollution, noise or physical activity. The exposure leads to different risks of certain health effects and eventually morbidity and mortality.

The model deliberately does not include all determinants and health outcomes but focuses on those for which economic valuation can be carried out, owing to the availability of exposure–response functions. The items discussed further in this report are dotted in Fig. 1.

Fig 1. Conceptual framework for transport-related health effects for this report



2.2 Overview of recent epidemiological literature on the health end-points considered

The aim of this part of the report was to identify relevant health end-points in relation to road transport for which sufficient evidence exists for including them in economic valuation of the health effects of road noise, transport-related air pollution, road traffic injuries and insufficient physical activity as well as other potential health outcomes. This was done for adults as well as children.

The literature on transport-related health effects is vast. Since systematically reviewing all transport-related health effects was beyond the scope of this project, the overview summarizes recent peer-reviewed literature reviews and major studies on the health effects of exposure to road noise, transport-related air pollution and insufficient physical activity with a focus on walking and cycling. In relation to road traffic injuries, health end-points are more clearly related to transport activities, but nevertheless an overview of key issues is presented. An international expert workshop discussed the selected literature and the advisory group reviewed it to ensure that the most important literature sources were taken into account.

After transport-related health effects were identified, health outcomes for inclusion in economic valuation were selected according to the following criteria:

- the availability of sufficient strength for an association between exposure and the health end-point; and
- the availability of a valid exposure–response function.

Some reviews have focused on the evidence provided by the results of epidemiological environmental studies investigating the relationships between road noise, transport-related air pollution and insufficient physical activity and health. To assess the strength of the association between exposure and a particular effect, the available evidence in these overviews was rated in terms of the categories proposed by the International Agency for Research on Cancer as “sufficient”, “limited”, “inadequate” or “lacking”. For this report, based on the available evidence on all these categories, the interest was in identifying the effects for which an association with transport-related air pollution, road noise or physical activity or inactivity is considered “sufficient”. For road noise and/or air pollution, only effects that are likely to occur at typical community levels were considered. Road traffic has also psychological and social effects on health and well-being. However, literature on these effects is scarce, and they are thus more difficult to quantify (Institute of Environmental Health, 2004). They are therefore not further discussed here, although they should be considered at least qualitatively in assessing the benefit or costs of transport interventions.

For each of the selected effects for which the strength of the evidence was sufficient, the availability of valid exposure–effect relations in the epidemiological literature was evaluated. According to WHO guidelines (WHO Regional Office for Europe, 2000), the exposure–effect relationships may be reported as a slope of a regression line or as a relative risk for a given change in exposure. Relationships can be used that are derived either from a quantitative analysis of published data (pooled analysis or meta-analysis) or, if not available, from single epidemiological studies.

Selecting a suitable set of exposure–effect relationships requires considering transferability: transfer of relative risk estimates to populations other than the study population from which the estimate has been derived. Although substantial differences in susceptibility to, for example, air pollution seem unlikely, factors that may affect transferability include differences in daily patterns of activity, climatic and housing conditions and other factors that would result in differences in exposure from the same ambient concentration; differences in the pollution mixture and differences in the importance of confounding factors that might not have been properly controlled for in the epidemiological studies; techniques for measuring the concentration of air pollution; and others (Martuzzi et al., 2003; Medina et al., 2004). Finally, conclusions were drawn on including the identified health end-points in economic valuation.

2.3 Overview of recent economic approaches to the health effects considered

The literature on the economic valuation of transport-related health effects is less extensive than the epidemiological literature. A more systematic approach could therefore be applied to this part of the literature analysis. The advisory group identified the literature reviewed in this project (see below) assisted by leading experts in transport-related health effects. To be included in the analysis, a study had to be:

- an economic valuation of transport-related health costs, including at least one of the categories considered;
- an international study (covering more than one country) or a national study that applied a very innovative approach and/or used special methods and for which a full description of the applied methods could be retrieved; and
- available in the public domain.

For studies on air pollution, exceptions to the first requirement were made since not all the studies included addressed the transport-related part of air pollution; they were included because they either applied a very innovative approach (the second requirement) or covered world regions otherwise not covered. The same exception was applied to some studies on insufficient physical activity, as very few studies have specifically examined transport-related activity.

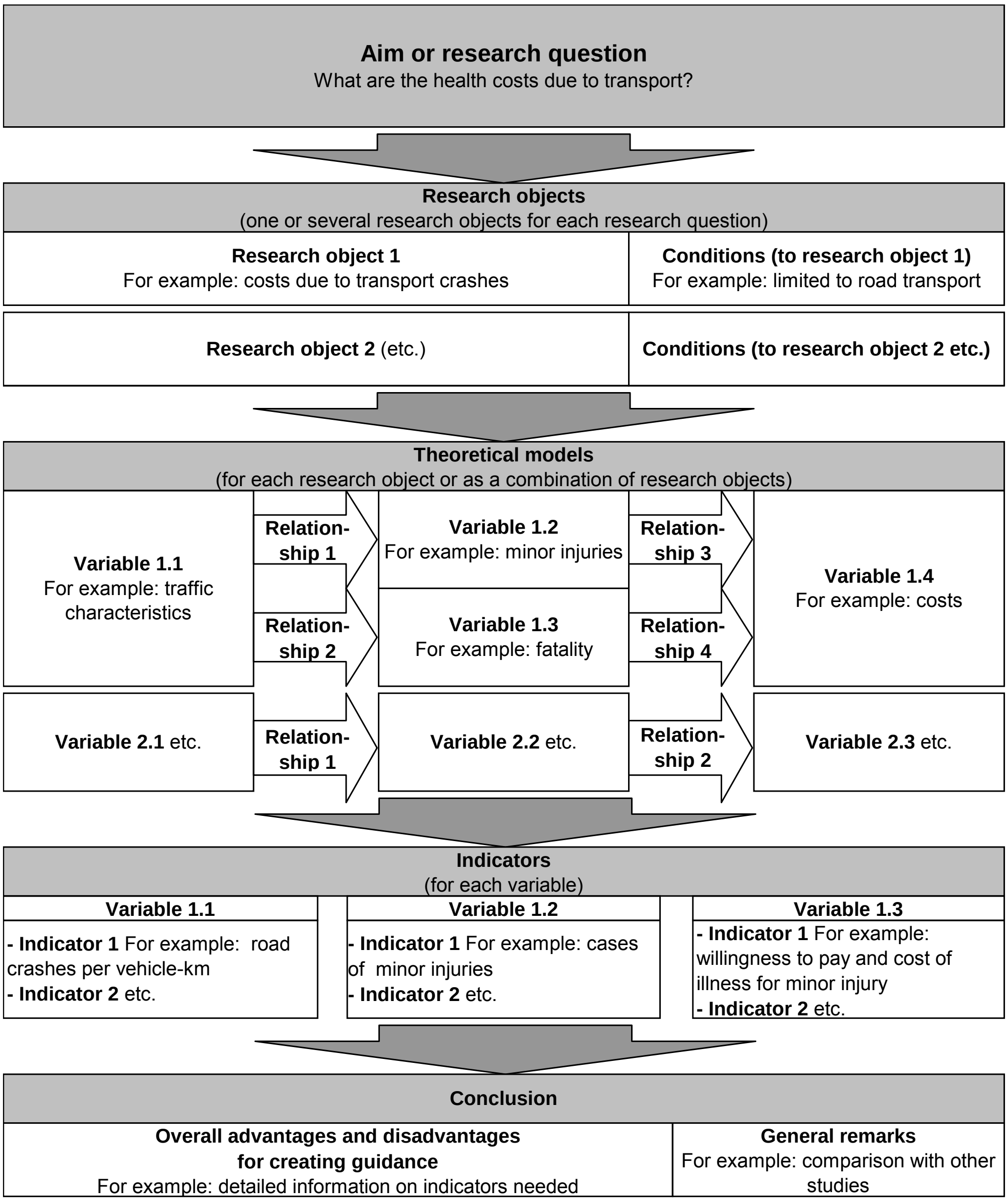
Based on this approach, 38 studies were included (Table 1). Annex 1 lists the studies and texts that were not included. Although academic studies discussing methodological approaches were considered, most studies of interest for this discussion represent practical attempts to develop economic valuation, overcoming limitations in data and methods, with the aim of developing a pragmatic approach that could serve as practical guidance to interested users.

Table 1. Studies on economic valuation of transport-related health effects selected for the literature analysis

	Road crashes	Air pollution	Noise	Physical inactivity
AEA Technology Environment (2005a)		x		
AEA Technology Environment (2005b)		x		
Allender et al. (2007)				x
Cavill et al. (2007)				x
Chenoweth (2005)				x
Colditz (1999)				x
Colman & Walker (2004)				x
Department for Culture, Media and Sport (2002)				x
DIW et al. (2000)	x		x	
Ecoplan (2002a)	x			
Ecoplan (2002b)	x			
Ecoplan et al. (2004a)		x		
Ecoplan et al. (2004b)			x	
Hammitt (2006)				
Holland & Watkiss (2002)		x		
Hunt & Oritz (2006a)				
Hunt & Ortiz (2006b)				
IER (1997)		x		
IER (2005)		x		
IER (2006)	x	x		
IER et al. (2000)		x	x	
IER et al. (2003)		x	x	
Infras & IWW (2004)	x	x	x	
Navrud (2002)			x	
Popkin et al. (2006)				x
Solomon et al. (2005)				
Sommer et al. (1999)		x		
Swedish National Road and Transport Research Institute (2000)	x			
Swiss Federal Office of Sport et al. (2001)				x
THE PEP (2004a)	x	x	x	x
Van Essen et al. (2007)	x	x	x	

Guidance in quantifying health costs comprises not only advice on how to calculate a certain variable. It also has to explain the broader context and the interconnections of these calculations; otherwise applying the correct elements for a specific research question is difficult. For this task, each text was analysed according to a standard analysis grid with a set of criteria (Annex 2) and summarized in a table. The analysis examined the basic research questions of each study and the detailed steps of the calculations, including variables and indicators (see Fig 2). Based on the results of this analysis as well as detailed discussion of the draft methods with an international panel of experts, a practical approach was developed that covers all these aspects (see Chapter 3).

Fig 2. Analysis steps for recent types of economic valuation and criteria applied



The analysis of each text started with the research question to obtain a fundamental understanding of the study. Next, the research objects were analysed to define which aspects the study included or excluded. Afterwards, the theoretical model for the calculation was analysed, including all variables and their interrelationships. The indicators used for each of the variables were identified where possible. As the last step, conclusions were drawn on the overall advantages and disadvantages of the study in deriving general guidance. For each step, special attention was given to the situation of children. What is already known about the situation of children? Do they have to be and can they be treated differently? Are there health effects that are only relevant for children?

3 Practical approaches for the valuation of health costs due to transport

3.1 Introduction

Before the practical approaches are presented, approaches to the economic valuation of health costs are introduced.

3.1.1 Overview of total health costs, their components and valuation approaches

The health effects of road transport involve not only mortality and morbidity but also aspects of the quality of life such as the aggravation of pre-existing disease symptoms, severe annoyance and perceived danger. Integrating such a diverse range of health effects is a challenging task. One method that facilitates the aggregation of different health effects is the use of aggregated health indicators such as disability-adjusted life-years (DALYs) or quality-adjusted life years (QALYs) (Knol & Staatsen, 2005). Health effects can also be expressed in monetary terms. This requires the difficult task of expressing loss of life, life-years or burden of disease in monetary units: valuating various health outcomes. However, using monetary units offers the advantage of comparing costs and benefits directly and assessing whether a proposed policy is worth its costs. Economic quantification of health effects also allows the results to be integrated into broader economic assessment (for example, of transport infrastructure) that does not use indicators such as DALYs or QALYs. Economic valuation also enhances the ability of decision-makers to understand the results of such assessments, although economic arguments should not be the sole basis for decision. Since expressing health effects in monetary terms always involves assumptions and simplifications, the underlying hypotheses and limitations need to be clearly communicated together with the results (WHO Regional Office for Europe, 2005a).

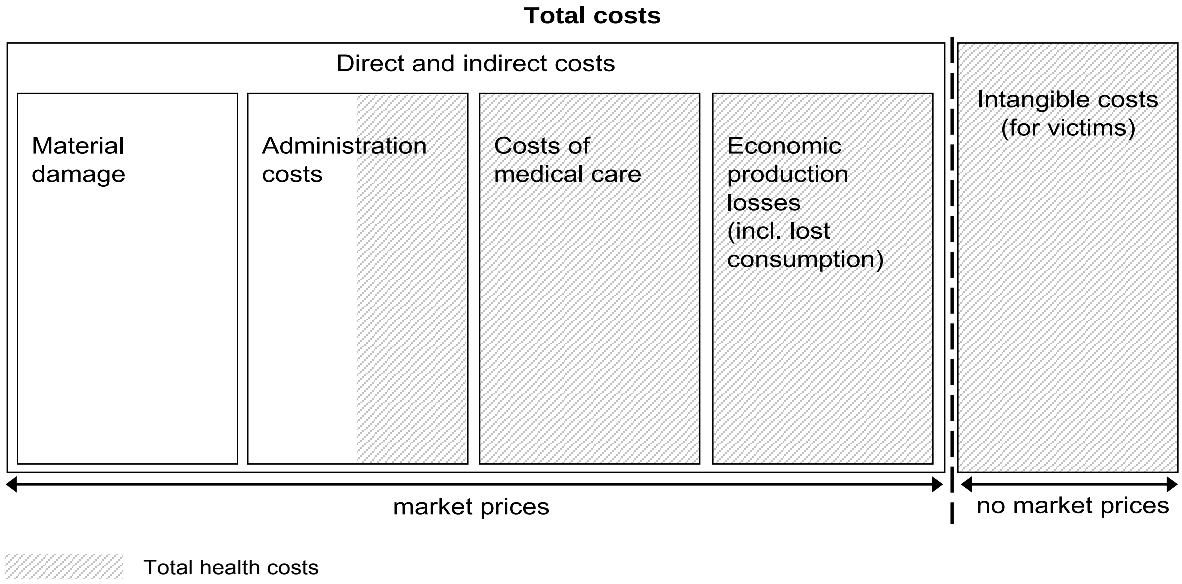
This section explains some general principles for valuating health effects in more detail. The general structure of the suggested approaches is presented, and the main sources of uncertainty for the different steps are discussed.

3.1.1.1 Total costs versus total health costs

Broadly speaking, total health costs comprise direct and indirect costs and intangible costs, which include suffering and grief. The main components of direct and indirect costs are the costs of material damage, costs of health care, administrative costs and economic production losses, whereas intangible costs are not usually subdivided further (Fig. 3). The most important difference between direct and indirect costs and intangible costs is that market prices are available for the first cost component (for example, the price of a specific medication or a health care consultation). Intangible costs have no market price. This difference greatly affects how total costs (and also total health costs) can be assessed in economic terms (see next section).

In contrast to total costs, total health costs include only the parts that are directly related to a health effect (the dashed area in Fig. 3). For example, if a health effect leads to economic production losses as the affected person is not able to work, these costs add to the total health costs. On the other hand, the costs of material damage and some of the administrative costs are not linked to any health effect but result, for example, from road crashes leading only to material damage or the damage to the historical heritage due to vibration and air pollution.

Fig. 3. Total costs of transport and their components and total health costs



3.1.1.2 Economic valuation of total health costs

Market prices for some of the cost components are used to estimate total health costs (administration costs, costs of health care and costs due to economic production losses). Hence, direct and indirect costs can easily be measured by applying these market prices to all costs that arise (Fig. 4). This valuation method is called the cost-of-illness approach. As mentioned, no such market prices exist for intangible costs, and so these costs cannot be valuated with the cost-of-illness method.

For the valuation of such intangible costs, a method called willingness to pay can be used. This method is based on the fact that society pays some of the total health costs and the victims pay some. For example, society pays a large share of the costs of health care, as taxes primarily fund most health care systems.[1] The intangible costs, in contrast, are only borne by the victim (see Fig. 4, indicated by the horizontal dashed line). The willingness-to-pay method measures how much a representative sample of individuals (potential victims) are willing to pay (in monetary terms) to reduce their risk such as a road crash or a specific illness or death.[2] Hence, willingness-to-pay cost figures are considered to include all costs borne by the victims:

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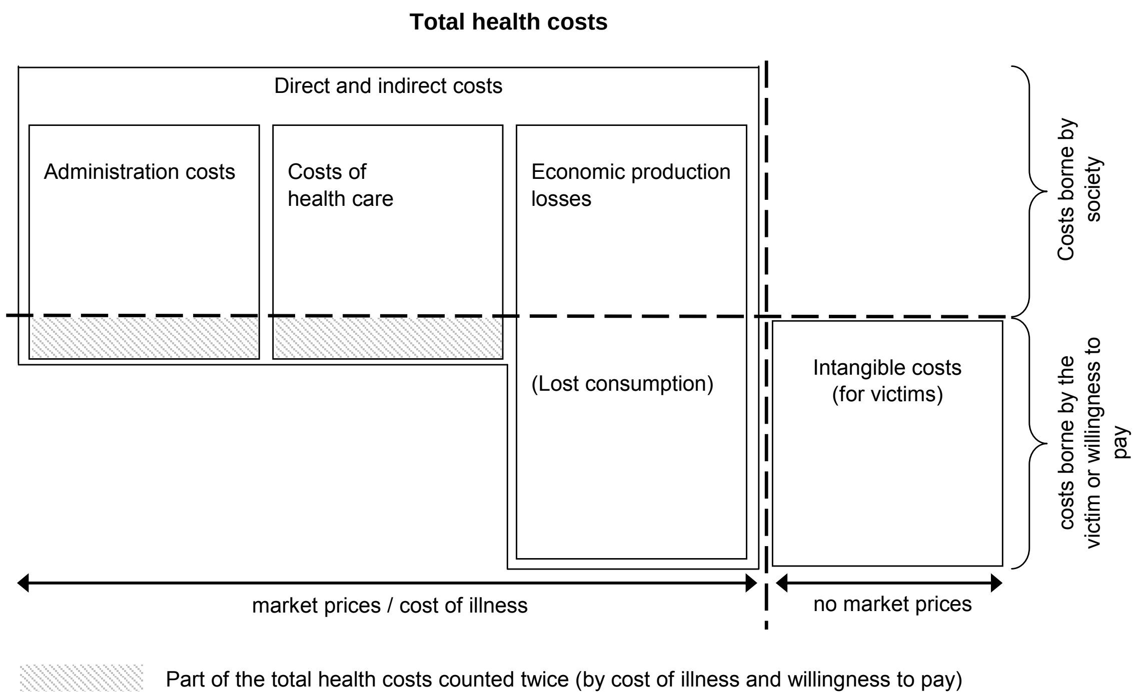
1 The share of cost components borne by the society or by the victims, respectively, depends on the legal and insurance system of a particular country. For example, this illustration assumed that the health insurance system includes components of social redistribution and not only covers individual risks.

2 An alternative method for measuring intangible costs is the willingness-to-accept approach. Similar to the willingness-to-pay approach, people are asked to express their willingness to accept to forego an improvement of a certain risk in monetary terms (for example the risk of fatality). However, the two methods differ in two important aspects: willingness-to-pay estimates an improvement to the current situation whereas the willingness-to-accept approach estimates the consent

intangible costs, lost consumption (the victim's share of economic production losses) and parts of the administration costs and costs of health care.

Theoretically, this applies to both adults and children. However, for practical reasons, no method is yet available for assessing willingness to pay among children (THE PEP, 2004a).

Fig. 4. Economic valuation of total health costs by the cost of illness and the willingness to pay



To estimate total health costs, the cost of illness and willingness to pay can be combined. However, summing up the values from both methods would overestimate total health costs because lost consumption and parts of the costs of health care and of the administration costs borne by the victims would be counted twice. However, each method alone considerably underestimates the real total health costs by the intangible costs or all costs borne by society, respectively. To overcome this dilemma between over- and underestimation, in the cost-of-illness method the net economic production losses can be calculated instead of gross economic production losses. This means that the lost (future) consumption is subtracted from the total economic production losses and is therefore excluded from the cost-of-illness valuation. Further double counting could arise from the costs of health care and administration costs if the costs borne by individuals are included in the cost-of-illness estimation (dotted area in Fig. 4). However, in many countries (such as

to the current risk situation. Further, willingness to pay binds respondents to their budget restriction (they cannot pay more money than they actually have), whereas there is no such limitation for willingness to accept. Hence, the values of willingness-to-accept measurements tend to be considerably higher than the ones derived with the willingness-to-pay method. As research commonly treats estimations with caution and therefore prefers conservative estimations ("at least"), using willingness-to-pay cost figures rather than willingness-to-accept cost figures has become established practice.

Switzerland), the cost-of-illness estimation does not include the costs borne by individuals and therefore no double counting occurs. Hence, this approach is the most precise way for cost calculations, as explained further below. Even in this case, a few of the costs of health care and of the administration costs are included in both the cost of illness and the willingness to pay and, hence, will be counted twice (the dotted area in Fig. 4). Nevertheless, this approach is still the most precise method for calculating costs, as explained further below.

Accuracy of combining the willingness to pay and the cost of illness

Some authors argue that the willingness-to-pay method covers all the costs of a health effect and therefore does not need to be complemented by the cost-of-illness method (Tolley et al., 1994). They claim that people expressing their willingness to pay for a certain health service (such as curing cancer) include in the cost figures all the benefits from this health service and therefore also health care expenditure and the costs of pharmaceuticals, productivity losses and administration costs. If this were the case, the combination of the willingness-to-pay method and the cost-of-illness method would considerably overestimate the total health costs. However, this argument does not contradict the suggestions made above. As explained, the two methods cover some of the same cost components (the area below the horizontal dashed line and to the left of the vertical dashed line), which is why lost consumption should be excluded from cost-of-illness measures. But as people express by their willingness to pay only the direct personal benefits from a health service, all the costs borne by society are not covered (the part above the horizontal dashed line). Only the cost-of-illness method can account for these costs. Hence, as long as society bears some health costs, the willingness-to-pay method alone does not fully cover the total health costs. Nevertheless, the more an individual (victim) bears the costs of health care and administration costs, the more the two methods overlap.

3.1.1.3 Internal versus external costs

Total health costs can be further differentiated in terms of external or internal costs. External costs are costs that are not borne by the person causing these costs but a third party or the society at large. For example, society or taxpayers (public spending for the health system, economic production losses and insurance premiums) and the victim (intangible costs and lost consumption) mainly bear the costs arising from health effects due to transport-related air pollution. The people who cause these health costs, the drivers, do not bear these costs except for their share of the overall taxes.[3] In contrast, the people who causes internal costs fully bear them.

3.1.2 Aim and scope of the proposed practical approaches

Transport is a crucial factor for any economy and contributes essentially to everyone’s wealth. However, as discussed in the previous chapters it also causes large costs by, for example, leading to road crashes, polluting the air, emitting noise and affecting opportunities to be physically active. Reliably quantifying

3 Whether the society and/or the victim has to pay (external) costs depends on how the health system is funded and whether a mechanism exists to charge the people who cause health effects for these costs.

these negative effects of road transport requires following a clear method and considering specific elements. Relying on the results of the literature review presented in Annex 1, practical approaches were developed on how to calculate health costs due to road crashes (section 3.2), noise (section 3.3) and air pollution (section 3.4). European Commission (2007) provides another example of practical guidance on how to measure transport-related health effects. Considerations on possible approaches to valuating health costs due to insufficient physical inactivity (and the key issues that need to be resolved) follow in section 3.5, as the currently available evidence does not yet allow a generic model for calculating economic costs to be developed.

The approach for road crashes, air pollution and noise is the same. Apart from some exceptions, the approach follows the same steps and needs the same input data. This basic structure of the underlying model is therefore explained first (section 3.1.3). Section 3.1.4 discusses the uncertainties in such cost calculations. Afterwards, the specific details for of the different health effects are presented (sections 3.2, 3.3, 3.4 and 3.5). In addition, the necessary input data and their sources are described. The elements of the estimation that contribute most to total health costs and are therefore of greatest importance in a study are also explained.[4] The last part of each section focuses on the situation of children.

This approach allows the total health costs due to road transport to be estimated for an entire country or region by calculating the total health costs of each effect and summing them up (see Chapter 4). Hence this approach is also useful for assessing the health effects in cost–benefit calculations for a specific infrastructure project. As Table 2 shows, health costs are one component of the elements of the costs of road crashes or environmental costs, respectively. Each of these health effects can also be calculated separately, allowing a focus on the costs of a specific health effect, such as the total noise costs due to road traffic in a country or the reduction of road crash–related costs because of improved road safety. The other three main elements of economic valuation are: the costs of investment in infrastructure, maintenance and the operating costs of infrastructures and effects on productivity and economic growth.

4 This information is useful if a survey is only intended to provide a first rough estimate of the dimensions of the costs by limited input of resources. However, in case of broader and more detailed cost valuation, other elements might be more important (that is, will influence the final results most) and, hence, need more resources, such as exposure assessment, concentration–response functions or evaluation of the value of a life-year saved and other economic cost figures (Sanderson & Hurley, 2005).

Table 2: Elements of economic valuation including health effects[5] (in grey)

Costs of investment in infrastructure	
	Planning costs
	Construction costs
	Costs for land
Maintenance and operating costs of infrastructure	
Costs for transport (change to current situation)	
	Time costs and congestion
	Fuel costs
	Parking costs
Costs of road crashes (change to current situation)	
	Material damages
	Health costs
Environmental costs, such as those due to noise or air pollution (change to current situation)	
	Health costs
	Building damages
	Costs due to climate change
	Pollution of agricultural and horticultural production
	Soil and water pollution
	Nuclear risks
Effects on productivity and economic growth (change to current situation)	

The practical approach presented can be also be used to calculate average costs (total costs relative to the volume of road traffic). The practical approach can also calculate marginal costs, which are needed for projects on the internalization of external costs (polluters have to pay the costs they caused by emitting certain pollutants, for example through tolls). However, for noise and road crashes, the relationship between the volume of road traffic and health costs is not linear (as it is for air pollution, which means that the marginal costs are equal to the average costs). Hence, this relationship has to be determined first or a realistic assumption has to be made.

The basic structure of calculating total health costs due to transport-related emissions presented in Fig. 5 cannot be applied to every country or region without some adjustment. Due to country- or region-specific constellations and due to restrictions in data availability, this model might have to be modified, as explained in the specific models presented in sections 3.1, 3.2 and 3.3. .

3.1.3 Basic structure of the proposed approaches

All three topics (road crashes, transport-related air pollution and noise) follow the same basic structure as shown in Fig. 5, consisting of four steps.

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5 Even though all elements of the table are explicitly or implicitly named “costs”, this does not mean at all that, in a cost–benefit analysis, these elements always have to be negative. Imagine, for example, a cost–benefit analysis of a road project that leads to a reduction in road crashes. In this case, the change in the costs of road crashes is actually a benefit, as these costs decrease due to the project. Hence, whether an element of the analysis is a cost or a benefit depends on the specific case.

Step 1. As a first step, the traffic characteristics for each mode of transport considered and all types of vehicles involved need to be determined. As the respective exposure situation strongly depends on the traffic characteristics (such as the volumes of traffic, the share of diesel vehicles and the quality of the transport infrastructure), this is a necessary first step in valuation. In this step, it also has to be decided which mode of transport should be considered in calculating health costs (such as road and/or rail transport). This decision depends on the aim of the study and on data availability and quality.

Step 2. Then, for air pollution and noise, total emissions can be calculated using information on the emissions of each mode of transport and type of vehicle. Applying dispersion models and using demographic data, the number of people exposed to a certain pollutant or level of noise can then be estimated. For road crashes, the number of people exposed (victims) is derived directly from the statistical data sources (see section 3.2.3 and Annex section I.1.2). For noise, an alternative way of estimating exposure is based on the number of homes exposed to harmful noise levels.

Step 3. The health effects due to this exposure are estimated by using exposure–response functions for the exposed part of the population. With this function, the population-attributable fraction (PAF)[6] can be calculated: the proportion of a health effect attributable to a certain exposure. There are two methods of calculating the number of cases attributable to this exposure. One method is based on life tables. This approach follows up a study population over time into the future (taking into account the probability of each age band dying), comparing a baseline scenario with scenario where the exposure changes (such as assuming no transport-related air pollution or noise exposure) (Hurley et al., 2005). The life-table method is based on a matrix defined simultaneously by (1) calendar years into the future and (2) the age distribution of the study population. The effect of a specific exposure on health is given by the differences between the two matrices (between the exposure-changed scenario and the baseline). Ecoplan & Infras (2008) and Ecoplan et al. (2004a) used this method. The examples presented below were also calculated using this method.

The other method uses a static approach, as follows:
Annual death rate × Study population size × % increase in health effect per increase in exposure (such as the % change in number of deaths associated with a 10 $\mu g/m^3$ change in $PM_{2.5}$ concentration) × Change in exposure (such as $PM_{2.5}$ concentration).

Künzli et al. (2000) and the CAFÉ cost–benefit analysis study (Hurley et al., 2005) used this method.

For some health end-points, the available evidence is not sufficient enough to firmly recommend inclusion, but they can be included if the aim is a more comprehensive rather than a conservative estimate of transport-related health costs and the larger uncertainty related to the other morbidity end-points is accepted and clearly acknowledged. These are specified in the respective subchapters.

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6 PAF = Pe(RR – 1)/(1 + (Pe(RR – 1)))
where Pe is the proportion of the population exposed to the risk factor, and RR is the relative risk of the health effect associated with the exposure in question.

Step 4. This step consists of valuating these effects by applying economic cost figures. As explained in section 3.1.1.1, health costs comprise direct and indirect costs and intangible costs (of the victim).[7] In the first category, the following elements should be considered: costs of health care, economic production losses and administration costs. It is suggested to use factor prices that exclude indirect taxes (such as the value-added tax, petrol taxes and other subsidies) rather than market prices, since these indirect taxes vary strongly between countries. Using market prices that include indirect taxes therefore makes comparing the results from different countries difficult. Since willingness-to-pay figures express market prices, they have to be converted into factor prices.[8] As the goal is to determine total health costs, all these cost components must be directly linked to a health effect. Non-health-related costs such as material damage and indirect costs such as costs due to travel delays, congestion and risk avoidance are not included here.

These two cost components, direct and indirect costs and intangible costs, are best valuated by using the cost-of-illness method in combination with the willingness-to-pay method including only net economic production losses (that is, without the value of lost consumption, because otherwise production losses will be counted twice), as explained in section 3.1.1. Both of these components strongly depend on gross domestic product per capita, income elasticity and sociocultural determinants (IER, 2006; Navrud, 2002; Sommer et al., 1999; THE PEP, 2004a) and on the organization of the health system in a country. Therefore, no uniform values can be proposed for use throughout Europe, as they can vary considerably from country to country.

The approach of the value of life-years lost should be used for the economic valuation of mortality as it takes the life expectancy of a victim into account.[9] This is of particular importance if children are to be included in a study. However, values of statistical life can be used, as they can be easily transformed into values of life-years lost by applying a discounting factor and the average of the statistical life expectation (see Box 1).

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7 As in all texts reviewed, intangible costs for relatives and friends are not considered here (see section I.1.3.5). As it can currently not be excluded that individuals' willingness to pay also covers to a certain extent intangible costs for relatives and friends, including it separately might lead to double-counting.

8 For example, in Switzerland, the average indirect tax rate is 7.7% (Ecoplan, 2002a). So the willingness-to-pay-figures have to be divided by 1.077 to derive factor prices. For more information on the exact procedure of the conversion and the necessary data, see IER (2006).

9 So far, the available evidence does not allow a final conclusion as to whether the relationship between the value of life-years lost and age is linear or non-linear; empirical evidence suggests that the relationship is complex (Ecoplan et al. 2004a). For the sake of simplicity and due to lack of clear evidence suggesting otherwise, it is recommended to assume a linear relationship, meaning that the value of a life-year lost is the same in all age groups.

Box 1. Converting values of statistical life into values of life-years lost

The following equation can be used to convert values of statistical life (VOSL) into the values of life-years lost (VLYL) (Ecoplan et al., 2004a):

$$VOSL = VLYL * \sum_{i=a}^{T} \frac{{}_{a}P_{i}}{(1+r)^{i-a}}$$

With T = the maximum age a person can reach (such as 110 years)

a = age of the person for whom the value of a statistical life was estimated (such as 40 years)

${}_{a}P_{i}$ = probability that a person with the age *a* will reach the age *i* in country A

r = discount rate (such as 2%).

Thus, the value of a statistical life is equal to the sum of all life-years of the future, discounted towards today's consumption and with the probability of becoming a year older (decreasing over age).

A growth in the real wage rate also leads to an increase in the willingness to pay for the value of life-years lost. Hence, depending on the expected growth rate of real wages, the discount rate has to be reduced according to the same number (for example, a discount rate of 2% and a growth rate of real wages of 1% gives a corrected discount rate of 1%).

However, these values might not be available for all countries. In this case, either a specific survey has to be carried out or, if resources are limited, it can be derived from existing valuation figures from another country. When valuation figures from other country are being used, considering the previously mentioned differences between the countries is important.

If neither local values nor values from a similar country are available, the values from European studies in Table 3 can be considered. However, the value of a life-year lost must be adjusted for the purchasing power parity (PPP) of the country under consideration (IER, 2006), and the results have to be interpreted and communicated with considerable caution.

Table 3. Examples of the value life years lost used in past analyses

Source	Year and method of pricing	Value of life years lost	Comments
Holland et al. (1998)	1995, market prices	€84 000–112 000 €141 000–169 000 €98 000 €84 000	0% discount rate, short-term effects 3% discount rate, short-term effects 0% discount rate, long-term effects 3% discount rate, long-term effects
Nellthorp et al., 2001	1998, market prices	€47 500[a] €75 000[a] €95 000[a] €150 000[a]	0% discount rate, accident context 3% discount rate, accident context 0% discount rate, environmental context 3% discount rate, environmental context All values are based on a value of a statistical life of €1.5 million[b]
IER et al., 2004	2002, market prices	€50 000[c] €125 000[c]	Median, not rounded €55 800 Mean The median is recommended for use based on results for the willingness to pay for reducing the mortality risk by 5 in 1000, over 10 years, in three European countries (France, Italy and the United Kingdom)
IER (2006)	2002, factor prices	€60 500 €40 300	Acute mortality Chronic mortality (based on IER (2006), median value) These values are valid in the environmental context and correspond to a value of a statistical life of about €1.0 million.[d] For road crashes, a value of a statistical life of €1.5 million (1998 market prices) is used[b].
United States Environmental Protection Agency (2006)	1999 prices for 1990, market prices		US$ 5.5 million for the value of a statistical life (no value of a life-year lost given). However, this value is based on meta-analysis of wage-risk studies of the value of a statistical life, which can lead to different values compared to stated preference studies for road crashes[a,b].

[a]In UNITE, only the higher values for the environmental context are given as the costs of road crashes are calculated directly from the value of a statistical life. In the environmental context, UNITE doubles the value of a life-year lost derived from road crashes because evidence suggests that the willingness to pay for reducing environmental mortality risks is higher than for road crash risks.

[b]Many recent projects have assigned €1.5 million as the value of a statistical life: EU: HEATCO (IER et al., 2006), GRACE (Lindberg, 2006), IMPACT (Maibach et al., 2008); five studies in Switzerland (on road crash and external costs); and one study in Belgium (De Brabander & Vereeck, 2007). Thus, the EU project GRACE concluded that a growing consensus on the method to estimate the value of a statistical life seems to be emerging.

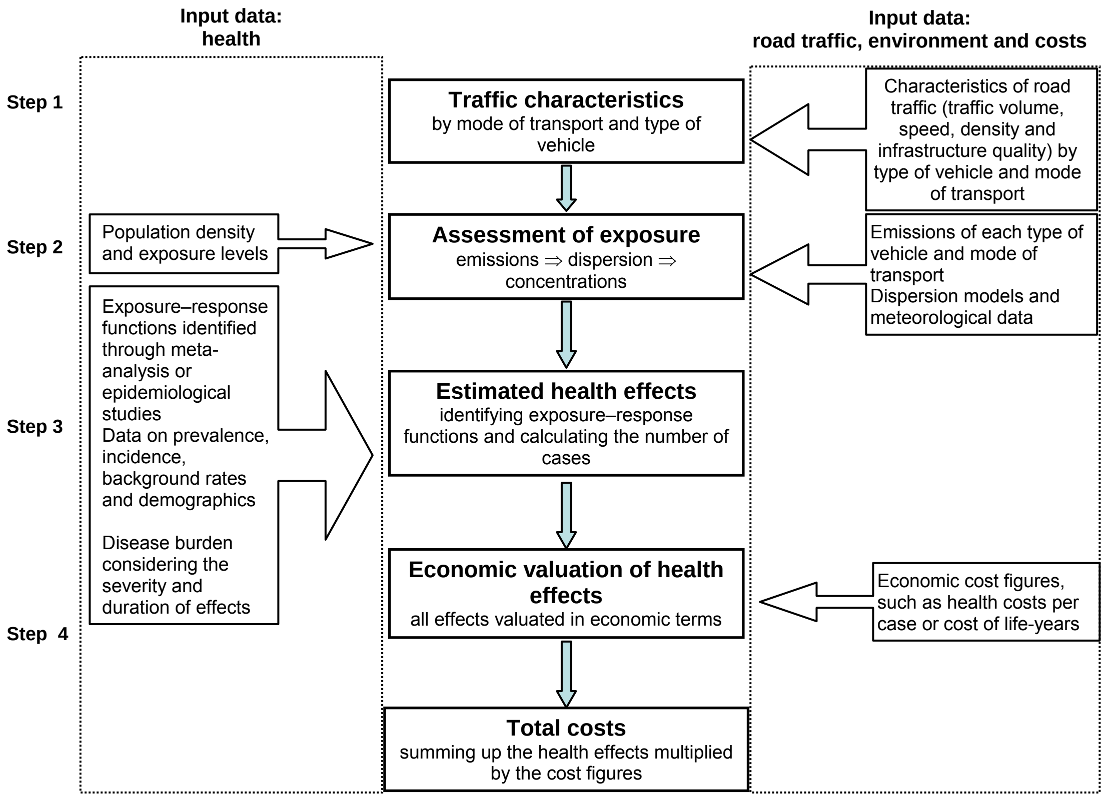
[c]These values are also used in the project CAFE (Clean Air for Europe): IER et al. (2004).

[d]These values are also used in Maibach et al. (2008).

Based on the number of people with certain health effects (or exposed homes in the case of noise) and the cost of each of these effects, the total health costs[10] can be calculated by multiplying the two and summing up the results obtained for each type of effect.[11]

[10] The term “total health costs” refers in the whole text to the sum of all different costs of a health effect considered in this study. As this study does not consider some costs of a health effect, for example, because they cannot yet be monetized, the term “total health costs” as used here does not cover all possible total health costs that occur in reality.

Fig. 5. General model for the valuation of transport-related health effects



3.1.4 Uncertainty

The following discussion relates to calculating the costs of the health effects of road crashes, air pollution and noise. However, the fundamental ideas on uncertainty and most of the types of uncertainty presented are also true for estimating total health costs due to insufficient physical activity.

3.1.4.1 General considerations on uncertainty in calculating costs

Any estimation of total health costs has to deal with the problem of uncertainty regarding results representing reality. Major sources of uncertainty are excessive simplification or shortcomings of the models used, incorrect assumptions, lack of data, errors in the data and failures in statistical and mathematical procedures and data processing (Ecoplan & Infras, 2008; Sanderson & Hurley, 2005). Uncertainty is the result of either incomplete knowledge on the situation in question or human error. Uncertainty is involved in all steps of the practical approach and input data needed (Ecoplan & Infras, 2008).

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[11] An immediate reduction in the noise level or in the level of air pollution does not lead to an immediate decrease in health effects because there is usually a time lag between exposure and health effects. Instead, the effect of reduced exposure will only be monitored over a longer time period (Ecoplan et al., 2004a; Sanderson & Hurley, 2005).

Although applying a sound quality assurance system might avert most uncertainty due to human error, uncertainty due to incomplete knowledge cannot be avoided. Hence, discussing these types of uncertainty in all steps of a cost calculation is crucial (Sanderson & Hurley, 2005) as is assessing, for example with sensitivity analysis,[12] their possible effects on the final results (AEA Technology Environment, 2005a; IER, 2005). Further, applying a conservative ("at least") approach is best. Such an approach will not make the results more reliable but far more transparent.

Some of the most important types of uncertainty in estimating total health costs are presented here. For this task, all 38 studies of the literature review (section 2.3) were analysed for how they handled uncertainty.[13]

First, the types of uncertainty that are relevant to all health effects are discussed. Afterwards, the types of uncertainty that are specific to one of the health effects are outlined (sections 3.1.4.3, 3.1.4.4 and 3.1.4.5). However, depending on the data, the aim of the study and other specific conditions, additional types of uncertainty might occur and different types of uncertainty might be most crucial.

3.1.4.2 Most significant types of uncertainty in calculating costs

In the calculation of costs due to health effects, an important source of uncertainty are the values of willingness to pay in general and the values of life-years lost in particular (Ecoplan, 2002a; Ecoplan & Infras, 2008; Lindberg, 2000; McCubbin & Delucchi, 1999; United States Environmental Protection Agency, 2000). The reason for this is twofold. First, these values are based on interviews with people reporting what they are willing to pay to avoid a specific risk (see section 3.1.1). As studies have shown, the results of these interviews (willingness-to-pay values) vary greatly and are sensitive to how the interviews are carried out (Ecoplan, 2002a; Ecoplan & Infras, 2008; Lindberg, 2000; McCubbin & Delucchi, 1999; United States Environmental Protection Agency, 2000). Further, these values might depend on additional variables such as income and age (Chanel & Luchini, 2008; Department for Environment, Food and Rural Affairs, 2004; United States Environmental Protection Agency, 1999; Vassandumrongdee & Shunji, 2005) and probably differ between some health effects (Chanel & Luchini, 2008; Department for Environment, Food and Rural Affairs, 2004; Ecoplan et al., 2004b; Vassandumrongdee & Shunji, 2005), especially between sudden death e.g. from a road crash and a premature death from a long-term exposure. Hence, the extent to which willingness-to-pay measurements represent the true values is uncertain. Second, for all three health effects (road crashes, air pollution and noise), intangible costs account for the biggest share of total health costs (see sections 3.2.4, 3.3.3 and 3.4.4). Uncertainty in the willingness-to-pay values therefore substantially affects total health costs.

The cost-of-illness approach also involves some uncertainty (Ecoplan, 2002a; Ecoplan & Infras, 2008), but this is less significant than that of willingness-to-pay measurements. This relates to proper definitions and accurate data (Ecoplan, 2002a; Ecoplan & Infras, 2008). In addition, as already explained in section 3.1.1,

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12 Probably the most accurate (but also very complex) instrument to do this is Monte Carlo analysis (see Ecoplan & Infras, 2008). Further possible instruments to estimate the effects of uncertainties are comparing one's own results with those of similar surveys (Ecoplan et al., 2004a), the use of different measurement methods (Sanderson & Hurley, 2005) and calculating the variance of the results (Sanderson & Hurley, 2005).

13 Due to many references for some points of the following discussion, selected authors are cited.

how much the two approaches (cost of illness and willingness to pay) overlap and hence how much the results might be overestimated are uncertain.

There is further uncertainty about the health effects of air pollution and noise (for further details, especially for single pollutants and health effects, see the discussion in Annex 1, sections I.1.2 and I.2.2). The only health effects considered in calculating costs are those for which sufficient scientific evidence on a relationship with exposure exists (see Annex 1, sections I.2.2 and I.3.3). Several health effects due to air pollution and noise are therefore probably not taken into account, and the results are likely to underestimate the total health costs. Even if there is sufficient evidence on the association between a health effect and a specific type of exposure, there still might be some doubt about causality (Ecoplan et al., 2004a; IER, 2005; United States Environmental Protection Agency, 1999, 2000; Van Essen et al., 2007) and the impact of time lags (United States Environmental Protection Agency, 2000).[14] Further, many types of uncertainty exist on the exact shape and slope of dose–response functions (such as linear or exponential) (Ecoplan et al., 2004a; IER, 2005; United States Environmental Protection Agency, 1999, 2000; Van Essen et al., 2007), whether there is a threshold for effects (Ecoplan et al., 2004a; IER, 2005; United States Environmental Protection Agency, 1999, 2000; Van Essen et al., 2007) and whether they can be applied to different geographical areas (such as a city, mountain area or shore) without any adaptation (IER, 1997).

Additional uncertainty is related to integrating the health effects on children into the calculation of total health costs. As discussed in more detail in the literature review (see Annex 1), research results on children are lacking in many cases, and they often cannot be treated differently from adults. This is especially true in terms of specific health effects (THE PEP, 2004a) and willingness-to-pay values (AEA Technology Environment, 2005b).

The choice of the discount rate used for valuating future costs and benefits is of less importance but still significant, as even a difference of just 1 percentage point changes the overall result of total health costs (Ecoplan et al., 2004b).

3.1.4.3 Specific types of uncertainty regarding health costs due to road crashes

Road crashes mainly pose one additional specific type of uncertainty: in most countries the number of cases not reported in official statistics is not known or can only be roughly estimated (Ecoplan, 2002a; see also section 3.2.2.2. Hence, this number is highly uncertain and affects the final result to a certain degree. Serious injuries and fatalities would have the strongest influence (see section 3.1.4.2), but the underreporting is likely to be highest on less serious injuries with a lower influence on health costs.

3.1.4.4 Specific types of uncertainty regarding health costs due to air pollution

There are two additional areas of uncertainty in calculating total health costs due to air pollution: the choice of the air pollutants and the estimation of the number of exposed people.

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14 An immediate reduction in the noise level or in the level of air pollution does not lead to an immediate decrease in health effects because there is usually a time lag between exposure and health effects. Instead, the effect of reduced exposure will only be monitored over a longer time period (Ecoplan et al., 2004a; Sanderson & Hurley, 2005).

As explained in section 3.4.2 and in more detail in Annex 1, section I.3.2, only a few proxy pollutants are considered in calculating costs instead of the entire mixture of air pollutants. Hence, there is uncertainty about the extent to which this approach reflects the real exposure or whether a more complex model should be used (Sanderson & Hurley, 2005). The exclusion of most pollutants also means that the cost calculation probably does not consider certain health effects (Ecoplan et al., 2004a). Further, the extent to which some pollutants or effects of a different mix of pollutants on human health correlate with each other is not always known (IER, 2005). Hence, the choice of the air pollutants is crucial.

The uncertainties in terms of choosing the right pollutants also relate to primary and secondary pollutants, as the relationships between them are also somewhat unclear (McCubbin & Delucchi, 1999). This is reinforced by the fact that, in the transformation and dispersion models, the associations between emissions and concentrations are probably often oversimplified (Ecoplan & Infras, 2008; Holland & Watkiss, 2002; United States Environmental Protection Agency, 1999). In addition, uncertainty arises in connection with the inclusion of weather data in these models (such as related to the reliability of the model, area dependence and lack of data) and with the possible effects of unknown interactions between pollutants (Ecoplan & Infras, 2008; United States Environmental Protection Agency, 1999) (see section 3.1.4.2). All these types of uncertainty affect the estimation of the total number of exposed people. In addition, the exact proportion of total air pollution that is transport-related air pollution is difficult to define (DIW et al., 2000).

3.1.4.5 Specific types of uncertainty regarding health costs due to noise

Similar to the problems connected to the transformation and dispersion models of air pollution (see section 3.1.4.4), there is also some uncertainty in the dispersion models of noise. Besides the general question of whether the models are reliable enough, there is the specific difficulty that noise levels depend on local weather and geographical circumstances, and most models only partly reflect this (THE PEP, 2004a). Hence, estimating the number of people exposed to noise is difficult for calculating the health costs due to noise.

Further uncertainty arises from the choice of the threshold for noise levels. There is no clear guidance on which level to choose as threshold (such as 50 or 55 dB(A)), although this choice strongly influences the results of the total health effects due to noise (Ecoplan et al., 2004b; Van Essen et al., 2007).

In addition, even though the correlations between noise and the reduction in rent and the causality of the effect are proven, the strength of this relationship is uncertain (Ecoplan & Infras, 2008).

3.1.5 Conclusions

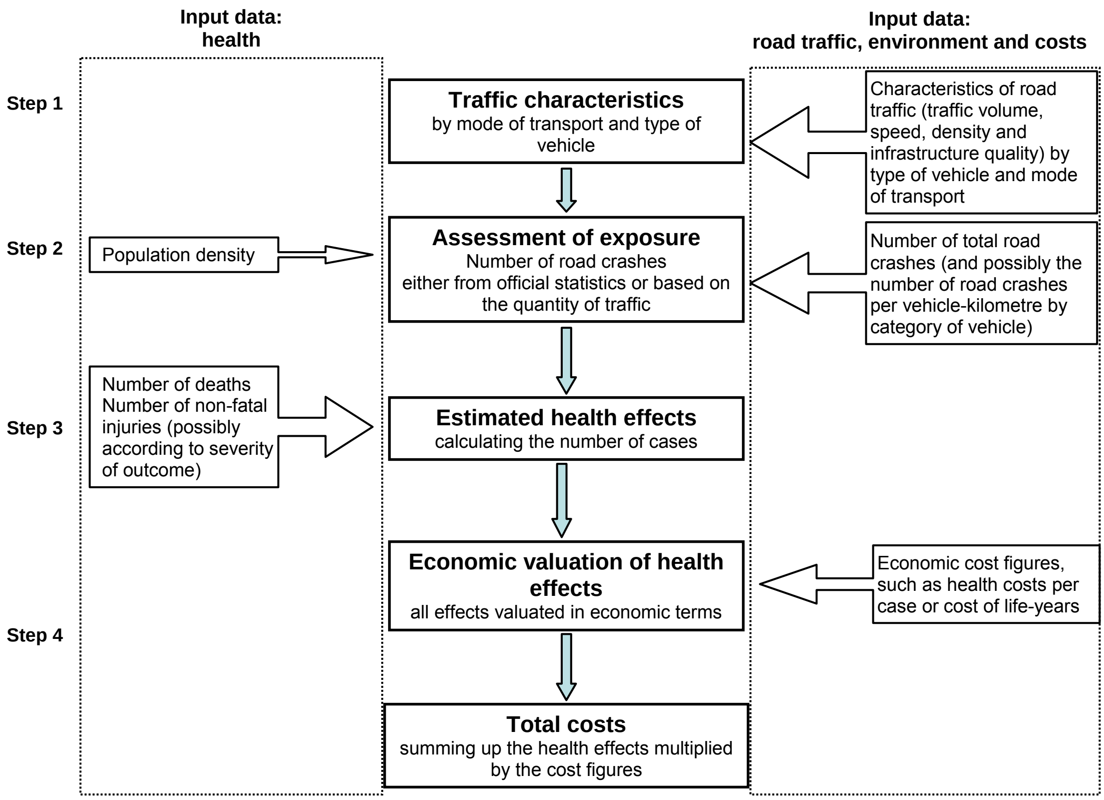
As the sections above show, several types of uncertainty exist in the estimation of total health costs. However, the results of such calculations show the dimension of these costs and sensitivity analysis can show the possible range of results and their probability. For example, a web tool developed within the GRACE project (Ricci et al., 2008) allows to estimate the marginal social costs of transport for a given section of the European transport network and to carry out sensitivity analyses on all major variables entering into the calculation, including road and vehicle type and emission and damage factors.

3.2 A practical approach for valuating health costs due to road crashes

3.2.1 The basic model

Following the general approach outlined in section 3.1, the steps to measure the costs of road crashes (defined in Annex 1, section I.1.1) are as follows (Fig. 6). First, the traffic characteristics for each mode of transport considered and by all types of vehicle involved need to be identified. Based on the number of road crashes per vehicle-km for each mode of transport and type of vehicle, the total number of road crashes and hence people involved in road crashes (victims) can be calculated. If reliable official data on total road crashes and victims exist for the time period in question, the first two steps can be left out (see section 3.2.1). In the next step, the health effects due to road crashes are summed up for all victims from the road crashes considered. Then economic cost figures are applied to these effects. Finally, total health costs[15] are calculated by multiplying the number of road crashes by the economic values.

Fig. 6. Steps to apply in economic valuation of the costs of road crashes



15 The term "total health costs" refers in the whole text to the sum of all different costs of a health effect considered in this study. As this study does not consider some costs of a health effect, for example, because they cannot yet be monetized, the term "total health costs" as used here does not cover all possible total health costs that occur in reality.

3.2.2 Steps for road crashes

3.2.2.1 Step 1: road traffic characteristics

If reliable official data on the number of road crashes are available, step 1 can be skipped. Otherwise road traffic characteristics have to be estimated for each mode of transport and for each type of vehicle.

3.2.2.2 Step 2: the affected people

If no reliable official data on the number of road crashes are available, the ratio of the number of road crashes to vehicle-km for each type of vehicle and mode of transport has to be estimated. If the number of victims can be taken from official statistics, no further estimation is needed in this step.

In this step, it has to be decided which kind of road crashes to consider in calculating total health costs. As the literature review shows (see Annex 1, section I.1.3.2), single-vehicle crashes and cases not reported in official statistics are usually included whereas suicides, road crashes involving only pedestrians (such as pedestrians falling on roads) and road crashes due to construction are excluded. In the case of road crashes with only cyclists, the literature review does not lead to a clear conclusion. However, this choice also depends on the aim of the study and on the availability of reliable data.

To include the cases not reported in official statistics in the cost calculation, this number (a correction factor) has to be estimated through a specific survey if possible. Underreporting is usually less of a problem for serious health effects and casualties but can have a larger effect for slight injuries (Ecoplan, 2002b; IER, 2006). Such estimations exist already for some countries and regions (for example, for Europe: IER, 2006; for Switzerland: Ecoplan, 2002b), but for most countries this is not the case. If no country-specific data are available, using the numbers from a similar country can be a second-best solution, but possible differences have to be considered due to differences in legal systems and in the regulations on reporting road crashes.

3.2.2.3 Step 3: health effects

Based on the epidemiological studies reviewed (see Annex 1, section I.1.2), two health effects due to road crashes should be differentiated:

- non-fatal injuries
- fatalities.

Depending on the availability and quality of data, non-fatal injuries can further be subdivided into slight injuries and serious injuries, allowing total health costs to be estimated more precisely. However, the definition of “slight” and “serious” can differ considerably between countries, which limits the comparability of cost values across countries.

Road crashes leading only to material damage, a further effect due to road crashes mentioned in some of the economic texts reviewed (see Annex 1, section I.1.3.4), are not considered here, as only total health costs are taken into account and not total costs.

3.2.2.4 Step 4: economic valuation

Valuating fatalities requires taking the victims' age into account (see also section 3.2.5).

3.2.3 Input data and data sources

Table 4 summarizes all input data necessary for estimating total health costs due to road crashes. The first column lists the best possible data (state of the art). The second column lists a second-best approach if some data might not be available; this sometimes means applying data from other countries, which might cause additional uncertainty. However, in most cases data has to be surveyed in the country or region of interest itself. The number of health effects needs to be available in any case.

Table 4. Input data for estimating the costs of road crashes and the second-best approach

State of the art	Second best
Number of total road crashes (including the age of each victim)	Road traffic characteristics by each type of vehicle for each mode of transport Number of road crashes per vehicle-km for each type of vehicle in each mode of transport (including the age of each victim)
Cases not reported in official statistics	Numbers from a similar country
Number of each health effect for total road crashes	–
Economic cost figures: willingness to pay, cost of illness and value of life-years lost	Cost figures from a similar country

3.2.4 Elements contributing most to total health costs[16]

The element contributing most to the health costs of road crashes is serious injuries, followed by minor injuries (Sommer et al., 2007). However, the intangible costs in any case account for the biggest share of total health costs (Sommer et al., 2007).

3.2.5 Special focus on children

As explained in Annex 1, section I.1.3.9, two of the reviewed studies included children in calculating the costs of health effects due to road crashes. In these studies, the value of life-years lost for adult was applied to children by taking their longer life expectancy into account, and specific information on the number of health effects in children was taken from official statistics (road crashes according to severity and age). Based on these two elements, costs due to road crashes affecting children are relatively easy to calculate.

16 The information in this chapter refers to Federal Office for Spatial Development (2007) and therefore reflects the situation in Switzerland. However, as comparisons with other studies and countries have shown, the main findings are similar in most cases. Nevertheless, depending on the road traffic situation, on the judicial system and laws and on the types of vehicles, the statements need to be viewed with caution.

3.2.6 A practical example for estimating the health costs of road crashes

This section applies the practical approach to calculate the health costs in Switzerland related to road crashes in 2005. The calculations in the example are based on input data derived from a study by Ecoplan & Infras (2008).

In 2005, Switzerland had 7.4 million inhabitants. Table 5 shows that slightly more than 60 billion vehicle-km were driven in 2005, resulting in 93 billion passenger-km and 15.8 million tonne-km.

Table 5. Underlying transport data for Switzerland, 2005

	Passenger transport									Freight transport				
	Car	Public bus	Public electric bus	Tram	Private coach	Motorbike	Scooter	Bicycle	Total (excluding bicycle)	Delivery van	Heavy goods vehicle	Articu-lated lorry	Tractor, machine	Total
Millions of vehicle-km	52 080	229	30	41	106	2 059	146	1 957	54 691	3 301	1 422	705	NA	5 428
Millions of passenger- km	83 348	2 780	722	1 458	2 235	2 222	146	1 957	92 910					
Millions of tonne-km										893	7 728	7 214	NA	15 835

NA: not available

Source: Ecoplan & Infras (2008).

Following the practical approach explained above, the example does not consider road crashes leading only to material damage, as it focuses on illustrating the calculation of total health costs. Similarly, police and legal costs were excluded here. Ecoplan & Infras (2008) derived an example of material damage as well as police and legal costs for Switzerland, where they amounted to US$ 1930 million and US$ 389 million, respectively.

3.2.6.1 Steps 1 and 2: transport characteristics and people affected

Step 1 of the practical approach introduced in section 3.2.2.1 on transport characteristics can be skipped for Switzerland, as the number of people affected (step 2) can be derived directly from the official data of the Swiss Federal Office of Statistics. These data are based on crashes reported to the police. Injuries and fatalities are distinguished, and the victims could be differentiated by the category of vehicle causing the road crash (for each accident the culpable party[17] has been determined). As shown in the second row of Table 6, 409 fatalities and 26 751 injured people (including children) were reported to the police. This number includes all transport crashes reported to the police except those caused by rail transport (which are not considered here as the focus is on road transport).

The Swiss Council for Accident Prevention has estimated the total number of crash victims, including those not reported to the police, to be 94 000: 3.5 times more than reported to the police (see first row of Table

[17] The culpable vehicle and the victim of a crash can belong to different vehicle categories (such as when a car driver injures a pedestrian). In most countries (including Switzerland), accident statistics use the monitoring principle: they show which vehicle category the victim used. In contrast, we show here which vehicle category caused the accident (although the victim might have used another vehicle category). This required additional information from the official data on the guilty party.

6).[18] These data are based on the evaluation of the data from the Central Statistical Office of Accident Insurance to which all accidents of the adult workforce are reported (Hubacher, 1994; Hubacher & Ewert, 1997).

Table 6. Number of casualties (fatal and non-fatal injuries) in Switzerland by vehicle category culpable for the crash, 2005

Type of casualty	Total	Vehicle category of the culpable party										
		Car	Private coach	Public bus	Delivery van	Heavy goods vehicle	Articulated lorry	Tractor, machine	Bicycle	Scooter	Motorbike	Pedestrian
Fatalities	415	226	14	1	16	21	7	9	27	10	54	29
Injuries	93 996	42 930	1 137	690	2 909	815	504	612	22 872	5 713	10 896	4 918
Fatalities	409	223	14	1	16	21	7	9	26	10	53	28
Injuries	26 751	17 711	148	131	1 009	320	196	169	2 116	1 192	2 540	1 218

Source: based on data from the Swiss Federal Office of Statistics and the Swiss Council for Accident Prevention.

3.2.6.2 Step 3: health effects

Table 6 shows that the necessary distinction between fatalities and non-fatal injuries in step 3 has already been made for Switzerland. However, another study also allows a further differentiation between four types of injury severity (Sommer et al., 2007): disability (the victim has lasting injuries and is entitled to disability pension), severe injury (hospital stay of at least seven days), moderate injury (hospital stay of one to six days and no lasting injury) and slight injury (outpatient treatment only). This split is also based on the data of the Central Statistical Office of Accident Insurance. The result is shown in Table 7. The large majority (88%) of all injured people are slightly injured, 4% moderately injured, 7% severely injured and 2% disabled. These percentages are used for all vehicle categories, as the available data on severity do not allow differentiation between the vehicle categories because the data are presented based on culpable party rather than based on the victims, but distinguishing between vehicle categories would be possible. However, this represents only an approximation and is likely to underestimate the severity of crashes occurring in certain vehicle categories, especially motorcycles.

[18] Also for fatalities there are a few unreported deaths, not because these crashes are not reported but because these victims die more than 30 days after the crash and are therefore not reported as victims of a road crash in the official statistics. For the distribution of the crashes according to culpable vehicle category, see Ecoplan & Infras (2008).

Table 7. Number of casualties in Switzerland by injury severity and culpable vehicle category, 2005

Injury severity	Total	Vehicle category of the culpable party: Car	Private coach	Public bus	Delivery van	Heavy goods vehicle	Articulated lorry	Tractor, machine	Bicycle	Scooter	Motorbike	Pedestrian
Fatalities	415	226	14	1	16	21	7	9	27	10	54	29
Disability cases	1 549	707	19	11	48	13	8	10	377	94	180	81
Severely injured	6 169	2 817	75	45	191	54	33	40	1 501	375	715	323
Moderately injured	3 669	1 676	44	27	114	32	20	24	893	223	425	192
Slightly injured	82 609	37 729	999	607	2 557	717	443	538	20 101	5 021	9 576	4 322

3.2.6.3 Step 4: Economic valuation

The standard approach was taken for economic valuation. To derive results in United States dollars, we have transformed the results in Swiss francs using the official exchange rate for 2005 of the Swiss Federal Office for Statistics (Sw.fr. 1.246 = US$ 1). All results are in factor (not market) prices (indirect taxes and subsidies are excluded, see section 3.1.3). Table 8 and Fig. 7 show the results by injury severity and health cost category in United States dollars. The following cost categories were included.

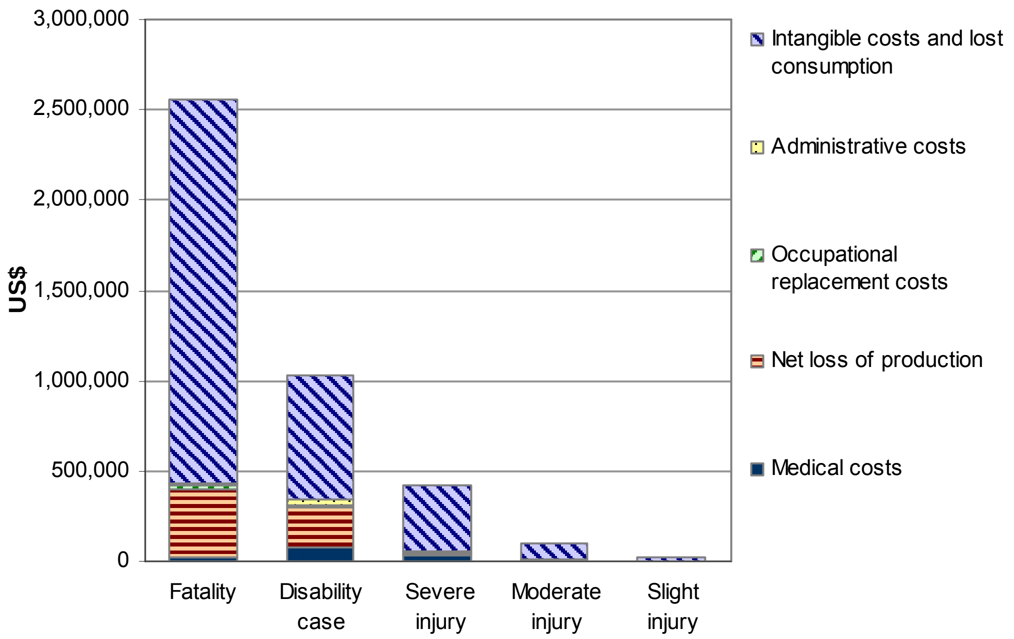
- Health care costs were based on the data from the Central Statistical Office of Accident Insurance.
- Net loss of production: the cost rates stemmed from official data of the Swiss Federal Office of Statistics (net loss per day), and the number of lost working days was based on data from the Central Statistical Office of Accident Insurance. For fatalities, the calculation applied the lost years of employment in the workforce (see below).
- Occupational replacement costs: after the death or disability of an employee, a successor has to be found, which causes costs of about 50% of one yearly wage of the employee.
- Administrative costs: these costs were based on data from the Central Statistical Office of Accident Insurance and official data on the ratio of administrative costs to total costs per insurance type (health insurance, accident insurance and disability insurance) in Switzerland.
- Intangible costs and lost consumption are based on the willingness-to-pay approach. The value of a statistical life is based on the value of €1.5 million proposed by the EU projects UNITE and HEATCO (Nellthorp et al., 2001), equivalent to about US$ 2.2 million. As proposed in section 3.1.1, the actual calculations are based on the value of life-years lost and the exact age of the victims (see below). The value of life-years lost is derived according to Box 1, which results in a value of US$ 74 200. For disabilities, 32% of the value for fatalities was used based on willingness-to-pay studies (Nellthorp et al., 2001). The literature also shows that the intangible costs of slight injury amount to 1% of the value of a statistical life (Nellthorp et al., 2001; Sommer et al., 2007). The values for intangible costs for moderately and severely injured people were derived from the value for slight injuries based on the number of lost workdays (from the Central Statistical Office of Accident Insurance data) for each category of injury severity.

Table 8. Health costs from road crashes in Switzerland in United States dollars per casualty by injury severity and health cost category

	Health care costs	Net loss of production	Occupational replacement costs	Administrative costs	Intangible costs and lost consumption	Total
Fatality	19 233	380 248	16 740	11 983	2 133 129	2 561 333
Disability case	81 356	216 989	16 198	30 159	681 818	1 026 520
Severe injury	37 785	11 116	–	3 020	371 838	423 759
Moderate injury	6 718	2 747	–	580	86 762	96 807
Slight injury	1 197	743	–	134	24 789	26 863
Averages						
Per injury	5 134	5 065	267	836	60 810	72 112
Per victim (including fatalities)	5 196	6 714	339	885	69 921	83 056

Source: Ecoplan & Infras (2008).

Fig. 7. Health costs from road crashes in Switzerland per casualty by injury severity and health cost category



Intangible costs and lost consumption were calculated in detail for fatalities using data on the exact age of the victims at the time of death. To derive the theoretically remaining years of life if the crash had not happened, the official data on the probability of survival for that age was used (in one-year age classes, differentiated by sex). Further, the expected economic growth of 1% per year and the discount rate of 2% per year were taken into account to calculate the total intangible costs of a fatality at a certain age. Using the approach of the value of life-years lost, the discounted costs of a death thus decrease with age, and the highest costs per death result from child fatalities. The net production losses were derived similarly. The only two differences were that only the years until retirement were taken into account (thus no production loss for the death of senior citizens), and instead of the value of life-years lost, the yearly net production loss was used (see also section 3.1.1).

As Table 8 and Fig. 7 show, the costs vary between US$ 26 900 and US$ 2 560 000 depending on the severity of the crash. About 84% of these costs consist of intangible costs and lost consumption.

3.2.6.4 Step 5, results: total health costs due to road crashes

Multiplying the number of casualties in Table 7 by the costs per casualty in Table 8 leads to the health costs due to road crashes. Table 9 shows that road crashes incur overall health costs of US$ 7800 million: 84% of these costs are due to intangible costs and lost consumption, 8% net production losses and 6% health care costs, whereas administrative costs and occupational replacement costs play a minor role. As expected, cars cause by far the largest share of crash costs (almost half), followed by bicycles (22%) and motorbikes (12%), and the other single-vehicle categories cause less than 6% of the costs.

Table 9. Health costs due to road crashes in Switzerland by health cost category and culpable vehicle category in millions of United States dollars, 2005

	Car	Private coach	Public bus	Delivery van	Heavy goods vehicle	Articulated lorry	Tractor, machine	Bicycle	Scooter	Motorbike	Pedestrian	Total
Health care costs	225	6	4	15	5	3	3	118	30	57	26	491
Net loss of production	303	11	4	21	12	5	6	126	33	76	36	634
Replacement costs	15	1	0	1	1	0	0	7	2	4	2	32
Administrative costs	39	1	1	3	1	1	1	19	5	10	4	84
Intangible costs and lost consumption	3 093	100	45	211	95	46	56	1 448	369	777	362	6 601
Total health costs	3 675	119	53	251	113	54	67	1 718	438	923	430	7 841
Percentage of total	47%	2%	1%	3%	1%	1%	1%	22%	6%	12%	5%	100%
Millions of vehicle-km	52 080	106	300	3 301	1 422	705	NA	1 957	146	2 059	NA	62 076
Cost in US dollars per vehicle-km	0.07	1.12	0.18	0.08	0.08	0.08	NA	0.88	3.00	0.45	NA	0.12

NA: not available.

Table 9 also shows the costs per vehicle-km (except for pedestrians, tractors and work machines, which are also excluded in the calculation of the average of US$ 0.12 per vehicle-km). As can be seen, in particular scooters cause high costs per vehicle-km, followed by private coaches (due to the high occupancy rate), bicycles and motorbikes. Cars and freight transport cause costs between US$ 0.07 and US$ 0.08 per vehicle-km.

Sommer et al. (2007) calculated the costs of road crashes for children, adults and senior citizens separately, showing that 13% of the costs are related to child casualties.

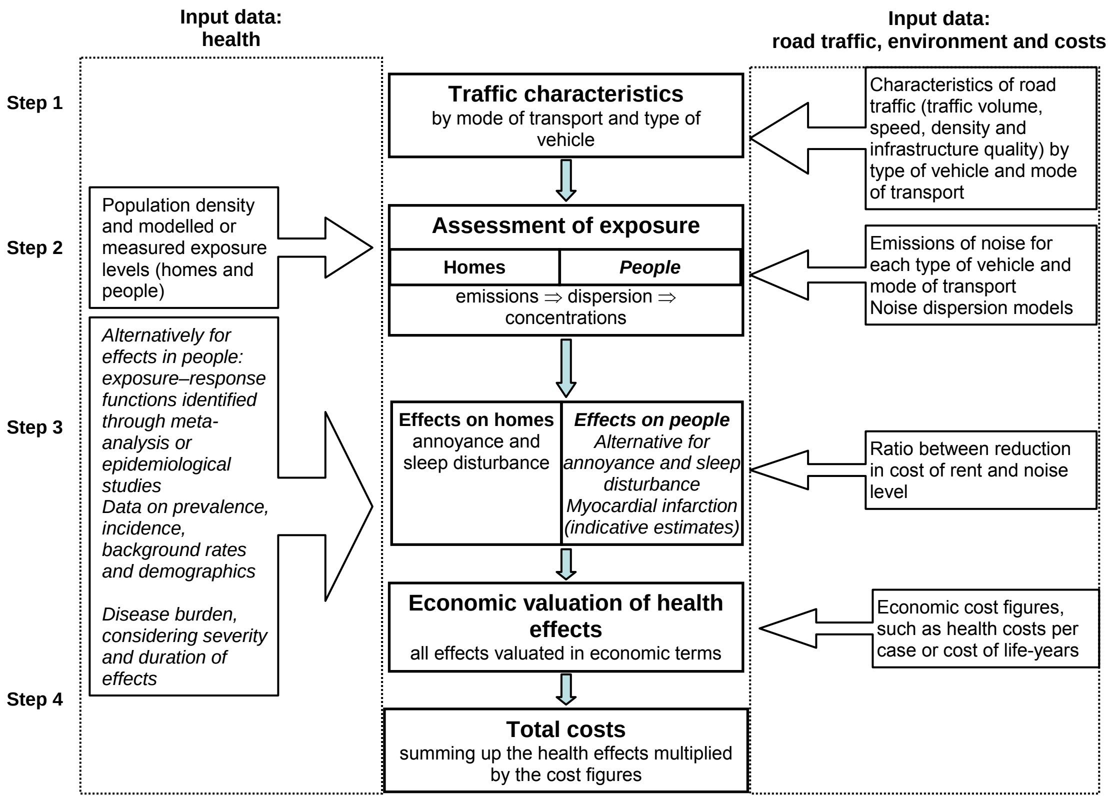
To assess the precision of the estimated cost figures, a sensitivity analysis for changes in certain important assumptions or a Monte Carlo simulation could be applied. For example, Ecoplan & Infras (2008) conducted a Monte Carlo simulation. The results show that the 95% confidence interval of the social accident costs (including material damages as well as police and legal costs) is between 34% lower and 73% higher costs.

3.3 A practical approach for valuating health costs due to noise

3.3.1 The basic model

The approach to measuring noise costs (Fig. 8) differs slightly from the general model presented in section 3.1. First, the road traffic characteristics for each mode of transport considered and by all types of vehicle involved need to be identified. Next, based on the noise emissions of each mode of transport and type of vehicle, total emissions can be modelled. Then, home addresses can be linked to modelled road noise levels, and if needed (see below), the noise exposure of the population under study can be estimated based on the noise level at the home address. In step 3, the effects on people's health due to this noise exposure are estimated by using exposure–response functions. The effects can then be summed up for exposed homes or for exposed people, respectively. In step 4, cost figures can be applied to these effects, and total health costs[19] are calculated by multiplying the number of health effects by the economic values.

Fig. 8. Steps to apply in economic valuation of noise costs (with alternative options in *italics*)



[19] The term "total health costs" refers in the whole text to the sum of all different costs of a health effect considered in this study. As this study does not consider some costs of a health effect, for example, because they cannot yet be monetized, the term "total health costs" as used here does not cover all possible total health costs that occur in reality.

3.3.2 Steps for noise

3.3.2.1 Steps 1 and 2: road transport characteristics and levels of exposure

The aim of this step is to determine individual exposure to road noise. The noise exposure of the population can be assessed by linking home addresses to modelled road noise levels. Address coordinates and data about the number of people living at a specific address can be entered into a geographical information system and combined with the modelled road noise levels, which are based on the local road traffic characteristics (traffic volume and composition, infrastructure etc.).

Assessment of road noise is not very uniform among countries; almost every country in Europe has its own national noise model for road noise levels. Nevertheless, in recent years much progress has been made in harmonizing noise indicators, calculations and mapping. Based on the EU Directive on Environmental Noise (European Commission, 2002), all EU countries are required to develop strategic noise maps.[20] The Directive selected Lden and Lnight as the common noise indicators, and methods for noise modelling are described. The European Commission Working Group Assessment of Exposure to Noise (2007) published the *Good practice guide for strategic noise mapping and the production of associated data on noise exposure* to facilitate the production and comparability of road noise mapping. Although locally used models are assumed to be more tailored to the local situation, collecting high-quality input data is important. To overcome this, the *Good practice guide* contains a toolkit that allows the validity and accuracy of the relevant input data to be assessed (Jarup et al., 2005). The European Commission is planning to provide a database with the information provided by EU countries in 2009. Noise maps are already becoming available (European Commission, 2008).

For assessing road noise levels, it is recommended to use the noise model that is commonly applied in the country or region if it is of good quality. The quality of input data should be evaluated using the good practice guide. If a noise model is not available, information on exposure in EU countries is available from the European Commission (2008) web site.

One important problem in noise modelling is that the accuracy of the calculated levels declines at lower noise levels. Input data such as road traffic intensity can be so low that relatively small deviations from the actual flow may greatly affect the calculated noise levels. To minimize the impact of such inaccuracy on the noise levels, a cut-off level is often introduced at the lower end of the noise spectrum. This cut-off value differs between countries (Jarup et al., 2008). Similar to the values that are mentioned in the good practice guide, it is recommended to use a cut-off value of 50 dB(A) (L_{den}). However, some health effects already occur at about 40 dB(A), particularly severe annoyance and sleep disturbance (see Annex 1, section I.2.2). With such a cut-off value, some of the homes and population will therefore not be included in the calculated exposure distribution, meaning that certain health effects are not included in the quantification, possibly leading to underestimation.

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20 Strategic noise maps must be developed for all agglomerations with more than 250 000 inhabitants and for all major roads that have more than 6 million vehicle passages per year, major railways with more than 60 000 train passages per year and major airports within their territories (European Environment Agency Working Group on the Assessment of Exposure to Noise, 2008).

3.3.2.2 Step 3: effects (people and homes)

According to the analysis of the epidemiological literature, sufficient evidence is only available for severe annoyance and severe sleep disturbance (and, for indicative estimates, myocardial infarction) to propose including them as health end-points in adults related to road noise (see Annex 1, section I.2.2). For all other health effects, including those mentioned in the studies from the economic literature (see Annex 1, section I.2.3.7), the strength of currently available evidence is not considered sufficient for inclusion and/or no valid exposure–response relationships are available.

The number of severely annoyed and severely sleep disturbed people can be estimated by combining the number of exposed people with the exposure-effect relationships from Table 10, as explained in the basic model in section 3.1.3.

Table 10. Health end-points and exposure–effect relationships for road noise

Health end-point	Applicable range	Formula	Source
Severe annoyance	L_{den} 45–75 dB(A)[a]	%HA = $9.868 \times 10^{-4} (L_{den} - 42)^3 - 1.436 \times 10^{-2} (L_{den} - 42)^2 + 0.5118 (L_{den} - 42)$	Miedema & Oudshoorn (2001)
Severe sleep disturbance	L_{night} 45–65 dB(A)[b]	%HSD = $-1.05 \times L_{night} + 0.01486 \times L_{night}^2$	Miedema et al. (2003)

L_{den}: The day-evening-night level is the equivalent sound level over 24 hours, with a penalty of 5 dB(A) for evening levels (19:00–23:00) and a penalty of 10 dB(A) during the night (23:00–07:00). L_{night}: the equivalent sound level during the night (23:00–07:00). HA: highly annoyed people. HSD: highly sleep-disturbed people.
[a]Outdoor noise exposure levels.
[b]Outdoor noise exposure levels of the most exposed façade.

If a study wishes to include indicative estimates for myocardial infarction, taking into account the greater uncertainty around the relative-risk function and possible double counting with air pollution effects (see section 3.4.2.3, Chapter 4 and Annex 1, section I.3.3), the odds ratios for different noise categories from Babisch (2006) can be used (see Table 11).

Table 11. Odds ratio and 95% confidence intervals (CI) to use for indicative estimates of incident cases of myocardial infarction attributable to road noise per noise category

Road noise ($L_{Aeq,\ 6–22\ h}$) in dB(A)	Odds ratio	95% CI
≤60	1.00	0.86–1.29
60–65	1.05	0.90–1.34
65–70	1.09	0.90–1.57
70–75	1.19	0.79–2.76
>75	1.47	0.86–1.29

Source: based on Babisch (2006).

Depending on the approach chosen for economic valuation (see below), exposed homes instead of people can alternatively be used to assess the effects.

3.3.2.3 Step 4: economic valuation and threshold levels

In this step, the health effects are valuated with the proposed combination of the willingness-to-pay and cost-of-illness approaches (see sections 3.1.1 and 3.1.3). However, this approach can be difficult to apply in practice to annoyance and sleep disturbance since very few studies on road noise have presented annual willingness-to-pay figures for annoyed and/or sleep-disturbed people.

Since modelling noise exposure below 50 dB(A) (Lden) is difficult, a threshold of 50 or 55 dB(A) (Lden) is suggested for costing annoyance and sleep disturbance. However, severe annoyance and sleep disturbance already occur at about 40 dB(A). Applying a threshold of 50 or 55 dB(A) is therefore likely to underestimate the total health costs. However, the costs per case are probably not the same at all exposure levels and are probably lower at the lower end of the exposure–response curve. In addition, the level of uncertainty of the willingness to pay increases below 55 dB(A), and modelling noise at levels below 50 dB(A) is difficult.

For incident cases of myocardial infarction, the threshold for indicative estimations of costs can be set at 60 dB(A), based on Babisch (2006) (see Annex 1, section I.2.2).

Since commonly available willingness-to-pay values for health outcomes are lacking, a good alternative is to use homes rather than people as the unit, as several of the economic studies reviewed have done (see Annex 1, section I.2.3.6). In contrast to the first approach, this is based on market prices expressed by a reduction in rent due to road noise, indicating people's willingness to pay to reduce noise disturbance. Severe sleep disturbance and severe annoyance should be combined for this exercise, as both health effects can be assumed to be reflected in the reduced rent. The exposure–response function is given by the relationship between the noise level and rent (Ecoplan et al., 2004b). The overall reduction in rent due to noise can be calculated for a specific area by using this exposure–response function and an average rent per home. This exposure–response function is derived from hedonistic pricing analysis using regression analysis (Ecoplan et al., 2004b). As this is not yet available in many countries, a specific survey has to be carried out first or, as a second-best approach, existing figures from other countries can be adapted (see also section 4.1.2 on the adaptation of willingness-to-pay estimations).

Based on this exposure–response function, a threshold of 55 dB(A) (daytime) should be applied for costing these two health end-points, as also suggested by other authors (Maibach et al., 2008). If market prices are applied, no further threshold is needed for night-time levels, as the market prices reflect effects occurring during the day and during the night.

3.3.2.4 Input data and data sources

Table 12 lists the data needed for estimating total health costs due to transport-related noise, summarizing sections 3.1.3, 3.3.1 and 3.3.2. The first column presents the best possible data (state-of-the-art approach). The three items in italics can be adapted from other countries or general statistics, but all other input data have to be surveyed in the country itself. In the second column, second-best data are presented that could serve as an alternative if the means to develop the state-of-the-art option are not available.

The Working Group on Health and Socio-Economic Aspects (2003) of the EU mentioned the only international value for willingness-to-pay for annoyance from road transport based on a review (Navrud, 2002). They suggested using a median willingness-to-pay value of €25 per household and year per perceived dB(A) (in L_{den}) decrease in noise, which can be applied for the range from 50/55 to 70/75 dB(A) (L_{den}). However, deriving a local value for willingness to pay would be preferable in any case, as explained in the introduction in section 3.1.3.

Table 12. Input data for cost estimations of noise and the second-best approach

State of the art	Second best
Traffic characteristics by each type of vehicle for each mode of transport	Numbers from a similar country
Emissions of each type of vehicle for each mode of transport (air pollution indicators)	–
Dispersion models	–
Demographics and concentration of homes	–
Exposure–response functions	–
Ratio between reduction in rent and noise level	Numbers from a similar country
Economic cost figures: willingness to pay, cost of illness and value of life-years lost	Cost figures from a similar country

3.3.3 Elements contributing most to total health costs

Based on the examples available[21], the costs due to annoyance contribute the largest share of total health costs due to noise measured by reduced rent (Federal Office for Spatial Development, 2007). All other health costs are lower but still significant.

3.3.4 Special focus on children

The overview of epidemiological literature concluded that, although evidence indicated that noise affects children, especially annoyance and cognitive functioning (see Annex 1, section I.2.2.5), no specific exposure–response functions are available yet. Hence, children's unique health situation can still not be considered appropriately. As in the economic studies reviewed, either they have to be treated as adults or left out altogether from the cost calculation, which is the less preferable of the two options as it considerably underestimates the total health costs. More epidemiological studies are needed on health effects among children exposed to noise.

3.3.5 A practical example for estimating the health costs of noise

This section calculates the health costs due to severe annoyance and severe sleep disturbance caused by road transport–related noise by applying the practical approach explained above. This example is based on exposed homes, so hedonic pricing studies are used to derive the reductions in rent due to noise. The noise

[21] The information given in this section refers to Federal Office for Spatial Development (2007) and therefore presents the situation in Switzerland. However, as comparisons with other studies and countries have shown, the main findings are similar in most cases. Nevertheless, depending on the road traffic situation, on the judicial system and laws and on the types of vehicles, the statements need to be viewed with caution.

costs due to myocardial infarction are not included, as they have been suggested only for indicative estimates and because the necessary input data for the calculation are not readily available.[22]

3.3.5.1 Steps 1–3: road traffic characteristics, levels of exposure and effects (homes)

In Switzerland, a noise databank SonBase was developed recently (Swiss Federal Office for the Environment, 2008). SonBase is a geographical information system application that allows noise exposure due to transport to be calculated by estimating noise emissions, noise propagation and the number of homes or people exposed to these noise levels on a very detailed level for the whole of Switzerland. Steps 1 to 3 of the calculation were therefore carried out in SonBase. Table 13 shows the results for road traffic. Almost half the homes in Switzerland (49.8%) are exposed to noise levels above 55 dB(A). The largest category of homes exposed is 50–55 dB(A). For higher (and lower) noise levels, the number of houses decreases.

Table 13. Number of homes exposed to certain levels of road noise (LAeq, 6–22 h) in Switzerland, 2005

Noise level L_r in dB(A)	Number of homes
< 45.0	349 093
45.0–49.9	591 744
50.0–54.9	936 901
55.0–59.9	772 850
60.0–64.9	782 363
65.0–69.9	265 385
70.0–74.9	41 925
≥75.0	2 569
Total	3 742 830
Total above 55 dB(A)	1 865 093

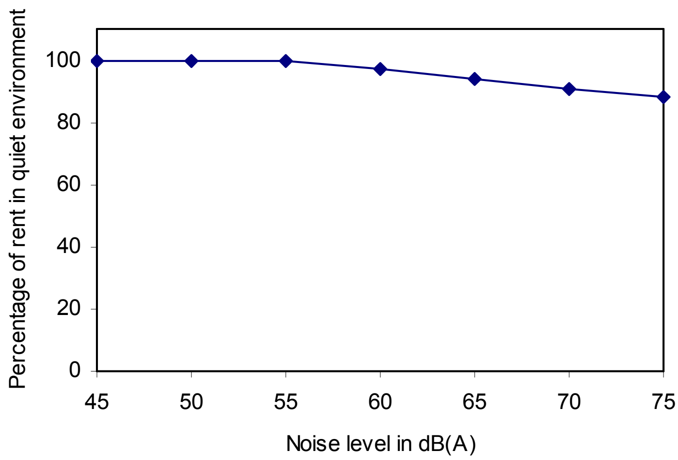
Source: Swiss Federal Office for the Environment (2008). The calculations below are based on the exact data for 1-dB(A) noise categories.

3.3.5.2 Step 4: economic valuation and threshold effects

As described in section 3.2.2.3, reduced rent is calculated for daytime noise levels above 55 dB(A). In Switzerland, there is rich literature on the hedonic pricing effects of noise on rent. Based on the results of nine studies for Switzerland (Ecoplan & Infras, 2008), a dose–response curve was derived that shows that, above 55 dB(A), the rent decreases by a constant percentage per dB(A), with an average reduction of 0.6% per dB(A) daytime noise (Fig. **9**). The average rent level in Switzerland in 2005 amounts to about US$ 12 000 per year (US$ 1000 per month (Ecoplan & Infras, 2008)). Thus the rent reduction per dB(A) is US$ 72 per year (0.6% × US$ 12 000).

[22] Another example in Ecoplan & Infras (2008) considered health costs due to ischaemic heart diseases (including myocardial infarction) and hypertension, which caused costs of US$ 25 and US$57 million, respectively. In total, these health outcomes amounted to 10% of the rent reduction.

Fig. 9. Relationship between noise level and rent in Switzerland



3.3.5.3 Step 5: results

The combination of the exposed homes in Fig. **9** and the calculated rent reduction of US$ 72 per dB(A) per year from step 4 leads to the results presented in Table 14.[23] In total, health costs related to road noise amount to about US$ 800 million. Almost half of these costs are due to homes exposed to daytime noise levels between 60 and 65 dB(A). This noise exposure category includes a large number of homes, and the related noise costs are already considerable. Detailed results per 1-dB(A) category show that the highest costs occur in the category of 61–62 dB(A). On average, the total rent in Switzerland is reduced by 1.8% due to road noise.

Table 14. Reduced rent (in millions of US dollars) due to road noise in Switzerland, 2005

Noise level L_r in dB(A)	Noise costs (millions of US dollars)	%
50.0–54.9	–	0.0
55.0–59.9	126.7	15.8
60.0–64.9	392.5	49.0
65.0–69.9	228.9	28.5
70.0–74.9	49.6	6.2
≥75.0	4.0	0.5
Total	801.7	100.0

To allocate the total noise costs to the vehicle categories, the method applied in Ecoplan & Infras (2008) was used. It is based on noise emissions and the total vehicle-km driven per category. This leads to the share of each vehicle category shown in Table 15. Passenger transport is responsible for almost 70% (US$ 560

23 The results in Table 14 were calculated based on exact data per 1-dB(A) noise increments and not directly from Table 13

million) of the noise costs, and freight transport accounts for the remaining 30% (US$ 240 million). Cars cause almost half the total costs (46% or US$ 365 million), followed by lorries larger than 3.5 tonnes (21% or US$ 170 million) and motorbikes (21% or US$ 165 million).

Table 15. Noise costs in Switzerland by vehicle category

	Passenger transport								Freight transport				Total
	Car	Public bus[a]	Trolley bus	Tram	Private coach	Motor-bike	Moped	Subtotal	Delivery van	Heavy goods vehicle	Articulated lorry	Subtotal	
Noise costs (in millions of US dollars)	364.6	18.4	0.2	0.9	8.5	165.5	1.0	559.1	71.6	114.3	56.7	242.5	801.7
%	45	2	0	0	1	21	0	70	9	14	7	30	100
Millions of vehicle-km	52 080	229	30	41	106	2 059	146	54 691	3 301	1 422	705	5 428	60 119
Costs in US dollars per vehicle-km	0.007	0.080	0.007	0.022	0.080	0.080	0.007	0.010	0.022	0.080	0.080	0.045	0.013

[a]Including all bus types (such as diesel, agrofuels etc.) except trolley buses.

Table 15 also shows the costs per vehicle-km, which vary between US$ 0.007 and 0.08. As can be seen, the costs are identical for several vehicle categories. This is due to the assumption that, for example lorries, non-electric buses, coaches and motorbikes cause the same noise emissions. According to Ecoplan & Infras (2008), they are 11.5 times noisier than cars, whereas tramways and delivery vans are only 3 times noisier.

Children are not treated differently from adults of any age. However, population groups may not be evenly distributed across a city or country. For example, families with children might try to find a flat in a quiet environment to protect their children, or parents with low income might be glad to find a larger but still affordable flat in a noisy environment.

To assess the precision of the estimated cost figures, a sensitivity analysis for changes in certain important assumptions or a Monte Carlo simulation could be applied. For example, Ecoplan & Infras (2008) conducted a Monte Carlo simulation. The results showed that the 95% confidence interval of noise costs due to lost rent ranged from 78% lower costs to 136% higher costs. The large confidence interval is due to uncertainty concerning the measurement of noise exposure and the reduction of rent per dB(A).

However, the calculations underestimate the true health costs due to road noise for the following reasons.

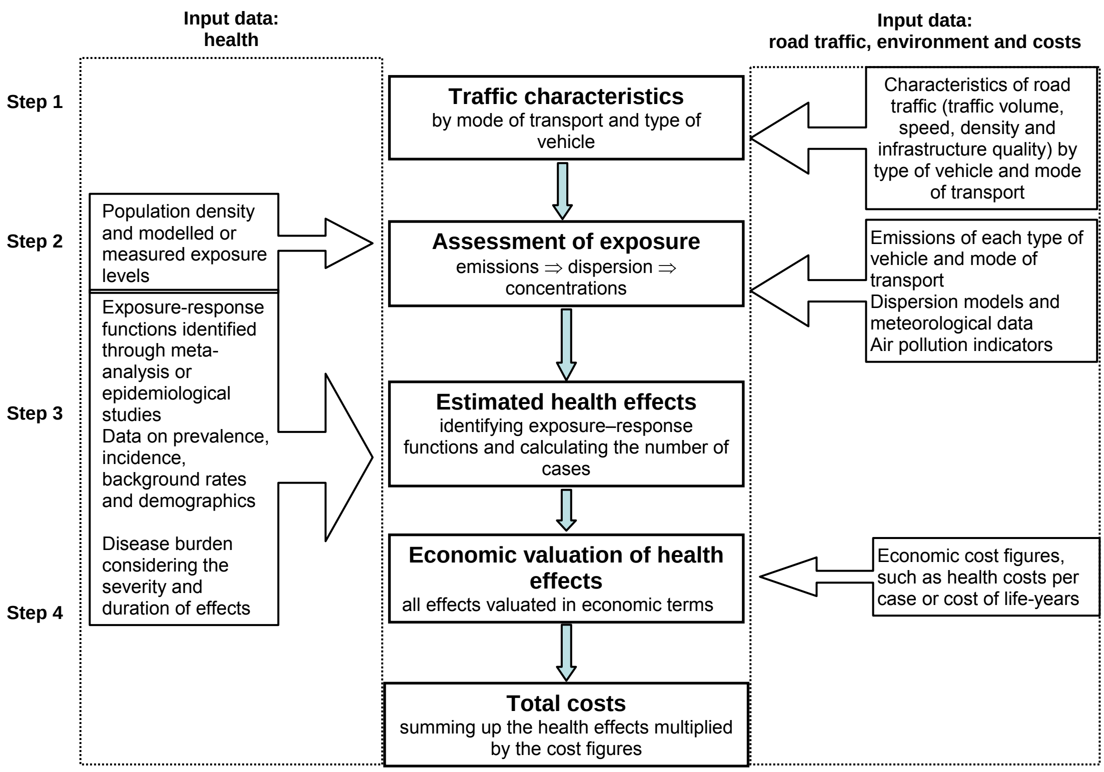
- Only noise costs at the place of residence were considered. Effects from workplace exposure were not taken into account.
- Costs related to myocardial infarction and other possible health outcomes for which scientific evidence is not yet sufficient were not included (see Annex 1, section I.2.2).
- As explained in sections 3.3.2.2 and 3.3.2.3, using the threshold of 55 dB(A) is likely to underestimate noise costs because annoyance and sleep disturbance already occur at lower noise exposure levels. A sensitivity analysis with a threshold of 50 dB(A) was therefore conducted. The results show that, with the lower threshold, the noise costs more than double (+107%). However, as also explained above, the level of uncertainty regarding the willingness to pay increases below 55 dB(A), and applying this threshold is therefore a conservative approach to avoid overestimating the costs. Further research is necessary in this area.

3.4 A practical approach for the valuation of health costs due to air pollution

3.4.1 The basic model

The steps to measure air pollution costs are the same as presented in the general approach in section 3.1 (see Fig. **10**). First, the road traffic characteristics for each mode of transport considered and all types of vehicle involved have to be determined. Next, total emissions can be calculated based on the air pollution emissions of each mode of transport and type of vehicle. Dispersion and transformation models can be used to estimate air pollution concentrations and the number of people exposed to a certain concentration. In step 3, the different health effects due to these air pollution concentrations are estimated by using exposure–response functions. These health effects are then valuated using economic cost figures. Finally, total health costs[24] are calculated by multiplying the number of health effects by the economic values and summing them.

Fig. 10. Steps to apply in economic valuation of air pollution costs



24 The term "total health costs" refers in the whole text to the sum of all different costs of a health effect considered in this study. As this study does not consider some costs of a health effect, for example, because they cannot yet be monetized, the term "total health costs" as used here does not cover all possible total health costs that occur in reality.

3.4.2 Steps for air pollution

3.4.2.1 Step 1: road traffic characteristics

Step 1 of calculating the cost of air pollution is identical to the general approach as presented in section 3.1.3). The respective emission situation depends strongly on the road traffic characteristics (such as road traffic volumes and circulation speeds) and fleet characteristics (such as the share of diesel vehicles and age and technology of the vehicles).

3.4.2.2 Step 2: levels of exposure

The dispersion and possible transformations of the emissions are estimated by using dispersion and transformation models. A whole range of modelling techniques has been developed in recent years to estimate road transport–related air pollution. The selection of a specific model strongly depends on the needs for the application in terms of exposure and health effects. All models are based on similar underlying assumptions (such as Gaussian or semi-Gaussian descriptions of dispersion processes) and have generally similar data requirements. Input data are generally: (1) road traffic characteristics (from step 1); (2) vehicle emissions; (3) weather data; (4) terrain geometry; and (5) receptor locations (such as houses). Models are typically applied on three different spatial scales: (1) local (only primary pollutants, such as CO, NO_x and SO_x), (2) urban (mainly primary pollutants) and (3) regional (primary as well as secondary pollutants such as NO_2, O_3 and secondary particles). Local and urban models are appropriate for modelling air quality, mainly influenced by local polluters such as line sources: road traffic on a roadway. Box models have been developed for the regional scale to include secondary pollutants. In addition, semi-empirical approaches to air pollution modelling have also been developed, for example, by Dirks et al. (2003). More recently, air quality monitoring systems based on satellite remote sensing have become available (Dimosthenis & Saisana, 2007). Enhanced computer capacity now allows advanced modelling for extended networks providing concentrations at high spatial and temporal resolution for a large number of point receptors. Models of the new generation can also handle complex topography, rural or urban terrain and different regional scales and include algorithms for building effects and turbulent atmospheric conditions. Temporal resolution can range from one hour to five years. These models are linked to geographical information systems, and the results can be mapped as a set of air pollution contour maps (isopleths) but they cannot yet replace monitoring to accurately describe ground levels of air pollution.

Numerous reviews and comparisons of these various models and methods have been carried out (Carruthers et al., 2001; Chang & Hanna, 2004; McHugh et al., 1999). The European Topic Centre on Air and Climate Change (2008) has set up a catalogue of air pollution models and a meta-database with information on more than 100 models to provide guidance for selecting the most appropriate model for a given situation.

If no air pollution models are available, alternatively there is a less complex approach: air pollution exposure can be estimated based on the measured ambient levels. Information on these levels is usually available from air pollution monitoring networks. Although this approach is less cost-intensive, it is second best, as the results are less precise. The steps for the estimation are the following.

First, assess the ambient concentrations of air pollution at several sites within an area of interest (for example, a city or an entire region). Then the fraction of air pollution attributable to traffic exhaust has to be determined. This can be obtained from the proportion of emissions due to different sources established in inventories. If this information is not available, a certain proportion (such as 30–40%) can be assumed, based on another city or region or on published literature. The differences in road traffic composition and air pollution situation between studies should be as small as possible (Krzyzanowski et al., 2005; Künzli et al., 2000). However, this approach might not result in very accurate estimates, since not only the road traffic composition should be similar but also the emission patterns from all other sources.

Whichever method is used, this step will assess the road traffic–related fraction of total exposure of the population. Both methods assume that there is no threshold for health effects to occur (Hurley et al., 2005).

Next, it has to be decided which air pollutants should be considered in the analysis. Table 41 shows that most of the economic studies reviewed used $PM_{2.5}$ and/or PM_{10}. There are also other air pollutants to be taken into account, as discussed in more detail in Annex 1, sections I.3.2 and I.3.3. As explained there, based on the conclusions from the recent epidemiological literature, currently, $PM_{2.5}$ and black smoke are considered the best indicators for transport-related emissions. In case no estimates (or exposure–response curves, see Table 16) for PM2.5 or black smoke are available; either PM10 can be used instead or the PM10 values can be converted into PM2.5 levels using adequate conversion ratios from a study done in a comparable location measuring both PM2.5 and PM10 (see Annex 1, section I.3.2). Some of the economic studies reviewed (see Annex 1, section I.3.4) have also included O3. However, O3 is considered a less good indicator of road transport–related air pollution (see Annex 1, section I.3.2). Where road transport is a main source of NO2 emissions, such as in Stockholm (Bellander et al., 1999), it would be a good indicator of transport-related air pollution. However, it also has limitations in predicting health effects (see also Annex 1, section I.3.3), and in a recent review, the United States Environmental Protection Agency only considered the association between short-term NO2 exposure and respiratory health effects as likely (United States Environmental Protection Agency, 2008a). In addition, no exposure–response curve from a meta-analysis is available yet that could be suggested for use. It is therefore suggested to use NO2 solely as an alternative for quantifying transport-related premature mortality from short-term exposure if: (1) no exposure data for particles are available in the particular study area; (2) road transport is a main source of NO2 in the study area; and (3) an exposure–response curve is available from local research.

Table 16 summarizes all suggested indicators and relative risk estimates that can be used in economic valuation.

3.4.2.3 Step 3: quantification of health effects

The suggested practical approach regarding health effects is mainly based on the Clean Air for Europe (CAFE) programme.

Health effects for particles are calculated based on the assumption that the exposure–response functions derived from epidemiological studies examining the whole mixture of ambient air pollution (and not specifically road transport–related air pollution) can also be used for estimating road transport–related health effects and that the related particles are as toxic as the background aerosol mixture.

Selection of risk estimates for mortality

The evidence base for an association of premature mortality with $PM_{2.5}$, PM_{10} and black smoke can be considered sufficient for including them in the quantification of health effects. For premature mortality associated with long-term exposure, the annual average of $PM_{2.5}$, if available, should be used for measuring exposure, since it indicates long-term exposure to road traffic and is consistent with exposure indicators used in the largest available cohort study of the American Cancer Society (Pope et al., 2002). If this is not available, PM_{10} or black smoke can be used in the practical approach presented below or the PM_{10} values can be converted into $PM_{2.5}$ values.

The most inclusive estimates of both attributable numbers of deaths and average reduction in lifespan are those based on cohort studies (WHO, 2006). The exposure–response function we recommend to use is based on the American Cancer Society cohort study (Pope et al., 2002), which was also used in a European cost–benefit analysis of ambient air pollution (CAFE cost–benefit analysis; Hurley et al., 2005). Since this cohort study was conducted in the United States, the generalization of these results to populations in Europe is subject to uncertainty. A large European cohort study (ESCAPE) has just started that may provide new estimates that can be used in future health impact assessment (Brunekreef, 2008).

If cause-specific mortality from long-term exposure is preferred for an assessment instead of all all-cause mortality, the evidence for cardiopulmonary mortality is considered sufficient and again, the dose-response function from Pope et al. (2002) is suggested to use. In case a study wishes to focus on the short-term effects of air pollution on mortality, sufficient evidence and dose-response functions for all-cause mortality and for all-cause and for cardiovascular and respiratory mortality are available (see Table 16). Only either cause-specific or all-cause mortality from either long- or short-term exposure should be included to avoid double counting.

Selection of risk estimates for air pollution–related morbidity

As noted in a report of the AIRNET project (Hurley & Sanderson, 2005) and in the cost–benefit analysis of the CAFE programme (Hurley et al., 2005), there are two different traditions in quantifying air pollution–related morbidity effects, reflecting different purposes and uses (WHO, 2006).

One approach, for example applied by the Committee on the Medical Effects of Air Pollutants (2007) and in the APHEIS project (European Commission, 2004), quantifies only the end-points for which sufficiently reliable data exist for both concentration–response functions and background rates. This approach is useful in demonstrating a public health problem of at least the magnitude quantified. If it is used in cost–benefit analysis, however, it may underestimate the overall health costs related to air pollution.

The second approach (ETSU & Metroeconomica, 1995; Künzli et al., 2000) aims to quantify all end-points for which, on the balance of probabilities, the relevant air pollutant has an effect. Thus, for some of the effects included in this approach, there is even greater uncertainty in the concentration–response function and especially in the background rates than would be acceptable under the first, more restrictive, approach. Nevertheless, the second approach as a whole probably more realistically assesses the possible range of air pollution effects and is thus an appropriate strategy for comparing the costs and benefits of specific policies or developments that affect air pollution, as long as the included relative risks have an acceptable level of certainty. It was therefore adopted for CAFE cost–benefit analysis (Hurley et al., 2005) and is also largely followed here. Further, Hurley et al. (2005) focused on studies of incidence rather than prevalence, so that the benefits of reducing pollution could more easily be expressed as annual benefits, for comparison with annual costs.

As discussed in Annex 1, section I.3.3, for morbidity, except for hospital admissions, the evidence is not yet considered sufficient to firmly suggest exposure–response functions (Table 48 and Table 49). However, following the second approach above, selected estimates of other effects on morbidity can be included in such analysis (Table 16) if the aim is a more comprehensive rather than a conservative estimate of transport-related health costs and the larger uncertainty related to the other morbidity health end-points is accepted and clearly acknowledged. For some of the health outcomes included in CAFE, the risk estimates are very small or the background disease rates are uncertain or the end-point cannot be valuated, such as effects of exposure on lung functioning, asthma or medication use; these are not included here (WHO, 2006).

In some cases, where the background rates are not easily available, the exposure–response functions and background rates have been combined into an impact function, expressed as the number of (new) cases or events per unit population (such as 100 000) per unit of pollutant per unit of time (such as a day or a year) (Hurley et al., 2005).

Table 16. Summary of suggested health end-points, age group and pollutant (in order of proposed priority if more than one per health end-point) and suggested relative risk estimates (and 95% confidence intervals (CI)) for a 10 μg/m3 increase in pollutant and source

Health end-point	Age group	Pollutant	Relative risk estimate / impact function (95% CI)	Sources
Mortality				
Mortality from long-term exposure				
All causes	>30 years	$PM_{2.5}$	1.06 (1.02–1.10)	Pope et al. (2002)
	>30 years	PM_{10}	1.04 (1.03–1.06)	Künzli et al. (2000)
	Infants (1–12 months)	PM_{10}	1.04 (1.02–1.07)	Woodruff et al. (1997)
Cardiopulmonary	>30 years	$PM_{2.5}$	1.08 (1.02–1.14)	Pope et al. (2002); WHO Regional Office for Europe (2006)
Mortality from short-term exposure				
All cause	All ages	PM_{10}	1.006 (1.004–1.008)	Anderson et al. (2004); WHO Regional Office for Europe (2006)
	All ages	Black smoke	1.006 (1.004–1.008)	Anderson et al. (2004)
Cardiovascular	All ages	PM_{10}	1.009 (1.005–1.13)	Anderson et al. (2004); WHO Regional Office for Europe (2006)
	All ages	Black smoke	1.004 (1.002–1.007)	Anderson et al. (2004)
Respiratory	All ages	PM_{10}	1.013 (1.005–1.20)	Anderson et al. (2004); WHO Regional Office for Europe (2006)
	All ages	Black smoke	1.006 (0.998–1.015)	Anderson et al. (2004)
Morbidity				
Hospital admissions				
Respiratory (annual rate)	All ages	PM_{10}	7.03 (3.38–10.30) per 100 000 population[a]	Hurley et al. (2005)
Cardiac (annual rate)	All ages	PM_{10}	4.34 (2.17–6.51) per 100 000 population[b]	Hurley et al. (2005)
Lower respiratory symptoms[c] *including cough*[d] (annual increase)	5–14 years	PM_{10}	1.86 (0.92–2.77) extra symptom-days per year and child[e]	Hurley et al. (2005)
	Adults	PM_{10}	1.30 (0.15–2.43) extra symptom-days per adult with chronic respiratory symptoms (about 30% of the adult population)[f]	Hurley et al. (2005)
Chronic bronchitis[c] (new cases/year)	>27 years	PM_{10}	26.5 (1.9–54.1) per 100 000 population[g]	Hurley et al. (2005)
Restricted activity day[c]	15–64 years	$PM_{2.5}$	902 (792–1013) per 1000 population[h]	Ostro (2006)
Working-loss day[c]	15–64 years	$PM_{2.5}$	207 (176–238) per 1000 population[i]	Hurley et al. (2005)

[a]Background rate used: 617 per 100 000 population.

[b]Background rate used: 723 per 100 000 population.

[c]The related *health end-points* are suggested to be only used for indicative estimates; see the text above.

[d]Defined as wheeze, chest tightness, shortness of breath or phlegm production.

[e]Background rate used: 15%.

[f]Background rate used: 30%.

[g]Background incident rate used: 0.378%.

[h]Background rate used: 19 per person per year.

[i]Background rate used: 4.5 per person per year.

3.4.2.4 Step 4: economic valuation

For step 4 of the estimation of total health costs due to air pollution, the general practical approach can be followed (see section 3.1.3). However, the health effects of long-term exposure occur with a certain delay. Hence, a reduction in air pollution does not immediately lead to a complete reduction in effects (as is the case with acute effects). This time lag regarding the long-term health effects has to be taken into account (Chanel et al., 2006; Ecoplan et al., 2004a; United States Environmental Protection Agency, 1999).

3.4.3 Input data and data sources

Table 17 presents the input data needed for estimating total health costs due to transport-related air pollution. Whereas the first column shows the state-of-the-art approach, the second column presents a second-best solution in case the financial and/or technical means to implement the state-of-the-art approach are not available. However, for road traffic characteristics and weather data, finding accurate surrogate data is difficult, as differences between countries are likely to be substantial. Demographic data also need be available from the study region. In contrast, emissions of each vehicle type and mode of transport, dispersion and transformation models and concentration–response functions can all be adopted from general data sources as explained above (indicated in italics in the table). Economic cost figures should also be derived locally, as explained in section 3.2.2.4. As a second-best approach, numbers from a country that is as similar as possible could be used, but differences in gross domestic product and organization of the health system could create difficulty in adopting non-local values. Table 3 on page 30 presents figures from various international studies as an indication.

Table 17. Input data for estimating the costs of air pollution: state-of-the-art and second-best approach (in *italics*: items that can be taken from general data sources)

State of the art	Second best
Road traffic characteristics by each type of vehicle and each mode of transport	(Numbers from a similar country)
Emissions of each vehicle type of each mode of transport (air pollution indicators)	–
Dispersion and transformation models	–
Weather data	–
Demographics	–
Baseline health data	–
Concentration–response functions	–
Economic cost figures (willingness to pay, cost of illness, value of life-years lost)	Cost figures from a similar country

3.4.4 Elements contributing most to total health costs

For the total health costs due to air pollution, the most important component[25] by far is the value of life-years lost due to premature mortality from long-term exposure, followed by costs due to chronic bronchitis (Federal Office for Spatial Development, 2007). All other health effects contribute much less to total health costs (Federal Office for Spatial Development, 2007).[26]

Again, the intangible costs account for the largest share of total health costs (Sommer et al., 2007).

3.4.5 Special focus on children

Children have not often been the focus in studies on health effects due to air pollution. Nevertheless, the literature has some concentration–response functions (see Annex 1, sections I.3.3 and I.3.4.11). Further, different values for a statistical life and life-years lost are available for children exposed to air pollution (AEA Technology Environment, 2005a, b). Hence, at least for some health effects, the cost of air pollution can be calculated for children as well. Nevertheless, more research is needed.

In addition, to allow applying concentration–response functions to children, the number of people exposed to air pollution needs to be available by age or age groups.

3.4.6 A practical example for estimating the health costs of air pollution

This section calculates the health costs due to road transport–related air pollution in Switzerland in 2005 as an example. Künzli et al. (2000) and Ostro (2006) provide other examples of health impact assessment of transport-related air pollution.

Mortality (all causes) from long-term exposure as well as respiratory and cardiac hospital admissions were taken into account. The other health end-points listed in Table 16 (lower respiratory symptoms, chronic bronchitis, restricted-activity day and working-loss days) are not considered here, as it has been suggested to use them only for indicative estimates and because the necessary input data for the calculation were not readily available.[27]

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25 The information in this chapter refers to Federal Office for Spatial Development (2007) and therefore reflects the situation in Switzerland. However, as comparisons with other studies and countries have shown, the main findings are similar in most cases. Nevertheless, depending on the road traffic situation, on the judicial system and laws and on the types of vehicles, the statements need to be viewed with caution.

26 This applies to both road and rail transport (Federal Office for Spatial Development, 2007).

27 Ecoplan & Infras (2008) also considered health costs due to chronic bronchitis (US$ 139 million), asthma attacks (US$ 1 million) and restricted activity days (US$ 116 million) among adults and acute bronchitis among children (US$ 4 million). The dose–response functions used for these calculations were different from those in Table 16. In total, these health end-points amounted to 30% of the health costs considered below.

3.4.6.1 Step 1: road traffic characteristics and step 2: different levels of exposure

In Switzerland, the population exposure to PM10 has been extensively calculated. For other pollutants, no data of similar quality were available. PM10 was therefore used as an exposure indicator. A Gaussian dispersion model available for Switzerland was used to calculate the population exposure to PM10. The model is based on a grid of cells 200 m by 200 m. In each cell, the primary and secondary PM10 emissions and resulting immissions are calculated. Based on population data, then population exposure was determined. The results showed that, in 2005, each person was exposed to an average of 4.34 µg/m3 of PM10 emitted by road transport.[28] Health end-points were calculated separately for infants (aged 0–1 years) and adults (older than 30 years of age), as specified in Table 16 and based on their exposure, which was 4.16 µg/m3 (infants)[29] and 4.37 µg/m3 (adults), respectively.

3.4.6.2 Step 3: health effects

The respiratory and cardiac hospital admissions can be calculated based on:

- the dose–response function per 10 µg/m3 in Table 16
- the number of hospital admissions per 100 000 people;[30]
- the population in Switzerland in 2005 (7.4 million people); and
- the average population exposure of 4.34 µg/m^3.

Table 18 shows the result. By coincidence, the number of hospital admissions due to road transport–related air pollution is almost identical for respiratory and cardiac symptoms (each 257 admissions per year).

The calculation of the years of life lost is more complicated (see also section 3.2.6 on road crashes). The starting-point was Switzerland's population in 2005 in one-year age classes, differentiated by sex. For the baseline scenario, the official data on the probability of survival were used (again for one-year age classes by sex) and followed everyone from 2005 until their expected death. The same calculation was repeated under the assumption that road transport caused no air pollution in 2005, so the probability of survival would be higher.[31] The difference between the two calculations then shows how many years of life are lost due to air pollution from road transport.

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28 The total exposure to PM_{10} from all sources was 19.67 µg/m^3 (Ecoplan & Infras, 2008).

29 This value is actually valid for children aged 1–14 years and is used as an approximation for age 0–1 years, as no data were available for infants.

30 Background rates for Switzerland were used instead of the background rates listed in Table 16. The number of hospital admissions per 10 µg/m^3 and per 100 000 population has therefore been adjusted linearly to 7.95 and 7.97 for respiratory and cardiac hospital admissions, respectively.

31 The relative risk from Table 16 was applied to the probability of survival after eliminating deaths caused by accidents and violence.

Table 18. Years of life lost and hospital admissions due to road transport–related air pollution in Switzerland, 2005

Respiratory hospital admissions	256.8
Cardiac hospital admissions	257.4
Years of life lost	12 944

This calculation needs to consider that the long-term effects of air pollution do not materialize within one year but accumulate over several years. The calculation of the effects of the air pollution in 2005 assumed that the effects decrease exponentially over 10 years.[32] The actual reduction is shown in Table 19; 64% of the effects occur in the first two years.

Table 19. Development of the effects of mortality from long-term exposure over time

Time (years)	1	2	3	4	5	6	7	8	9	10
Share of total effect	39.6%	24.0%	14.6%	8.8%	5.4%	3.3%	2.0%	1.2%	0.7%	0.4%

Source: Röösli et al. (2005). The results have been adjusted to total 100%.

The results show that 12 944 years of life were lost due to road transport–related air pollution (see Table 18). The calculations were done separately for adults older than 30 years of age and for infants younger than 1 year of age (see Table 21): infants lost 3.3% or 432 years of life and adults 96.7% (12 512 years).

3.4.6.3 Step 4: economic valuation

For the economic valuation, the standard approach was taken, which is very similar to the approach taken for road crashes (see section 3.2.6). Numbers were calculated in United States dollars using factor prices. Administrative costs were been considered because Ecoplan (1996) has shown that they account for less than 0.5% of the costs. The following health cost categories were included (Table 20).

- Health care costs were based on Ecoplan & Infras (2008).
- Net loss of production: the same cost rates per lost day as for road crashes were used, based on official data from the Swiss Federal Office of Statistics. For hospital admissions, the number of lost working days caused by an average hospital stay was calculated (respiratory 10.9 days, cardiac 12.1 days (Ecoplan et al., 2004a)) and multiplied by two to incorporate the recovery time spent at home before returning to the workplace. Further, the calculation took into account the fact that most people requiring hospital treatment due to air pollution are older people, as only 10.7% of the hospital admissions are due to people in the labour force (work-years lost as a percentage of total life-years lost). For fatality cases, the calculation considered only the lost years of employment (instead of all years of life lost).

32 The function used is $e^{-0.5 \cdot t}$.

- Occupational replacement costs: similar to road crashes, the cost of about 50% of the yearly earnings of the employee to be replaced due to death or disability was used.
- Intangible costs and lost consumption: the same cost rates were used as for road crashes (see section 3.2.6): US$ 74 200 per life-year lost. The willingness to pay (covering intangible costs and lost consumption) for hospital admissions was taken from Ecoplan & Infras (2008) with the value per hospital day multiplied by the average length of the hospital stay mentioned above.

Table 20. Costs in United States dollars per health end-point and health cost category

	Willingness to pay	Health care costs	Net loss of production	Total
Years of life lost	74 217	-	4 283[a]	78 500
Respiratory hospital admission	7 303	7 441	233	14 977
Cardiac hospital admission	8 107	10 580	259	18 945

[a]Including occupational replacement costs.

3.4.6.4 Step 5: results

When the final results are calculated, the health effects occurring in future years (Table 19) need to be discounted. A discount rate of 2% per year was used (according to Swiss Standard SN 641 821 (Swiss Association of Road and Transportation Experts, 2006)). Further, it is assumed that real wages grow by 1% per year. Thus, the actual discounting of future years of life lost is 1% (exactly 0.99% = 1.02/1.01).

First, the discounted years of life lost were calculated, which amounted to 11 564 (or 89% of the undiscounted ones, see Table 18). These were multiplied by the costs per health end-point and health cost category from Table 20. Table 21 presents the final results. In total, health costs amounted to US$ 920 million; 99% of these costs were due to lost life-years and only 1% were due to respiratory and cardiac hospital admissions. Of the costs due to lost life-years, 2.6% occurred among infants, and the large majority of these costs were related to adults older than 30 years of age.

Table 21. Health costs due to road transport–related air pollution in Switzerland in millions of United States dollars, 2005

	Willingness to pay	Health care costs	Net loss of production	Total
Years of life lost	858.2	–	49.5[a]	907.8
>30 years of age	836.1	–	48.3	884.3
0–1 years of age)	22.1	–	1.3	23.4
Respiratory hospital admissions	1.9	1.9	0.1	3.8
Cardiac hospital admissions	2.1	2.7	0.1	4.9
Total	862.2	4.6	49.7	916.5

[a]Including occupational replacement costs.

A two-step procedure was used to allocate the costs to the categories of vehicles. First, the air pollutant dispersion model was calculated separately for passenger and freight transport, and health effects were calculated separately for passenger and freight transport. Second, the allocation to the vehicle categories was based on the PM10 emissions as shown in Table 22.

Table 22. Health costs due to air pollution by vehicle categories

	Passenger transport						Freight transport				Total
	Car	Public bus	Trolley bus	Private coach	Moped or motorbike	Total	Delivery van	Heavy goods vehicle	Articu-lated lorry	Total	
PM_{10} emissions in tonnes	2 594	184	16	44	106	2 944	374	523	270	1 167	4 111
Share of emissions	88.1%	6.3%	0.5%	1.5%	3.6%	100%	32.0%	44.8%	23.2%	100%	
Health costs in millions of US dollars	461	33	3	8	19	523	126	176	91	393	916
Share of health costs	50.3%	3.6%	0.3%	0.8%	2.1%	57.1%	13.7%	19.2%	9.9%	42.9%	100%
Millions of vehicle-km	52 080	229	30	106	2 205	54 649	3 301	1 422	705	5 428	60 078
Cost in US dollars per vehicle-km	0.0089	0.143	0.096	0.073	0.0085	0.96	0.038	0.12	0.13	0.072	0.0153

The results for trams are not available. Source for PM_{10} emissions: German Federal Office for the Environment and Swiss Federal Office for the Environment, Forest and Landscape (2004).

Passenger transport caused slightly more half of the costs (57%). Cars caused about half the total health costs; the other vehicle categories of passenger transport were less important. In freight transport, heavy goods vehicles dominated with 19%, and delivery vans (14%) and articulated lorries (10%) also caused considerable shares of the total costs.

The costs per vehicle-km (also shown in Table 22) varied between US$ 0.009 and US$ 0.143. The most polluting vehicle category was public buses because they often stop (higher fuel consumption and more abrasion) and usually drive around in densely populated areas and thus have a higher share in the population exposure. Lorries exceeding 3.5 tonnes also caused high costs per vehicle-km (US$ 0.126 per vehicle-km).

As discussed in sections 3.4.2 and Annex 1, I.3.3, no dose–response functions are available from meta-analysis for mortality among children except for infants. As shown, only 2.6% of the costs due to mortality from long-term exposure occurred among infants. The costs of hospital admissions were calculated for people of all ages, but one can assume that the share of children was also rather low as older people are likely to be more affected.[33]

Ecoplan & Infras (2008) analysed the precision of the estimation using a Monte Carlo simulation. Since the health costs considered here are largely determined by years of life lost, only this health outcome is analysed here. The resulting 95% confidence interval was between 60% lower and 135% higher costs. The large confidence interval stems from uncertainty concerning the value of a life-year lost, the confidence intervals of the dose–response function and the calculation of the population exposure.

[33] Of the four additional heath end-points Ecoplan & Infras (2008) considered, only acute asthma concerned children for which the costs amounted only to US$ 1 million or 0.1% of the four health end-points mentioned previously (chronic bronchitis, asthma attacks and restricted activity days among adults and acute bronchitis among children).

However, the calculations above underestimate the true health costs for the following reasons (Ecoplan & Infras, 2008):

- As explained above, many health end-points were not considered. The four additional health end-points considered in Ecoplan & Infras (2008) (chronic bronchitis, asthma attacks and restricted activity days among adults and acute bronchitis among children) would increase the health costs by 30%, and there are even more health end-points for which scientific evidence was not yet considered sufficient (see Annex 1, section I.3.3).
- Only health effects due to PM_{10} are considered, whereas health problems due to other air pollutants are not included, as explained in Annex 1, section I.3.3.
- The air pollution emissions of road vehicles might be more toxic than emissions from other sources, but all emissions were treated equally in the calculations.
- Morbidity relates in most cases only to the short-term effects of air pollution. The long-term effects have not yet been studied to a sufficient extent due to complex methods. However, such effects are likely. For mortality, the effects from long-term exposure are about seven times as high as those from short-term exposure (Table 13).
- The effects of air pollution have been calculated only for the age groups for which dose–response functions are available. For example, the mortality of 1- to 29-year-olds is not included because this age group has not yet been the subject of long-term studies. Nevertheless, air pollution is also expected to increase mortality rates in this age group. For example, applying the identical dose–response function for adults older than 30 years of age to the 1- to 29-year-olds would increase the costs by 3%.
- The value of the willingness to pay for years of life lost (which accounts for 94% of the total costs) may have to be doubled, as the willingness to pay has been taken from an accident context. However, there are preliminary indications that individuals are willing to pay twice (or even three times) as much for the involuntary and uncontrollable risk of air pollution as for the perceived comparably more voluntary and controllable risk of road crashes (Ecoplan et al., 2004a; Friedrich & Bickel, 2001; Hunt, 2001; Jones-Lee et al., 1998; Kenkel, 2000; Nellthorp et al., 2001; Sommer et al., 1999).
- The calculations of years of life lost do not factor in the population’s increasing life expectancy in the future.

3.5 Considerations on valuating health costs due to insufficient physical activity

This section focuses on the costs of insufficient physical activity in accordance with the other sections focusing on the costs of road transport. As explained in more detail in Annex 1, section I.4.2, for all-cause mortality sufficient evidence is available to suggest inclusion in economic valuations. For some cause-specific types of mortality (coronary heart diseases, ischemic stroke, Type 2 diabetes, colon and breast cancer), sufficient evidence is also available but it is nevertheless suggested to focus on all-cause mortality as it takes account of all deaths and data is more readily available and less prone to misclassification.

As current evidence was not yet considered sufficient to allow proposing a complete model for this topic, two main issues – the apportionment of transport-related insufficient activity and the calculation of morbidity costs – and ideas on calculating the costs of all-cause mortality are discussed here.

3.5.1 Apportionment of the share of insufficient activity to road transport

For air pollution and noise and definitely for road crashes, several studies have determined the share of negative effects that can be attributed to road transport with reasonable certainty. This discussion is quite new for insufficient physical activity: the lack of walking and cycling.

One important element of evidence available that can be helpful in this question relates to the large potential for active travel in European urban transport systems. In European cities, more than 30% of trips by car are shorter than 3 km and 10% shorter than 1 km (European Commission, 2000): distances that could be conveniently covered by cycling or walking. Up to half of these short trips could potentially be replaced by cycling and walking (Stangeby, 1997). The research project WALCYNG (Hydén et al., 1998) showed that cycling and walking trips are predominantly short trips up to about 5 km (Solheim & Stangeby, 1997).

Nevertheless, some activity substitution could occur when people commute actively: for example, they could replace jogging after work by commuting by bicycle. Cavill et al. (2007) showed that economic analysis frequently did not account for this, often assuming that any observed increase in cycling or walking automatically leads to an increase in total physical activity. This may overestimate the benefits, as some evidence indicates that substitution could take place (Rodriguez et al., 2006). Economic analysis should account for activity substitution as far as possible and not assume that an increase in cycling or walking automatically leads to a similar increase in total physical activity. Taking activity substitution into account results in more conservative estimates. Ideally, comprehensive data on physical activity behaviour in different life domains should be available to address this issue (leisure, transport, work and home).

Different approaches to the apportionment of transport-related insufficient activity have been discussed in this project. It was decided to provide examples of existing interventions or situations from which practitioners can derive a share of transport-related inactivity that is reasonable for their particular situation.

3.5.1.1 Theoretical baseline scenario with no motorized transport

As usually done in apportioning air pollution (see section 3.4.6), the reference scenario used to quantify transport-related health effects has no motorized transport. Within the current transport system, the European Commission (1999) concluded that up to half the trips up to 5 km could potentially be replaced by cycling and walking. To derive a theoretical baseline scenario for cycling and walking, it could therefore be assumed that 75% to 100% of the trips up to 5 km could be replaced by active modes of transport. Without any motorized means of transport, overall travel behaviour would change dramatically. The length of trips people would be willing to undertake by cycling and walking would therefore be likely to increase, and 5 km is probably a conservative assumption. An assumption should also be made on the occurrence of activity substitution, at least as part of sensitivity analysis (Cavill et al., 2007). In this scenario, transport-related physical activity would replace much if not all current leisure physical activity.

3.5.1.2 Intervention effects on cycling

Instead of assuming no motorized transport at all, the effect of interventions can be taken as a basis for assumptions on possible changes that can realistically be achieved. Two evaluated examples are helpful as input for this approach: the Odense cycling city project and the London congestion charge project. In Odense, between 1999 and 2002 significant investment (€2.7 million) was made in a comprehensive cycling promotion programme with about 50 projects, including physical improvements (such as upgrading junctions, bicycle lanes and parking), information and publicity campaigns and changes in regulations. By the end of 2002, cycling journeys had increased by 20% (Troelsen et al., 2004). In London, the congestion charge was introduced in February 2003, accompanied by a comprehensive evaluation programme (Mayor of London – Transport for London, 2007). The assessment showed that, compared with pre-charging conditions, cycling kilometres increased by 43% and their overall share of kilometres travelled increased from 4% to 7% until 2006. Apart from the congestion charging, no specific measures targeted cyclists. These two examples show that interventions in the transport system can significantly increase cycling.

Assumptions on the transport-related share of inactivity can then be based on the lack of cycling in the absence of such investment. Based on these examples, 10–40% of cycling kilometres seems reasonable, depending on the baseline level of cycling (the lower the baseline, the higher the share of insufficient physical activity due to the transport environment can be assumed to be) and the existence of supportive measures (the fewer measures, the higher the share of insufficient physical activity due to the transport environment can be assumed to be). An assumption should also be made on the occurrence of activity substitution, at least as part of sensitivity analysis.

3.5.1.3 Effects of pedestrian-friendly environments on walking

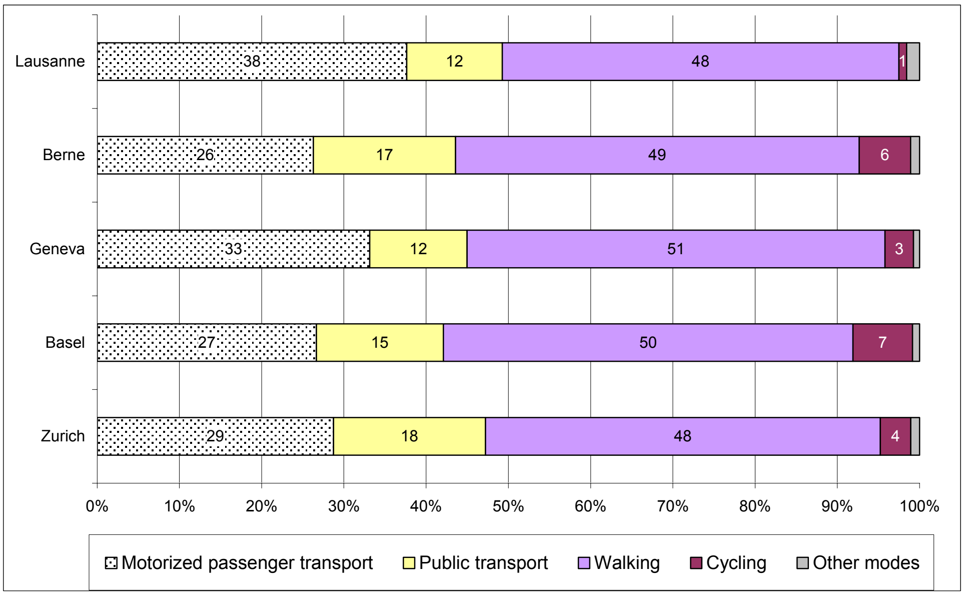
A recent systematic review of interventions to promote walking (Ogilvie et al., 2007) did not identify any environmental intervention of sufficiently high quality to be recommended here. A review of effective policies to increase physical activity (Ross et al., 2006) identified three studies that compared pedestrian-friendly environments with non-pedestrian-friendly ones (Kitamura et al., 1997; McNally & Kulkarni 1997; 1000 Friends of Oregon, 1993). Parameters studied included the ease of crossing streets, local street characteristics and the continuity of sidewalks or topography. The studies reported between 25% and 200% difference in either walking and/or cycling and/or total physical activity. However, the lack of detail in reporting the results does not allow any assumption on the transport-related share of walking inactivity in the absence of such supportive environments.

3.5.1.4 Differences in transport-related insufficient physical activity between cities

The observed differences in walking and cycling between cities that offer more or less supportive environments for active travel can also provide a basis to derive an assumption on the share of transport-related insufficient physical activity.

As Fig. 11 shows, the level of cycling differs clearly cities in Switzerland. Basle and Berne, which can be considered particularly cycling- and walking-friendly, have higher modal shares of cycling (6–7%), whereas the shares in Zurich, Geneva and particularly in the hilly Lausanne are lower (1–4%). However, there are no clear differences in walking.

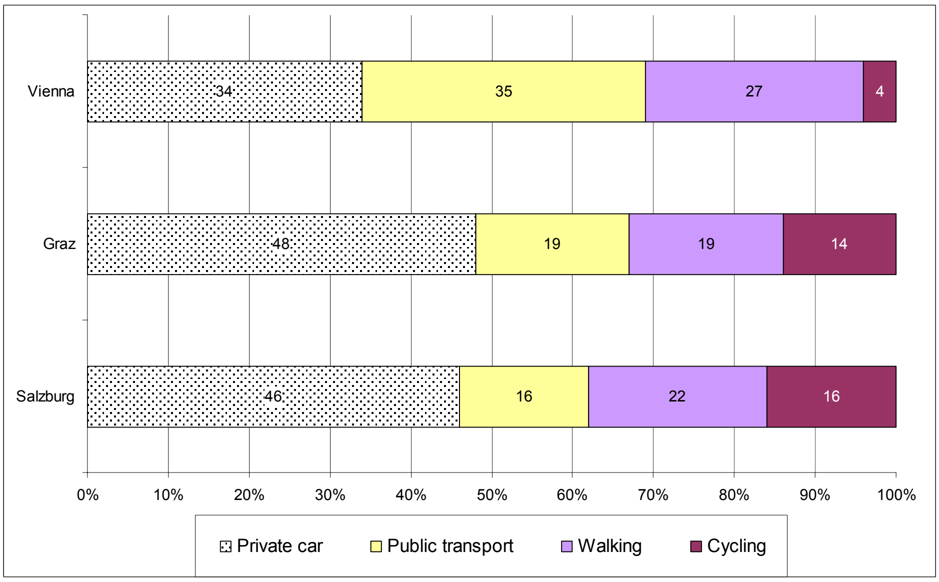
Fig. 11. Shares of modes of transport in cities in Switzerland (as a percentage of all legs[34])



Source: Swiss Federal Offices of Statistics and of Spatial Development (2007).

In Austria (Fig. 12), the share of cycling is also clearly higher (and rather similar) in the more cycling- and walking-friendly cities of Graz and Salzburg (14% and 16%, respectively) than in Vienna (4%). However, in Vienna the level of walking is clearly higher than in Graz and Salzburg, probably often in connection with using public transport, which has a higher share in Vienna as well, and possibly also in connection with longer commuting distances.

Fig. 12. Shares of modes of transport in cities in Austria (as a percentage of all journeys)

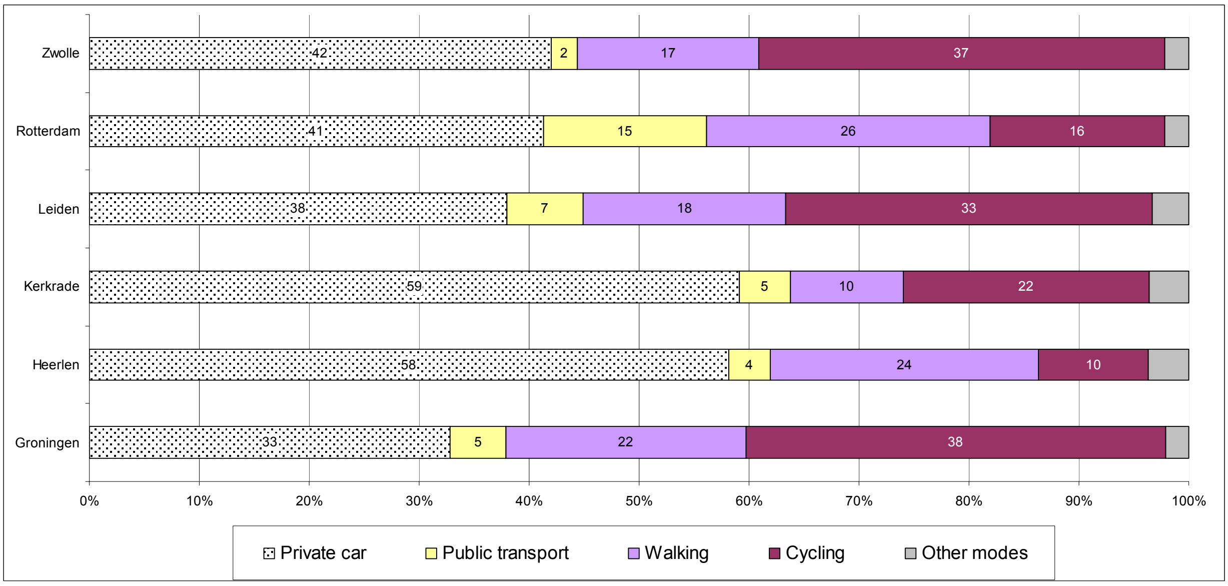


Sources: Salzburg and Vienna: Herry et al. (2007); Graz: Municipality of Graz (2008).

[34] The Swiss Travel Survey uses the concept of “legs”, i.e. it assesses the different parts of a journey separately.

Overall higher levels of cycling can be observed in the Netherlands (Fig. 13). Rotterdam, Heerlen and Kerkrade, which have been rated as being among the least cycling-friendly larger Dutch cities (Fietserbond, 2003), have a cycling share which is comparable with the most cycling friendly cities in the previous two examples (16%, 10% and 22%, respectively). The most cycling-friendly cities of Zwolle, Groningen and Leiden reach around one third of the share of total trips made by bicycle. But as in the other two examples, the levels in the more cycling-friendly was about two to three times as high as in the less cycling-friendly cities. As opposed to the other two examples, in this case also the walking levels were clearly higher in those cities rated more cycling friendly.

Fig. 13. Shares of modes of transport in cities in the Netherlands (as a percentage of all journeys)



Sources: Statistics Netherlands (2006).

Based on these examples, it can be assumed that transport-related cycling could be doubled or tripled by providing a supportive transport system, depending on the baseline level of cycling (the lower the baseline, the higher the share of insufficient physical activity due to the transport environment can be assumed to be). Due to the inconsistent evidence, no assumption could be derived for walking. With regard to cycling, the differences between the transport systems in Switzerland, Austria, the Netherlands and the respective study area should be considered, with assumptions being largely based on locally available data on the modal share of transport. An assumption should also be made on activity substitution, at least as part of sensitivity analysis.

3.5.1.5 A modeling approach to walking, urban design and health

Boarnet et al. (2008) have presented an approach for modelling the health benefits that can be expected from different levels of walking depending on the characteristics of the neighbourhood where people live using regression analysis. Based on this model, inference could be made on the health effects to be expected from a hypothetical or foreseen intervention.

The model is based on travel diary data from Portland, Oregon, collecting data on individual travel patterns (mode, purpose, distance, time etc.). Boarnet et al. modelled the amount of walking reported, including sociodemographic variables[35] and variables to measure the urban design and land-use characteristics.[36] They assumed that the walking behaviour persists long enough to realize health benefits and that no activity substitution occurs (but did sensitivity analysis based on the latter assumption). Based on this model, they could show the change in distance walked if the land-use variables change (for example, from an average level represented by the median value to a higher level, such as the 75th or 95th percentile of the value). The change in walking distance ranged from 0.0019 to 2.51 more miles walked associated with a change in the least to the most important variable (Boarnet et al., 2008). Based on selected health end-points, such a change in behaviour was then translated into changes in health.

The strength of this approach is that it is based on reported, individual travel data and objective indicators for the quality of the neighbourhoods. It also allows changes in transport-related physical activity behaviour to be modelled based on changes in single variables rather than a global change to the global transport environment, such as based on comparing different cities as in section 3.5.1.4. Its main weakness is the fact that it is based on data from the United States, which is likely to differ in several ways from Europe. However, the approach could be adapted by using national or local data, if available.

3.5.2 Valuation of costs from mortality

As explained in Annex 1, section I.4.2, all-cause mortality associated with the lack of commuter cycling from the study by Andersen et al. (2000) is currently the most suitable dose–response relationship to be included in economic assessments of transport-related insufficient activity. This study also takes activity substitution into account (see also Annex 1, section I.4.4). Although high-quality studies exist for walking (Hakim et al., 1999; Manson et al., 2002), they cannot yet readily be used for developing a calculation model.

As for the other topics discussed, the approach of the value of life-years lost, taking life expectancy into account, should be used as it is more accurate than the approach of the value of a statistical life. If the necessary data are available, alternatively an approach based on years of (disability-adjusted) life lost could be applied as well, which allows more comprehensive assessment of health effects. However, the approach of the value of life-years lost is more commonly applied in transport and might therefore facilitate the communication of results.

A model for calculating the economic savings due to health benefits from cycling has been developed that can serve as guidance for developing a similar model for the costs of transport-related insufficient activity (WHO Regional Office for Europe, 2007d).

[35] Sex, number of children in the household, interaction term sex by number of children, age, income, whether the diary included a workday and the existence of any physical handicap.

[36] Population density, total employment and retail employment density, number of intersections, distance from home to the nearest light rail station, distance to the central business district and the quality of the pedestrian environment (measured as a pedestrian environment factor developed in an earlier study).

3.5.3 Morbidity-related costs: an alternative approach

Most of the economic studies reviewed also calculated the costs of different morbidity outcomes (see Annex 1, section I.4.4.5). However, from the viewpoint of the epidemiological literature (see Annex 1, section I.4.2), including selected causes of disease into the cost calculation might not be the optimal approach. First, although strong evidence indicates associations between several morbidity outcomes and insufficient physical activity, overall data on morbidity are still less strong than data on mortality. Second, data on morbidity are more prone to reporting and classification error, whereas data on mortality are usually more easily available and more reliable. Third, selecting only a subset of disease outcomes underestimates the costs related to insufficient physical activity, since they are also likely to be associated with other health outcomes than those selected.

Thus, an alternative approach could be taken: instead of calculating new estimates for different health outcomes, a ratio could be applied to the mortality costs since a significant proportion of total health costs can be safely assumed to be due to morbidity. For example, a burden-of-disease study (Public Health Group, 2005) showed that, except for stroke, the morbidity measured in years of life with disability due to the main causes was between 1.2 and 8.8 times as high as the burden associated with mortality (Table 23). The costs from all-cause mortality could therefore be multiplied by a certain ratio to produce a combined estimate of costs related to mortality and morbidity. Although the costs of morbidity strongly depend on the income level of a country and the type and organization of the health system, a ratio of 1:1 could be seen as a reasonable approach.

Unfortunately, none of the reviewed economic studies on physical (in)activity (see Table 3 and Annex 1, section I.4.4) allowed a similar comparison, as none used years of life (lost or with disability) as measures for both morbidity and mortality as did the study in Australia (Public Health Group, 2005). In any case, appropriate sensitivity analysis should be carried out when this approach is applied.

Table 23. Overview of the share of morbidity and mortality attributed to selected causes associated with physical inactivity in a study in Victoria, Australia

Causes	Morbidity Years of life with disease	Mortality Years of life lost	Factor
Ischaemic heart disease	60 790	52 986	1.2
Stroke	13 141	20 618	0.6
Road crashes	11 505	9 306	1.2
Diabetes	29 183	8 565	3.4
Mental health[a]	53 436	6 048	8.8
Hypertensive heart disease	Not available	1 436	-
Breast cancer	16 182	9 797	1.7
Total	184 237	108 756	1.7

[a]Alzheimer's disease or other dementia or depression.
Source: Public Health Group (2005).

3.5.4 Special focus on children

More recently, evidence has become available for a relationship between physical activity and/or fitness and health in children as well as a clustering of risk factors, particularly cardiovascular ones. According to recent reviews (see section Annex 1, I.4.2), the evidence base is strongest for the beneficial effects of physical activity on aerobic fitness, musculoskeletal health, weight loss among overweight youth and cardiovascular and metabolic health. However, the vast majority of epidemiological studies have been conducted on adults and as yet, very little evidence on exposure–response relationships for children is available. No specific dose–response functions for calculating the costs transport-related insufficient physical activity among children can therefore be suggested.

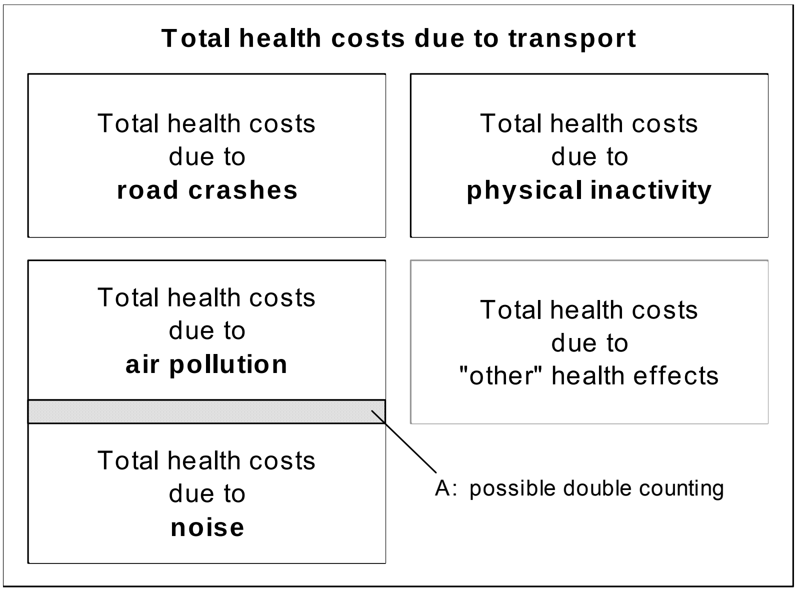
4 Bringing it all together

4.1 Total costs of all health effects together

Chapter 3 presented practical approaches on how to measure total health costs[37] due to road crashes, transport-related air pollution and transport-related noise. Key issues related to transport-related insufficient physical activity were discussed. This chapter discusses how to estimate total health costs due to road transport considering several different health effects.

As illustrated in Fig. 14, the overall health costs due to road transport comprise total health costs due to the four components discussed in this report: road crashes, air pollution, noise and insufficient physical activity. A fifth component is health costs due to other health effects. Annex 1, section I.5 presents a list of such possible "other" health effects as discussed in some of the texts reviewed. However, further health effects due to road transport probably occur (such as negative effects on the development of motor skills of children due to reduced opportunities for free play and physical activity in some urban environments with heavy road traffic); however, they are not discussed in further detail.

Fig. 14. Bringing it all together: total health costs due to transport



The total health costs due to transport seem to be the sum of the total health costs of each health effect. However, two parts of these health costs, those related to air pollution and to noise, might overlap to some extent (dashed area A in Fig. 14) if cardiovascular effects are included as health end-points for both air

[37] The term "total health costs" refers in the whole text to the sum of all different costs of a health effect considered in this study. As some costs of a health effect are not considered in this study, for example because they cannot yet be monetized, the term "total health costs" as used here does not cover all possible total health costs that occur in reality.

pollution (suggested, see sections 3.4.2 and Annex 1, I.3.3) as well as noise (for indicative estimates, see sections 3.3.2.2 and Annex 1, I.2.2). Hence, as indicated by some empirical studies, summing up these two health effects might count the number of people with cardiovascular diseases twice since the same people are usually exposed to both transport-related air pollution and noise. The available evidence does not yet allow clear allocation of the share of the cardiovascular effects to either of these types of exposure. The risk of a certain double counting therefore has to be kept in mind if cardiovascular effects due to noise are included as a health end-point in cost calculations. Further research is needed to clarify this issue.

As discussed in Annex 1, section I.4.2, the lack of transport-related physical activity not only probably causes direct health effects but probably contributes indirectly to higher health costs due to more road crashes, increased air pollution and a higher noise level. With regard to economic valuation of health effects, however, these side effects are accounted for in the costs of the other topics and no further adjustments are necessary.

Besides double counting, a further point has to be considered in calculating the total health costs due to road transport. Obtaining consistent results requires that all health cost components be estimated on the same geographical scale (for example a region or a country) and for the same time period. Otherwise the cost figures will not be comparable and cannot be added.

Share of total health costs: an example

A recent study on total transport-related health costs in Switzerland found that the costs due to road crashes and to air pollution were approximately equal, and the health costs due to noise were about half these amounts (Federal Office for Spatial Development, 2007). The study did not calculate the share of transport-related insufficient physical activity in the total health costs due to transport. These results might differ for other countries or regions due to differences in the road traffic situation, in the judicial system and laws and in the types of motor vehicles involved.

Summary of the practical examples

Table 25 summarizes the results of practical examples of calculations of health costs from road crashes and road transport–related noise and air pollution in sections 3.2.6, 3.3.5 and 3.4.6. The calculated examples of health costs due to road crashes and road transport–related noise and air pollution in Switzerland in 2005 amounted to US$ 7345 million. This only takes into account the costs related to motorized road transport and not the costs of road crashes related to bicycles, pedestrians, tractors and work machines. Based on Ecoplan & Infras (2008), including these costs would increase the total by US$ 2215 million or 30%. Thus, the conclusions drawn below are only valid for motorized transport and not for total transport.

Road crashes are clearly the dominant source of health costs due to motorized road transport (77%). The remaining costs are more or less evenly split between air pollution and noise, with air pollution being somewhat more important. Compared with road crashes, the total costs as well as the costs per vehicle-km were on average about one seventh for noise. The difference was particularly large for mopeds (a factor of more than 400). For cars, the most important vehicle category, the factor was 10. Only for lorries exceeding 3.5 tonnes were noise costs marginally higher than crash costs.

For air pollution and noise, the total health costs (or the costs per vehicle-km) were comparable. For freight transport, air pollution–related costs were about 60% higher than noise-related costs (for all vehicle categories), and the air pollution–related costs of passenger transport were 6% lower than noise-related costs. This was mainly due to motorbikes, which caused about 10 times higher noise-related costs than air pollution–related costs. For cars, air pollution–related costs were 26% higher.

Table 24. Summary of health costs due road crashes and traffic-related air pollution and noise in Switzerland, 2005

	Passenger transport								Freight transport				Total
	Car	Public bus	Trolley	Tram	Private coach	Motor-bike	Moped or scooter	Total	Delivery van	Heavy goods vehicle	Articulated lorry	Total	
Costs in millions of US dollars													
Road crashes	3675		53[a]		119	923	438	5208	251	113	54	419	5627
Air pollution	461	33	3	NA	8	19[b]		523	126	176	91	393	916
Noise	365	18	0	1	9	165	1	559	72	114	57	243	802
Total	4470		108[a]		135	1547[b]		6290	449	404	202	1054	7345
Costs in US dollars per vehicle-km								Average					Average
Road crashes	0.071		0.177[a]		1.12	0.449	2.99	0.095	0.076	0.079	0.077	7.7	0.094
Air pollution	0.009	0.143	0.096	N.A.	0.073	0.009[b]		0.010	0.038	0.124	0.129	7.2	0.015
Noise	0.007	0.08	0.007	0.022	0.08	0.080	0.007	0.010	0.022	0.080	0.08.0	4.5	0.013
Total	0.087		0.361		1.273	0.701[b]		0.115	0.14	0.283	0.286	18.9	0.122

Cycles, pedestrians, tractors and work machines, which have been taken into account for crashes, are disregarded here as no data for air pollution and noise are available for these vehicle categories.
NA: not available.
[a] average of public and trolley buses and tram
[b] average of motorbike and moped or scooter

4.2 Conclusions

The practical approach presented in this report has several implications of possible interest:

- it combines state-of-art knowledge in the fields of economics and epidemiology in relation to transport-related health effects;
- it facilitates the inclusion of health effects in economic valuations of interventions and policies related to transport;
- it can be used by non-health experts;
- it is applicable to different spatial scales, as well as to different types of studies and research objectives. For example, in addition to estimating total health costs due to road transport, this approach can also be used to evaluate the effect of a specific measure on people's health (such as introducing a different speed limit for road traffic), as well as to measure health costs in a cost–benefit analysis for a road transport project; and
- its modular approach can fit different studies needs, and support the valuation of different health effects related to transport either separately, or in combination, providing a coherent framework for the combination of costs arising from different effects.

Although several questions for further research were identified and uncertainty exists on several issues, this approach will derive values based on the best available evidence that will show the dimension of these costs.

Transport-related health effects include also effects for which monetization is not feasible based on current knowledge (such as community severance and other mental effects) (THE PEP, 2004d). These aspects are important, as they are often implicit determinants of preferences and choices. Their influence should be acknowledged and captured in assessments of transport interventions even if economic tools cannot be used, for example by applying qualitative approaches.

I Annex 1: Overview and analysis of recent literature

The first step in developing a practical approach for the economic valuation of health costs due to transport is to analyse the most important recent research studies and reviews in this field. This chapter summarizes the results of this analysis. For each category of health costs considered – transport crashes, air pollution, noise and insufficient physical activity and a group of "other health effects" – the results of the literature were compared and discussed, first regarding possible health end-points to be included based on the epidemiological literature and then regarding recent approaches taken in economic valuation. The overview presents the different methods applied so far and the research gaps and open issues.

I.1 Overview of the literature on the costs of transport crashes

I.1.1 Introduction

Historically, the relationship between transport activities and the deaths and injuries resulting from crashes has been the first transport-related health effect to be clearly and unambiguously identified and the only one for which all European countries produce and share statistics on the number of deaths and injuries attributable to road crashes, frequently disaggregated by sex, age and mode of road transport involved in the crash. Road traffic injuries are also the first transport-related health effect economists studied, and many studies exist providing national and international estimates of the costs of road traffic injuries.

Injury accidents in road transport included in official statistics are internationally defined as "any accident involving at least one road vehicle in motion on a public road or private road to which the public has right of access, resulting in at least one injured or killed person" (UNECE Intersecretariat Working Group on Transport Statistics, 2003). As for outcomes of accidents, a "person killed" includes "any person killed immediately or dying within 30 days as a result of an injury accident", and a "person injured" is defined as "any person not killed, but who sustained an injury as result of an injury accident, normally needing medical treatment".

Although road transport statistics are relatively easily available across Europe, several methodological issues need to be taken into account when estimating the number of deaths and injuries resulting from road crashes. First, countries interpret "injured" differently, which makes international comparisons of statistics on non-fatal outcomes of road crashes difficult. Second, statistics on road traffic injuries are available from different data sources, notably road police statistics, health authorities (based on death certificates and on records from health care systems) and insurance companies. These data sets are often inconsistent: studies from different European countries indicate that road police data tend to significantly underestimate the non-fatal outcomes, particularly single-road-vehicle crashes involving cyclists, who often are treated by emergency and health care facilities without being recorded by road police (Racioppi et al., 2004; WHO, 2006).

Several reviews have shown the extent of the burden of road traffic injuries, both at the global and European levels (Peden et al., 2004; Racioppi et al., 2004; Sethi et al., 2006).

I.1.2 Health end-points

I.1.2.1 Severity of road traffic injuries as a function of mode of transport

The mode of transport used influences the severity of the injury sustained. The risk of dying for users of motorized two-wheelers is 20 times that of car occupants on average. The risk for cyclists and pedestrians per distance travelled is 7–9 times those for motor vehicle occupants (Peden et al., 2004a). However, two thirds of crashes occur in urban areas, where there is a greater mixture of vulnerable road users and motor vehicles. The level and proportion of mortality among pedestrians vary between countries, reflecting differences in both exposure and safety. They are lowest in the Nordic countries and highest in eastern Europe, the Caucasus and central Asia and the Baltic countries. Pedestrian safety should be considered because children and older people are more vulnerable to sustaining severe injuries when struck. Older people account for nearly half the pedestrian deaths in the WHO European Region (Sethi et al., 2006).

For non-fatal injuries, accident statistics distinguish between minor and severe injuries. However, the respective definitions can differ significantly between countries, which makes comparison difficult (Table 8).

I.1.2.2 Risk factors

The main risk factors for road traffic injuries are speed, alcohol, exposing vulnerable road users to motorized traffic, poor visibility and not using protective equipment (Racioppi et al., 2004).

The probability of a pedestrian being killed increases eightfold as the speed of impact rises from 30 to 50 km/h (Pasenen & Salmivaara, 1993).

Alcohol is an important risk factor for all road users, and young drivers and passengers 18–25 years old are particularly at risk of crashing (Mathijessen, 2002). The likelihood of crashing increases with blood alcohol concentration, especially at concentrations above 0.04 g/dl (Racioppi et al., 2004). At a concentration of 0.08 g/dl, the risk is twice that at 0.05 g/dl and, at 0.10 g/dl, it is three times as high (Mathijessen, 2002). The legal limit is 0.05 g/dl in most countries. Driving under the influence of drugs is also a risk factor, and mixing drugs with alcohol amplifies driving impairment (Racioppi et al., 2004).

I.1.2.3 Road traffic injuries among children and young people

An estimated 32 000 children and young people 0 24 years old lost their lives to road traffic injuries in 2002 (Sethi et al., 2007). Vulnerability to road crash deaths increases with age, being highest among people 20–24 years old. Three fourths of the people 0–24 years old who are killed in road crashes are male, and the increased risk for males relative to females increases with age. The increase with age reflects changes in exposure to risk resulting from differences in travel patterns.

Compared with the general population of Europe, more people younger than 25 years who die from road crashes die as car users (54% of road traffic deaths versus 47% for the general population) and as users of motorized two-wheelers (17% of road traffic deaths versus 11% for the general population).

However, closer examination of the mortality data in Europe by type of road user reveals important differences in mortality, indicating changes in exposure and risk from childhood into adolescence and young adulthood. Among children younger than 15 years, the leading causes of road traffic deaths are as pedestrians (48%), followed by car occupants (32%) and bicycle riders and passengers (8%). At 15–24 years, this changes considerably, and the leading causes become as car drivers or occupants (59%), riders and passengers of motorized two-wheelers (19%) and pedestrians (17%). These differences reflect greater exposure to risk as pedestrians and cyclists among children and as car drivers or occupants and riders and passengers of motorized two-wheelers among those 15–24 years old (Sethi et al., 2007).

Table 25. Overview of selected health end points related to road traffic injuries among children and adults

Health end-point	Weight of the evidence	Existence of biologically plausible mechanism(s)?	Sufficient information to allow quantification?	Valid exposure–effect relationship available?
Mortality from road traffic injury	Sufficient	Yes	Yes	Yes
Non-fatal injuries[a]	Sufficient	Yes	Yes	Yes
Post-traumatic stress disorders	Sufficient	Yes	No	No

[a]ICD-9 codes E810–E819, E826–E829 and E929.

I.1.3 Economic approaches

Nine of the 38 studies analysed for this report focused on health costs due to transport crashes (see Table 1).

I.1.3.1 The model used in the studies analysed

All nine studies basically used the same theoretical model for calculating health costs due to transport crashes:

Total health costs[38] = number of transport crashes × cost of transport crashes

However, the studies also differed in the details of this calculation. The following sections discuss these differences. The concluding section I.1.3.9 focuses on the special situation of children in transport crashes.

Three of the studies discussed here were literature reviews themselves (Swedish National Road and Transport Research Institute, 2006; THE PEP, 2004a; Van Essen et al., 2007). Hence, the information they give might not relate to one study or text but to several.

I.1.3.2 Types of road crashes considered

The theoretical model shows that the number of road crashes is key for calculating total health costs. In addition to the “normal” road crashes involving two or more vehicles in the nine studies analysed, five other types were mentioned: single-vehicle crashes, suicides, road crashes involving only pedestrians and cyclists, road crashes due

38 The term “total health costs” refers in the whole text to the sum of all different costs of a health effect considered in this study. As this study does not consider some costs of a health effect, for example, because they cannot yet be monetized, the term “total health costs” as used here does not cover all possible total health costs that occur in reality.

to construction and cases not reported in official statistics (Table 26). Three texts included single-vehicle crashes but not suicides and road crashes due to construction (Table 26). Road crashes involving only pedestrians and cyclists were not in cluded or only partly included in most studies analysed (see Table 26: Ecoplan, 2002b included road crashes involving cyclists only but not the ones involving pedestrians only). Three of the studies considered cases not reported in official statistics.

Table 26. Types of road crashes considered in the studies on road crashes

	Study								
	Van Essen et al. (2007)	DIW et al. (2000)	Ecoplan (2002a)	Ecoplan (2002b)	IER (2006)	Infras & IWW (2004)	Swedish National Road and Transport Research Institute (2000)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)
Literature review – more than one study considered	x							x	x
Road crashes considered									
Single-vehicle crashes		+	+	+		+			
Suicides		–	–	–		–	–		
Road crashes involving only pedestrians and cyclists		–	(+)	(+)			–		
Road crashes due to construction		–	–	–			–		
Cases not reported in official statistics		–	+	+	+	+			

+ : included
– : excluded (explicitly)
: no information given
( ) : special case or exception

I.1.3.3 Mode of transport considered

For the number of transport crashes, not only the nature of the transport crash itself is of importance (see section I.1.3.2) but also the modes of transport considered. Table 27 shows that nearly all studies reviewed considered rail transport in addition to road transport. Two studies included sailing and fishing, whereas two studies excluded water transport. Which modes should be included is mainly a question of data availability and the importance of a certain mode of transport for the specific country. This is also true for air transport, which two studies considered (only private air transport).

Table 27. Modes of transport considered in the studies on transport crashes

	Study								
	Van Essen et al. (2007)	DIW et al. (2000)	Ecoplan (2002a)	Ecoplan (2002b)	IER (2006)	Infras & IWW (2004)	Swedish National Road and Transport Research Institute (2000)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)
Literature review – more than one study considered	x							x	x
Mode of transport considered									
Road transport		+	+	+	+	+	+	+	+
Rail transport		+	+	+		+	+	+	
Air transport		(+)	–	–			(+)		
Sailing and fishing		+					+		
Water transport			–	–					

+ : included
– : excluded (explicitly)
: no information given
( ) : special case or exception

I.1.3.4 Severity of injuries

The next point of analysis concerned the severity of injuries, which is crucial for valuating the transport crash costs. However, the nine transport crash studies did not differ much (Table 28): most texts distinguished between fatality, minor and serious injury and crashes involving only material damage.

Table 28. Different grades of severity of injuries considered in the studies on transport crashes

	Study								
	Van Essen et al. (2007)	DIW et al. (2000)	Ecoplan (2002a)	Ecoplan (2002b)	IER (2006)	Infras & IWW (2004)	Swedish National Road and Transport Research Institute (2006)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)
Literature review – more than one study considered	x							x	x
Severity of injury									
Crashes involving only material damage		+	+	+	+		+		
Minor injury: no hospital treatment required or the effect of the injury subsides quickly		+	+	+	+	+	+		
Serious injury: requires hospital treatment or lasting injuries		+	+	+	+	+	+		
Fatality		+	+	+	+	+	+		

+ : included
– : excluded (explicitly)
  : no information given
( ) : special case or exception

I.1.3.5 Costs considered: injuries

Two main aspects are considered in valuating the costs of transport crashes.[39] The first part includes the direct and indirect costs of a transport injury. As Table 29 shows, this includes the costs of health care, economic production losses, material damage and administration costs. One study considered travel delays or congestion costs. However, other studies exclude them (or did not mention them). Three studies explicitly excluded cost for risk avoidance, and the other studies did not explicitly include this.

The second part of the total health cost is the intangible costs for the victim and for friends and relatives. As shown in Table 29, all nine studies included the intangible costs for victims but not for friends and relatives.

[39] The legislative framework of liability insurance companies is essential to define the relevant elements of the costs (for example, transfer payments) (DIW et al., 2000). Hence, a slightly different set of cost categories might be included for each country.

Table 29. Types of costs considered in the studies on transport crashes

	Study								
	Van Essen et al. (2007)	DIW et al. (2000)	Ecoplan (2002a)	Ecoplan (2002b)	IER (2006)	Infras & IWW (2004)	Swedish National Road and Transport Research Institute (2000)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)
Literature review – more than one study considered	x							x	x
Costs considered									
Direct and indirect costs									
Costs of health care (including rehabilitation and reintegration)	+	+	+	+	+	+	+	+	+
Economic production losses (including replacement costs)	+	+	+	+	+	+	+	+	+
Material damage	+	+	+	+	+	+	+	+	+
Administration costs	+	+	+	+	+	+	+	+	+
Travel delays and congestion costs		+	–	–			–		
Costs for avoiding risk			–	–			–		
Intangible costs									
For victims (suffering and grief)	+	+	+	+	+	+	+	+	+
For relatives and friends							–		

\+ : included
– : excluded (explicitly)
  : no information given
( ) : special case or exception

I.1.3.6 Method applied to measure the costs

As explained in section 3.1.1, costs can be measured in two ways: the cost-of-illness or the willingness-to-pay approach. Either of these methods can be applied separately, which underestimates the real costs, or the two methods can be combined. All the studies reviewed on transport crashes combined the methods (Table 30).

Table 30. Different ways of measuring costs in the studies on transport crashes

	Study								
	Van Essen et al. (2007)	DIW et al. (2000)	Ecoplan (2002a)	Ecoplan (2002b)	IER (2006)	Infras & IWW (2004)	Swedish National Road and Transport Research Institute (2000)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)
Literature review – more than one study considered	x							x	x
Measuring costs									
Cost of illness and willingness to pay		+	+	+	+	+	+	+	+
Willingness to pay only									

+ : included
– : excluded (explicitly)
: no information given
( ) : special case or exception

I.1.3.7 Economic production losses: net versus gross values

This section on economic production losses is linked to the discussion in section I.1.3.6 on methods of measuring costs. As explained in section 3.1.1, calculating total costs by combining both methods (willingness to pay and cost of illness) poses a risk of counting lost consumption twice. Hence, all studies using this valuation approach must examine whether economic production losses are using net (excluding lost consumption) or gross values (including lost consumption). The information from the nine studies on transport crashes indicates that all used net economic production losses (Table 31), therefore ruling out any double counting of lost consumption.

Table 31. Net or gross economic production losses considered in the studies on transport crashes

	Study								
	Van Essen et al. (2007)	DIW et al. (2000)	Ecoplan (2002a)	Ecoplan (2002b)	IER (2006)	Infras & IWW (2004)	Swedish National Road and Transport Research Institute (2000)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)
Literature review – more than one study considered	x							x	x
Economic production losses: net versus gross									
Net, without the value of lost consumption		+	+	+	+	+	+		
Gross, including the value of lost consumption									

+ : included
– : excluded (explicitly)
: no information given
( ) : special case or exception

I.1.3.8 Valuating mortality

The last point of analysis deals with how to valuate mortality. There are two approaches: the willingness of a middle-aged person to pay to avoid a death (value of a statistical life), and the second one is based on the life expectancy of a victim attaching a price to each life-year lost by using willingness-to-pay figures (value of life-years lost). All nine studies valuated mortality by using the value of a statistical life method (Table 32).

Table 32. Ways of valuating mortality in the studies on transport crashes

	Study								
	Van Essen et al. (2007)	DIW et al. (2000)	Ecoplan (2002a)	Ecoplan (2002b)	IER (2006)	Infras & IWW (2004)	Swedish National Road and Transport Research Institute (2000)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)
Literature review – more than one study considered	x							x	x
Method used in valuating mortality									
Value of a statistical life	+	+	+	+	+	+	+	+	+
Value of life-years lost									

+ : Included
– : excluded (explicitly)
: no information given
( ) : special case or exception

I.1.3.9 Specific focus on children

Only two studies (Ecoplan, 2002a, b) included children in calculating the health costs due to transport crashes and valuated the life-years lost among children. All the other studies either treated children as adults, probably in most cases (but there was not always a clear statement in this respect) or left them out altogether from the calculations. Only THE PEP (2004a) referred to children, saying that the willingness to pay, and therefore the intangible costs, cannot be measured for children as they cannot properly express their willingness to pay to avoid a certain risk. However, as most data on road crashes are differentiated by age or age groups, the costs of road crashes for children can be calculated separately. In this case, the value of life-years lost available for adults can be applied to children while taking their longer life expectancy into account (Ecoplan, 2002a, b).

I.1.3.10 Comparing results

Given the variety of approaches applied in the studies (based on the discussion above), it is not surprising that the studies reviewed differ substantially in the calculated total health costs due to transport crashes. Beside these differences, the studies vary according to the geographical area considered (such as countries, regions or communities), the year of the survey, further definitions and assumptions (such as external costs or the internalization of risks) and the type of costs measured (such as marginal or total health costs). All four dimensions exert additional influence on the final assessment of total health costs due to transport crashes. However, to give an idea of the dimension of these costs but also to show their variance, Table 33 presents the results of two studies.

Table 33. Overview of selected results for total health costs due to transport crashes

Source	Mode	Geographical area	Year	Definitions and assumptions	Average health costs (in euros per passenger-km)	Total health costs (in millions of euros)
Infras & IWW (2004)	Road transport	EU-15 plus Norway and Switzerland	2000	Only passenger cars	0.0309	114 191.0
	Rail transport	EU-15 plus Norway and Switzerland	2000	Only passenger trains	0.0008	262.0
Ecoplan (2002a)	Road transport	Switzerland (entire country)	1998	Exchange rate Sw.fr./€ 0.6, only passenger cars	0.0534	1 395.6
	Rail transport	Switzerland (entire country)	1998	Exchange rate Sw.fr./€ 0.6, for average health costs: only passenger trains	0.0042	79.0

I.2 Overview of results on transport-related noise

I.2.1 Introduction

In Europe, road traffic is an important source of community noise (Staatsen et al., 2004). Recent studies show that much of the population is exposed to road traffic noise: an estimated 20% of the population of the EU countries is exposed to noise exceeding 65 dB(A) during daytime and more than 30% is exposed to levels exceeding 55 dB(A) during night-time (WHO Regional Office for Europe, 2007a). Scientists consider such noise levels to adversely affect human health and well-being (Berglund et al., 2000).

I.2.1.1 Noise standards and threshold values

In 2002, the EU Directive on Environmental Noise (European Commission, 2002) was issued, aiming at harmonizing the assessment and management of environmental noise. It prescribes methods for noise calculation, noise indices and the development of strategic noise maps. The Directive does not provide specific noise standards: since they are regarded as a matter of subsidiarity, defining them is left to the individual EU countries (Nijland & van Wee, 2005). Noise standards differ across the EU countries since they are mainly based on noise assessment using national calculation models. Especially in recent years, noise exposure levels at the population level have been estimated by means of noise models incorporated in geographical information systems. These noise models predict equivalent noise levels at user-defined outdoor points. From this, conventional noise indicators (such as L_{den} and L_{night}) can be calculated. In some cases, monitoring systems are used to estimate population noise exposure. However, the results of noise measurements strongly depend on local factors such as the shielding and reflection of buildings and weather conditions (Fast, 2004).

Threshold values for health effects are based on scientific knowledge on the shape of exposure–response functions, and national noise standards also consider other regulatory factors. As discussed below, a threshold value or no-observed-effect level for noise exposure is still under discussion. The threshold is also likely to be different for different health end-points.

I.2.2 Health end-points

Long-term noise exposure is associated with several effects on health and well-being. These include not only community responses such as annoyance and sleep disturbance but also physiological effects such as effects on the cardiovascular system. The WHO definition of health as “a state of complete physical, mental and social well-being and not merely the absence of disease and infirmity” embraces the concept of well-being. Based on this definition, effects of noise such as annoyance and sleep disturbance are also part of the overall health effects. Based on recent reviews of the effects of environmental noise, a set of outcomes related to health and well-being that are often reported in relation to exposure to transport noise can be identified: behavioural responses such as coping strategies and complaints; social responses such as annoyance; acute physiological responses; cognitive responses such as task interference and learning; chronic physiological responses; and clinical effects including mental and cardiovascular symptoms (Babisch, 2006; Health Council of the Netherlands, 1994, 2004; Staatsen et al., 2004; Van Kempen et al., 2002; WHO, 2000; WHO Regional Office for Europe, 2007b).

Some of these effects on health and well-being have been found to be associated with long-term exposure to road traffic noise. For this publication, we used major peer-reviewed reviews (such as those of WHO and the Health Council of the Netherlands) to select the effects for which evidence is sufficient for an association with transport-related noise exposure. These include:

- annoyance
- sleep quality
- sleep disturbance
- insomnia
- hypertension and ischaemic heart disease
- reduced cognitive functioning.

Hearing loss or hearing impairment was not included as a possible health end-point for the economic valuation of transport-related health effects since typical exposure to road traffic noise is unlikely to cause hearing damage. The WHO community guidelines for noise state that hearing impairment is not expected to occur at L_{Aeq} 8-h levels at or below 75 dB(A). Environmental and leisure noise with an $L_{Aeq,\,24\,h}$ at or below 70 dB(A) is not expected to cause hearing impairment in the large majority of people, even after lifetime exposure (WHO, 2000).

I.2.2.1 Annoyance

Annoyance is one of the most widespread and well-documented responses to noise. It is a collective term for several negative reactions such as irritation, dissatisfaction or anger, which occur when noise disturbs someone’s daily activities (WHO, 2000). In observational studies, annoyance is usually measured by means of one or more questions as part of a questionnaire or interview.

However, the answer categories from which the respondents can choose for reporting their degree of annoyance differ between studies. Although the percentage of highly or severely annoyed respondents is usually reported, studies sometimes report the mean annoyance or the percentage annoyed. The International Commission on Biological Effects of Noise and the International Organization for Standardization have recently made efforts towards the use of standardized questions about the degree of annoyance. To determine the percentage of “annoyed” and “highly annoyed” people, cut-off values of 50 and 72 on a scale of 0 to 100 are often used, respectively (Miedema & Oudshoorn, 2001). In estimating the disease burden (expressed in DALYs), only the number of severely annoyed respondents is included (Knol & Staatsen, 2005), considering "severe annoyance" as a health state in which people's daily functioning might be affected.

For adults, the dose–response relationship for the association between road traffic noise and annoyance, derived from a meta-analysis by Miedema & Oudshoorn (2001), is the best available. Based on this study, the percentage of people "severely" annoyed by road noise as a function of the outdoor L_{den} at the most exposed façade can be derived. Although the relationship by Miedema & Oudshoorn was derived from probably the most extensive international database currently available, methodological differences in the original studies may have influenced the observed relationship. Despite these methodological issues, the relationship is suitable to assess the fraction of people highly annoyed due to road traffic noise in a strategic assessment.

I.2.2.2 Effects on sleep

Night-time noise can affect people's sleep. These effects can manifest themselves in various ways: in the sleeping behaviour, in the structure of the sleep, as physiological responses or as effects during the day. Three types of studies investigate the relationship between night-time noise exposure and sleep: (1) studies on the reactions to noise events; (2) studies on the effects before, during and after a night of sleep (investigating how night-time noise during a sleep period affects the sleep stages, sleep quality and awakening); and (3) studies on the effects of long-term noise exposure on health and well-being (such as decreased sleep quality, sleep disturbance and insomnia) (Van Kempen et al., 2005).

This report solely focuses on the effects of long-term exposure to road traffic noise (expressed as L_{night}: the equivalent sound level during night-time (23:00–07:00) on health and well-being. Three effects with sufficient evidence for an association with road traffic noise were identified: decrease in sleep quality, sleep disturbance and insomnia.

A decrease in sleep quality includes difficulty in falling asleep, difficulty in sleeping through the night, waking up during the night and shorter sleeping time (Fast, 2004; Health Council of the Netherlands, 2004). Unfortunately, no exposure–response relationships are available yet for the association between long-term exposure to road traffic noise and a decrease in sleep quality.

Insomnia can be defined as the inability to sleep or abnormal wakefulness. Three studies have investigated the association between exposure to night-time noise from road traffic and insomnia (Kageyama et al. 1997; Kawada et al. 2003; Passchier-Vermeer et al. 2007). The results of these studies are inconsistent. Kageyama et al. (1997) found an exposure–response relationship, but the other two studies did not. However, since Kageyama et al. (1997) only studied women, the generalization of the exposure–response relationship to other situations remains questionable.

Sleep disturbance is regarded as a particular form of annoyance and is usually measured by means of a direct question as part of an interview or questionnaire. Similar to annoyance, sleep disturbance questions vary between studies, not only the number of response categories but the wording (Van Kempen et al., 2005).

For adults, the relationship for the association between noise from night-time road traffic (L_{night}) and sleep disturbance derived by Miedema et al. (2003) is the best currently available. It is based on reanalysis of individual data from 14 studies (11 from Europe, 2 from Canada and 1 from Japan) from 1975 to 2001. The exposure–

response functions derived provide the percentage of highly sleep-disturbed, sleep-disturbed and (at least) slightly sleep-disturbed people due to road traffic noise as a function of the outdoor L_{night} at the most exposed façade.

For annoyance, only the number of severely sleep-disturbed people has been included in calculations of the disease burden (Knol & Staatsen, 2005).

I.2.2.3 Effects on the cardiovascular system

The biological mechanism of noise exposure leading to cardiovascular effects seems plausible, although it is quite complex: noise-induced cardiovascular effects can be regarded as the consequence of stress, which can arise in several ways in relation to noise. Studies investigating the effects of noise exposure on the cardiovascular system have reported a broad range of effects: (1) differences in blood pressure; (2) changes in the occurrence (prevalence and incidence) of hypertension, angina pectoris and myocardial infarction; and (3) changes in the number of hospital admissions, medication use, visiting a general practitioner or specialist and/or mortality due to cardiovascular disease. The effects found were usually small (Babisch, 2006; Van Kempen et al., 2005).

The observed effects on the cardiovascular system might also be attributable to factors other than exposure to road traffic noise. The observed associations between exposure to road traffic noise and cardiovascular disease could also be explained by the exposure to transport-related air pollution (Grazuleviciene et al., 2004; Rosenlund et al., 2006).

Some risk estimates for the association between road traffic noise and myocardial infarction (ICD-9: 410) are available and could be indicatively used for health impact assessment (Babisch, 2006; Van Kempen et al., 2002).

I.2.2.4 Conclusions

Sufficient evidence indicates that noise exposure is associated with annoyance and sleep disturbance among adults. As the most reliable exposure–response functions are available for severe annoyance and severe sleep disturbance, it is suggested to use these outcomes as health end-points rather than annoyance and sleep disturbance in general. The evidence for cardiovascular disease risk is limited, although a biological mechanism is plausible (Table 34).

Although effects will occur at lower levels based on the available exposure–response function, for practical reasons, 50 or 55 dB(A) (L_{den}, outdoors) could be used as a threshold value for health impact assessment (see section 3.3) for severe annoyance and severe sleep disturbance. Evidence is not yet sufficient to define a clear threshold value for the relationship between exposure to transport noise and cardiovascular disease (see section 3.4.2.3). In a meta-analysis, Babisch (2006) assumed that exposure to road traffic noise has no effects below 60 dB(A) (LAeq, 6–22 h). So researchers deciding to indicatively estimate the number of incident cases of myocardial infarction could use the relationship derived by Babisch and apply the threshold of 60 dB(A) (LAeq, 6–22 h).

Table 34. Overview of selected health end-points related to exposure to road traffic noise in adults

Health end-point	Exposure indicator	Strength of the evidence[a]	Valid exposure-effect relationship available?
Severe annoyance	L_{den}	Sufficient	Yes
Sleep quality	L_{night}	Sufficient	No
Severe sleep disturbance	L_{night}	Sufficient	Yes
Insomnia	L_{night}	Sufficient	No
Ischaemic heart disease[b] (including myocardial infarction)	$L_{Aeq,\ 16\ h}$	Limited	Only indicatively[c]

[a] Strength of the evidence for an association with noise exposure according to WHO (2000) and/or the Health Council of the Netherlands (1994, 2004).
[b] ICD-9 codes 410–414.
[c] Exposure–effect relationships are being updated. Until this update is available, the relationships of Van Kempen et al. (2002) and/or Babisch (2006) can be used (see also section 3.3.2.2).

I.2.2.5 Health effects among children

In past decades, substantial research has focused on the effects of noise on children. A broad range of effects has been reported on hearing, cognition, motivation and the cardiovascular and endocrine systems. Effects on mental health, annoyance, self-reported health and sleep have also been investigated (Table 18). Only a few studies investigated the effects of noise exposure on congenital abnormalities, birth weight or disorders related to the immune system (Bistrup et al., 2001).

Most studies focused on the effects or air transport noise; only a few studies have investigated the effects of road traffic noise. The main effects investigated were cognitive functioning, annoyance and blood pressure (Cohen et al., 1986; Lercher, 2003; Stansfeld et al., 2000).

Only a few studies investigated the effects of exposure to road traffic noise on children's cognitive functioning and blood pressure (Van Kempen et al., 2006). These included mainly cross-sectional studies. The conclusions that can be drawn from these studies are limited and inconsistent, and no exposure–effect relationships are available.

Although exposure–effect relationships for noise annoyance among adults have been widely studied, only a few studies have assessed residential annoyance in children quantitatively and systematically (Boman & Enmarker, 2004; Evans et al., 1995, 1998; Haines et al., 2001a, b, c; Lercher et al., 2000; Stansfeld et al., 2005; Sukowski et al., 2000). In these studies, children living in noisier areas in their community were significantly more annoyed by noise than children living in quieter areas. Two studies derived exposure–response relationships for the association between road traffic noise and annoyance (Lercher et al., 2000; Stansfeld et al., 2005). In addition, both studies demonstrated that the shape of the curve describing the relationship between noise exposure and annoyance in the mothers was very comparable with that of their children. In both studies, the mothers were more annoyed at higher noise levels. Since similar questions were used to measure annoyance among children and their parents, these findings may be regarded as relatively significant. In addition, there are also indications that noise annoyance among children pertains to the same constructs as among adults (Boman & Enmarker, 2004; Haines et al., 2003).

There is almost no knowledge on how exposure to road traffic noise affects children's sleep. Very few studies are available on how sensitive children are to the effects of road traffic noise compared with adults. The results of these studies are inconsistent.

Table 35. Overview of selected end-points related to traffic noise exposure in children

Health end-point	Strength of the evidence[a]	Valid exposure–effect relationship available?
Annoyance	Sufficient	No
Cognitive functioning	Limited to sufficient	No

[a] Strength of the evidence for an association with noise exposure according to WHO (2000) and/or the Health Council of the Netherlands (1994, 2004).

In conclusion, sufficient evidence indicates that exposure to road traffic noise is associated with annoyance in children. The evidence for an association between road traffic noise and effects on cognitive functioning is more limited (Table 35).

I.2.3 Economic approaches

Ten studies focused on health costs due to transport-related noise (Table 1).

I.2.3.1 Model used in the studies analysed

To calculate transport-related noise costs, 10 studies on noise costs reviewed all used the following equation:

Total health costs[40] = number of people or cases with health effects due (or attributed to) to exposure to transport-related noise × costs per case

However, the studies differed clearly in the calculation of total health costs. Sections I.2.3.2 to I.2.3.11 discuss these differences. Section I.2.3.12 focuses on the special situation of children in the context of transport-related noise.

The four studies by Infras & IWW (2004), Swedish National Road and Transport Research Institute (2006), THE PEP (2004a) and Van Essen et al. (2007) are literature reviews themselves, so they provide information based on more than one study. Some results presented from these studies therefore seem to contradict each other (two ways of dealing with annoyance by THE PEP (2004a) as shown in section I.2.3.7). However, this is only due to information coming from different sources discussed in these reviews (that is, one text considered only health effects and excluded annoyance; the other one treated annoyance as one of the health effects).

I.2.3.2 Dispersion models considered

To calculate total noise costs, the number of people exposed to different levels of noise has to be known, which depends on the emission and dispersion of noise. Noise emissions are estimated based on the characteristics of

40 The term "total health costs" refers in the whole text to the sum of all different costs of a health effect considered in this study. As this study does not consider some costs of a health effect, for example, because they cannot yet be monetized, the term "total health costs" as used here does not cover all possible total health costs that occur in reality.

road traffic. Several models estimate the dispersion of noise. In the 10 studies on noise reviewed, four models were mentioned (Ecoplan et al., 2004b; IER et al., 2000, 2003) (Table 36).[41],[42]

Table 36. Models of dispersion considered in the studies on noise

	Study									
	DIW et al. (2000)	Ecoplan et al. (2004b)	IER (2006)	IER et al. (2000)	IER et al. (2003)	Infras & IWW (2004)	Navrud (2002)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)	Van Essen et al. (2007)
Literature review – more than one study considered						x		x	x	x
Models of dispersion considered			(+)				(+)		(+)	
RLS90					+					
Schall03					+					
CadnaA		+								
Weather factors				+						

+ : included
– : excluded (explicitly)
: no information given
( ) : special case or exception

I.2.3.3 Threshold used: daytime

Another important factor concerning the number of people exposed to noise is the threshold used, indicating the point at which sound emissions become annoying for people and therefore cause costs. As Table 37 shows, the studies reviewed used four different thresholds: 50, 55, 60 and 65 dB(A),[43] respectively (all values refer to outdoor measurements at the front of a building). Given that an increase from 50 to 55 dB(A) might cause at least a doubling of noise costs because the decibel scale is logarithmic (Ecoplan et al., 2004b; Van Essen et al., 2007) a threshold difference of 5 or 10 dB(A) is important for the total health cost figures. In the studies by Ecoplan et al. (2004b) and Infras & IWW (2004), more than one threshold was applied because they used different thresholds for different health effects (see also section I.2.3.7).

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41 Three texts merely stated that such models need to be used without saying which of them in particular (IER, 2006; Navrud, 2002; THE PEP, 2004a).

42 In four models, noise levels were estimated in L_{eq} dB(A) (adjusted noise level for 24 hours) or in $L_{(den)}$ dB(A) (adjusted noise level for a certain time of a day).

43 Van Essen et al. (2007) and the Swedish National Road and Transport Research Institute (2006) mentioned this discussion regarding a threshold without applying a specific one.

Table 37. Threshold used for daytime in the studies on noise

	Study									
	DIW et al. (2000)	Ecoplan et al. (2004b)	IER (2006)	IER et al. (2000)	IER et al. (2003)	Infras & IWW (2004)	Navrud (2002)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)	Van Essen et al. (2007)
Literature review – more than one study considered						x		x	x	x
Threshold used (daytime)								(+)		(+)
50 dB(A)			+			+	+			
55 dB(A)			+							
60 dB(A)						+				
65 dB(A)			+							

+ : included
– : excluded (explicitly)
 : no information given
( ) : special case or exception

I.2.3.4 Difference between daytime and night-time thresholds

Normally, thresholds applied during daytime are higher than those during night-time to take the night-time rest period into account and to reflect the fact that some health effects are especially relevant during night-time (see also section I.2.2 and section I.2.3.7). Most of the studies on noise reviewed made this differentiation as well (Table 38).

Table 38. Differentiation between thresholds during daytime and night-time in the studies on noise

	Study									
	DIW et al. (2000)	Ecoplan et al. (2004b)	IER (2006)	IER et al. (2000)	IER et al. (2003)	Infras & IWW (2004)	Navrud (2002)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)	Van Essen et al. (2007)
Literature review – more than one study considered						x		x	x	x
Difference between daytime and night-time										
yes		+	+		+	+	+	+	+	
no										

+ : included
– : excluded (explicitly)
: no information given
( ) : special case or exception

I.2.3.5 Modes of transport considered

In calculating costs due to transport crashes, air pollution or noise, determining which mode of transport should be considered is important for valuating the number of people exposed to a certain level of noise. Most of the 10 studies on noise considered road traffic, rail transport and air transport (Table 39).

Table 39. Modes of transport considered in the studies on noise

	Study									
	Van Essen et al. (2007)	DIW et al. (2000)	Ecoplan et al. (2004b)	IER (2006)	IER et al. (2000)	IER et al. (2003)	Infras & IWW (2004)	Navrud (2002)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)
Literature review – more than one study considered						x		x	x	x
Transport mode considered										
Road transport		+	+	+	+	+	+	+	+	+
Rail transport		+	+	+	+	+	+	+	+	+
Air transport		–	+	+	+		+		+	+
Water transport		–								

\+ : included
– : excluded (explicitly)
  : no information given
( ) : special case or exception

I.2.3.6 Affected unit: people or homes

The studies analysed applied as the affected unit either the population exposed to transport noise or homes exposed to noise as a proxy for exposed people for calculating noise costs. In this case, a reduction of rent due to noise exposure indicates people's willingness to pay to avoid annoying sound: a person is willing to pay more rent in return for a less noisy home. The dose–response relationship is derived from hedonistic pricing analysis using regression analysis (Ecoplan et al., 2004b). Of the texts reviewed, only Ecoplan et al. (2004b), the Swedish National Road and Transport Research Institute (2006) and Van Essen et al. (2007) used this approach, considering homes as the affected unit (Table 40).

Table 40. Affected unit (people or homes) considered in the studies on noise

	Study									
	DIW et al. (2000)	Ecoplan et al. (2004b)	IER (2006)	IER et al. (2000)	IER et al. (2003)	Infras & IWW (2004)	Navrud (2002)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)	Van Essen et al. (2007)
Literature review – more than one study considered						x		x	x	x
Affected unit										
People		+	+	+	+	+	+	+	+	+
Homes		+						+		+

+ : included
– : excluded (explicitly)
 : no information given
( ) : special case or exception

I.2.3.7 Health effects considered

The economic literature often distinguishes between two main types of health effects: annoyance and other health effects such as sleep disturbance, increased blood pressure or myocardial infarction. However, not all studies considered both types (Table 41). One study reviewed by THE PEP (2004a) did not include annoyance. Others used different combinations of annoyance and other health effects (such as IER et al. (2000, 2003), Navrud (2002) and one of the studies reviewed by THE PEP (2004a)). These different approaches have implications for the cost figures, as some risk double-counting effects and others neglect some of the overall effects. Six studies on noise in the literature review differentiated at least between annoyance and "other health effects".

Table 41. Health effects considered in the studies on noise: annoyance versus other health effects

	Study									
	DIW et al. (2000)	Ecoplan et al. (2004b)	IER (2006)	IER et al. (2000)	IER et al. (2003)	Infras & IWW (2004)	Navrud (2002)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)	Van Essen et al. (2007)
Literature review – more than one study considered						x		x	x	x
Effects considered										
Annoyance	+	+	+			+		+		+
Health effects	+	+	+			+		+	+	+
Health effects as part of annoyance				+						
Annoyance as part of the health effects					+		+		+	

+ : included
– : excluded (explicitly)
: no information given
( ) : special case or exception

The 10 studies on noise differentiated eight health effects (Table 42).[44] Most studies considered sleep disturbance, ischaemic heart disease and hypertension, and other health effects were less often included.

44 As explained in section I.2.3.7, four studies did not differentiate between annoyance and "other" health effects, so both are marked in Table 42.

Table 42. Detailed health effects considered in the studies on noise

	Study									
	DIW et al. (2000)	Ecoplan et al. (2004b)	IER (2006)	IER et al. (2000)	IER et al. (2003)	Infras & IWW (2004)	Navrud (2002)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)	Van Essen et al. (2007)
Literature review – more than one study considered						x		x	x	x
Health effects considered										
Interference (communication)				+		+	+			
Annoyance				+	+		+		+	
Sleep disturbance			+	+	+		+	+	+	
Hypertension			+		+		+	+	+	
Ischaemic heart disease		+	+	+	+	+	+			
Myocardial infarction		+			+	+		+	+	
Angina pectoris		+			+				+	
Psychological effects on children							+			

\+ : included
– : excluded (explicitly)
  : no information given
( ) : special case or exception

I.2.3.8 Costs of health effects considered

Only two cost components are relevant for the direct and indirect costs of noise (Table 43; for discussion, see sections 3.1.1 and I.1.3.5). The other part of total health costs is the intangible costs for the victim. All but two "noise" studies used both cost components (Table 43). No information was available for the three remaining studies.

Table 43. Types of costs of health effects considered in the studies on noise

	Study									
	DIW et al. (2000)	Ecoplan et al. (2004b)	IER (2006)	IER et al. (2000)	IER et al. (2003)	Infras & IWW (2004)	Navrud (2002)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)	Van Essen et al. (2007)
Literature review – more than one study considered						x		x	x	x
Costs of the health effects considered										
Direct and indirect costs										
Costs of health care (including rehabilitation and reintegration, and administration costs)		+			+	+	+		+	
Economic production losses (including replacement costs)		+			+		+		+	
Intangible costs										
For victims (suffering and grief)		+	+	+	+	+	+		+	

\+ : Included
– : excluded (explicitly)
  : no information given
( ) : special case or exception

I.2.3.9 Methods for measuring costs

As explained in section 3.1.1.2, two approaches can be differentiated to valuate total health costs: using the willingness-to-pay method for all cost categories (as far as it covers them) or applying both methods in combination (see section 3.1.1). Of the studies on noise reviewed, most chose the second possibility (Table 44) [45], and only three used the willingness-to-pay method alone.

45 Navrud (2002) and THE PEP (2004a) suggested using only the willingness-to-pay method if overlapping of the valuations of both methods combined cannot be averted (see also section I.2.3.10.

Table 44. Different ways of measuring costs in the studies on noise

	Study									
	DIW et al. (2000)	Ecoplan et al. (2004b)	IER (2006)	IER et al. (2000)	IER et al. (2003)	Infras & IWW (2004)	Navrud (2002)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)	Van Essen et al. (2007)
Literature review – more than one study considered						x		x	x	x
Measuring costs										
Cost of illness and willingness to pay		+			+	+	+		+	+
Willingness to pay only			+	+				+		

+ : Included
– : excluded (explicitly)
: no information given
( ) : special case or exception

I.2.3.10 Economic production losses: net versus gross values

If both valuation methods (cost of illness and willingness to pay) are applied in combination (see section I.2.3.9), there is a risk of double-counting the lost consumption of a victim (see section 3.1.1) if gross instead of net economic production losses are used. As shown in section I.2.3.8, five of the studies on noise reviewed applied both valuation methods. Hence, net and gross economic production losses must be differentiated. Table 45 shows that Ecoplan et al. (2004b), Navrud (2002) and THE PEP (2004a) calculated net economic production losses, whereas two studies did not provide further information (IER et al., 2003; Infras & IWW, 2004).

Table 45. Net or gross economic production losses considered in the studies on noise

	Study									
	DIW et al. (2000)	Ecoplan et al. (2004b)	IER (2006)	IER et al. (2000)	IER et al. (2003)	Infras & IWW (2004)	Navrud (2002)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)	Van Essen et al. (2007)
Literature review – more than one study considered						x		x	x	x
Economic production losses: net versus gross										
Net, without the value of lost consumption		+					+		+	
Gross, including the value of lost consumption										

+ : Included
– : excluded (explicitly)
 : no information given
( ) : special case or exception

I.2.3.11 Valuating mortality

The studies on noise reviewed only used the approach of the value of life-years lost (see also section I.1.3.8 to valuate mortality (Table 46).

Table 46. Ways of valuating mortality in the studies on noise

	Study: DIW et al. (2000)	Ecoplan et al. (2004b)	IER (2006)	IER et al. (2000)	IER et al. (2003)	Infras & IWW (2004)	Navrud (2002)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)	Van Essen et al. (2007)
Literature review – more than one study considered						x		x	x	x
Ways of valuating mortality										
Value of a statistical life										
Value of life-years lost		+		+	+				+	

+ : Included
– : excluded (explicitly)
 : no information given
( ) : special case or exception

I.2.3.12 Specific focus on children

None of the 10 studies reviewed discussed specifically the situation of children affected by noise or provided separate exposure–response functions and cost figures for children. Either they were treated as adults or left out of the calculation of noise costs altogether, but most of the studies made no clear statement in this regard. Only the study by THE PEP (2004a) referred to children, saying that the willingness to pay, and therefore the intangible costs, cannot be measured for children (as they are not able to properly express their willingness to pay to avoid a certain risk). Currently, from an epidemiological viewpoint, health effects among children related to exposure to road traffic noise cannot be quantified due to the lack of exposure–response functions.

I.2.3.13 Comparing results

The studies analysed and discussed in this chapter not only differ in ways similar to the studies in the previous sections but also according to the geographical area considered (such as countries, regions or communities), the year of the survey, further definitions and assumptions (such as external costs and special vehicle types) and the type of costs measured (such as average or total health costs). Hence, this variety among the studies makes the results for total health costs due to noise differ. As an illustration, Table 47 presents the results of selected studies, which show not only this variation but also the dimension of total health costs due to noise.

Table 47. Selected results for total health costs due to noise

Source	Mode	Geographical area	Year	Definitions and assumptions	Average health costs (in euros per vehicle-km)	Total health costs (in millions of euros)
Ecoplan et al. (2004b)	Road transport	Switzerland (entire country)	2000	Exchange rate Sw.fr./€ = 0.6 for average health costs: only passenger cars	0.0046	521.6
	Rail transport	Switzerland (entire country)	2000	Exchange rate Sw.fr./€ = 0.6 for average health costs: only passenger trains	0.0041[a]	77.2
IER et al. (2003)	Road transport	Helsinki (urban area)	2003	Only passenger cars, daytime	0.0022	–
		Stuttgart (urban area)	2003	Only passenger cars, daytime	0.0150	–
		Berlin (urban area)	2003	Only passenger cars, daytime	0.0047	–
		Basel-Karlsruhe (interurban area)	2003	Only passenger cars, daytime	0.0002	–
		Strasbourg-Neubrandenburg (interurban area)	2003	Only passenger cars, daytime	0.0012	–
		Milan-Chiasso (interurban area)	2003	Only passenger cars, daytime	0.0001	–
		Bologna-Brennero (interurban area)	2003	Only passenger cars, daytime	0.00001	–

[a]The measurement unit is not vehicle-km but passenger-km.

I.3 Overview of results on transport-related air pollution

I.3.1 Introduction

Motor vehicle emissions include hundreds of compounds that are released into the atmosphere. It is only partly known which are toxic and which are not toxic. Road transport contributes to a range of gaseous air pollutants and to suspended PM of different sizes and composition. For a selection of relevant health end-points and available exposure–response functions to be used in health impact assessment and cost-benefit analysis of traffic impacts, the main recent key publications were used, including the WHO Air Quality Guidelines, an update of the most recent scientific evidence on the adverse health effects of PM and NO2 and other air pollutants (WHO, 2006; WHO Regional Office for Europe, 2006), a meta-analysis carried out for WHO (Anderson et al., 2004), a cost-benefit analysis carried out in the framework of the Clean Air for Europe (CAFE) programme (Hurley et al., 2005) and a review on NO2 by the United States Environmental Protection Agency (2008a).

I.3.2 Exposure to transport-related air pollution

Road transport contributes to a range of gaseous air pollutants such as NO_2 and PM of different sizes and composition, including PM_{10}, $PM_{2.5}$, black smoke or soot and ultrafine particles ($PM_{0.1}$). Tailpipe emissions of particles from road transport may account for up to 30% of fine particulate matter ($PM_{2.5}$) (Krzyzanowski et al., 2005). Other emissions related to road transport (such as resuspended road dust and wear of tyres and brakes) are the most important source of the coarse fraction of PM (2.5–10 μm). In some eastern European countries, sulfur dioxide and lead emissions from motorized vehicles may still be a problem, depending on the type of fuel used. There is some evidence that diesel exhaust emissions are more relevant for adverse health effects than gasoline emissions (Nel, 2005).

I.3.2.1 Indicators for exposure to road transport–related air pollution

It is still unclear which of the constituents of transport emissions are responsible for the observed adverse effects on people's health. Most epidemiological studies have concentrated on the classical air pollutants such as black smoke, NO_2 or PM_{10} and $PM_{2.5}$. Only a few studies have investigated the role of ultrafine particles (Krzyzanowski et al., 2005) and almost none the role of the different constituents of PM.

The volume and spatial distribution of the emissions as well as dispersion conditions affect pollution levels. Population exposure is also determined by the number of people in polluted areas, how long they stay there and what activities they carry out. Residence or working near busy roads and time spent in road traffic are critical for population exposure. Travellers are often exposed to levels that can be three times as high as the background levels. In-vehicle exposure is especially high for primary exhaust gases and PM (Krzyzanowski et al., 2005).

Exposure to transport-related air pollution cannot be measured directly, and none of the pollutants measured in most standard (fixed) monitoring systems are solely caused by road traffic. Depending on the study scale, epidemiologists therefore use different approaches to estimate exposure to road transport–related air pollution in their studies:

- selecting a balanced set of fixed monitoring stations and using one air pollutant such as $PM_{2.5}$ (or PM_{10}) or NO_2 as a marker for the mixture of road transport–related emissions;
- using proxies determined either by the distance between residences and busy roads or road traffic count data at streets;
- using a road traffic index combining distance and traffic intensity: or
- using statistical or dispersion models to assess specifically exposure to road transport–related air pollution (such as PM_{10}, NO_2 or black smoke).

However, none of these exposure measures can be characterized as a gold standard, and the availability of data is often the main reason why researchers decide to choose one of the measures.

Street canyons are valley-like spaces created by tall buildings on both sides of a street in which air exchange is reduced with heavy road traffic. Their concentrations of transport-related pollutants are much higher than in areas not affected by major urban streets (with urban background levels). In a 500-metre-wide belt along major urban highways, the concentrations of NO_2, black smoke (or soot) and $PM_{0.1}$ are markedly higher than in areas with less road traffic. Several other road transport-related pollutants, however, are spread more uniformly over large areas of a city (Krzyzanowski et al., 2005). A study in the Netherlands compared the results from air pollution measurements near roads and model calculations, showing that the contribution of a busy road to the (road transport–related) air pollution mixture depends on the pollutant (Fischer et al., 2007). The contribution to PM_{10} and primary $PM_{2.5}$ is relatively limited, and the contribution to levels of soot, elementary carbon (a marker for carbon compounds), NO_2 and $PM_{0.1}$ is much larger. Further, the influence of road traffic is not limited to a few hundred metres along the road but can be observed up to 1000 m from the road (Fischer et al., 2007).

The choice of an exposure indicator depends on the scope and the available data. Possible indicators of exposure to PM from traffic in urban areas include PM_{10}, $PM_{2.5}$, black smoke and $PM_{0.1}$. Using only PM_{10} would not provide a complete picture of the health risks. In 2006, WHO recommended using $PM_{2.5}$ as an indicator for health effects caused by particulate pollution to supplement the most commonly used PM_{10} (WHO Regional Office for Europe, 2006). Black smoke is a good indicator of diesel emissions as long as there are no other major sources of air pollution in the area. NO_2 could also be used as an indicator of emissions from road transport. In some studies, O_3 has been used as an indicator for overall air pollution (Institute for Economic and Environmental Studies, 2003). However, O_3 is a secondary pollutant that requires some residence time in the atmosphere to be formed and is therefore not a good local indicator of local road transport–related air pollution. It is further recommended to consider either NO_2 or one PM measure ($PM_{2.5}$ or, preferably, black smoke) as an indicator for diesel emissions. If both NO_2 and PM are used, the effects due to road transport–related air pollution might be double counted, unless the study from which the relative risks are derived considered both pollutants simultaneously.

However, for the quantification of road transport–related health effects, indicators that would be ideal from a strictly exposure-related viewpoint of view may not be able to be used due to limited data availability, as discussed further below.

I.3.3 Health end-points

Recent WHO reviews indicate that road transport–related air pollution is clearly associated with a number of health outcomes (Krzyzanowski et al., 2005; WHO, 2006; WHO Regional Office for Europe, 2006). Evidence from epidemiological and toxicological studies on the effects of road transport–related air pollution has increased substantially, although it is only a fraction of the total evidence on the effects of urban air pollution on health. Road transport–related air pollution has been associated with several adverse health outcomes, including premature mortality from both acute exposure (Gehring et al., 2006; Krzyzanowski et al., 2005; Nafstad et al., 2004; Rosenlund et al., 2006) and long-term exposure to PM (Finkelstein et al., 2004; Hoek et al., 2002; Laden et al., 2006; Pope 1995, 2002, 2004; Dockery et al., 1993), allergic and non-allergic respiratory diseases (Bayer-Oglesby et al., 2006), cardiovascular and respiratory diseases (Hoffmann et al., 2006, 2007; Peters et al., 2004; Tonne et al., 2007), cancer (Nafstad et al., 2003; Nyberg et al., 2000) and birth outcomes (Slama et al., 2007; Wilhelm & Ritz, 2003). Laboratory studies indicate that road transport–related air pollution may increase the risk of developing allergy and can exacerbate symptoms, particularly in susceptible subgroups (such as people with asthma). One way that such a relationship may show is through increased use of medication such as bronchodilators. The evidence from population studies, however, does not clearly support this observation (Krzyzanowski et al., 2005). The results from controlled human exposure studies and animal experiments indicate that road transport–related air pollution is linked to changes in the formation of reactive oxygen species, changes in antioxidant defence and increased inflammation (Krzyzanowski et al., 2005).

The few studies on cardiovascular and respiratory morbidity report a significant increase in the risk of myocardial infarction following exposure to air pollutants form road transport, but the observed effects may not solely be attributable to road transport–related air pollution. The observed associations between air pollution exposure and cardiovascular disease could also be partly explained by the exposure related to road traffic–related noise (see section I.2.2.3) (Babisch, 2006; Van Kempen et al., 2002). Some studies suggest that road transport–related air pollution also causes adverse outcomes in pregnancy (such as low birth weight and premature birth), but the available evidence is inconsistent (Bobak, 2000; Gouveia et al., 2004; Ritz et al., 2000). A few studies suggested an increased incidence of lung cancer among groups with higher or longer than average levels of exposure, but evidence is insufficient to draw firm conclusions (Krzyzanowski et al., 2005).

However, the strength of the available evidence differs for the various exposure indicators presented above. In addition, the effects observed in epidemiological studies often cannot be attributed solely to the specific pollution indicator used in the respective study but are more likely to be linked to a mixture of pollutants. Fine PM (including black smoke) is associated with increased risks of mortality and respiratory morbidity, whereas exposure to NO_2 and coarse PM has been linked to allergic responses. The United States Environmental Protection Agency (2008a) concluded that short-term NO_2 exposure is likely to be associated with respiratory health effects.[46] The effects include a wide range of symptoms, such as decreased lung functioning, asthma and hospital admissions for respiratory problems. The evidence for the effects of long-term exposure to NO_2 is less strong. For other health outcomes, the evidence for an association with NO_2 is inconclusive, especially for mortality related to cardiovascular and respiratory disease, for which also the underlying mechanisms are unclear. NO_2 is therefore a good indicator for road transport–related exposure but is not considered a good predictor of health effects. It is therefore suggested not to use it for quantifying and valuating road transport–related health effects.

I.3.3.1 Health effects among children

Several studies have assessed the relationship between living near busy road traffic routes and respiratory diseases among children. Studies in California (Kim et al., 2004), Germany (Duhme et al., 1996; Hirsch et al., 1999; Weiland et al., 1994), Italy (Ciccone et al., 1998), Japan (Shima et al., 2003), the Netherlands (Oosterlee et al., 1996) and the United Kingdom (Venn et al., 2001) found that more children with airway complaints (reported by the parents) lived in the busy streets than in less busy streets.

Studies in Germany (Krämer et al., 2000) and Switzerland (Braun-Fahrländer et al., 1997) found that more children were sensitized to pollen if they had been more exposed to road transport–related air pollution. A study from the United Kingdom suggested that children admitted to a hospital with an asthma diagnosis were significantly more likely to live in an area with high road traffic flow located along the nearest segment of main road than were children admitted for non-respiratory reasons or children from the community (Edwards et al., 1994). A study in California seemed to confirm this result, suggesting that higher road traffic flows may be related to an increase in repeated health care visits for children with asthma (English et al., 1999).

Most of these studies relied on the distance to major roads or on measures of road traffic intensity as a proxy for exposure to air pollutants, suggesting that living near busy roads leads to adverse respiratory health effects among children.

Fewer studies have modelled the concentrations of or directly measured air pollution. Lung functioning decrements have been observed among children in relation to the exposure to road transport–related air pollution measured at their schools and to the daily number of lorries passing on the motorway near the school (Brunekreef et al., 1997; Gauderman et al., 2007; van Vliet et al., 1997). Respiratory symptoms have been associated with modelled or

46 When the associations observed in epidemiological and experimental toxicological studies are strong, consistent, coherent, and plausible, the United States Environmental Protection Agency concludes that the relationship is “likely causal.”

measured road transport–related air pollutants in several studies (Brauer et al., 2003, 2007; Gehring et al., 2002; Janssen et al., 2003; Nicolai et al., 2003).

In summary, there is sufficient evidence that exposure to road transport–related air pollution is associated with respiratory disorders in children (WHO Regional Office for Europe, 2005b).

I.3.3.2 Evaluation of available exposure–response relationships

Although the information from epidemiological and toxicological studies clearly identifies the hazards of road transport–related air pollution, deriving a well-based exposure–response function is difficult, and this is needed to reliably quantify the adverse effects of road transport–related air pollution (Krzyzanowski et al., 2005). As discussed above, the available epidemiological studies use different exposure and outcome indicators, which limits the possibility of comparing their results quantitatively and estimating a common risk function, as has been done for studies of ambient air pollution (Anderson et al., 2004). Further, isolating the population's exposure to road transport–related air pollution from the exposure to ambient pollution is still difficult, which limits the precision with which road transport–related effects can be quantified.

Since WHO published *Health effects of transport-related air pollution* (Krzyzanowski, 2005), several additional studies on the health effects of road transport–related air pollution have been published. A main feature of these studies is the use of geographical information systems for assessing exposure measures. Several studies used the distance of the home address to the nearest major road as a proxy for personal exposure to road transport–related air pollution. Respiratory symptoms among adults (Bayer-Oglesby et al., 2006), the prevalence of coronary heart disease (Hoffmann et al., 2006), the degree of coronary atherosclerosis (Hoffmann et al., 2007), respiratory health among very young (one- to two-year-old) children (Morgenstern et al., 2007) and cardiovascular mortality (Beelen et al., 2008; Gehring et al., 2006; Hoek et al., 2002) were all associated with living close to major roads. However, due to the different road traffic proxies used in the studies, an exposure–response function with distance to the road as proxy for exposure to road transport–related air pollution cannot yet be derived.

Instead of using exposure–response functions from studies on road transport–related air pollution, the exposure–response functions derived from epidemiological studies looking at the whole mixture of air pollution and not specifically at road transport–related air pollution can alternatively be applied for assessing road transport–related health effects. The assumption is that the road traffic emissions are as toxic as the background aerosol mixture, which is probably an underestimation.

Based on best currently available evidence, it is therefore proposed to use a selection of risk functions derived from ambient air pollution studies until better estimates from road traffic studies become available.

I.3.3.3 Selection of health end-points

Adults

Exposure to outdoor air pollution is associated with a broad range of acute and chronic health effects. Although all these outcomes are potentially relevant for health impact assessment, WHO proposes long-term all-cause mortality associated with PM2.5 as the essential health effect for health impact assessment (WHO, 2006; WHO Regional Office for Europe, 2006). If only PM10 levels are available, they can be converted into PM2.5 levels using a ratio from a study done in a comparable location measuring both PM2.5 and PM10. The main reasons to choose all-cause mortality are the strength of evidence, the well-defined end-point[47] and the availability of baseline prevalence rates for most countries. Although fewer studies are conducted on the effects of long-term exposure than on acute exposure, long-term exposure effects should be included because the relative risk estimates from long-term exposure are about one order of magnitude larger than the estimates from short-term exposure (see section 3.4.2.3). Mortality, expressed as numbers of deaths or reduction in life expectancy, is the most important health effect for disability-adjusted life years (DALYs) (de Hollander et al., 1999) and the monetary value of the impact (Künzli et al., 2000). Cause-specific mortality can be chosen instead of all-cause mortality if background rates are available. Considering specific causes (such as cardiovascular and respiratory mortality and lung cancer) can improve the precision of the estimates. To avoid double counting, either all-cause or cause-specific mortality should be used.

Quantifying the effects of PM exposure on morbidity is more difficult than that for mortality, since the evidence on concentration–response functions and background rates of health end-points is scarcer. However, selected estimates of the effects on morbidity can be included in such analyses (WHO, 2006; WHO Regional Office for Europe, 2006). In particular, the evidence allows a valid exposure–effect relation to be derived for respiratory and cardiovascular hospital admissions. Other morbidity health end-points can be included if the aim is a more comprehensive rather than a conservative estimate of transport-related health costs and the greater uncertainty related to the other morbidity end-points is accepted and clearly acknowledged**.** Many of the recent economic studies have considered morbidity (see section 3.3.4). In addition to the level of conservatism to be applied in a study, inclusion of morbidity depends on data availability (such as on baseline prevalence rates).

If more comprehensive estimation is desired, health end-points that can be considered for indicative estimates of costs from morbidity are those related to lower respiratory symptoms and chronic bronchitis, working-loss days or days with restricted activity (Table 48). However, the evidence for these health end-points is less well established than for mortality and well-founded concentration–response functions (such as those derived from meta-analysis or pooled studies) are lacking for road transport–related exposure indicators but may be derived from studies on general ambient air pollution (see above and section 3.4.2).

47 Mortality is a well-defined health end-point that is registered in all European countries. Using all-cause mortality eliminates errors due to misclassification of causes. Other outcomes (such as bronchitis) are subject to substantial variation in severity. Definitions of end-points such as restricted activity days or use of primary care services are likely to vary between countries and health care systems.

Table 48. Overview of selected health end-points associated with road transport–related air pollution among adults

Health end-point	Strength of the evidence	Existence of biologically plausible mechanisms?	Sufficient information to allow quantification?	Valid exposure–response relationship available?
Mortality				
Mortality from short-term exposure (all cause)	Sufficient	Yes	PM_{10}, black smoke	Yes
Mortality from long-term exposure				
All causes	Sufficient	Yes	PM_{10}, $PM_{2.5}$	Yes
Cardiovascular and respiratory	Sufficient	Yes	PM_{10}, black smoke	Yes
Morbidity				
Hospital admissions				
Respiratory	Sufficient	Yes	PM_{10}, black smoke	Yes
Cardiovascular	Sufficient	Yes	PM_{10}	Yes
Lower respiratory symptoms including cough	Limited	Yes	PM_{10}	Only indicatively[a]
Chronic bronchitis	Limited	Yes	PM_{10}	Only indicatively[a]
Medication use[b]	Limited	Yes	PM_{10}	No
Restricted activity day	Limited	Yes	$PM_{2.5}$	Only indicatively[a]
Working-loss day	Limited	Yes	$PM_{2.5}$	Only indicatively[a]

[a] Indicatively: an exposure–response function is available, but the risk estimates are very small, not statistically significant or derived from one study only. They can be used for estimating total disease burden, but one should be aware of the weaker evidence base behind these calculations.
[b] Bronchodilator use.

Table 16 in chapter 3 on the practical approaches presents an overview of summary estimates that may be used in economic valuations of health effects from road transport–related air pollution.

When air pollution effect estimates derived from one study population are used to estimate the effects in another population, the implicit assumption is that both populations are equally sensitive to the effects of air pollution. Differences in baseline health status, the distribution of causes of death, socioeconomic status and lifestyle factors between the two populations in question may limit this approach (Ostro, 2006). Regional characteristics (such as sources of PM, including sea salt or desert sand) may modify the composition and effects of air pollution (WHO Regional Office for Europe, 2004b). This could be important, as the characteristics of populations, environments and air pollution (including particle size distribution and composition) vary throughout Europe. However, the currently available evidence from different parts of Europe does not yet allow the development of specific subregional guidance for assessing and valuating the health effects of road transport–related air pollutants.

Children

Substantial evidence indicates that road transport–related air pollution adversely affects a wide range of measures of fetal and infant health, including short-term infant mortality (WHO, 2006). WHO concludes that evidence is sufficient to assume a causal relationship between air pollution exposure and aggravation of asthma among children, shown through increased use of medication (WHO Regional Office for Europe, 2004), but meta-analyses do not provide child-specific risk estimates for respiratory hospital admissions (Anderson et al., 2004) or for health end-points such as changes in lung functioning.

Table 49. Overview of selected health end-points associated with road transport–related air pollution among children

Health end-point	Strength of the evidence	Existence of biologically plausible mechanism(s)?	Sufficient information to allow quantification?	Valid exposure–effect relationship available?
Mortality				
Mortality (all causes, infants 1–12 months of age)	Sufficient	Yes	PM_{10}	Yes
Morbidity				
Hospital admissions	Limited	Yes	No	No
Cough	Sufficient	Yes	PM_{10}, black smoke	Only indicatively[a]
Medication use[b]	Sufficient	Yes	PM_{10}, black smoke	No

[a] Small but statistically non-significant increase; see section 3.4.2.3.
[b] Bronchodilator use.

Section 3.4.2.3 shows the exposure–response relationships that can be used for these health end-points for practical application in the economic valuation of road transport-related health effects.

When this report was published, several research projects were still ongoing or had just started that were aiming at further developing and selecting summary estimates and exposure–response functions of road transport–related air pollution, such as projects of the Health Effect Institute in the United States and EU-funded projects such as INTARESE (Integrated Assessment of Health Risks from Environmental Stressors in Europe (2008)) and an international cohort study (European Study of Cohorts for Air Pollution Effects – ESCAPE) (Brunekreef, 2008). It is recommended to closely follow the developments of these projects and adjust the approach for assessing the health effects of road transport–related air pollution, if warranted.

I.3.4 Summary of the review of economic approaches

Health costs due to road transport–related air pollution were analysed in 18 of the 38 economic studies considered in this part of the literature review (see Table 1, page 18).

I.3.4.1 The model used in the studies analysed

The basic model applied to calculate heath costs due to road transport–related air pollution can be expressed as:

Total health costs[48] = the number of cases with health effects attributed to road transport–related air pollution times costs per case

48 The term “total health costs” refers in the whole text to the sum of all different costs of a health effect considered in this study. As this study does not consider some costs of a health effect, for example, because they cannot yet be monetized, the term “total health costs” as used here does not cover all possible total health costs that occur in reality.

All 18 studies base their calculation on this model.

Even though all "air pollution" studies referred to this basic model, they differed regarding some of the steps and factors of the costs calculation. Sections I.3.4.2 to I.3.4.10 present and discuss these differences. The last section of this part of the report focuses on the special situation of health effects on children due to road transport–related air pollution.

Some studies discussed here are literature reviews (AEA Technology Environment, 2005a, b; IER, 2005; IER et al., 2000;[49] Infras & IWW, 2004; Swedish National Road and Transport Research Institute, 2006;[50] THE PEP, 2004a; Van Essen et al., 2007). Hence, the information derived from them might relate to several studies.

I.3.4.2 Type of air pollution considered

Air pollutants can be divided into criteria pollutants (United States Environmental Protection Agency, 2008b) and toxics. The former include PM, CO, O_3, NO_2, SO_2 and lead, which are ubiquitous and are the result of almost any combustion process, whereas toxics are more process-specific.

Table 50 shows that the studies considered a wide range of air pollutants.[51] Criteria pollutants are included much more frequently than toxics, and most studies included only three: PM2.5, PM10 and O3. Air pollution cost estimations tend to focusing solely on these three pollutants, as PM2.5 and PM10 are often considered as indicators for all the other pollutants except O3. Some studies also included greenhouse gases such as CO2 that do not have direct health effects.

49 IER et al. (2000) is not a literature review in the strict sense. However, there are many examples from other studies used, which results in a kind of literature review.

50 Swedish National Road and Transport Research Institute (2006) presents four case studies and summarizes their findings, and this text can therefore be regarded as a literature review.

51 Some of the air pollutants listed in Table 41 are primary pollutants, which will transform into secondary pollutants (see section I.3.4.4) and, hence, will only then be regarded as harmful.

Table 50. Indicators of air pollution considered in the studies on air pollution

	Study																	
	AEA Technology Environment (2005a)	AEA Technology Environment (2005b)	Ecoplan et al. (2004a)	Holland & Watkiss (2002)	IEES (2003)	IER (1997)	IER (2005)	IER (2006)	IER et al. (2000)	IER et al. (2003)	Infras & IWW (2004)	McCubbin & Delucchi (1999)	Sommer et al. (1999)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)	United States Environmental Protection Agency (1999)	United States Environmental Protection Agency (2000)	Van Essen et al. (2007)
Literature review – more than one study considered	x	x					x		(x)		x			(x)	x			x
Type of air pollution considered																		
Indicator pollutants																		
$PM_{2.5}$	+		+			+	+	+		+	+			+	+	+	+	+
PM_{10}	+	+	+		+	+		+	+	+	+	+	+	+	+	+	+	+
Diesel particles					+					+	+			+				
O_3	+		+	+	+	+	+	+		+	+	-		+	+	+	+	
CO					+						+			+	+	+	+	
NO_2 and other nitrates					+	+		+		+	+		+	+	+	+	+	+
SO_2 and other sulfates			+		+	+		+		+	-		+	+	+	+	+	+
Heavy metals (including lead)					+	+					-	-		+				
Toxics																		
Ammonia													+					
VOC (volatile organic compounds)					+								+	+	+			+
Hydrocarbons (including polyaromatic and non-methane hydrocarbons and butadine)					+	+					+					+		
Dioxins						+											+	
Aldehydes (including formaldehyde and acrolein)					+	+					+							
Ethylene oxide					+													
Ethene					+													
Benzene											+							
Methyl tertiary-butyl ether					+													
Aromatic compounds					+	+				+				+				
Other air pollutants																		
CO_2													+					

\+ : included
– : excluded (explicitly)
  : no information given
( ) : special case or exception

I.3.4.3 Transport mode considered

As shown in Table 51, most of the 18 “air pollution” studies took road transport into account. Rail and water transport were also often included. Some studies related their calculation to air pollution in general, not distinguishing between modes of transport. Nevertheless, the method of these types of cost calculations is identical to the other ones, except that they did not need to apportion the share of air pollution to a specific mode of transport.

Table 51. Types of modes of transport considered in the studies on air pollution

	Study																	
	AEA Technology Environment (2005a)	AEA Technology Environment (2005b)	Ecoplan et al. (2004a)	Holland & Watkiss (2002)	IEES (2003)	IER (1997)	IER (2005)	IER (2006)	IER et al. (2000)	IER et al. (2003)	Infras & IWW (2004)	McCubbin & Delucchi (1999)	Sommer et al. (1999)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)	United States Environmental Protection Agency (1999)	United States Environmental Protection Agency (2000)	Van Essen et al. (2007)
Literature review – more than one study considered	x	x					x		(x)		x			(x)	x			x
Transport mode considered																		
Road transport			+			+		+	+	+	+	+	+	+	+		+	+
Rail transport			+			+			+	+	+		–	+				
Air transport									+		+		–					
Water transport				+		+			+	+			–					
Air pollution in general	+	+	+	+	+		+									+		

+ : included
– : excluded (explicitly)
: no information given
( ) : special case or exception

I.3.4.4 Dispersion and transformation model used

Another important factor to estimate the number of people exposed to air pollution is the concentration of specific air pollutants. Deriving these figures requires estimating the emission due to the road traffic characteristics and modelling the dispersion and transformation of the air pollutants. The 18 "air pollution" studies used 17 models of dispersion and transformation (Table 52).[52] However, not all can be applied to all air pollutants: for example, some models can solely be used for the dispersion of O3. Section 3.4.2 provides more information on models.

52 Some of the texts only mentioned that dispersion and transformation models have to be used without specifying them (AEA Technology Environment, 2005a; IER, 2006; Sommer et al., 1999; THE PEP, 2004a; Van Essen et al., 2007).

Table 52. Models of dispersion or transformation considered in the studies on air pollution

	Study																	
	AEA Technology Environment (2005a)	AEA Technology Environment (2005b)	Ecoplan et al. (2004a)	Holland & Watkiss (2002)	IEES (2003)	IER (1997)	IER (2005)	IER (2006)	IER et al. (2000)	IER et al. (2003)	Infras & IWW (2004)	McCubbin & Delucchi (1999)	Sommer et al. (1999)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)	United States Environmental Protection Agency (1999)	United States Environmental Protection Agency (2000)	Van Essen et al. (2007)
Literature review – more than one study considered	x	x					x		(x)		x			(x)	x			x
Models of dispersion and transformation considered	(+)							(+)					(+)		(+)			(+)
ROADPOL						+				+				+				
Lagrangian ozone model or source–receptor ozone model							+		+	+				+				
Lagrangian trajectory model or windrose trajectory model							+		+	+				+				
Gaussian dispersion models			+				+		+									
Industrial source complex model							+											
Uniform world model							+											
WATSON							+											
ECOSENSE						+												
EMEP and Harwell						+												
Urban airshed model – variable (UAM-V)																+	+	
Regulatory modelling system for aerosols and deposition (REMSAD)																+	+	
Regional acid deposition model (RADM)																+	+	
Regional particulate model (RPM)																+		
NCAR/Penn State mesoscale model (MM5)														+				
Linear scaling																+		
Global scale model											+							
Inhalation and ingestion dose								+										

+ : included
– : excluded (explicitly)
  : no information given
( ) : special case or exception

I.3.4.5 Health effects considered: mortality

Premature mortality can be classified according to three dimensions (Table 53):

- mortality from short-term exposure versus mortality from long-term exposure;
- all-cause versus cause-specific mortality: all cases of mortality regardless of the causes or only mortality related to a certain health effect; and
- whether mortality among infants (defined here as being 1–12 months of age (IER, 2006)) was treated separately.

The review of the 18 studies shows that most concentrated on all-cause mortality from both acute and long-term exposure. Only two studies considered mortality from specific causes (AEA Technology Environment, 2005a, b). The same is true for infant mortality: just four studies treated them separately from cases of adults.

Table 53. Health effects: types of mortality differentiated in the studies on air pollution

	Study																	
	AEA Technology Environment (2005a)	AEA Technology Environment (2005b)	Ecoplan et al. (2004a)	Holland & Watkiss (2002)	IEES (2003)	IER (1997)	IER (2005)	IER (2006)	IER et al. (2000)	IER et al. (2003)	Infras & IWW (2004)	McCubbin & Delucchi (1999)	Sommer et al. (1999)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)	United States Environmental Protection Agency (1999)	United States Environmental Protection Agency (2000)	Van Essen et al. (2007)
Literature review – more than one study considered	x	x					x		(x)		x			(x)	x			x
Health effects: mortality																		
Acute mortality																		
All causes				+		+	+	+	+	+			–	+	+	+	+	
Specific causes	+	+										+	–					
Chronic mortality																		
All causes			+	+		+	+	+	+	+	+	+	+	+	+	+	+	
Specific causes	+	+													+			
Infant mortality	+	+	+				+						–					

+ : included
– : excluded (explicitly)
  : no information given
( ) : special case or exception

I.3.4.6 Health effects considered: morbidity end-points

The 18 "air pollution" studies analysed considered 27 types of morbidity end-points (Table 54).[53] As explained in section I.3.3, not all air pollutants are associated with the same health effects. Therefore, which health effect is considered depends on the air pollutant chosen and on the available knowledge of health effects (concentration–response functions) and in the availability of local health data. Table 54 lists all identified health effects without differentiation for the air pollutant considered in the respective study. Further, some health effects are only relevant for children (C) or adults (A), respectively.[54] Three texts treat school absences and work-loss days, respectively, as a form of health effects (IEES, 2003; United States Environmental Protection Agency, 1999, 2000).

53 Holland & Watkiss (2002) only advised to consider acute and chronic morbidity without going more into detail regarding specific health end-points.

54 The studies do not use the same age to distinguish between children and adults (range from 15 years to 30 years, depending on the health effect in question).

Table 54. Health effects: types of morbidity considered in the studies on air pollution

	Study																	
	AEA Technology Environment (2005a)	AEA Technology Environment (2005b)	Ecoplan et al. (2004a)	Holland & Watkiss (2002)	IEES (2003)	IER (1997)	IER (2005)	IER (2006)	IER et al. (2000)	IER et al. (2003)	Infras & IWW (2004)	McCubbin & Delucchi (1999)	Sommer et al. (1999)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)	United States Environmental Protection Agency (1999)	United States Environmental Protection Agency (2000)	Van Essen et al. (2007)
Literature review – more than one study considered	x	x					x		(x)		x			(x)				x
Health effects: morbidity				(+)														
Illnesses																		
Chronic obstructive pulmonary disease									+									
Congestive heart failure										A					+			
Lung cancer													–		+			
Cancer							+		+	+			–		+	+[a]		
Leukaemia															+			
Respiratory symptoms	A/C	A/C				+	A/C	+	+	A/C		+	–	+	+	+	C	
Asthma attacks			A/C			+			+		A/C	+	A/C		A/C	+	+	
Cough	A/C	A/C					A/C	+	C	A/C				+	+			
Chronic bronchitis	+	+	A				+	+	C	A	A		A	+	+	+	A	
Acute bronchitis			C								C		C		C	+	C	
Headache												+						
Sore throat												+						
Excess phlegm												+						
Symptom days	+	+							+	+					+			
Eye irritation												+						
Shortness of breath, chest tightness, wheeze																+		
Chronic morbidity						+						+						
Restriction of activities																		
Restricted activity days	A	A	+			+	A	+	+	A	A	+	A	+	A	+	A	
School absence or work-loss day					C											A	A	
Consumption of services																		
Respiratory hospital admissions	+	+	+		C	+	A/C	+	+	+	A/C		A/C	+	A/C	+	+	
Cardiovascular hospital admissions	+	+	+			+		+			A/C		A/C	+	A/C	+	+	
Cerebrovascular hospital admissions						+			+	+					+			
Emergency room visits					C[b]	+	–						–			+[b]	+[b]	
Bronchodilator usage								+		A/C					+	+[a]		
Use of respiratory medication (asthma)	A/C	A/C					A/C							+				
Consultations for allergic rhinitis							A/C											
Consultations with primary care physicians	+	+					A/C											

+ : included
– : excluded (explicitly)
  : no information given
( ) : special case or exception
[a]Skin cancer due to a loss of stratospheric ozone.
[b]Only asthma-related emergency room visits.
A: adults. C: children. A/C: adults and children.

I.3.4.7 Costs considered

Total health costs consist of two different cost categories: direct and indirect costs and intangible costs (Table 55). The first category includes the costs of health care and economic production losses (see section 3.1.1). The second category includes intangible costs for the victim such as pain and suffering.

In contrast to the economic costs due to road crashes (see chapter I.1.3.5), only two cost figures are relevant for the direct and indirect costs as the others (material damage, travel delays or congestion costs and administration costs) do not occur for health effects due to air pollution or are very difficult to assess (such as costs for risk avoidance in the form of averting behaviour: for example, when physical activities are restricted or reduced due to high levels of air pollution). Hence, the cost of health care and economic production losses are the only considered components of direct and indirect costs in this case.

IEES (2003) said that, besides the direct and indirect costs, intangible costs are considered as well. However, as only the cost-of-illness method is used for the cost calculation (see section I.3.4.8), it is not clear how intangible costs can actually be estimated.

Table 55. Types of costs considered in the studies on air pollution

	Study																	
	AEA Technology Environment (2005a)	AEA Technology Environment (2005b)	Ecoplan et al. (2004a)	Holland & Watkiss (2002)	IEES (2003)	IER (1997)	IER (2005)	IER (2006)	IER et al. (2000)	IER et al. (2003)	Infras & IWW (2004)	McCubbin & Delucchi (1999)	Sommer et al. (1999)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)	United States Environmental Protection Agency (1999)	United States Environmental Protection Agency (2000)	Van Essen et al. (2007)
Literature review – more than one study considered	x	x					x		(x)		x			(x)	x		(x)	x
Costs considered																		
Direct and indirect costs																		
Costs of health care (including rehabilitation and reintegration)		+	+		+		+	+				+	+	+	+	+	+	
Economic production losses (including replacement costs)		+	+		+		+	+						+		+	+	
Intangible costs																		
For victims (suffering and grief)	+	+	+	+	(+)	+	+	+	+	+	+		+	+	+	+	+	+

+ : included
– : excluded (explicitly)
: no information given
( ) : special case or exception

I.3.4.8 Methods of measuring costs

As explained in section I.1.3.5, two approaches can be differentiated to valuate total health costs. Either the willingness-to-pay method is used for all cost categories (as far as it covers them), or both methods are applied in combination (see section 3.1.1). As Table 56 shows, more than half the reviewed studies assessed health costs using the willingness-to-pay method only. Six texts combined both methods. Four studies (IEES, 2003; McCubbin

& Delucchi, 1999; Sommer et al., 1999; United States Environmental Protection Agency, 2000) used the cost-of-illness approach without the willingness-to-pay approach (for United States Environmental Protection Agency (2000) only if willingness to pay could not be applied).

Table 56. Ways of measuring costs in the studies on air pollution

	Study: AEA Technology Environment (2005a)	AEA Technology Environment (2005b)	Ecoplan et al. (2004a)	Holland & Watkiss (2002)	IEES (2003)	IER (1997)	IER (2005)	IER (2006)	IER et al. (2000)	IER et al. (2003)	Infras & IWW (2004)	McCubbin & Delucchi (1999)	Sommer et al. (1999)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)	United States Environmental Protection Agency (1999)	United States Environmental Protection Agency (2000)	Van Essen et al. (2007)
Literature review – more than one study considered	x	x					x		(x)		x			(x)	x			x
Measuring costs																		
Direct and indirect costs																		
Cost of illness and willingness to pay		+	+				+	+					+	+	+			
Cost of illness only					+							+					(+)	
Willingness to pay only	+			+		+			+	+	+					+	+	+

+ : included
– : excluded (explicitly)
: no information given
( ) : special case or exception

I.3.4.9 Economic production losses: net versus gross values

The discussion on economic production losses is closely related to the section on methods of measuring costs (see section I.3.4.8). If both methods, willingness to pay and the cost of illness, are applied in combination, lost consumption should be excluded from calculation of the economic production losses (using net economic production losses) to avoid double counting (see section 3.1.1). However, only one of the texts reviewed applying the combined valuation approach provided information on this. In the others it was not clear whether the total health costs of these studies might have been overestimated by applying gross instead of net values.

I.3.4.10 Valuating mortality

There are two approaches for valuating premature mortality effects: valuating a statistical life or valuating the years of life lost due to each death (see section I.1.3.7). The 18 texts on air pollution reviewed were divided into two groups (Table 57). In addition, three studies (AEA Technology Environment, 2005a, b; IER/Infras et al., 2000) gave cost figures for both approaches, allowing a comparison.

Table 57. Ways of valuating mortality in the studies on air pollution

	Study																	
	AEA Technology Environment (2005a)	AEA Technology Environment (2005b)	Ecoplan et al. (2004a)	Holland & Watkiss (2002)	IEES (2003)	IER (1997)	IER (2005)	IER (2006)	IER et al. (2000)	IER et al. (2003)	Infras & IWW (2004)	McCubbin & Delucchi (1999)	Sommer et al. (1999)	Swedish National Road and Transport Research Institute (2006)	THE PEP (2004a)	United States Environmental Protection Agency (1999)	United States Environmental Protection Agency (2000)	Van Essen et al. (2007)
Literature review – more than one study considered	x	x					x		(x)		x			(x)	x			x
Way of valuating mortality																		
Value of a statistical life	+	+		+					+		+		+			+	+	
Value of life-years lost	+	+	+			+	+	+	+	+					+			+

+ : included
– : excluded (explicitly)
  : no information given
( ) : special case or exception

I.3.4.11 Specific focus on children

As mentioned in section I.3.4.5, four of the reviewed texts on air pollution treated children separately from adults for mortality. Two applied different values of statistical life to children than adults (AEA Technology Environment, 2005a, b), and two others applied the value of life-years lost approach to children, taking their longer life expectancy into account (IER, 2005; IER et al., 2000) (see also section I.3.4.10). Several studies gave specific concentration–response functions for health effects due to air pollution for children and infants (AEA Technology Environment, 2005a, b; IEES, 2003; IER, 2005). However, this specific focus on children in some of the studies was limited to certain aspects of measuring costs due to road transport–related air pollution (except IEES (2003), which focuses solely on school absence). The epidemiological literature, however, provides evidence showing that exposure to road transport–related air pollution is associated with respiratory disorders among children (see also section I.3.3).

I.3.4.12 Comparing the results from selected studies

As the previous chapters have shown, the studies vary in many important aspects and approaches applied. Hence, they also differ concerning their results for total health costs due to air pollution. Further, the texts reviewed also consider different geographical areas (such as countries, regions and communities), years of the survey, further definitions and assumptions (such as external costs or specific types of fuel or vehicles) and methods for measuring costs (such as average or total health costs), which all influence the final result of total health costs, so the results are not easily comparable. It was therefore not possible within the scope of this report to compare the results of all the studies analysed. As an example to at least indicate the magnitude of the total health costs due to air pollution, Table 58 shows the results from three selected studies that used more comparable methods.

Table 58. Overview of selected results of total health costs due to transport–related air pollution

Source	Mode	Geographical area	Year	Methods	Average health costs (euros per vehicle-km)	Total health costs (millions of euros)
Ecoplan et al. (2004a)	Road transport	Switzerland (entire country)	2000	all modes (passenger cars, public and private buses, motorized two-wheeled vehicle) WTP approach	0.0101	915
	Rail transport	Switzerland (entire country)	2000	passenger and good trains WTP approach	0.0019[b]	60
IER et al. (2003)	Road transport	Helsinki (urban area)	2003	only car petrol EURO02 WTP approach	0.0012	–
		Stuttgart (urban area)	2003	only car petrol EURO02 WTP approach	0.0025	–
		Berlin (urban area)	2003	only car petrol EURO02 WTP approach	0.0015	–
		Basle-Karlsruhe (inter-urban area)	2003	only car petrol EURO02 WTP approach	0.0037	–
		Strasburg-Neubrandenburg (inter-urban area)	2003	only car petrol EURO02 WTP approach	0.0011	–
		Milan-Chiasso (inter-urban area)	2003	only car petrol EURO02 WTP approach	0.0025	–
		Bologna-Brennero (inter-urban area)	2003	only car petrol EURO02 WTP approach	0.0020	–
Sommer et al. (1999)	Road transport	Austria (entire country)	1996	all modes WTP approach	–	2 892
		France (entire country)	1996	all modes WTP approach	–	21 615
		Switzerland (entire country)	1996	all modes WTP approach	–	2 216

[a]For average health costs.
bThe measurement unit is not vehicle-km but passenger-km.

I.4 Overview of results on transport-related insufficient physical activity

I.4.1 Introduction

Physical inactivity is an important determinant of burden of disease. In the European Region alone, physical inactivity causes about 600 000 deaths per year (5–10% of total mortality, depending on the country) and leads to the loss of 5.3 million DALYs (WHO, 2002). Globally, physical inactivity causes an estimated 10–16% of cases of breast, colon and rectal cancer and diabetes mellitus and about 22% of ischaemic heart disease. The negative health effects of physical inactivity have been demonstrated for many diseases (WHO, 2004). Estimates of the burden of disease attributable to physical inactivity have been calculated for ischaemic heart disease, ischaemic stroke, type 2 diabetes and colon and breast cancer.

In addition to the negative effects of physical inactivity, much research has gone into assessing the positive effects of physical activity in preventing a range of noncommunicable diseases and premature mortality due to the above and other causes. The benefits of physical activity are not limited to preventing or limiting the progression of disease but also include improving physical fitness, muscular strength, independence at old age and the quality of life. The following sections draw on evidence reviewed by Bull et al. (2004) and published by WHO and, more recently, the report of the Physical Activity Guidelines Advisory Committee (2008) published by the United States Department of Health and Human Services, other summaries such as that of the United States Institute of Medicine (2007) and other recent major studies and consensus statements, which were also discussed with a group of leading international experts contributing to this publication. The results are summarized below and in Table 59 and Table 60 with a focus on potential negative effects on inactivity in accordance with the rest of this report.

Transport plays a role in the level of physical activity by providing or hindering opportunities for cycling and walking. For the scope of this report, cycling and walking are understood as contributing to overall physical activity similar to other forms such as organized sport, dancing and gardening (Haskell et al., 2007). Specific evidence on the health benefits of transport-related cycling and walking is presented when available. However, few studies have looked at specific benefits for particular types of physical activity behaviour; instead, the outcome is often total physical activity (Physical Activity Guidelines Advisory Committee, 2008). Given this limitation, this report assumes that the evidence on overall physical activity is also valid for activity undertaken as transport-related cycling and walking.

I.4.2 Health end-points

I.4.2.1 All-cause mortality

The strongest and clearest evidence exists for the association between physical inactivity and an increased risk of death, as shown in numerous reviews and studies (Kesaniemi et al., 2001; Physical Activity Guidelines Advisory Committee, 2008; United States Department of Health and Human Services, 1996). Few studies have assessed all-cause mortality and commuter walking or cycling. One is a prospective study in Denmark that found a 40% increase in the risk of death among those who did not commute to work by bicycle compared with those who did (Andersen et al., 2000). A study reported similar results among women in China (Matthews et al., 2007). For walking, Hakim et al. (1999) found a decrease in all-cause mortality among men who walked 1–2 miles (1.6–3.2 km) per day. Evidence suggests a curvilinear dose–response relationship, and there is no evidence of a threshold (Physical Activity Guidelines Advisory Committee, 2008).

I.4.2.2 Cardiovascular disease

Strong evidence for the benefits of physical activity pertains to the reduction of the risk of mortality and morbidity from cardiovascular disease, particularly acute myocardial infarction and other forms of ischaemic heart disease (Bull et al., 2004; Institute of Medicine, 2007; Physical Activity Guidelines Advisory Committee, 2008). Biological plausible mechanisms for the positive effects of moderate and/or vigorous intensity of physical activity on ischaemic heart disease most likely function by improving the related risk factors, including atherosclerosis, lipid profile, ischaemia, blood pressure, thrombosis and fibrinolytic activity. Besides these intermediate effects, physical activity also acts as an independent risk factor. There is an inverse dose–response relationship between physical activity and cardiovascular disease incidence and mortality (Kesaniemi et al., 2001; Kohl 2001; Lee & Skerrett, 2001; Oja, 2001; Physical Activity Guidelines Advisory Committee, 2008). The exact shape of the dose–response curve is still being discussed, but assuming a linear shape could be seen as a conservative approach. Manson et al. (2002) demonstrated that walking was associated with a similar risk reduction for cardiovascular events (including both mortality and morbidity) as vigorous exercise. A meta-analysis by Murphy et al. (2007) also showed that taking up regular brisk walking significantly reduces cardiovascular risk factors among previously sedentary adults. A meta-analysis by Hamer & Chida (2008) also confirmed these positive findings of active commuting (combining walking and cycling) on cardiovascular end-points.

I.4.2.3 Stroke

The Physical Activity Guidelines Advisory Committee (2008) concluded that physical activity is inversely related to stroke (both ischaemic and for hemorrhagic stroke) but that data on stroke subtypes are still quite limited. This confirms an earlier conclusion by Kohl (2001) that the evidence for an association between physical activity and ischaemic stroke was equivocal. The biological mechanisms are thought, however, to be related to decreased atherosclerosis and hypertensive disease. The unclear picture might be partly due to the fact that many studies did not differentiate between ischaemic and haemorrhagic strokes (Bull et al., 2004), and studies that have looked at these outcomes separately were hampered by the decrease in the number of events, which impedes definite conclusions. A more recent review on preventing ischaemic stroke (Goldstein et al., 2006) concluded that a sedentary lifestyle is associated with an increased risk, with a level of evidence grouped as B (data derived from a single randomized trial or nonrandomized studies) on a scale from A to C.

I.4.2.4 Cancer

Physical activity is associated with a reduction in the overall risk of certain types of cancer (Bull et al., 2004; Institute of Medicine, 2007; Physical Activity Guidelines Advisory Committee 2008). Numerous studies have shown the protective effect of physical activity on the risk of colon cancer (Thune & Furberg, 2001). Biological mechanisms that have been suggested to explain this relationship include effects on prostaglandin levels, which act on colon mucosal cell proliferation, and effects on insulin-like growth factors. Based on a review of 41 studies, Thune & Furberg (2001) observed a graded inverse dose–response association between physical activity and colon cancer.

Most studies investigating the benefits of physical activity and breast cancer reported a risk reduction ranging from 20% to 80% among physically active women (Physical Activity Guidelines Advisory Committee, 2008). Biological mechanisms for this association are likely to be related to levels of female sex hormones. Thune & Furberg (2001) concluded that physical activity and breast cancer are inversely associated, with a dose–response relationship. Evidence for other types of cancer such as colorectal, lung or prostate cancer is increasing but still less conclusive (Physical Activity Guidelines Advisory Committee, 2008).

Matthews et al. (2007) found that both walking and cycling protect against mortality from cancer.

I.4.2.5 Type 2 diabetes

Both observational and experimental research studies strongly support the benefits of physical activity in preventing type 2 diabetes as well as related cardiovascular events and death (Bull et al., 2004; Institute of Medicine, 2007; Physical Activity Guidelines Advisory Committee, 2008). The biological mechanisms for this protection are related to the increased insulin sensitivity of trained muscle cells, increased glucose clearance (Dela et al., 1996), positive changes in the insulin cascade as well as more efficient transport of glucose from the blood into the muscle cells through training (Franch et al., 1999).

Regular physical activity is also an important component in treating type 2 diabetes.

I.4.2.6 Bone health

Recent evidence also suggests a strong relationship between physical activity and bone health (Institute of Medicine, 2007; Physical Activity Guidelines Advisory Committee, 2008).

I.4.2.7 Other health outcomes

Physical inactivity has recognized harmful effects on mental health (particularly depression), muscular health, functional health and quality of life. Less conclusive evidence indicates adverse effects on several other health outcomes, including sleep quality, Alzheimer’s disease, Parkinson’s disease, multiple sclerosis, other mental health conditions (including suicide) and chronic fatigue syndrome and cognition. For all these harmful effects, evidence is insufficient to compute the magnitude of risk reduction and the attributable burden associated with physical inactivity.

Since obesity is considered to be a separate risk factor, it is not included here. However, the available literature on how physical activity affects weight shows that habitual physical activity over lifetime can attenuate the increase in weight normally associated with increasing age. Participation in appropriate amounts of activity can prevent weight gain, lead to weight maintenance and support weight loss (Physical Activity Guidelines Advisory Committee, 2008). Murphy et al. (2007) also showed that sedentary adults taking up brisk walking positively affected body weight, body mass index and percentage body fat, among other outcomes.

I.4.2.8 Conclusions on adults

According to the literature presented, strong evidence indicates that physical activity reduces the risk of all-cause mortality as well as mortality and morbidity related to cardiovascular disease, stroke, breast and colon cancer, type 2 diabetes and morbidity related to bone health, anxiety and depression.

Dose–response relationships would be available for cause-specific mortality for cardiovascular disease, stroke, breast and colon cancer and type 2 diabetes (Bull et al., 2004). However, all-cause mortality is currently the most suitable health outcome to be included in economic assessment for several reasons:

- death from any cause is a stronger measure, as it takes account of all deaths and does not restrict the study to a specific predetermined subset of diseases;
- data on total mortality are likely to be relatively readily available in most countries, including at the local level, and to be less influenced by possible misclassification of the underlying cause of death; and

- the use of one simple parameter reduces the possibility of error in applying the model.

Economic valuation of physical inactivity should therefore be based on all-cause mortality since studies are available that allow the quantification of health effects among the people who do not carry out transport-related physical activity (Andersen et al., 2000). This is also in accordance with the conclusions of a recent project on economic valuation of cycling and walking (WHO Regional Office for Europe, 2007c).

From a population health viewpoint, morbidity benefits materialize more quickly than reductions in mortality. They can also be important individual motivators for walking and cycling, as people may be more likely to increase physical activity to improve their immediate health and well-being rather than to prolong their life.

However, evidence on morbidity is still less strong than evidence on mortality. Including the impact of morbidity in economic appraisal would therefore lead to greater uncertainty. One pragmatic option might be to include the notion that the morbidity burden represents an agreed proportion of the calculated mortality burden and to attach an appropriate monetary value.

Table 59. Overview of selected health end-points related to physical activity in adults

Health end-point	Strength of the evidence	Existence of biologically plausible mechanism(s)?	Sufficient information to allow quantification?	Valid exposure–effect relationship available?
Mortality				
All-cause mortality	Sufficient	Yes	Yes	Yes
Coronary heart disease	Sufficient	Yes	Yes	[a]
Ischaemic stroke	Sufficient	Yes	Yes	[a]
Type 2 diabetes	Sufficient	Yes	Yes	[a]
Colon cancer	Sufficient	Yes	Yes	[a]
Breast cancer	Sufficient	Yes	Yes	[a]
Morbidity				
Coronary heart disease	Sufficient	Yes	Yes	b
Ischaemic stroke	Sufficient	Yes	Yes	b
Type 2 diabetes	Sufficient	Yes	Yes	b
Colon cancer	Sufficient	Yes	Yes	b
Breast cancer	Sufficient	Yes	Yes	b
Prostate cancer	Inadequate	Uncertain	Limited	No
Rectal cancer	Inadequate	Uncertain	Limited	No
Musculoskeletal health	Limited	Uncertain	Limited	No
Bone health	Sufficient	Yes	No	No
Anxiety	Sufficient	Yes	No	No
Depression	Sufficient	Yes	No	No

[a]Available from Bull et al. (2004), but using all-cause mortality is suggested.
[b]Available from Bull et al. (2004), but it is suggested either not to include morbidity for the time being or to apply a certain proportion of the all-cause mortality burden (see section 3.5.3).
Sources: Bull et al. (2004), Institute of Medicine (2007) and Physical Activity Guidelines Advisory Committee (2008).

I.4.2.9 Children

Physical activity in children has been reported to affect several indicators of health and well-being, including musculoskeletal health and fitness, obesity, type 2 diabetes, mental effects and predictors of cardiovascular disease (metabolic syndrome, cholesterol and triglyceride levels, blood pressure and cardiovascular fitness). In addition, developmental and other quality of life effects on such factors as academic performance have been observed (Physical Activity Guidelines Advisory Committee, 2008; Strong et al., 2005).

An earlier analysis found the evidence base for these health and well-being effects of physical activity on children to be less convincing than the results for adults (Cavill et al., 2001), but more recent evidence based on improved definitions of health outcomes found strong evidence for a relationship between physical activity and/or fitness and a clustering of risk factors, particularly cardiovascular ones (Andersen et al., 2006; Anderssen et al., 2007; Ekelund et al., 2006).

According to more recent reviews, the evidence base is strongest for the beneficial effects of physical activity on aerobic fitness, musculoskeletal health, weight loss among overweight youth and cardiovascular and metabolic health (Institute of Medicine, 2007; Physical Activity Guidelines Advisory Committee, 2008; Strong et al., 2005). Evidence is suggestive but limited for mental health, including self-concept and anxiety and depression symptoms (Institute of Medicine, 2007; Strong et al., 2005).

The vast majority of epidemiological studies have been conducted on adults, however, mainly because the disease end-points (such as coronary heart disease) are rare in children. Further, demonstrating positive effects is difficult, since most young people are healthy and remain so during childhood and adolescence. In addition, young people are the most active segment of society. In addition, outcome variables change with normal growth, maturation and development, whether or not the individual is physically active regularly (Institute of Medicine, 2007; Physical Activity Guidelines Advisory Committee, 2008). As a consequence, very little evidence is available on dose–response relationships among children.

Table 60. Overview of selected health end-points related to physical activity in children

Health end-point	Strength of the evidence	Existence of biologically plausible mechanism(s)?	Sufficient information to allow quantification?	Valid exposure-effect relationship available?
Aerobic fitness	Sufficient	Yes	No	No
Strength and endurance	Sufficient	Yes	No	No
Skeletal health	Limited	Yes	No	No
Cardiovascular health	Sufficient	Yes	No	No
Blood pressure and metabolic indicators	Sufficient	Yes	No	No
Mental health	Limited	Uncertain	no	No

Sources: adapted from Institute of Medicine (2007) and Physical Activity Guidelines Advisory Committee (2008) and based on expert opinion.

I.4.3 Interactions between transport-related physical activity, air pollution, noise and road traffic injuries

With the introduction of transport-related physical inactivity and activity as a new topic into the discussion of transport-related health effects, the question arises on possible interactions between exercise through cycling and walking and exposure to ambient air pollution as well as road traffic injuries.

I.4.3.1 Physical activity and air pollution

A potentially higher exposure of active commuters to air pollution could be related to a higher personal uptake due to higher breathing rates, especially for cyclists, or to higher exposure to ambient pollutants. Unfortunately, no review on the benefits from active transport as a form of physical activity has considered the possible negative effects of ambient air pollution. Several individual studies have assessed this topic, such as O'Donoghue et al. (2007), showing that concentrations of hydrocarbons were significantly higher in buses than for cycling commuters. The levels became slightly higher for cyclists when respiration rates and travel times were considered. Two studies on exposure to benzene, toluene, ethylbenzene and xylenes showed, in contrast, that concentrations in cars were 2–4 times higher than for cycling and walking commuters and concluded that, even after accounting for the increased respiration rate of active commuters, car drivers seemed more exposed (Chertok et al., 2004; Rank et al., 2001). Another study including three of the four components of benzene, toluene, ethylbenzene and xylenes also found much higher concentrations in personal samples of car drivers (van Wijnen et al., 1995). After the measured ventilation of cycling study subjects was included (which was 2.3 times that of car drivers), uptake only sometimes approached that of car drivers. A study solely examining benzene found that car drivers the highest exposure (Kingham et al., 1998). Chertok et al. (2004) also assessed NO_2 exposure and found that cycling and walking commuters had significantly lower exposure than bus commuters and exposure comparable to that of car commuters. van Wijnen et al. (1995) found that concentrations of NO_2 differed little between car drivers, cyclists and pedestrians, but when ventilation was taken into account, cyclists' uptake of NO_2 was higher than that of drivers.

Adams et al. (2001) carried out a study on personal exposure to fine particles ($PM_{2.5}$) in central London and found that cyclists had the lowest exposure; bus and car were slightly higher and the underground rail system was considerably higher. They also showed that personal exposure in modes of road transport was about twice that of an urban background fixed-site monitor and that between-route variation was significant. Kingham et al. (1998) also looked at inhalable particles and found that car drivers and cyclists had comparable exposure on general roads but that cyclists were significantly less exposed on cycle paths. They also found that wind speed significantly influenced pollution levels. Gulliver & Briggs (2004) also found that car drivers and walkers have similar personal exposure to different types of particles when the routes are comparable.

Some studies applied a more clinical approach, such as the one by Mills et al. (2007), a chamber study among 20 men with a history of myocardial infarction exposed to either dilute diesel exhaust or filtered air that found elevated risks. Although these studies often have limited potential for generalization and this one did not report the fitness or levels of physical activity of the subjects, they have to be taken into account with regard to biological pathways.

Although some studies indicate, as discussed above, that cyclists are exposed to lower average concentrations, their journey times maybe be longer, thus compensating largely for the reduced average exposure (Krzyzanowski et al., 2005). However, this might not take into account the fact that the travel speed of cars is often comparable to that of bicycles, especially in city centres where most cycling trips are made or – in case of congestion – even slower than that of bicycles. In recent examples, the average speeds in city centres in France were about 15 km/h (Doll & Maibach, 2006). In addition, time spent in cars to find a parking space is often not taken into account.

As yet, available evidence from these air pollution studies, which were often based on small sample sizes, does not allow clear conclusions on the extent to which the negative effect of long-term air pollution influences the significant positive effects of commuter cycling (Andersen et al., 2000; Matthews et al., 2007). The results differed depending on the type of air pollutant studied, and cyclists seem to have only had clearly higher uptake of NO_2 on general roads. Since the air pollution studies were not longitudinal in design and did not examine health outcomes, the extent to which short episodes of increased exposure influence long-term health outcomes compared with long-term background exposure also remains open. One exception is days with high O_3 levels, which have been shown to have negative but reversible effects on pulmonary functioning. However, this is short-term peak exposure and must be clearly distinguished from long-term exposure and chronic health effects.

The two longitudinal studies on commuter cycling and mortality (Andersen et al., 2000; Matthews et al., 2007) did not assess the individual exposure of their subjects. However, participants in the study in China were probably exposed to significantly higher levels of air pollution than those in Copenhagen (World Bank, 2007). These studies indicate that the benefits from exercise related to active commuting are probably more important for health than the possible negative effects of long-term air pollution, as also concluded in the recent review by the Physical Activity Guidelines Advisory Committee (2008).

I.4.3.2 Physical activity and road noise

To our knowledge, no study has assessed the possible effects of cycling and walking on noise levels. It can be assumed, however, that increasing active commuting could lead to lower noise levels if cycling and walking replace car trips. There is potential for such substitution, as many car trips in Europe are short (European Commission, 1999). Such a modal shift could also positively affect air pollution.

I.4.3.3 Physical activity and road traffic injuries

Another possible offset to the benefits from commuter cycling and walking could come from a higher risk of road crashes. First, the number of deaths associated with physical inactivity in the European Region is estimated to be about five times as high as those caused by road crashes (WHO, 2002). Nevertheless, as explained in section I.1.2.1, the risks for cyclists and pedestrians per kilometre travelled are on average considerably higher than those for vehicle occupants. Nevertheless, car drivers travel many kilometres on safe motorway infrastructure. If the kilometres driven in urban centres are compared, the risk of crashes is much more comparable between car drivers and cyclists (European Commission, 1999). On the other hand, comparisons between countries in Europe do not support the suggestion that more cycling and walking could increase the number of road crashes (Ensink & Zeegers, 2005; Jacobsen, 2003). Increased active transport appears to be linked to reduced road crash deaths, implying that the increasing presence of pedestrians and cyclists improves the awareness of motor vehicle drivers or that policies to separate motorized from non-motorized transport are effective. An analysis in Australia confirmed these findings (Robinson, 2005), and this conclusion is also supported by the comparison of fatality and injury rates in Germany and the Netherlands, with relatively high levels of cycling and walking compared with those in the United States (Pucher & Dijkstra, 2003). Additional evidence comes from the Odense Cycling City project (Troelsen et al., 2004), where the number of road crashes involving cyclists decreased despite large increases in cycling over time. The trends from the London Congestion Charge project (Mayor of London – Transport for London, 2007) are not conclusive in this regard but could indicate a slight increase in cycle crashes (Noland et al., 2008). The reasons are not yet well understood; one possible explanation, however, could be the lack of measures taken to provide appropriate infrastructure for the increasing number of cyclists and possibly more inexperienced cyclists. Another reason could be that different population groups use cars and bicycles:

younger age groups are more often active commuters and are also generally at higher risk for road crashes due to less awareness of the dangers.

In addition, the Physical Activity Guidelines Advisory Committee (2008) concluded that injury rates from commuting activity were much lower than for other forms of physical activity such as sports and that the benefits of regular physical activity outweigh the inherent risk of adverse events, including injuries.

I.4.3.4 Conclusions

Overall, evidence suggests that, if promotion of active commuting is accompanied by suitable transport planning and safety measures (which could lead to decreased air pollution exposure if cycling can be done further away from main roads), active commuters are likely to benefit from the safety-in-numbers effect. Robinson (2005) suggested the formula: (injuries)/(amount of cycling) is proportional to (amount of cycling) – 0.6 and reported that, if cycling doubles, the risk of death and serious injuries falls by about 34%; conversely, if cycling halves, the risk per kilometre will be about 50% higher. In a conservative model, an assumption could be made that the risk of road traffic injuries remains unchanged, as done by Rutter (2006) in calculating the economic benefits from commuter cycling through reduced mortality.

I.4.4 Economic approaches

Of the texts reviewed, 11 discussed the health effects and costs due to insufficient physical activity or to physical activity, respectively (see also Table 1 for an overview of all analysed economic studies).

I.4.4.1 The model used in the studies analysed – activity and insufficient activity

The model used in 9 of the 11 studies analysed for their measurement of costs due to insufficient physical activity was the following:

Total health costs[55] = (number of insufficient physically active people × costs per related case of illness)/mortality

Two studies examined the benefits from physical activity instead of the costs of insufficient activity (total and per case), and two looked at both costs and benefits (Table 61).

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55 The term “total health costs” refers in the whole text to the sum of all different costs of a health effect considered in this study. As this study does not consider some costs of a health effect, for example, because they cannot yet be monetized, the term “total health costs” as used here does not cover all possible total health costs that occur in reality..

Table 61. Focus regarding activity or inactivity (insufficient activity) considered in the studies reviewed

	Study										
	Allender et al. (2007)	Cavill et al. (2007)	Chenowith et al. (2005)	Colditz (1999)	Colman & Walker (2004)	Department for Culture, Media and sport (2002)	Katzmarzyk & Janssen (2004)	Popkin et al. (2006)	Stephenson et al. (2000)	Swiss Federal Office of Sport et al. (2001)	THE PEP (2004a)
Literature review – more than one study considered		x		x				x			x
Physical activity versus physical inactivity											
Physical activity	+			+					+		+
Physical inactivity		+	+	+	+	+	+		+	+	

+ : included
– : excluded (explicitly)
 : no information given
( ) : special case or exception

Nevertheless, the two approaches are comparable in most ways and basically deal with the same questions. In a simplifying manner, the two models can be seen as two identical equations with different signs: inactive people (insufficient active) versus active people, costs versus benefits. Thus, the costs of insufficient physical activity are calculated in terms of health care and economic production losses (see section I.4.4.7), and reduced health costs and absenteeism are considered in the valuation of physical activity. The same pattern applies to most health effects (see sections I.4.4.4 and I.4.4.5): exposure–response functions indicate, for example, that physical activity reduces mortality or insufficient physical activity increases mortality, respectively. Hence, the texts on physical activity and on physical inactivity (insufficient activity) are discussed and compared together and are referred to as "texts on insufficient physical activity" except if the difference has important implications for the issue discussed.

Four texts (Cavill et al., 2007; Colditz, 1999; Popkin et al., 2006; THE PEP, 2004a) are literature reviews themselves. Hence, the information in these texts originates from several different studies. The information presented from these four studies therefore might seem contradictory (the different mode of transport considered by Cavill et al. (2007); see section I.4.4.2). However, this is just due to information originating from different sources: one text of the review considered walking only, whereas another focused on both walking and cycling (see section I.4.4.2).

I.4.4.2 Type of physical activity considered

As with the other three topics, the first question is related to the decision on which mode of transport was considered. The studies on insufficient physical activity investigated either insufficient activity in general or two different modes (walking and cycling) or combinations of these modes (Table 62).

Table 62. Modes of transport considered in the studies on insufficient physical activity

	Study										
	Allender et al. (2007)	Cavill et al. (2007)	Chenoweth (2005)	Colditz (1999)	Colman & Walker (2004)	Department for Culture, Media and Sport (2002)	Katzmarzyk & Janssen (2004)	Popkin et al. (2006)	Stephenson et al. (2000)	Swiss Federal Office of Sport et al. (2001)	THE PEP (2004a)
Literature review – more than one study considered		x		x				x			x
Type of physical activity considered											
Walking only		+									
Cycling only		+									
Walking and cycling		+									+
Physical (in-)activity in general	+		+	+	+	+	+	+	+	+	

+ : included
– : excluded (explicitly)
  : no information given
( ) : special case or exception

I.4.4.3 Effects of substitution

For the two studies that investigated a specific type of physical activity (Cavill et al., 2007; THE PEP, 2004a), the effects of substitution have to be taken into account to determine the total number of transport-related insufficiently physically active people. For example, an increase in commuter cycling does not necessarily mean that people's overall physical activity increases as well, as they might reduce physical activity in other settings (for example, less leisure sport). The same is true the other way round: some people changing from cycling and walking to the use of motorized transport might be as physically active as before as they could increase leisure physical activity. Hence, statements on the total number of physically active or inactive people in relation to transport must always consider overall physical activity and possible substitution. This can be done either by using a relative risk that controls for other types of physical activity or by incorporating a factor into the calculations.

Of the two studies for which the question of substitution was relevant, Cavill et al. (2007) discussed this issue, reporting that some of the studies reviewed had considered this but that most had not (Table 63).

Table 63. Effects of substitution considered in the studies on insufficient physical activity

	Study										
	Allender & Foster et al. (2007)	Cavill et al. (2007)	Chenoweth (2005)	Colditz (1999)	Colman & Walker (2004)	Department for Culture, Media and Sport (2002)	Katzmarzyk & Janssen (2004)	Popkin et al. (2006)	Stephenson et al. (2000)	Swiss Federal Office of Sport et al. (2001)	THE PEP (2004a)
Literature review – more than one study considered		x		x				x			x
Effects of substitution considered											
Yes		+									
No											

\+ : included
– : excluded (explicitly)
 : no information given
( ) : special case or exception

I.4.4.4 Health effects considered: mortality

As discussed in section I.4.2, numerous epidemiological studies have shown that insufficient physical activity causes several health effects, including mortality. Table 64 presents an overview of the health effects related to mortality considered in the economic studies reviewed. Except for two effects (angina pectoris and myocardial infarction), most texts included more or less the same causes of mortality. As an option, all-cause mortality was considered instead of cause-specific mortality (Cavill et al., 2007; Stephenson et al., 2000).

Table 64. Health effects: causes of mortality considered in the studies on insufficient physical activity

	Study										
	Allender & Foster et al. (2007)	Cavill et al. (2007)	Chenoweth (2005)	Colditz (1999)	Colman & Walker (2004)	Department for Culture, Media and Sport (2002)	Katzmarzyk & Janssen (2004)	Popkin et al. (2006)	Stephenson et al. (2000)	Swiss Federal Office of Sport et al. (2001)	THE PEP (2004a)
Literature review – more than one study considered		x	x	x				x			x
Health effects: mortality											
All-cause mortality		+							(+)[a]		
Cause-specific mortality											
Cardiovascular disease: in general								+		+	
Coronary or ischaemic heart disease	+	+			+				+		
(Ischaemic) stroke	+	+			+	+		+	+		
Angina pectoris						+					
Myocardial infarction						+					
Cancer (colon or rectal, breast cancer)	+	+			+	+			+	+	
Type 2 diabetes	+	+			+	+			+	+	

[a]Only for comparison of the cost-of-illness approach.

+ included

- excluded (explicitly)

no information given

( ) special case or exception

I.4.4.5 Health effects considered: morbidity

The 11 economic studies reviewed differentiated up to 12 health effects due to insufficient physical activity (Table **65**).

Table 65. Health effects: types of morbidity differentiated in the studies on insufficient physical activity

	Study										
	Allender & Foster et al. (2007)	Cavill et al. (2007)	Chenoweth (2005)	Colditz (1999)	Colman & Walker (2004)	Department for Culture, Media and Sport (2002)	Katzmarzyk & Janssen (2004)	Popkin et al. (2006)	Stephenson et al. (2000)	Swiss Federal Office of Sport et al. (2001)	THE PEP (2004a)
Literature review – more than one study considered		x		x				x			x
Health effects: morbidity											
Coronary heart disease		(+)	+	+	+		+	+	+		
Stroke	+				+		+	+	+		
Hypertension		(+)		+	+	+		+	+	+	+
Type 2 diabetes	+	(+)	+[a]	+	+		+	+	+		+
Gallbladder disease				+				+			
Cancer[b]	+		+	+	+		+		+		+
Mental health, depression		(+)	+				+			+	
Nervous system diseases			+								
Musculoskeletal health[c]		(+)	+	+	+	+		+	+		+
Overweight			+					+			
Obesity		(+)	+	+				+	+		
Other health effects (ill-defined)			+								

[a]Including gout and impaired immune response.
[b]Including breast cancer, colon and rectum cancer and endometrial cancer (Popkin et al., 2006) or cancer in general.
[c]Including osteoporosis (hip fracture for Popkin et al. (2006)),
osteoarthritis, back pain and other musculoskeletal problems.
+ included
– excluded (explicitly)
no information given
( ) special case or exception

The review of epidemiological literature on insufficient physical activity (section I.4.2) concluded that strong evidence indicates that physical activity reduces the risk of all-cause mortality as well as mortality and morbidity related to cardiovascular disease, stroke, breast and colon cancer, type 2 diabetes and bone health and morbidity from anxiety and depression. However, all-cause mortality was suggested to be the most suitable health outcome for now to be included in economic assessment, as including morbidity would lead to greater uncertainty. One pragmatic option might be to include the notion that morbidity benefits represent an agreed proportion of the calculated mortality benefits and to attach an appropriate monetary value.

I.4.4.6 Other effects considered

Some of the studies on physical activity investigated several other effects, such as road crashes,[56] air pollution, noise, parking costs and congestion and travel time (Table 66). However, most of the studies reviewed considered none or only a few of these effects. One study also looked at effects from unhealthy diets and low birth weight (Popkin et al., 2006).

Section I.4.2 discusses interactions between insufficient physical activity, air pollution, noise and road crashes.

Table 66. Other effects considered in the studies on insufficient physical activity

	Study										
	Allender & Foster et al. (2007)	Cavill et al. (2007)	Chenoweth (2005)	Colditz (1999)	Colman & Walker (2004)	Department for Culture, Media and Sport (2002)	Katzmarzyk & Janssen (2004)	Popkin et al. (2006)	Stephenson et al. (2000)	Swiss Federal Office of Sport et al. (2001)	THE PEP (2004a)
Literature review – more than one study considered		x		x				x			x
Other effects					–						
Dietary factors								+			
Low birth weight								+			
Road crashes		+				(+)				(+)	
Air pollution		+									+
Noise											+
Parking costs											+
Congestion and travel time											+

\+ : included
– : excluded (explicitly)
 : no information given
( ) : special case or exception

I.4.4.7 Costs considered

As explained previously, total health costs are composed of direct and indirect costs as well as intangible costs (see sections I.1.3.5, I.3.4.7 and I.2.3.8).[57] All studies in the literature reviewed on insufficient physical activity

[56] In the studies by the Swiss Federal Office of Sport et al. (2001) and Department for Culture, Media and Sport (2002), not road crashes but sport accidents were considered as they measured total health costs due to insufficient physical activity in general and not related to transport (see section I.4.4.2).

[57] Depending on the focus of a study, physical activity versus insufficient physical activity, here one would rather talk of benefits instead of costs. However, as already explained in section I.4.4.2**,** the different components of benefits can generally be easily transformed into cost components.

considered the direct and indirect costs (Table 67), although some only focused on the costs of health care or economic production losses, respectively. Only four studies considered intangible costs.[58]

Table 67. Types of costs considered in the studies on insufficient physical activity

	Study										
	Allender & Foster et al. (2007)	Cavill et al. (2007)	Chenoweth (2005)	Colditz (1999)	Colman & Walker (2004)	Department for Culture, Media and Sport (2002)	Katzmarzyk & Janssen (2004)	Popkin et al. (2006)	Stephenson et al. (2000)	Swiss Federal Office of Sport et al. (2001)	THE PEP (2004a)
Literature review – more than one study considered		x		x				x			x
Costs considered											
Direct and indirect costs											
Costs of health care (including rehabilitation and reintegration)	+	+		+	+	+	+	+	+	+	+
Economic production losses (including replacement costs)		+	+	+	+	+		+	+	+	+
Intangible costs											
For victims (suffering and grief)	(+)	+	(+)								+

+	:	included
–	:	excluded (explicitly)
	:	no information given
( )	:	special case or exception

I.4.4.8 Methods of measuring costs

As explained in section 3.1.1**,** total health costs can be estimated by using the willingness-to-pay approach only or in combination with the cost-of-illness approach. Of the studies reviewed, only three applied the combination method (Allender et al., 2007; Chenoweth, 2005; THE PEP, 2004a) (Table 68). The others used the cost-of-illness approach only. This is due to the fact that these studies did not consider intangible costs (see section I.4.4.7).

58 To estimate the intangible costs, Allender et al. (2007) and Chenoweth (2005) applied the disability-adjusted life-years (DALYs) method.

Table 68. Different ways of measuring costs in the studies on insufficient physical activity

	Study										
	Allender & Foster et al. (2007)	Cavill et al. (2007)	Chenoweth (2005)	Colditz (1999)	Colman & Walker (2004)	Department for Culture, Media and Sport (2002)	Katzmarzyk & Janssen (2004)	Popkin et al. (2006)	Stephenson et al. (2000)	Swiss Federal Office of Sport et al. (2001)	THE PEP (2004a)
Literature review – more than one study considered		x		x				x			x
Measuring costs											
Cost of illness and willingness to pay	+		+								+
Cost of illness only		+		+	+	+	+	+	+	+	
Willingness to pay only											

+ : included
– : excluded (explicitly)
: no information given
( ) : special case or exception

The three studies that apply the combined method of cost of illness and willingness to pay to calculate total health costs due to insufficient physical activity (Allender et al., 2007; Chenoweth, 2005; THE PEP, 2004a) provided no information on whether they used net or gross economic production losses (compare section 3.1.1 and sections I.1.3.7, I.3.4.9 and I.2.3.10).

I.4.4.9 Valuating mortality

Mortality can be valuated in two different ways: either as the value of a statistical life or as the value of life-years lost (see also sections I.1.3.8, I.3.4.10 and I.2.3.11). Of the 11 studies on insufficient physical activity, only 4 gave information on this. Colman & Walker (2004) and Stephenson et al. (2000) thereby did not use the value of life-years lost but the total number of life-years lost due to insufficient physical activity.

Table 69. Different ways of valuating mortality in the studies on insufficient physical activity

	Study										
	Allender & Foster et al. (2007)	Cavill et al. (2007)	Chenoweth (2005)	Colditz (1999)	Colman & Walker (2004)	Department for Culture, Media and Sport (2002)	Katzmarzyk & Janssen (2004)	Popkin et al. (2006)	Stephenson et al. (2000)	Swiss Federal Office of Sport et al. (2001)	THE PEP (2004a)
Literature review – more than one study considered		x		x				x			x
Valuating mortality											
Value of a statistical life		+									
Value of life-years lost		+			(+)				(+)		+

+ : included
– : excluded (explicitly)
  : no information given
( ) : special case or exception

I.4.4.10 Specific focus on children

All the studies on insufficient physical activity reviewed focused solely on adults. Ideally, children should be included in such cost estimations as the values could be considerable, also since insufficient physical activity is one of the reasons for increasing overweight and obesity in children. However, children should be treated differently than adults in terms of health effects due to insufficient physical activity as the relative risk and exposure–response curves are likely to be different for children than for adults (see section I.4.2). However, children's health costs related to insufficient physical activity cannot yet be measured, as very little evidence on exposure–response relationships for children is available as yet.

I.5 Overview of results on other transport-related effects

Four health effects due to road transport have been discussed: road crashes, air pollution, noise and insufficient physical activity (see sections I.1, I.2, I.3 and I.4). However, some studies mentioned further transport-related effects with implications for human health. Table 70 presents an overview of these effects (studies that did not mention "other health effects" are not listed here). Most often, effects related to climate change and global warming and to nuclear risks were included.

Table 70. Other transport-related effects with implications for human health considered in the studies reviewed

	Study									
	AEA Technology Environment (2005a)	DIW et al. (2000)	IER (1997)	IER (2005)	IER (2006)	IER et al. (2000)	IER et al. (2003)	Infras & IWW (2004)	Van Essen et al. (2007)	WHO (2004)
Literature review – more than one study considered	x			x		(x)		x	x	x
Other health effects considered										
Climate change and global warming	+	+	+	+	+	+	+	+	+	+
Vibration		+								
Mental effects										+
Nuclear risks		+	+			+		+	+	
Major accidents in non-nuclear fuel chains			+	+						

+ : included
– : excluded (explicitly)
  : no information given
( ) : special case or exception

Annex 2. Literature identified that was not included in the review of economic approaches

Annesi-Maesano I et al. (2007). Residential proximity fine particles related to allergic sensitisation and asthma in primary school children. *Respiratory Medicine*, 101:1721–1729. A study on the long-term health impacts of nitrogen oxides and fine particle emissions on more than 7000 children in six French cities. No economic valuation; the topic is covered in the health end-points part of the review.

Carthy T et al. (1999). On the contingent valuation of safety and the safety of contingent valuation. Part 2. The CV/SG "chained" approach. *Journal of Risk and Uncertainty*, 17:187–214. Focuses only on how to valuate life years lost from mortality.

Committee on the Medical Effects of Air Pollutants (2002). *The quantification of the effects of air pollution on health in the United Kingdom*. London, Committee on the Medical Effects of Air Pollutants. Single-country study and the long-term effects are left out although they are most important.

DeJong G et al. (2003). *Executive summary – the economic cost of physical inactivity in Michigan*. Lansing, MI, Governor's Council on Physical Fitness, Health & Sports and the Michigan Fitness Foundation (http://www.michiganfitness.org/Publications/documents/CostofInactivity.pdf, accessed 25 August 2008). Only a four-page summary available; the aspects included were the same as in Chenoweth et al. (2005), but the methods are described much more briefly.

Garrett NA et al. (2004). Physical inactivity: direct cost to a health plan. *American Journal of Preventive Medicine*, 27:304–309. Calculated only health care expenditure based on example of one health plan in Minnesota, USA.

Hubbell BJ et al. (2005). Health-related benefits of attaining the 8-hr ozone standard. *Environmental Health Perspectives*, 113:73–82. The health effects considered are not specifically transport-related.

Lindberg G (2006). *Marginal cost case studies for road and rail transport*. Leeds, ITS, University of Leeds. Mainly about infrastructure costs and other kinds of costs of no relevance in this context.

Maibach M et al. (2008). *Handbook on estimation of external costs in the transport sector*. Produced within the study Internalisation Measures and Policies for all External Costs of Transport (IMPACT). Delft, CE Delft (http://ec.europa.eu/transport/costs/handbook/doc/2008_01_15handbook_external_cost_en.pdf, accessed 25 August 2008). No additional information, as the suggestions in the handbook are mainly based on the surveys of CAFE and UNITE.

METHODEX (methods and data on environmental and health externalities) news reports [web site]. Oxon, METHODEX, 2008 (http://www.methodex.ord/news.htm, accessed 25 August 2008). None of the texts were included in the literature review (focus only on marginal costs, health costs of minor importance, single-country studies, etc.).

Minnesota Department of Health (2002). *Health care costs of physical inactivity in Minnesota*. Minneapolis, Minnesota Department of Health (http://www.health.state.mn.us/divs/hpcd/chp/obesity/pdf/costsofphysicalinactivity_fact.pdf, accessed 25 August 2008). Only a three-page fact sheet but the full report is not available. Looking at health care costs only; the methods were similar to Chenoweth et al. (2005).

NEEDS – New Energy Externalities Developments for Sustainability project reports [web site]. Rome, NEEDS, 2006 (http://www.needs_project.org/index.php?option=com_content&task=viewed=42itemid=66, accessed 25 August 2008). Very little information on how the empirical surveys were conducted.

Organization for Economic Co-operation and Development (OECD) (2006): Economic valuation of environmental health risks to children. Paris, OECD. Extensive discussion of relevant issues but not transport-related.

Swedish Environmental Protection Agency (2005). *Den samhällsekonomiska nyttan av cykeltrafikåtgärder. Förbättring av beslutsunderlag [The macroeconomic benefits of cycling. Improving the decision-making basis]*. Stockholm, Swedish Environmental Protection Agency Study on the benefits of physical activity not the costs of insufficient activity.

United States Environmental Protection Agency (2003). *The benefits and costs of the Clean Air Act, 1990–2020*. Washington, DC, United States Environmental Protection Agency. Single-country study mainly focusing on scenarios (trends in air pollution and its costs in the future).

United States Environmental Protection Agency (2004). *Final regulatory analysis – control of emissions from nonroad diesel engines*. Only health effects due to non-road air pollution are considered.

Wang G et al. (2004). Physical activity, cardiovascular disease, and medical expenditures in U.S. adults. *Annals of Behavioral Medicine*, 28:88–94. Looking only at one specific disease and health care expenditure.

Annex 3. Analysis grid for the review of economic literature

Criteria	Characteristics	Further remarks
Year of the publication Year of the data		
Research question or aim	Knowledge on a certain policy issue	
	A certain policy outcome to achieve	
Research objects (one or several research objects for each research question)	Each research object targets one specific sub-research question	
	Each research object is further defined by certain conditions (including or excluding) and (geographical and socioeconomic) scope	Of special interest are age groups (such as children and adults)
Theoretical model (for each research object or as a combination of research objects)	Each model consists of variables and their relationships to each other	Of special interest are conditional relations and, if part of the model, the type of health effects
Indicators (for each variable)	Indicators define the way on how a variable is (empirically) measured	Of special interest are the indicators for measuring illnesses and costs
	Each variable has one or several indicators	If there is more then one indicator, the relationship between the indicators is also important (weight and arithmetic)
Overall advantages and disadvantages	Overall advantages and disadvantages to create guidance	
General remarks		

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AEA Technology Environment (2005b). *Methodology for the cost-benefit analysis for CAFE*. Volume 2. *Health impact assessment*. Oxon, AEA Technology Environment.

Alberini A, Hunt A, Markandya A (2006). Willingness to pay to reduce mortality risks: evidence from a three-country contingent validation study. *Environmental and Resource Economics*, 33:251–264.

Allender S et al. (2007). The burden of physical activity-related ill health in the UK. *Journal of Epidemiology and Community Health*, 61:344–348.

Andersen LB et al. (2000). All-cause mortality associated with physical activity during leisure time, work, sports and cycling to work. *Archives of Internal Medicine*, 160:1621–1628.

Andersen LB et al. (2006). Physical activity and clustered cardiovascular risk in children: a cross-sectional study (the European Youth Heart Study). *Lancet*, 368:299–304.

Anderson HR et al. (2004) *Meta-analysis of time-series studies and panel studies of particulate matter (PM) and ozone (O_3). Report of a WHO task group*. Copenhagen, WHO Regional Office for Europe (document EUR/04/5042688; http://www.euro.who.int/document/e82792.pdf, accessed 25 August 2008).

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Bistrup ML et al. (2001). *Health effects of noise on children and perception of the risk of noise*. Copenhagen, National Institute of Public Health (http://www.niph.dk/upload/health-effects-noise-children.pdf, accessed 25 August 2008).

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