5 Valuing the health impact of air pollution: Deaths, DALYs or Dollars?[i]

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In this chapter we investigate the feasibility of different indicators to quantify the health impact of air pollution: 1) attributable mortality *risk*, numbers of fatalities, 2) attributable *burden of disease* (disability-adjusted life-years: DALYs), and 3) monetary evaluation of health loss (*'willingness to pay'*). We focus on the health loss associated with the major air pollution phenomena in the Netherlands (indoor as well as outdoor): particulates (long and short term), ozone, ETS, indoor radon, and dampness. Our calculations are based on health impact estimates for the Dutch population, which were produced in the framework of the fifth National Environmental Outlook.

[i] Based on Hollander AEM de, Melse JM. Valuing the health impacts of air pollution: deaths, DALYs or dollars. In: Ayers J, Maynard B (eds) Air pollution and Health. Londen: Imperial College Press (in press, 2004).

5.1 Fifty years on

Fifty years after the infamous London smog of 1952 killed 4,000 to 12,000 inhabitants air pollution might still be a significant health risk factor[1,2]. Since those days air pollution in Western Europe has changed greatly, with regard to both composition and concentrations. In the 19th century, despite some legislation to prevent smoke nuisance from industrial emissions, levels of fossil fuel-related pollutants, such as smoke and sulphur dioxide, were at least tenfold higher than nowadays. Until the 1960s, in large cities such as London, Paris or Amsterdam, the bulk of air pollution came from domestic sources. Since then the widespread use of coal, especially for domestic fires has disappeared almost completely in highly developed countries. High sulphur coal of varying quality was replaced by natural gas of which abundant stocks were discovered in the sixties, e.g. in the North Sea and north-eastern parts of the Netherlands. However, in the same period the volume of road traffic grew exponentially and in spite of impressive, concurrent development of clean engine technology, road traffic is now the most important source of air pollution, accounting for more than one third of fine particulate matter (PM_{10}) emissions (for the smaller fractions, $PM_{2,5}$, more than half), and more than half of nitrogen dioxide emissions[3,4].

It is generally accepted modern day ambient air pollution still affects public health, although the precise causal fraction of the air pollutant mix remains a subject of fierce debate[5,6]. As we spend the greater part of our days inside houses and buildings, indoor air pollution is inevitably of great relevance to public health as well, in particular exposure to radon, second hand tobacco smoke and dampness related allergens all of which contribute significantly to disease burden[7]. In most Western European countries environmental policy is not very much concerned with indoor air quality, at least not in the light of recent discussions on (proposed) legislation with respect to ambient air pollution. Nevertheless in this chapter health impacts of indoor air pollution will be considered as a reference to put the ambient air pollution health risk debate in a wider perspective.

5.2 Problems for policy makers

The rapid advancements in statistical methodology have not really made life easier for policy makers involved in air pollution control. In the 'good old days' risk managers would judge the air quality by compliance with health-based standards. As long as concentrations of air pollutants were below these standards the air was 'safe' to breathe. Whenever concentrations started to exceed standards regularly public health was at stake and risk-reducing measures had to be considered. These standards were - and still are - often primarily based on guidelines derived by expert committees after careful consideration of available toxicological and epidemiological evidence. These 'compound by compound' evaluations are completed with proposals for safe ambient air pollution concentrations using simple, quantitative models. In these models science, societal preferences and policy are elegantly mixed, as establishing air quality guidelines requires several normative choices to be made[8,9]. For instance, one has to define critical toxicological end-points and decide on the extent of the safety margins, given the quantity and quality of the exposure-response data.

Current epidemiological insights do not comply very well with this type of quality standard-based risk management as clear evidence of a threshold for the health end-points considered appears to be lacking, at least at realistic levels of ambient air pollutant concentrations[6]. Furthermore, the observed end-points include a large variety of health effects, ranging from mild, reversible lung function deficits, slightly restricted potential for physical activity to hospital admission and mortality among the susceptible (figure 4-1). Obviously the nature and severity of the health effects depend largely on the individual health status. In this situation drawing the line between trivial and health threatening responses is difficult (see chapter 2).

In the absence of clearly defined safe or virtually safe levels there is no easy way out of discussions on the acceptability of health responses associated with population exposure to air pollution[10,11]. In the last few decades much effort has been put into controlling air pollution associated with key activities, such as transport, energy production, industry, and waste treatment. Current levels are largely determined by best available technology and further emission reductions to meet more stringent standards require ever-bigger resources. Beside constraints on the opportunities for economic activity that might produce more wealth and well being, these resources cannot be used for other things we value, such as health care or the quality of the urban environment. Furthermore, several analyses have cast some doubt on the efficiency of standard-based health risk management, as very expensive measures often appear to yield fairly minor public health gains (and vice versa)[12]. Some authors even suggest that very expensive life-saving regulations might be counterproductive, as a drop in the aggregate national income of around 8 million Euro will theoretically induce one extra fatality[13]. Table 3.3 in chapter 3 shows a remarkable variation in costs per healthy life-year that is saved for a broad range of life saving interventions, of which environmental measures are often found at the more expensive end of the distribution[14].

5.3 Putting money where public health profits most

As pure standard-based decision rules no longer seem to apply fully, a shift is being made towards rules based on 'utility', putting the money where public health profits most[15]. This is not without significance as environmental quality standards traditionally stand for *equity*, the right to protection from adverse effects for everybody, regardless of age, health, and susceptibility (right-based decision rules). On the other hand, from an utilitarian perspective maximisation of utility (efficiency) may require a more skewed distribution of health risk over societal groups, simply because it would be more cost-effective. Or, in other words: guaranteeing the last citizen in town the general level of health protection can be extremely expensive[16]. Here, according to neo-classical economic theory, the trade-off between efficiency and equity becomes an issue.[17]

From a utilitarian perspective, when considering air pollution, certain questions have to be addressed:

- How bad is this environmental exposure, e.g. compared to other environmental exposures or other health risks in general?
- How does public health benefit from policy measures to reduce public exposure?
- What policy measures are most efficient or what is the optimal deployment of available resources in terms of health gains?

To describe and compare the health impact of various environmental exposures and, eventually, to perform cost effectiveness analysis of options for environmental policies obviously some sort of 'denominator' is required. Generally speaking three ways of characterising potential health benefits are being used: health risk reduction (numbers), (health adjusted) life-years (e.g. QALYs or DALYs) or money ('monetarised' health endpoints) .

5.3.1 Numbers

To characterise the health impact of air pollution or any other risk factor, one can simply calculate the number of cases of health damage associated with a certain exposure distribution, such as deaths, hospital admissions or number of asthma attacks. In traditional quantitative risk analysis, health risks are measured and, often implicitly, compared in terms of annual mortality risk: numbers per year. In several Western countries environmental regulation with respect to industrial safety (within or around the facility), radiation protection or chemical pollutants is based on a small 'accepted' annual mortality risk for each exposed individual. Often a individual risk criterion in the order of 10^{-5} or 10^{-6} is being used as a threshold of acceptability[19,20,21]. Such an approach at least guarantees everybody is treated in the same way, young and old, rich and poor, executive and unemployed; it prevents inequity as a result of 'unloading' health risk on smaller groups of individuals, often the low-cost solution[16].

5.3.2 Health adjusted life-years (e.g. DALYs)

However, it has become clear that one 'annual ten to the minus six' risk may differ substantially from another in several important aspects[22], such as loss of *life expectancy* and *non-lethal health outcomes*. For instance mortality during particulate air pollution episodes may at least in partly involve 'precipitation' of death among the old and weak, thus costing several months of unhealthy life at the most[23,24,25], while the impact associated with fatal accidents involving individuals with a 'random' age distribution may amount to a loss of many healthy years[26]. Interestingly, recent analysis of fatal traffic accidents in the Netherlands show that, apart from a well-known peak in traffic deaths around age 20, the age distribution is very similar to that of total mortality (primarily because old people are more frail and have lower survival probabilities than younger people)[27]. In addition, public health focus has gradually changed from life expectancy to health expectancy, i.e. postponing as long as possible or mitigating the functional limitations that come with chronic disease of older age and that affect the ability to cope with the demands of daily life[28,29]. More or less the same goes for the health impact of air pollution. In many cases these do not involve mortality, but rather aspects of the quality of life, such as

- aggravation of pre-existing disease symptoms, e.g. asthma, chronic bronchitis, cardiovascular or psychological disorders
- severe annoyance, sleep disturbance, as well as a reduced ability to concentrate, communicate or perform normal daily tasks
- feelings of insecurity or alienation, unfavourable health perception and stress in relation to poor quality of the local environment and perceived danger of large fatal accidents[30].

This implies that mortality risk might not always be the most appropriate indicator of environmental risk.

In an utilitarian approach to maximising cost-effectiveness for society, one would want to employ some sort of public health currency unit representing the full attributable health loss. Over the last few years much effort has been put into the development of metrics in which any type of morbidity or mortality is transformed into an equivalent number of life years (quantity plus quality of life). This type of health aggregating metric allows formal analysis of cost-effectiveness of environmental policy measures, which by now has more or less become common practice in medical technology assessment and public health research[3132,18].

5.3.3 Monetary value

Alongside the cost-effectiveness analysis, cost-benefit analysis is a form of evaluation in which the (health) benefits are also expressed in monetary terms. Efficiency calculations are made easier by putting cost and benefits under one heading, namely money. Furthermore, it is easier to include non-health aspects on the benefit side (equity, productivity, well-being) . In principle, investments in the health domain can be compared with investments outside, for instance transport safety, education, or ecological quality. Obviously this form of analysis requires the difficult task of expressing loss of life, life-years, or burden of disease in money. In such a 'hardcore' economic approach one seeks to attach a price tag to the incidence of different health end-points, e.g. by investigating willingness of people to pay (WTP) to prevent defined health endpoints, or the amount of money for which people are willing to accept (WTA) a certain level of health risk. In some studies primarily the costs of productivity loss and health care use are estimated[33,34,35,36,37]. Of course, the latter approach would not include the price of individual suffering.

In this chapter we investigate the feasibility of paradigms of risk management (mortality risk, numbers), attributable burden of disease (disability-adjusted life-years: DALYs) and monetary economic evaluation to support public health policy with respect to air pollution. We will focus on the health loss associated with the major air pollution phenomena in the Netherlands (indoor as well as outdoor). Our calculations are based on health impact estimates for the Dutch population, which were produced in the framework of the fifth National Environmental Outlook[38].

5.4 Healthy time as a metric

Accepting the fact that annual mortality or even loss of life expectancy do not fully represent the environmental health loss, we have applied an approach largely based on the 'burden of disease' measure developed by Murray and Lopez. To assess the global disease burden, and consequently the health policy priorities in different regions in the world, they used disability adjusted life years (DALYs). This health impact measure combines years of life lost and years lived with disability that are standardised by means of severity weights[28,39]. Our adaptation of the DALY-concept was inspired by the notion that the multiform health loss due to environmental exposure is fairly well characterised by three dominant aspects of public health, viz.

quantity of life (life expectancy)

quality of life, and

social magnitude (or number of people affected).

The diagram in figure 2-4 (chapter 2) sketches the basic idea behind our approach. At birth each of us may expect around a potential eighty years of healthy life. However, due to our genetic programming, our often-unfavourable life-styles, poverty, occupational or environmental conditions or just bad luck, most of us will encounter disease that will reduce the quality of part of our life-years. These diseases may manifest themselves in episodes, as chronic disease or even progressive disability until death. Some of us will die abruptly, for instance caused by an accident or an infectious disease. Thus, public health loss is defined as time spent with reduced quality of life, aggregated over the population involved. Based on this concept health loss attributable to air pollution can be assessed by:

defining responses that are associated with air pollution exposure

calculating the number of people affected (N),

estimating the average duration of the response (including loss of life expectancy as a consequence of premature mortality, D), and

attributing disability weights to each unfavourable health condition (Box 1)

Finally we estimate the number of disability adjusted life years that is lost per year of exposure, using the following equation:

$$DALY_{e.e.} = \sum_{i=1}^{n} \sum_{k} I_k * f_k(RR_i, C_i) * S_k * D_k$$

in which

$DALY_{e.e.}$ = health loss related to n environmental exposures, measured as disability or quality adjusted life-years per year of exposure,

$f_k(RR_i, C_i)$ = a set of functions (including exposure C_i and associated relative risk measures RR_i) representing the population attributable fraction (PAF) of condition k,

I_k = annual incidence of response variable (baseline risk) k, S_k = severity factor discounting time spent with the condition (see previous paragraph), D_k = duration of the condition; in case of premature mortality: loss of life expectancy.

To estimate the number of people affected we calculated PAFs by combining population exposure distributions with quantitative exposure-response information, applying the following equation (or one its derivatives):

$$PAF = \frac{\sum_{i>0} (RR_i - 1) * p_i}{\sum_{i \geq 0} RR_i * p_i}$$

in which RR_i = relative risk in exposure class i and p_i = exposure probability in class i.

Subsequently we estimated the number of people affected by a certain response by multiplying the PAF with annual incidence figures, obtained from Dutch health statistics, primary care registrations, or specific surveys.

5.5 Trading health for wealth or wealth for health

While DALYs measure health loss in units of time, applying the WTP (or WTA) approach is an effort to measure health loss in terms of money. Embedded in welfare economics it should be seen as the rate of substitution between health and wealth. Health is regarded as an economic good. An individual's preference for one health condition over a certain period of time can be represented by a change of income or wealth, or, in other words decreased possibilities to purchase other valued goods[17]. There are two broad ways to produce these estimates:

r*evealed preferences:* investigating how health risks that are related to certain risky occupations are allowed for in the differences in salary, or what extra amount people are prepared to pay for safer or healthier products (for example, cars with airbags, or houses in quieter surroundings);

stated preferences: using questionnaires to find out what people are prepared to pay for one extra life-year or one year free of disease or disabilities (contingency valuation).

Significant methodological objections are attached to both methods in connection with the transferability of implicit (behaviour) or explicit (survey) preferences of people from one circumstance to another, including a series of shortcomings that are characteristic of all questionnaire-based surveys[40,41,42]. In terms of the value of a statistical life, the outcomes of the above-mentioned methods are nevertheless reasonably consistent. On average Americans, Canadians or Europeans are willing to pay in the order of two million euros for a statistical life (1.5 to 7 million Euro), which is approximately 70,000 to 80,000 Euro a year at a discount rate of 3% (which for most of us is considerably more than our earning capacity, another commonly used proxy)[43,44].

5.6 Impact assessment

5.6.1 Health responses to exposures

Table 5-1, table 5-2, and table 5-3 present an overview of the population exposure-response functions for each air pollution type observed in a range of environmental and occupational epidemiological analyses. The associations for fine particulate matter (table 5-1) and ozone (table 5-2) are presented as increases in end-point incidence per 10 and 150 μg/m^3 increase of the pollutant concentration, respectively, either based on results of published Dutch studies or simple random effects meta-analysis[45,46].

In all tables an estimate is given of either incidence (including mortality) or prevalence of the health end-points in the Netherlands. These were based on available Dutch health statistics, summarised in the framework of the National Public Health Status and Forecast Report[29,47].

Table 5-1. Exposure response functions for health effects associated with particulate air pollution.

Health response *(particulate matter, PM_{10})*	**study design**	**Effect estimate (% per 10 μg/m^3)**	**Inc/Prev (p_0)**
Long term mortality	cohort[48,49,109]		
overall annual mortality		2.4 (0.1-6.1)[a]	0.0078
cardiopulmonary		4.3 (0.5-11.0)[a]	0.0032
lung cancer		2.0 (-4.5-8.1)	0.00054
chronic respiratory disease			
chron respir symp. children	cohort[50,51]	8.7 (0.7-16.6)	0.057
chronic bronchitis adults	cohort[52,53,54,55,56]	2.5 (1.2-3.8)	0.018
daily mortality	time series[57]		
total		0.36 (0.25-0.47)	0.000024
respiratory disease (COPD)		1.11 (0.64-1.61)	0.00000011
cardiovascular		0.25 (-0.09-0.42)	0.0000071
pneumonia		1.21 (0.65-1.80)	0.00000065
hospital admission	time series[58]		
respiratory		0.8 (0.5-1.1)	0.0000195
cardiovascular		0.7 (0.5-0.9)	0.0000384
emergency room visits	time series[77]		
respiratory		1.5 (0.5-2.6)	0.022
aggravation of asthma	panel studies[59,60,61,62]		
attacks		4.4 (-2.0-10.5)	0.00023
use of bronchodilators		7,.0 (0.64-12.9)	0.00059
respiratory symptoms	panel studies[77]		
upper respiratory tract		2.0 (-0.13-4.1)	0.19
lower respiratory tract		3.8 (0.3-7.1)	0.038

a. calculated for the fraction highly exposed living within 50, respectively 100 meters distance from a major inner-city road or a freeway, roughly comparable with a concentration difference of 10 mg/m^3 for black smoke, 30 mg/m^3.for NO_2 and 40 mg/m^3 for PM_{10}^{3} [109]

Table 5-2. Exposure response functions for health effects associated with ozone air pollution.

Health response *photochemical (ozone)*	**Study design**	**Effect estimate (% per 150 μg/m^3, 8h average)**	**Inc/Prev (p_0)**
daily mortality	time series[57]		
total		4.1 (2.0-5.9)	0.000024
respiratory disease		12.7 (4.8-21.1)	0.00000011
cardiovascular		3.2 (0.3- 6.1)	0.0000071
pneumonia		18.8 (9.0-29.9)	0.00000065
hospital admission	time series[58]		
respiratory		6.5 (3.9- 9.1)	0.0000195
emergency room visits	time series[45]		
respiratory		9.1 (2.9-13.8)	0.035

Population-weighted exposure distributions for ozone and particles were derived through linear interpolation of data from stationary air pollution monitors of the national network. As urban concentrations of particles tend to be higher adjustments were made based on the virtual diameter of the urban agglomerations (> 40.000 inhabitants), employing an empirically determined factor. Subsequently, the spatial distribution of air pollutant concentrations was combined with data on population density by means of Geographic Information Systems (table 5-4)[63,64].

Table 5-3. Exposure response functions for health effects associated with selected air pollutants.

Air pollutant	Health end-point	Effect estimate	P_o
damp houses[65]	asthma symptoms children	RR_{expo}: 1.41 (1.23-1.71)	0.135
	asthma prevalence adults	RR_{expo}: 1.56 (1.25-1.95)	0.012
ETS[66]	Lung cancer ns[a] men	RR_{expo}: 1.34 (1,00-1.84)	0.000016
	Lung cancer ns women	RR_{expo}: 1.24 (1.13-1.36)	0.00000074
	IHD[b] non-smoking men	RR_{expo}: 1.23 (1.03-1.47)	0.0006
	IHD non-smoking women	RR_{expo}: 1.19 (0.97-1.45)	0.0005
	aggravation of asthma	RR_{expo}: 1.63 (1.30-1.96)	0.004
	lower respir. symptoms	RR_{expo}: 1.46 (1.33-1.60)	0.037
	otitis media	RR_{expo}: 1.29 (1.05-1.35)	0.00012
	sudden infant death	RR_{expo}: 1.94 (1.55-2.43)	0.0000037
Radon[67,119]	lung cancer	Add. Risk 0.0004 (0.0002)/WML[c]	

[a] non-smoking
[b] ischaemic heart diseaese
[c] working month level (exposure measure)

We recognise the fact that this type of exposure characterisation only poorly reflects personal exposure, which is a function of time-activity patterns and micro-environmental concentrations. On the other hand epidemiological studies incorporate the same kind of relatively poorly defined ambient air pollution data. Furthermore, at least with respect to particles, a fairly consistent correlation has been shown between ambient and personal exposure indicators[68].

The prevalence of exposure to spousal environmental tobacco smoke was based on cross-sectional monitoring data[69], assuming smokers had a risk of living with a smoker three times as high as a non-smoker. Prevalence of damp homes was based on an investigation by the Dutch ministry of housing[30]. Based on an extensive monitoring program in more than 1,500 Dutch dwelling, (future) exposure to indoor radon of the Dutch population was modelled based on characteristics of the total stock of dwellings, such as air-tightness, building materials and ventilation behaviour[30,70].

5.6.2 Long-term versus short term mortality

For the sake of clarity (and simplicity) we make a distinction between exogenous risk factors that are involved in the onset and progression of (chronic) disease and risk factors that primarily 'accelerate' death among the weak. The first type of interaction can be shown in cohort studies in which populations are followed up during a sufficiently long period to see whether certain exposures affect the incidence or mortality of disease, allowing for gender, age, smoking, occupational status and diet. Examples of these are studies among workers exposed to hazardous substances in an occupational setting (radon, benzene, polycyclic aromatic hydrocarbons) and a number of American cohort studies retrospectively relating survival among citizens to air pollution levels[71,72,73,74]. Results of both the American Cancer Society Cohort

(ACS) and the Harvard Six Cities studies stood up to extensive scrutiny by the Health Effects Institute[75]. An extension of the follow-up of the ACS study yielded results that were consistent with earlier reports[76]. Therefore it is reasonable to assume that long term exposure to particulate air pollution indeed affects survival, especially by increasing the risk of cardiopulmonary disease and lung cancer. 'Harvesting', the bringing forward of death among the susceptible, for instance through aggravation of disease during unfavourable air pollution conditions, would go unnoticed in this type of study, given the extended 'time-window' of many years. This effect is especially seen in time series analysis, observing day-to-day variations (see figure 4-1)[24,77]. Of course just the accumulation of these day-to day health insults underlying changes in mortality and hospital admission may very well be a causal factor of chronic morbidity and survival loss[78].

Table 5-4. Exposure distributions for air pollution types.

Air pollutant	Metric	Mean (stDev)
PM10[63]	ann.av. 24-h ($\mu g/m^3$) ann.av. 8-hour ($\mu g/m^3$)	34.4 (3.6)
Ozone (8-hours $\mu g/m^3$)[63]		45.6 (6.8)
Environmental Tobacco Smoke	Prevalence	
non-smoking men 1970	smoking partner	0.26
non-smoking women 1970	smoking partner	0.60
non-smoking men 1990	smoking partner	0.22
non-smoking women 1990	smoking partner	0.30
children 1995	smoking parent	0.40
Damp homes[79]	prevalence	0.175 (0.07)
Radon[80]	average indoor concentration (Bq m^{-3})	23-24 Bq m^{-3}

To estimate loss of life-years we applied the concept of attributable risk, assuming that cause specific mortality due to environmental exposure is similar to any 'other' cause specific mortality, with respect to onset and time of dying. For each end-point we estimated the years of life lost by means of life-table analysis (the sum of number of cases per age-group times remaining life expectancy divided by total number of cases), similar to the way in which several authors have calculated the impact on *national* average life-expectancy[24,81,82]. We also investigated the sensitivity of the result to changing relative risks with age or implementing a certain length follow-up (table 5-5).

Several sophisticated epidemiological analyses have been undertaken to estimate the loss of life-years due to the short term mortality that is revealed by the analysis of time series of daily mortality and air pollution data. Most analyses suggest the average loss of lifetime would range from a few days to several months. In some cases the loss may extend to one year or more, for instance due to pneumonia or a heart attack[83,84,85,86]. Furthermore, several analyses have shown most of the air pollution associated deaths occur outside the hospital, implying that these effects are not limited to the terminally ill[87]. Here we will *arbitrarily* apply a rather non-informative subjective distribution ranging from one week to one years, with an average of three months (table 5-5).

Table 5-5. Estimated duration of effects.

Health response (particulate matter, PM_{10})	distribution applied in Monte Carlo analysis	parameters (years)
Long term mortality	**(Dutch life table)**	
overall annual mortality	no	10.9
cardiopulmonary	no	8.2
lung cancer	no	13.0
ischemic heart disease	**uniform**[a]	
	minimum	1
	maximum	11
sudden infant death	no	70
daily mortality	**subjective Beta**[b]	
	minimum	0.02
	most likely	0.25
	mean	0.5
	maximum	10
hospital admission	**subjective Beta**[c]	
respiratory	minimum	0.011
cardio-vascular	most likely	0.019
	mean	0.038
	maximum	0.167
emergency room visits	**normal**	
respiratory	mean	0.033
	stand. dev.	0.021
aggravation of asthma	**uniform**	
attacks	min	0.0027
use of bronchodilators	max	0.0055
respiratory symptoms	**point estimate**	
upper respiratory tract		0.019
lower respiratory tract		0.038
lung cancer morbidity	**point estimate**[d]	2.9

[a] A uniform distribution: minimum loss of life expectancy 1 year, maximum 11.3, being the loss based on life-table calculation (passive smoking as dominant cause of morbidity and death).

[b] A subjective BETA distribution allowing to incorporate the following quantitative assumptions: minimum loss of life expectancy 1 week, most likely 6 months, maximum 10 years, being the loss based on life-table calculations (e.g. air pollution impedes recovery from pneumonia).

[c] A subjective PERT distribution allowing to incorporate the following expert assumptions: minimum duration of disease aggravation episode 4 days, most likely two weeks, maximum two months.

[d] Based on Dutch data on incidence and prevalence.

As in the case of long-term mortality here we apply the concept of population attributable risk. The fraction attributable to air pollution exposure is assumed to be similar to the total morbidity load. To assess the time spend with a certain morbidity we use year prevalence data (asthma, ischaemic heart disease). In a stable situation by definition prevalence equals the number of new cases times the average duration of the condition[88].

Estimates for the duration of health care events were either derived from the literature, the Dutch health care registration (NIVEL) or from expert consultation[89,90,91,92,93]. An overview of duration estimates is presented in table 5-5.

Box 1. Severity weights for disease states

In the framework of the National Public health Status and Forecast Reports an estimate was produced of the burden of disease within the Dutch population[29]. To define 'Dutch' severity weights Stouthard et al selected 55 diagnoses of greatest public health significance in terms of number of patients, and years of (healthy) life lost. These diagnoses were divided in 176 health states of various severity (or disease stage)[94].
According to the protocol designed by Murray et al[95], physicians with ample clinical experience were invited to perform the weighting procedures, which consisted of two steps. At first they evaluated a selection of 16 representative indicator states, using two varieties of a person-trade-off approach. This first step of the valuation process was performed during workshops, as deliberation is an explicit part of the protocol. A visual analogue scale (VAS) was added as another instrument of valuation, mainly for the purpose of validation. Furthermore, a standardised classification of the indicator states according to EuroQol-5D+ was provided to assist panel members[96]. This classification instrument involves a three-point scale for six health dimensions, viz. mobility, self-care, daily activities, pain/discomfort, anxiety/depression and cognitive functions. Using the indicator states for 'calibration' the remaining health states were valued individually by means of interpolation (ranking health states similar to one or in between two consecutive indicator conditions, figure 5-4).
For air pollution exposure-related chronic disease morbidity for which different health states have been defined, such as asthma or ischaemic heart disease, a severity factor was composed as a prevalence-weighted average, assuming that the environmental exposure had no effect on disease prognoses[88] (e.g. mild, moderate and severe asthma). In some cases weights referred to transition from one severity state to another, for instance from mild to severe asthma ('aggravation of asthma').

5.6.3 The health market

To assess the loss of economic utility associated with the health end-points involved, one would require an economic valuation of the full quality of life impact to the affected individual. This would include expenses such as medical costs and lost income (often referred to as cost of illness, COI), and less tangible effects on well being such as pain, discomfort and restriction of everyday activities. One way of assessing the economic utility loss is to ascertain the individual maximum WTP for the reduced incidence of illness and adverse symptoms. To compile a set of WTP-values for the health impacts quantified here we drew upon a number of studies that reviewed the literature on WTP for avoiding changes in risk of death, chronic disease as well as milder morbidity effects[33,34,35,37,97,98]. These values were primarily derived from studies in which preferences of both healthy and infirm individuals were revealed through questionnaires, the so-called contingent valuation studies. In some cases, in the absence of reliable data, COI-estimates were used as a proxy, adjusted upwards by a factor of 2. Table 5-6 lists the WTP values derived from these reviews.

Table 5-6. Monetary values for health endpoints based on 'willingness to pay' (WTP).

Health response	distribution	Euros
Long term mortality	**discrete**	
overall annual mortality	lower (0.33)	1,950,000
cardiopulmonary	middle (0.5)	4,360,000
lung cancer	upper (0.17)	6,760,000
ischemic heart disease		
sudden infant death		
chronic bronchitis	**discrete**	
	lower (0.33)	150,000 [a]
	middle (0.33)	220,000
	upper (0.33)	390,000
asthma	**discrete**	
	lower (0.33)	65,000 [b]
	middle (0.33)	105,000
	upper (0.33)	130,000

Health response	distribution	Euros
chronic respiratory symptoms (1 year)	**discrete**	
	lower (0.33)	165 [c]
	middle (0.33)	330
	upper (0.33)	495
daily mortality	**subjective PERT**	
	minimum	1,200 [d]
	most likely	38,000
	mean	109,000
	maximum	4,360,000
hospital admission	**discrete**	
respiratory	lower (0.33)	1,100 [e]
	middle (0.33)	7,000
	upper (0.33)	14,000
cardiovascular	lower (0.33)	1,100 [e]
	middle (0.33)	7,000
	upper (0.33)	15,000
emergency room visits	**discrete**	
respiratory	lower (0.33)	260 [f]
	middle (0.33)	520
	upper (0.33)	780
aggravation of asthma	**discrete**	
attacks	lower (0.33)	35 [g]
use of bronchodilators	middle (0.33)	70
	upper (0.33)	140
respiratory symptoms	**discrete**	
upper respiratory tract	lower (0.33)	6 [h]
	middle (0.33)	12
	upper (0.33)	18
lower respiratory tract	**discrete**	
	lower (0.33)	6 [i]
	middle (0.33)	38
	upper (0.33)	330

a based on contingent valuation study in US[34,99]
b based on contingent valuation study in US[34,37]
c based on adjusted cost of illness (COI) for cases of acute bronchitis in US[34]
d see text[25]
e lower estimate based on empirical relationship between WTP and quality of life in accordance with ref. 25; central estimate based on contingent valuation study in US[37]; upper estimate adjusted COI in the US[34]
f based on WTP-study in US[34,37]
g lower estimate WTP from adjusted COI[34]; central estimate WTP non-asthmatic respondents, upper estimate WTP asthmatic respondents in Norwegian CVM-study
h based on CVM-estimates in US[34] and a Norwegian study[100]
i lower estimate based on CVM-study in US[34]; central estimate based on Norwegian study[100]; upper estimate WTP for acute bronchitis in US[34].

From a review of these studies we suggest a range for the value of a (statistical) death of 2-7 million Euro, with a central estimate of around 4,500,000 Euro, independent of age or disability. The contingency valuation method produces the highest estimates, while the consumer market studies yield the lowest values[37].

The economic valuation of daily mortality presents a problem as most reviews suggest that recorded deaths may often involve old people suffering from severe chronic disease. As compared to fatalities among young or middle aged, healthy individuals, a number of adjustments appear to be justified. These would concern the loss of life-years, the average age distribution, and the quality of life of the cases involved. Much in accordance with the report of the Ad-Hoc Group on the Economic Appraisal of the Health Effects of Air Pollution in the UK we propose a lower limit of 1,200 Euro for day-tot-day mortality. Here a maximum reduction factor is applied adjusting for loss of life-years, age and disability (0.083: assuming a minimum individual loss of life expectancy of one week instead of 12 years; 0.75 to adjust for a lower valuation of a prevented statistical fatality at older ages[37,101], and 0,2 as the lower bound of the disability weight for the very ill)[25]. The upper bound estimate of the value of a prevented statistical fatality would equal the unadjusted (central) estimate, given the unlikely possibility daily mortality would involve a healthy subject with an average life expectancy. For the 'most likely' monetary estimates the loss of life-time per case is assumed to be 6 months respectively; a factor of 0.6 is applied to adjust for age and disability[44,83].

As immediate benefits in general are more valuable to people than benefits some time in the future, many valuation systems used in cost benefit analysis apply discount rates, not just for costs but for health benefits as well (e.g. 3% annually). As we did not make a formal cost-effectiveness analysis, here we have not applied discount rates. In our exercise discount rates would apply to chronic effects, such as premature mortality or an extra case of chronic bronchitis. A discount rate of 3% over 10 years would reduce the valuation in DALYs or Euros by around 13%. This would fall well within the range of uncertainty we deal with in this analysis.

5.6.4 Uncertainty

We analysed the uncertainty in the calculations of air pollution related DALYs and Euros by means of Monte Carlo techniques[102,103,104]. In a Monte Carlo simulation model input parameters are treated as random variables. For each of the input parameters, such as population exposure, exposure-response function estimates, average duration of the response and discount factors, a probability distributions function was estimated, representing parameter uncertainty. Subsequently, an output distribution for the different health impact measures was estimated by iterative (Latin hypercube) sampling from each of the defined parameter distributions, followed by recalculation. Here we will present the 5- and 95-percentiles of this distribution as a measure of uncertainty.

5.7 Deaths, DALYs and dollars (Euros)

Point estimates as well as the 5 and 95-percentiles of the probability distribution for impact metrics are shown as a measure of uncertainty (table 5-7, figure 5-1). In most cases this uncertainty range is substantial, but less than one order of magnitude[102,103,104]. Strikingly, the rank order of the exposures with respect to attributable burden differs from one impact indicator to another, with the exception of long term exposure to particulate matter which is always on top of the list, outpacing the other factors.

In our calculations the health loss related to air pollution is substantially dominated by the long-term effects of particulate air pollution, which amounts to 40% of deaths, 65% of total DALY-loss and 75% of the total monetary loss related to air pollution (Table 5-7). The estimated 'monetarised' loss is huge, almost 16 billion Euro (5-95%-tiles: 7-25 billion), which is between one quarter and one third of the total annual budget for Health Care in the Netherlands, but it must be stressed that this is only an indicative figure as the WTP-amount of money is not 'real money'.

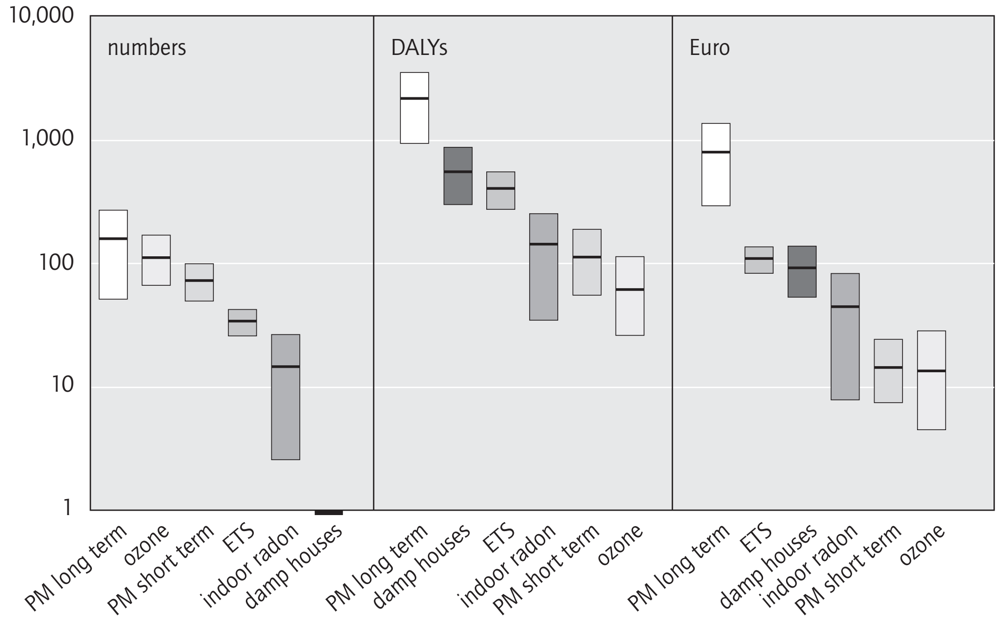

Figure 5-1. Annual health loss for a number of air pollution phenomenon expressed in terms of fatality number, DALYs and million Euro per million inhabitants (average and 5-95%-tiles of uncertainty distribution).

Table 5-7. Summary of health impact estimates for a number of air pollution phenomenon, expressed in fatalities, DALYs and Euros.

Environmental factor	Health outcome	#/million[a]	5-/95% -tile	DALYs/ million	5-/95% -tile	Euros (x106)/ million	5-/95% -tile
particulate	*mortality*						
air pollution	- cardiopulmonary	135	60-220	780	256-1480	585	230-1060
long-term	- lung cancer	30	-60-120	230	-450-975	115	-225-450
	morbidity						
	- chronic resp. sympt.	3,300	825-6,200	135	28-275	1.1	0.3-2.2
	- chronic bronchitis	400	90-710	1050	225-2030	100	20-190
total long term		*160[b]*	*52-275*	*2,200*	*950-3,560*	*800*	*295-1370*
particulate	*mortality*						
air pollution	- total	73	50-100	25	5-64	8	1-21
short-term	- respiratory	10	6-14	3.5	0.6-9.0	1.1	0.1-3.0
	- coronary heart dis	17	7-27	6	1-15	2.0	0.2-5.1
	- pneumonia	7	4-10	2.5	0.5-6.0	0.7	0.1-2.0
	Hospital admission						
	- respiratory	85	440-144	1.9	0.7-3.7	0.6	0.2-1.3
	- cardiovascular	86	42-150	2.1	0.8-4.1	0.7	0.2-1.4
	emergency room visits						
	- respiratory	1530	550-2900	24	4-61	0.8	0.3-1.6
	aggravation of asthma						
	- asthmatic attacks	4,7000	6,500-105,000	26	2-73	4	0.5-9.5
	- use of bronchodil.	51,000	6,800-116,000	28	2-82	5	0.5-10
	aggravation of resp. symptoms						
	- upper resp. tract	9,860	1250-19,100	8.2	0-24	0.1	0-0.2
	- lower resp. tract	3,930	870-7200	25	2-72	0.5	0.1-1.3
total short term		*73[b]*	*50-100*	*113*	*55-190*	*15*	*7.5-25*

Environmental factor	Health outcome	#/million[a]	5-/95% -tile	DALYs/ million	5-/95% -tile	Euros (x106)/ million	5-/95% -tile
ozone	*mortality*						
	- *total*	112	76-170	39	6-102	12	1.5-35
	- respiratory	16	8-28	6	1-15	1.8	0.2-5.0
	- cardiovascular	34	7-90	12	1-40	3.7	0.2-13.0
	- pneumonia	9	5-13	3.1	0.5-8.0	1.0	0.1-2.6
	hospital admission						
	- respiratory disease	72	46-98	2.0	1.0-3.5	0.5	0.2-1.1
	emergency room visits						
	- respiratory disease	1,290	520-2,300	21	3.4-62	0.8	0.3-1.6
total ozone		*112[b]*	*67-172*	*61*	*26-114*	*13.5*	*4.6-29*
damp houses	*Asthma*						
	- children	1665,	898-2622	133	58-235	41	21-68
	- adults	1814	680 –3430	145	47-302	45	16-87
	lover resp. disease						
	- children	661	300-1165	27	5.6-61	0.5	0.2-1.1
	- adults	6092	3140-9730	250	53-520	5	1.9-9.2
total dampness		*0*	*0*	*554*	*300-880*	*92*	*53-142*
ETS	*lung cancer* -morbid.	1,9	1,3-2,4	0.8	0.5-1.2		
	(female) -mortality	0.9	0.6-1.2	12.2	8.5-15.7	2.8	1.9-3.7
	lung cancer -morbid.	2.7	0.7-4.7	1.2	5- 45		
	(male) -mortality	1.3	0.3-2.3	17	4.5-30.5	3.9	1.0-7.1
	IHD						
	- morbidity females	118	75-162	34	17-55		
	- mortality females	16.5	10.4-23	90	22-175	50	32-71
	- morbidity males	116	73-162	34	17-54		
	- mortality males	15	9.5-21	83	21-161	47	29-66
	aggravation of asthma	91780	42720-1538200	51	9-118	3.4	1.3-6.3
	lower respir. sympt.	1900	1600-2150	12	1.5-28	0.2	0.0-0.5
	otitis media acuta	2700	1250-4100	30	1.1-76	0.3	0.1-0.8
	sudden infant death	0,6	0,1-1,1	43	5-76	1.9	0.2-3.4
total ETS		*35[b]*	*26-43*	*410*	*280-553*	*110*	*84-139*
radon	*lung cancer morbid*	*15*	*3-27*	*13*	*2.2-25*	*46*	*8-85*
	lung cancer mort	*15*	*3-27*	*132*	*23-241*		
total radon		*15*	*3-27*	*145*	*35-256*	*46*	*8-85*
overall		*395[b]*	*273-525*	*3,485*	*2,175-4,905*	*1,075*	*560-1,650*

[a] number of people affected
[b] number of deaths only

Over the past decade annual investments in traffic emission reduction in the Netherlands added up to somewhere between 250 and 350 million Euro, yielding an emission reduction of around 4% per year[105,106]. At the same time traffic volume has grown with over 3 % per year. If we assume there is a linear relation between emission and actual population exposure, the benefits of these investments in terms of monetarised health gain are in the same order of magnitude, even somewhat higher (7-8% of an annual disease burden of 13 billion Euro: 550 million Euro). Based on the same assumption the cost of air pollution control measures per DALY saved is around 50,000 Euro. In recent discussions among health economists an investment of 40,000 to 50,000 Euro for each equivalent year of perfect health gained by an intervention is regarded as 'acceptable value for money'[107]. The World Bank recently proposed 3 times the Gross Domestic Product per capita; for the Netherlands that would be around 75,000 Euro/DALY

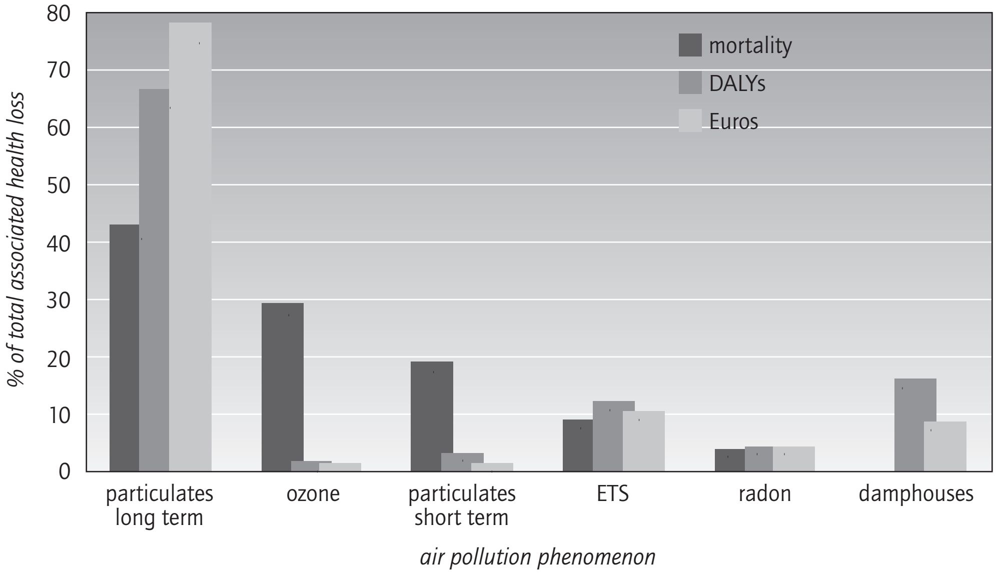

Figure 5-2. Overview of health impact of air pollution type as percentages of total air health loss expressed in terms of fatality numbers, DALYs and Euro.

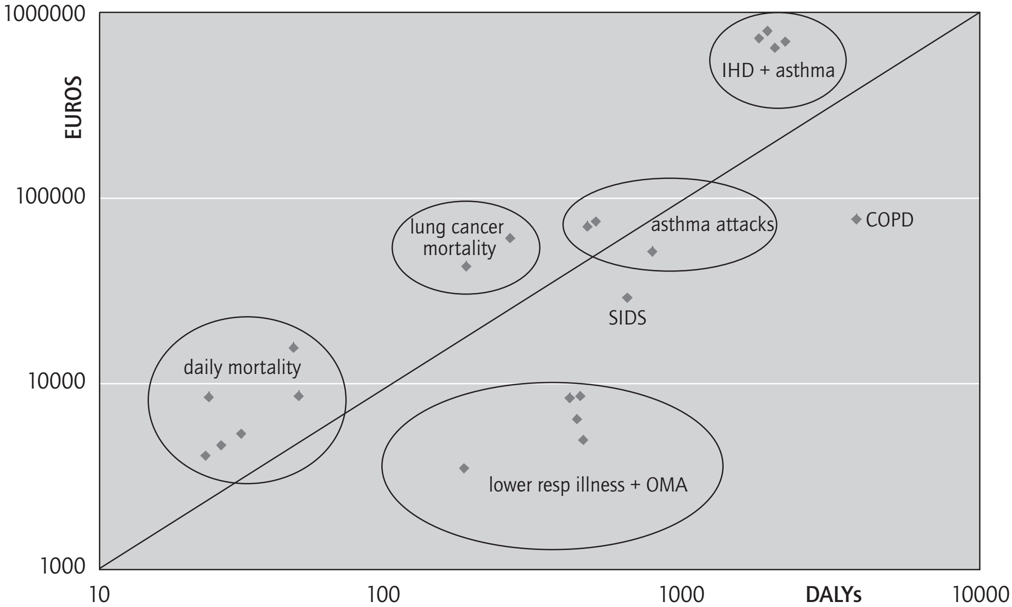

Figure 5-3. Response specific DALYs plotted against response specific euro.

saved. We emphasise that besides health gains the investments in emission reduction have yielded several other benefits, such as a substantial decrease in ecological impacts, improved and more energy-efficient engine technology. In sum, according to these calculations it was money well spent.

The total air pollution attributable disease burden is estimated to be in the order of 50,000 (5-95%-tiles: 25,000-75,000) DALYs annually. As expected the health loss attributable to environmental exposures is small but substantial in the Netherlands. Recently the total annual burden of disease was estimated to be in the order of 3 million DALYs[47]. According to the calculations presented here, the air pollution impact described here would contribute slightly less than 2 percent to this total disease burden. In the Netherlands life-style factors, such as smoking, physical inactivity cause a burden of disease of 14.7, and 4.5% of total disease burden (DALYs) respectively[14]. High cholesterol is an example of a single factor for which the attributable disease burden is in the same order of magnitude, 2.5%.

The metric used to express air pollution associated health loss, mortality risk, DALYs or Euros has an obvious effect on rank order of the different types of air pollution effect (figure 5-2). The mortality counts associated with air pollution episodes are not reflected in the amounts of lost DALYs and Euro, as a part of it is considered to be due to 'harvesting' of those with only a limited time to live. No mortality is calculated for damp houses, but chronic bronchitis associated with damp housing weighs heavily in terms of DALYs and Euro.

In figure 5-3 for each air pollution exposure the disease burden in DALYs is plotted against monetarised health loss on a log-scale. On average the DALY is worth around 300,000 Euro, but the plot is clearly very scattered.

5.8 Discussion

5.8.1 Health impact assessment

Several potential flaws are associated with this type of impact assessment. Most of these are not specific to the approaches and metrics we have proposed here, but concern health impact assessment in general. Recurring, almost inevitable shortcomings involve the imprecision of population exposure assessments, the unknown, and probably 'unknowable' shape of the exposure-response curves at low, environmental exposure levels, and the translation of exposure-response information from one species to another, as well as from one population to another.

Another important issue is the internal and external validity of epidemiological results. In view of the way in which our aggregated results are dominated by the long-term public health effects of particulate air pollution, we have to stress the fact that these estimates are based on the results of only two American cohort studies and a small Dutch pilot study[71,72,108,109]. Given the inherent shortcomings of this type of retrospective epidemiological study, especially with respect to the validity of the exposure indicators, the relatively large health impact calculated here still needs to be confirmed in other well-designed studies, such as the Dutch study in which exposure assessment is at least performed on an individual level. Even though most of Hill's criteria for causality are met, substantial uncertainties remain unsolved with respect to specific causative agents and mechanism of actions[18,19,110,111].

Most of the quantitative health impact assessments presented here are fairly sensitive to the choice of reference level of exposure. Suspended particles, ozone and other oxidants in outdoor air, radon or allergens in indoor air are a fact of life, even without any human activity. Here we have more or less applied the concept of feasible minimum exposure distribution. If we optimise the way we build our houses as well as our ventilation behaviour radon levels should not exceed 16 Bq/m^3 and damp houses would be very rare[112]. In principle a world without smokers is possible, so exposure may be zero. For particulates (PM_{10}) and ozone we tentatively applied annual concentrations of 10 and 15 mg/m^3 as baseline exposure levels.

The most fundamental problem we encountered was lack of complete and quantitative insights into how air pollutant exposures are involved in the onset and development of human disease. We have only incorporated exposure-response associations which have been studied comprehensively in epidemiological research and for which clearly defined health outcomes have been established. 'Health events' that are recorded in epidemiological surveys or health care registration may only be the tip of the iceberg, with many minor impairments remaining beneath the surface undetected (see figure 4-2)[30]. In particular, time-series analyses provide us with the means to detect relatively small elevations of mortality and hospital admission during episodes. It is almost inconceivable these relatively severe health events would occur without a greater part of the population suffering from transient lung function deficits, or asthma or angina attacks, resulting in an increase use of medication, visits to the general practitioner or even an emergency room among the more susceptible. However, obvious means of investigation, such as panel studies appear to lack sufficient power to detect an increased incidence of such endpoints[6,113].

5.8.2 Environmental health impacts on a scale

Attributing impact weights to air pollution related conditions involves societal and individual values and preferences which does not always agree well with the scientific traditions in several disciplines. At the same time, normative evaluation of health-endpoints is virtually inevitable in health risk assessment (chapter 2). Even using annual mortality risk as an impact measure is value-laden, as non-fatal health outcomes as well as age at death (loss of life expectancy) are implicitly ignored. The same applies for health-based exposure guidelines, for which value judgements have to be made regarding the health significance of toxicological or epidemiological response variables.

Whose values matter and how are these best assessed? A large number of studies have revealed that the health related quality weights attributed to certain health states may differ between patients and non-patients[14]. Health status weights for DALYs are mostly derived by calling in health professionals, who are able to assess the various dimensions of disease. WTP-values are determined with the help of large-scale surveys. Both have their own systematic biases, but policy decisions about resource allocation should adopt a societal perspective and thus may involve these types of generic preference classifications. Furthermore, it is important to notice that health preference measurements tend to be rather stable and reproducible, even across countries[38,41,115].

In health economics there are five main ways to elucidate preferences with respect to health status quality: standard gamble (choosing between probabilities of possible outcome), time trade-off (exchanging time with different health states), person trade-off (exchanging persons with different health states), contingent valuation and revealed preference (wage-risk, hedonic pricing etc.). Results produced by different methods may differ substantially indicating that many socio-psychological processes are not accounted for. An appealing example is that most people find it impossible to exchange relatively minor morbidity outcomes among many with death among few (e.g. reluctance to give up life). Several authors have claimed that real life preferences with respect to health and life expectancy are much too complicated to be covered by the simple concept of DALY. They argue that the plain observation that preferences are not stable over a life-time and will depend largely on whether health states will occur near or far into the future makes application of DALYs questionable. Others favour a more practical approach: there are simply no better alternatives[116].

Uncertainty analysis shows that altering weights within the variance seen in most weighting exercises does not substantially affect the overall picture. Compared with the huge uncertainties that are often connected with health impact estimates, the effect of the possible variance in attributed weights appears rather small. There is an important exception, however. The lower the disability weights attributed to health states, the more sensitive they are to variation. It is much easier to double the small weight given to severe noise annoyance than to double that of the terminal state of lung cancer. This is reflected in the results of the uncertainty analysis with respect to 'respiratory illness'. The variation of the output probability distribution is largely due to variation in the severity weights given by panel members. Since the less severe responses tend to affect the highest number of people, there is some room for 'manipulation' of results, e.g. increasing the public health significance of 'one's favourite health risk'.

5.8.3 Worth the money?

The calculations with respect to the monetary valuation of air pollution health impacts indicate an enormous loss in economic terms. However, the loss is primarily due to the long-term effects on survival associated with particle exposure. Regarding this association there is still substantial uncertainty on biological mechanisms and causal agents. This dilemma of uncertain health impacts yielding a huge health loss stresses the necessity to complete the economic appraisal with accepted methods for economic valuation of uncertainty[37,25].

As already mentioned, transfer of WTP-values from other studies can be tricky, since valuations may be largely determined by contextual variables, comprising risk perception or the respondent's perspective (individual versus collective, altruistic versus self-centred). An economic appraisal of air pollution health effects may profit from WTP-estimates based on European contingent valuation studies that directly concern air pollution health effects in the right context.

Air pollution attributable health loss expressed in DALYs is not completely proportional to the health loss in terms of Euros (Figure 7), although there is some convergence. Theoretically this might be explained by the fact that both measures are based on different principles. Monetary valuation may involve all possible dimensions of reduced health, trading off health and other valued services money can buy, while DALYs are an aggregate of only three formal dimensions, number of people involved, duration and severity. It is unlikely that the same dimensions are valued for all diseases. For instance, the mortality-aspect of DALYs (YLL) depends on life expectancy alone, while the non-linear relation between WTP-values and age at death suggests that individuals value other dimensions too. Health responses to exposures that are dominated by mortality, such as the impacts of ozone and particles (long-term) produce relatively higher health loss estimates for Euros than for DALYs and vice versa (short-term responses and morbidity due to particles). Of course one has to bear in mind that many shortcomings are connected to an economic appraisal of the prevention of statistical fatalities as well as morbidity risks. For instance the wage-risk method relies on the assumption of enough labour mobility for workers to really have a free choice, which is doubtful in most cases[37]. With respect to the contingent valuation method to unveil general WTP figures a number of objections have been raised. In the first place it is a highly hypothetical number, i.e. it is not real money simply what people claim they would spend in this 'virtual' market place. Respondents may also show 'strategic' behaviour. People with 'green' preferences may intentionally exaggerate their willingness to pay; others may provide interviewers with politically correct answers. Respondents may feel it is unethical to put a price tag on someone else's death or illness. Furthermore, given the often-complex nature of the questionnaires contingent valuation methods are vulnerable to many potential biases ('starting point bias', 'interviewer bias', and 'embedding bias'). Some health end-points may be rather hypothetical to most of the respondents, but at the same time familiar to others, e.g. asthmatic attacks. Familiarity with end-points such as asthmatic attacks, and chronic bronchitis appears to increase WTP[44,117].

Thus, there is reason for concern about the stability of the quantitative preferences of respondents into which we have only limited insight. For instance, many risk perception studies underline the importance of context variables, such as personal interest in risk source, perceived personal and institutional control over risk generating activities, degree of voluntariness of exposure, inequity with respect to distribution of risks and benefits[118]. Other socio-economic and demographic variables may also be important, such as base-line risk, or personal income and education. This type of uncertainty may not fully justify the transfer of WTP-measurements to health impact assessment of air pollution. In particular, most of the WTP-studies involving morbidity risks have been done in the US and there are systematic differences between the US and Europe in the cultural and socio-economic dimensions discussed here. Therefore the results of WTP-calculations should be regarded as crude estimates, allowing comparisons between different risks rather than being real money.

In the framework of this study we have not fully considered the feasibility or costs of measures to reduce air pollutant exposure. It is clear that current ambient air concentration of particles and ozone can only be reduced at high economic and societal costs, while a certain level must be considered as a fact of life. For environmental tobacco smoke and dampness in homes measures to reduce population exposure may be less expensive, for a comparable benefit although data on these indoor factors in this context are lacking. Measures to reduce radon levels in newly constructed houses, such as performance standards for building materials and facilities to increase the ventilation rate (including energy costs) would cost somewhere between 50,000 and 150,000 Euro/DALY (see chapter 2)[119].

5.8.4 Equity and efficiency

It is important to note here that the choice of an indicator to represent environmental health loss is not just an academic question. It is not just coincidence or lack of methodology that lies in the choice of mortality risk as the classical health loss indicator. Managing risk based on mortality guarantees everybody is treated equally, while simple application of health adjusted life-years as a measure of health impact is not without serious distribution or ethical consequences. In principle, society would benefit from passing health risks to the elderly since they have less life-years and health to lose. Furthermore, the use of DALYs implies that people with a disease count less than healthy people. In this respect some authors warn against double jeopardy. People with poor health suffer a disadvantage twice: first they are disabled, and secondly the saving of one year of their lives counts less than that of an healthy person. One can strive to maximise utility, but one can also strive to concentrate efforts on people with the worst health, or the greatest improvement potential[120].

WTP-values are also dependent on income. Application of WTP may thus violate equity principles, especially when locally derived WTP values are applied globally, typically giving less weight to third world health problems[18].

5.8.5 Conclusion

In spite of methodological and ethical problems, presenting health impact in terms of DALYs and/or money offers a useful framework for *explicit evaluation* and *comparison* of health loss associated with different environmental exposures, involving a wide variety of non-fatal health outcomes. It enables incorporation of the public health interest in decision making with respect to environmental quality and spatial planning (e.g. extensive infrastructure projects involving a range of diverging, often accumulating exposures). For instance, in scenario studies aggregates can be applied to explore the 'health' score of different options, evaluated from the perspective of different policy philosophies.

The calculation of DALYs associated with environmental exposures provides a comparative picture from the viewpoint of public health. However, we would not suggest an immediate change of environmental policy priorities based on this type of calculation. Policy priorities may be partly explained by dimensions of health risk perception, which are not captured in our approach, such as 'dread', voluntariness of exposure, the perceived controllability or familiarity of risk generating processes (e.g. traffic). In particular, the social distribution of risk and benefit[118] is not covered in our approach, as by definition a slight reduction of health amongst many may be equal to severely affected health amongst few. From the viewpoint of policy makers the principle of equity may prevail.

Application of monetary valuation of health-endpoints in principle allows formal cost-benefit analysis of policy options to improve the air pollution situation from different perspectives. As further air pollution control becomes more and more expensive and will increasingly affect individual behaviour rather than institutions (e.g. the energy or industrial sector), a crude indication of what we are willing to pay to avoid health impacts attributable to air pollution may improve the policy making process. However, this would necessitate more adequate WTP-estimates, preferably from European studies explicitly dealing with air pollution situations.

Application of aggregate health metrics, such as DALYs and WTP can be of help in environmental health policy as long as they are not considered as the ultimate 'health coin', and as long as other criteria, such as equity and solidarity are incorporated in decision making as well.

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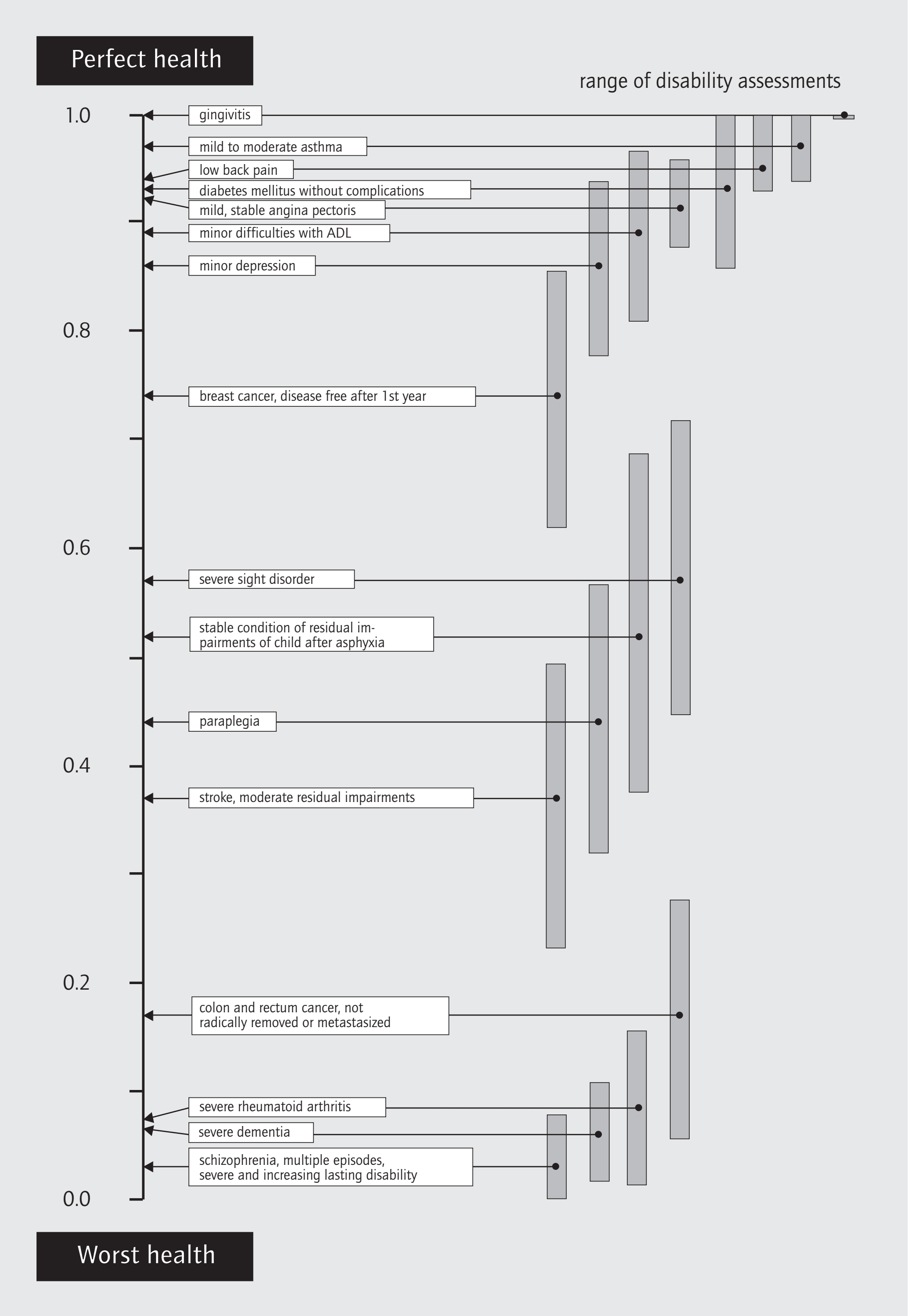

Figure 5-4. Scale of 'calibration' health states, employed to attribute weights to health states associated with environmental exposures.

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