Chapter 1

The Valuation of Environmental Health Risks

Environmental policy affects human health by reducing environmental risks that result in either premature mortality or non-fatal ill-health. People attach value to the reductions in health risk associated with environmental policies, and valuing such benefits can be undertaken using either revealed preference or stated preference methods. Depending on the nature of the environmental pressure and health impact, it has been found that health benefits can represent a majority of benefits of policy interventions. However, most such studies have been done using adult samples, and there is a need for similar estimates for children.

Introduction

Environmental policy affects human health by reducing environmental risks that result in premature mortality. Second, it may reduce the risk of acute non-fatal health impacts which are temporary in nature, or improve the health conditions of those living permanently with a disease or other health condition. These are known as morbidity benefits. Indeed, health-related benefits often dominate the benefits associated with the introduction of environmental policies.

A review (Pearce, Atkinson and Mourato 2006) of valuation studies undertaken in the European Union reveals that health benefits account for a minimum of one-third and a maximum of nearly 100% of overall benefits from pollution control.[1] The US EPA's (1997) assessment of the Clean Air Act (CAA found that the benefits of the Act (1970) and its amendments (1977) are dominated by health impacts. These can be as great as 99%, if effects on children's IQ are included. A prospective analysis (EPA 1999) of the CAA Amendments of 1990 found that health benefits represented over 96% of total estimated benefits.[2]

An analysis (Holland *et al.* 2005) of the benefits associated with the Clean Air for Europe (CAFÉ) programme reached comparable conclusions. Positing a set of scenarios based upon potential policy developments, it was found that health benefits relative to the baseline (current legislation) were between EUR 37 and EUR 160 billion per year in 2020, while non-health impacts were estimated to be less than EUR 1.0 billion. However, it is important to emphasise that the latter only includes damage to crops from ozone exposure and material damages from acid deposition.

Given their relative importance in total benefits, it is important to determine how best to ensure that values for health risks are estimated correctly if cost-benefit studies are to be a reliable input into policy-making processes.

Valuing health risks in general

As noted in the Introduction, there are two main approaches to estimating the WTP for a mortality risk reduction. The first approach, revealed preference studies, uses actual behaviors to infer the rate at which individuals trade off income for safety, and includes compensating wage studies, consumer behavior studies, and hedonic pricing approaches. For example, labor market studies (see Viscusi and Aldy, 2003) relate wage rates to the risk of fatal and non-fatal accidents on the job, reasoning that workers would be prepared to accept a riskier job only for higher pay.[3] Other studies have related the price of automobiles to the risk of dying in an accident associated with an automobile's safety features (Atkinson and Halvorsen, 1990; Andersson, 2005), or the value of a home to the risk of dying for environmental exposures in the neighborhood (Gayer *et al.*, 2000). In the case of child mortality, Jenkins *et al.* (2001) have used expenditures on bicycle helmets to infer the VSL for children of various ages and adults, and Blomquist *et al.* (1996) have relied on the time spent fastening car seatbelts. Davis (2004) uses a cluster of children's leukemia cases in a Nevada community and housing prices to infer the value of a statistical case of child leukemia.

The second approach to estimating the VSL – stated preference studies – queries individuals about what they would do under specified hypothetical circumstances. Stated preference methods include contingent valuation (CV) and conjoint choice experiment surveys. Unlike revealed preference studies, stated-preference studies can be designed to cater to any population and any risk of interest (see Bateman *et al.*, 2002 for a review). In addition, since they rely on hypothetical scenarios created by the researchers, stated preference studies can be designed to deal squarely with the issue of latent risks, in which there is a lag between exposure and the health impact. For these reasons it was decided to implement stated preference surveys in this study.

Once the value associated with a change in mortality risk is estimated, the risk change in question is divided by this value, which then gives the VSL. The social impacts of the policy can then be derived upon the basis of an assessment of the change in risk arising from some change in an environmental variable, say pollution concentrations (*e.g.* a dose-response function). This function can be used to estimate numbers of premature mortalities, and it is these mortalities that are multiplied by the VSL to give an aggregate measure of the social benefits associated with the introduction of the policy. The final equation is:

$$\mathrm{VSL} = \frac{dw}{dp} = \frac{u_a(w) - u_d(w)}{(1-p)u'_a(w) + pu'_d(w)}$$

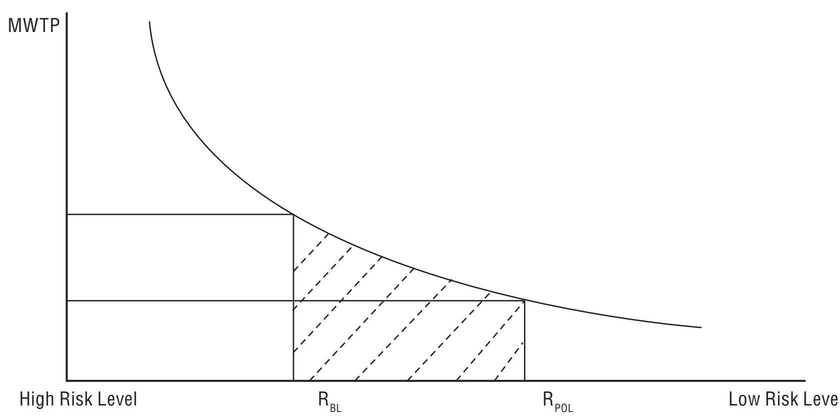
where w is wealth (which is often proxied by income), p is the probability of dying in the current period, (1 – p) is the probability of surviving the current period, u is utility. The subscripts "*a*" and "*d*" refer to survival and death respectively. The numerator thus shows the difference in utility between surviving and dying in the current period, while the denominator is the marginal utility of wealth conditional on survival or death (see Pearce *et al.* 2006 for a discussion).

As such, the equation gives the marginal rate of substitution between a risk of dying and wealth. VSL is necessarily positive since people attach a positive value to both survival and wealth. As such, both the numerator and the denominator are positive. Respondents are presented with changes in the risk of dying (*e.g.* through a public policy or a private purchase), and are requested to "trade off" this change in risk by their WTP for a public policy (*i.e.* tighter standards) or a private purchase which reduces the risk.

Figure 1.1 illustrates this relationship between marginal WTP (on the y-axis) and risk levels (on the x-axis). Risk levels are decreasing from left to right. WTP is expressed in marginal terms (MWTP) because this is what is elicited in valuation studies – *i.e.* what the respondent is WTP for a change in risk. The MWTP is assumed to be decreasing with risk levels, which implies that at very low levels of risk people are WTP relatively less for still further reductions in risk.[4]

In Figure 1.1 the baseline risk level is at point R_{BL} (*e.g.* 10 in 10 000). Suppose the policy measure in question reduces risk levels from the baseline level of risk to point R_{POL} to the right on the x-axis (*e.g.* 5 in 10 000), then the WTP for that risk reduction is equal to the shaded area under the marginal WTP curve between these two points. If the results of a valuation study indicate that the mean WTP to secure this risk reduction is USD 100. Then the VSL would be USD 200 000 [*i.e.* (USD 100 × 10 000) / (10 – 5)].

Figure 1.1. **Marginal WTP for a Risk Reduction**



On the basis of available empirical evidence WTP is affected by a number of factors, including quality of life of the period survived as a consequence of the risk reduction, *i.e.* WTP to reduce risks should be higher if the individual anticipates being in good health (apart from the risks in question), and lower if the individual expects to be in poor health. Some of the other factors which affect WTP for reductions in mortality risks are discussed below.

Latency and Discounting

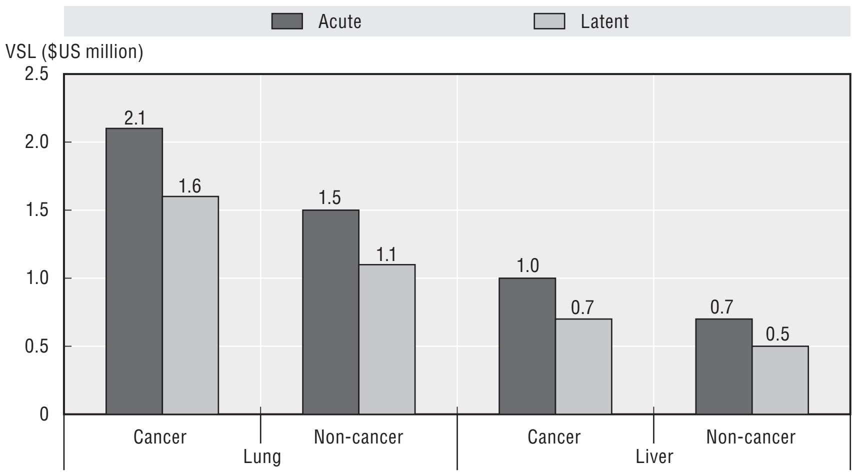
WTP is likely to be affected by the point in time at which the risk reduction is incurred. In the environmental health context, this would arise when the risk is latent, *i.e.* situations in which exposure now does not cause death (or ill-health) until some point in the future. The immediate risk would be relevant to, say, road or occupational accidents. What is sought in this context is the WTP to avoid that risk which could occur tomorrow or in the very near future, *i.e.* acute risks. However, in the case of air pollution, there may be a lag between the "dose" (air pollution concentrations) and the "response" (*e.g.* respiratory problems), *i.e.* there is a degree of latency. Depending upon the environmental pressure under consideration this lag can be very long.

According to standard economic theory, a good received today is valued more than a good received tomorrow. The discount rate is a measure of the extent to which delayed satisfaction differs from immediate satisfaction. While "private" discounting reflects the such inter-temporal trade-offs from the narrow perspective of the individual (or firm), the "social discount rate" should reflect such trade-offs at the level of society as a whole, and is thus more appropriate for cost-benefit analyses. However, the social discount rate applied in a given CBA should reflect the private discounting practices of those affected by the policy. Whether the rates differ in practice will depend upon factors such as the efficiency and taxation of capital markets. Policies with intergenerational impacts raise particular complications.[5]

Since reductions in risk are valued by individuals in a manner analogous to other goods and services, the point in time at which the benefits of such risk reductions are accrued should also be discounted. As such, it might be imagined that latent impacts would be valued less than immediate impacts. However, this may not be the case since latency implies: A) the date will be later; B) the person exposed will be older. The effect of A is reflected in the discount rate. However, since preferences for reducing risks depend on the perceived utility associated with different periods of life, the effect of B may result in latent impacts actually being valued more than immediate impacts.

For this reason empirical evidence is much-needed. A study by Hammitt and Liu (2004) for Taiwan finds that, irrespective of the organ affected, or whether the risk relates to cancer or not, with a proposed latency of 20 years the estimated VSL is at least 30% less than for equivalent acute risks (see Figure 1.2). They estimate a discount rate of approximately 1.5% per year. However, this is less than what was estimated in a number of other studies [*i.e.* 8% per year in Krupnick *et al.* (2002), and 4.5% per year in Alberini *et al.* (2006a), and as much as 17% in Itaoka *et al.* (2007)].

Figure 1.2. **Estimated Value per Statistical Life**



Source: Hammitt and Liu (2004).

Age and Life Expectancy

Early studies of the VSL made little or no reference to the age of the individuals at risk, perhaps because of the focus on road accident or occupational risks where the mean age of the person at risk is fairly constant. However, in the context of environmental policy the issue of age becomes more important for VSL since it is the very old and (perhaps) the very young which are most vulnerable. The implications of the very old have been examined, since it is well-known that pollution control policy reduces mortality amongst the elderly (Pope *et al.*, 1995; Krupnick *et al.*, 1999).

While there may be differences in risk for different age groups, whether or not WTP for the same risk reduction varies with age is less clear (Krupnick, 2007). The most evident impact of age on WTP for a risk reduction is that since older people have lower life expectancy, the benefit of any current reduction in risk declines. As such, one would expect VSL to decline. However, assuming that there are fewer alternative uses, the opportunity cost of spending money on a risk reduction declines as time goes by, and as a consequence, WTP for risk reduction may actually rise with age.

Which of these two effects dominates will depend upon many factors, and it is commonly asserted that it may follow an inverted-U, first increasing with age and then falling. One of the first studies to look at this issue (Jones-Lee *et al.* 1985) found VSL to be fairly flat, but increasing to mean age (about 40) and decreasing thereafter. Krupnick *et al.* (2002) found WTP flat from age 40 to 69 and decreasing from age 70 to 74. Based upon revealed preference evidence, Viscusi and Aldy (2007) find an inverted-U relationship, reaching a maximum in the mid-40s and then falling relatively sharply thereafter.

Risk characteristics and context

The precise nature of the risk may also have an influence on the WTP for risk reductions.[6] For instance, some risks may be particularly "dreaded", and thus for which risk reductions would be particularly highly valued. The "dread" aspect of a given risk can indeed have a significant impact on WTP, because it is generally associated with greater fear. Cancer risk is a notable example frequently discussed in the literature, and some studies which have sought to estimate the "cancer premium" (see van Houtven *et al.* 2008 for a recent example). Other types of risk which are thought to inspire "dread" include particular types of fatal accident.

Another important risk characteristic which appears to have an influence on WTP is "voluntariness", which can be understood as the choice people have of voluntarily exposing themselves to the risk in question. Research in both psychology and economics has shown that people are more concerned about risks that they perceive to be involuntary (*e.g.* exposure to air pollution) than about risks perceived to be voluntary (*e.g.* smoking) (Fischoff *et al.* 1978 and Slovic 1987). As such, they generally prefer voluntary risks to involuntary ones, suggesting that the degree of "risk voluntariness" could have an impact on the WTP. Closely related is the issue of "controllability", which reflects the extent to which people believe they are able to undertake preventive actions which reduce their exposure to risk.

In a study of Tokyo Metropolitan residents which examined risk characteristics in a systematic manner, Tsuge *et al.* (2005) examined four types of risks: accidents, cancer, heart disease, and general risks. The study showed that voluntariness, controllability, severity, public knowledge and exposure each had a significant and positive impact on the WTP to reduce a given risk. They found a small preference for avoiding cancer risks. Overall, respondents displayed the highest preference for the measures against cancer, and the lowest preference for measures against accidents.

Size of baseline risk and risk reductions

The VSL is usually derived by considering only the WTP for a risk change and the size of the risk change itself. However, WTP may also be influenced by other risks. That is, competing risk reduces the chance that the individual will benefit from the policy-related risk. This effect is likely to be most important for those most at risk of mortality in general. Given the generally low baseline risks in our study (mortality risks for children associated with environmental pressures), this is unlikely to be important. However, in other cases it may be important, *e.g.* for the elderly and/or those in poor health.

In addition, the size of the proposed risk reduction may affect WTP in a manner which is not strictly proportional, as predicted by theory. Hammitt and Graham (1999) test for two predicted relationships: a) that WTP increases with the size of the risk reduction, and b) for low risks WTP should be virtually proportional to the change in risk. For the 10 studies which contain sufficient information to test scope sensitivity, the studies confirm the first hypothesis that WTP varies with risk reduction, but proportionality is not observed. Overall, a significant minority of respondents report the same WTP regardless of the size of risk change.

While a number of arguments have been put forward to try and explain scope insensitivity, in the context of this study, one possible explanation relates to the problems of communicating low risk levels to respondents. In effect scope insensitivity may not reflect underlying preferences, but rather failings in study design. However, it is also clear from the literature that small risks are difficult for people to understand and judge.

Morbidity

As noted above, some of the most important health benefits associated with the introduction of environmental policies relate to improved health, and not reduced mortality risks *per se*. Clearly many of the issues raised above (*e.g.* context, baseline risks) are relevant to the valuation of morbidity risks. However, it is perhaps the nature of the risk characteristics which pose the most significant complications for the valuation of morbidity, and in particular issues related to dread concerning pain and suffering.

Table 1.1. **Marginal WTP for a Risk Reduction**

Health Endpoint	% attributable to pain and suffering
Respiratory hospital admission	25.87%
Cardiac hospital admission	21.33%
Respiratory emergency department visit	46.73%
Cardiac emergency department visit	23.15%
Reduced activity day	47.92%
Asthma symptom day	57.14%
Acute respiratory symptom day	7.69%

Source: Stieb *et al.* (2002).

The relative importance of these costs for different environment-related health end points can be assessed based on two studies. In one case, Stieb *et al.* (2002) estimate the economic benefits of reducing acute cardio-respiratory morbidity associated with air pollution in Canada (see Table 1.1)[7] In a contingent valuation study undertaken in Strasbourg, France, Rozan (2005) found that pain and suffering represented between 15%-100% of the total value of health impacts related to air pollution. Interestingly, the proportion is highest for children (and the elderly).

Conclusions

All of the issues raised above highlight the complexity of obtaining reliable estimates of WTP for health risk reductions for children. This is exacerbated by the fact than many of these factors are related in complicated ways. For instance, there is, a link between context and age. Indeed, much debate in the VSL literature has focussed on how the age of an individual matters in relation to different risk contexts. By and large this has involved assessing whether VSLs derived in accident contexts (especially road accidents and workplace accidents) are equally applicable to pollution contexts. Accidents tend to affect people of much lower average age than pollution.

In addition, there may well be a link between the degree of latency and age. For instance, the risk associated with air pollution may well be immediate for older people since we know that it is older people who tend to be most affected by air pollution, *i.e.* the risks they face are still acute. But for younger people the risk of immediate premature mortality will be considerably less. The benefit of reducing pollution will accrue to this younger group when they are much older. Distinguishing between age and latency is crucial to understanding the determinants of VSL.

And finally, latency and risk characteristics may also interact. If the latent risk is accompanied by a period of suffering which is "dreaded" then the respondent may well prefer to die immediately than pay for an intervention which increases his chances of surviving for a specific period. Preferences for reducing current and latent mortality risks cannot be divorced from the quality of life associated with the period "survived", and the results cited above concerning "pain and suffering" underscore this point.

Valuing health risks for children

Perhaps, the most important challenge in children's health valuation relates to the impossibility of directly eliciting preferences from children since they do not have command over resources to make trade-offs in actual markets, and may not have the maturity to make such trade-offs in a hypothetical market. Since it is not possible to directly elicit preferences from children, three alternative perspectives have been proposed to elicit children's preferences indirectly. The first approach is referred to as the "societal perspective", and consists in eliciting preferences from a representative sample of the population, including all adults. The "adult-as-child" perspective, in which the adult respondents are requested to place themselves in the "place" of children is another possibility. Finally, the "parental perspective" can be used, in which parents are asked about the value they place on their children's health.

None of the perspectives is ideal. The societal perspective may be affected by the capacity of the researcher to distinguish between different types of altruism, only some of which should be included in a measure of social WTP to avoid double counting.[8] The "adult-as-child" perspective is very demanding on the respondent, requiring them to think back to their own childhood and assess the risks they faced (and preferences they held) at that time. There is a general consensus in the literature that the parental perspective would appear to be the most promising approach (Viscusi *et al.*, 1987). Although the difficulties associated with properly accounting for people's altruism are also likely to be a major concern with this perspective, it has the advantage of asking the persons who have the interests of the child at heart, and who are used to making decisions on their behalf (see Dockins *et al.* 2002).

The valuation of children's health brings to the fore the problem that the valuation exercise does not take place in the traditional individual context where someone is asked to state a WTP for his/her own risk reduction, but rather in a household context where someone is asked to evaluate a risk reduction for another member of his/her household. As a consequence, the choice of the intra-household allocation model and household-related factors may affect the WTP estimates.

Two types of household allocation model can be used: a unitary model in which the household is treated as a unit and financial resources are pooled, or a collective model in which the individual utility functions of each household member (at least the adults) are pooled to obtain a collective decision, taking account of the differences in household members' preferences. Generally, children are considered as passive participants in family decision-making. But what happens when the child becomes adolescent and is in a better position to express his/her preferences? What about two parents having different preferences concerning their own children? Alternative approaches that could fit better to these particular contexts should also be considered and examined. For further details on the various household allocation models, see Dickie and Gerking (2006).

Irrespective of the model assumed, household-related factors may affect estimates of the value of risk reductions for children. As an example, the family structure and composition affect resource allocation and health outcomes experienced (Dickie and Ulery, 2002). Some studies have highlighted differences between children according to their health status, gender or age (Pitt and Rosenzweig, 1990; Hanushek, 1992; Liu *et al.*, 2000). Finally, altruism from parents toward their children may significantly affect the estimates and be a source of disparity between adults' values and children's values (Dickie and Ulery, 2001). These results suggest that applying a unique value for all children would lead to unreliable estimates of children's health.

Moreover, a number of the risk factors which are important for valuation in general (*i.e.* context and risk characteristics, age, latency, size of baseline risk and risk reduction, etc.) have particular resonance for the valuation of children's environmental health risks in particular. For instance, the non-linear relationship between age and WTP for risk reductions clearly has important implications for children. However, extrapolating this relationship to childhood would clearly be inappropriate, given that the studies were based only on adult samples. What determines the age-WTP relationship within childhood may be very different from the relationship within the adult population.

In addition, latency can have different implications for risks for children and for adults. On the one hand, there is evidence that parents discount latent impacts differently for themselves than for their children. On the other hand, the issue of latency has particular implications when exposure is incurred in childhood but the health impacts are realised much later as an adult. In the event that risk preferences differ between children and adults, do these differences relate primarily to differences associated with exposure or with response? As such, latent impacts which can manifest themselves ten or more years after the point of exposure raises particular complications for the researcher (and policymaker).

The degree of "voluntarism" of a given risk may also mean something very different for a 6-year old than for an adult. While respondents to a survey may perceive the risks associated with traffic to be voluntary for adults, the very same risks may be perceived as involuntary for children due to the more restricted options, *e.g.* in order to get to school.

Similarly, a risk which is perceived as "controllable" for an adult may be seen as uncontrollable for children. Even if a defensive expenditure is undertaken as a means to reduce risk, the parent may feel that they have "imperfect control" over its ability to protect their child from a given risk. Mitigation of the risk of skin cancer from UV rays through the application of suntan lotion may represent such a case. Another case might be the purchase of bicycle or motorcycle helmets.

And finally, the issue of dread may be understood very differently for children than for adults. It is quite possible that dread may be very different for a similar risk (in terms of context) which affects children than adults. For instance, the perception of welfare losses attributable to the pain and suffering associated with some types of risks may be different for children and adults.

Review of previous epidemiological and economic studies

Given these difficulties, it is hardly surprising that epidemiological and economic evidence on children's environmental health is limited. The lack of available data specific to children precludes an evaluation of the health impacts of existing environment-related health policies. More studies are necessary, particularly on specific health endpoints comparable to those for adults, such as chronic asthma morbidity. Therefore, priority should be given to the collection and assessment of epidemiological data to implement valuation studies to provide meaningful policy advice. However, improved epidemiological data of this sort is not sufficient. Ignoring valuation differences between adults and children could lead to biased estimates of health benefits associated with a reduction of environmental risk and therefore to inefficient and wasteful policies.

Some of the most important health impacts associated with air and water pollution are listed in Tables 1.2 and 1.3. However, these are based upon general epidemiological studies on adult populations. A paper prepared by Hunt and Arigoni Ortiz (2006a) for this project reviews the epidemiological evidence on the relationship between environmental exposures and adverse health impacts for children.[9] The review highlights the emphasis on air pollution (PM, NO_2, CO) in epidemiological research. However, there are some studies that relate to other environmental pressures (*e.g.* pesticides) and that find some evidence of adverse health impacts. The impacts of exposure to lead and other heavy metals on cognitive capacity have been the subject of numerous studies.

In general, the evidence from mortality studies is limited compared to that from morbidity studies. For instance, almost all of those studies that have been conducted in European countries have focused on morbidity, not mortality. Nonetheless, the evidence suggests that children are susceptible to exposure to environmental pollution, with the health endpoints of most importance being air pollution-induced mortality and respiratory symptoms, and perhaps cancers associated with pesticide use. (See Annex for a summary of some of the most important studies.)

Differences in the estimation of the benefits associated with the introduction of environmental policies arise not only from differences in the

Table 1.2. **Health Effects Associated With Selected Water Pollutants**

	Disease/Pollutant	Health impacts
Bacterial	Amoebic dysentery	Abdominal pain, diarrhoea, dysentery
	Capbylobacteriosis	Acute diarrhoea
	Cholera	Sudden diarrhoea, vomiting. Can be fatal if untreated
	Cryptosporidiosis	Stomach cramps, nausea, dehydration, headaches. Can be fatal for vulnerable populations.
Chemical	Lead	Impairs development of nervous system in children; adverse effects on gestational age and fetal weight; blood pressure
	Arsenic	Carcinogenic (skin and internal cancers)
	Nitrates and nitrites	Methaemoglobinaemia (blue baby syndrome)
	Mercury	For fetuses, infants, and children, the primary health effect of mercury (in the form of methylmercury) is impaired neurological development. At high doses, mercury is also known to induce higher incidences of kidney damage, some irreversible.
	Persistent organic pollutants	These chemicals can accumulate in fish and cause serious damage to human health. Where pesticides are used on a large-scale, groundwater gets contaminated and this leads to the chemical contamination of drinking water.

Source: EEA/WHO-Europe (2002).

Table 1.3. **Health Effects Associated With Selected Air Pollutants**

Pollutant	Short-term effects	Long-term effects
PM	– Increase in mortality – Increase in hospital admissions – Exacerbation of symptoms and increased use of therapy in asthma – Cardiovascular effects – Lung inflammatory reactions	– Increase in lower respiratory symptoms – Reduction in lung function in children and adults – Increase in chronic obstructive pulmonary disease – Increase in cardiopulmonary mortality and lung cancer – Diabetes effects – Increased risk for myocardial infarction – Endothelial and vascular dysfunction – Development of atherosclerosis
O_3	– Increase in mortality – Increase in hospital admissions – Effects on pulmonary function – Lung inflammatory reactions – Respiratory symptoms – Cardiovascular system effects	– Reduced lung function – Development of atherosclerosis – Development of asthma – Reduction in life expectancy
NO_2	– Effects on pulmonary structure and function (asthmatics) – Increase in allergic inflammatory reactions – Increase in hospital admissions – Increase in mortality	– Reduction in lung function – Increased probability of respiratory symptoms – Reproductive effects

Source: Adapted from WHO (2004b; 2006).

risks faced by different populations (*e.g.* adults and children), but also differences in the values which society attributes to risk reductions for different populations. While there are relatively few studies that have sought to value the benefits of health risk reductions for children which are explicitly related to environmental exposures, there are a number of studies which have estimated the WTP to reduce health risks associated with other causes for children and adults.

Although the evidence is mixed, most of the studies concluded that the WTP to reduce mortality risks to children was greater than the WTP to reduce similar risks to adults. Table 1.4 provides a summary of some recent studies in which values (mortality and morbidity) have been estimated for both adults and children, while the Annex discusses these and other relevant studies in more detail.

The objectives of the VERHI project

In the area of children's environmental health risks, policymakers have been forced to make decisions and set priorities on the basis of very limited evidence and limited information. This raises a question on the appropriateness of policies currently in place that have significant implications for children's health.

Environmental standards are generally based on evidence related to their impacts on adult populations, which may be quite different from those for children. Proper valuation of impacts on children may well result in standards which are different from those currently in place. Analogously, policy priorities across different environmental health impact areas are based on values obtained for adult populations which may be inappropriate for children. In such cases, governments are not allocating investments cost-effectively so as to avoid loss of lives or reduce ill-health. It is, therefore, important to obtain values for environmental health risk reductions specifically for children. Moreover, it is important that these values be comparable to those obtained for adult populations in order to set policy priorities in an optimal manner.

The rest of this document discusses how this was done in the context of the VERHI project. The next chapter reviews some of the main methodological concerns associated with addressing environmental health impacts for children. Chapter 3 summarises the survey development work which was undertaken in order to ensure that the surveys implemented generated credible estimates. Chapter 4 provides a summary of the main results of the project. The document concludes with a discussion of policy implications.

Table 1.4. **Estimates of VSL and WTP for Children and Adults**

Study	Country	Valuation Method	Benefits Measure	Value
Mortality				
Takeuchi *et al.* (2008)	Japan	Contingent valuation	Societal WTP to reduce fatality risks	VSL (in Yen billion)
				1.17 to 7.74 (child)
Mount *et al.* (2000)	United States	Averting behaviour – automobile safety purchases	Parental WTP to reduce fatality risks	VSL (in USD million)
				7.3 (child)
				7.2 (adult)
				5.2 (elderly)
Jenkins *et al.* (2001)	United States	Averting behaviour – child bicycle helmets	Parental WTP to reduce fatality risks to children	VSL (in USD million)
				2.9 (child of 5-9)
				2.8 (child of 10-14)
				4.3 (adult)
Hammitt and Haninger (2010)	United States	Contingent valuation	Parental WTP to reduce fatal-disease risks by consuming pesticide residues on food	VSL (in USD million)
				12.4 (child)
				7.5 (adult)
Morbidity				
Liu *et al.* (2000)	Taiwan	Contingent valuation	Mother's WTP for preventing a cold to her and her child	USD 57 (child)
				USD 37 (mother)
Agee and Crocker (2001)	United States	Contingent valuation	WTP for a 10% increase of the health status of the child and the respondent	USD 452 (child)
				USD 249 (adult)
Dickie and Ulery (2001)	United States	Contingent valuation	WTP to avoid seven days of one symptom	USD 150 to USD 350 (child)
				USD 100 to USD 165 (adult)
			WTP to avoid one-week incident of acute bronchitis	USD 400 (child)
				USD 200 (adult)
Dickie and Brent (2002)	United States	Contingent valuation	WTP to avoid one day of first symptom	USD 92 (child)
				USD 35 (adult)
Braun Kohlová and Scasny (2006)	Czech Republic	Contingent valuation	WTP to reduce mild bronchitis	EUR 38 (child)
				EUR 21 (adult)
Dickie and Gerking (2001)	United States	Contingent valuation	WTP for a 1% reduction in non-melanoma exposure to skin cancer risk	USD 3.18 (child)
				USD 1.29 (adult)

Notes

1. Studies included in the review include: Holland and Krewitt, 1997; Holland *et al.* 1999; Krewitt *et al.*, 1999; IVM, NILU and IIASA, 1998; Olsthoorn *et al.*, 1999.
2. It is important to note that both studies did not include monetised estimates of the benefits of certain health (*e.g.* from toxic pollutants) and non-health (ecosystem damage) impacts.
3. See Schnier *et al.* (2009) for a somewhat different approach, based on a commercial fishing vessel captain's decision to go fishing in the Alaskan red crab fisheries as a function of weather and policy variables intended to improve safety. Schnier *et al.* (2009) obtain VSL values of USD 4.6-4.9 million, and attempt to disentangle the value of crew members from that of the vessel's captain.
4. This is discussed below.
5. An example is climate policy, see Arrow *et al.* (1996) for a discussion.
6. US EPA (2000) lists the following pairs of risk characteristics: voluntary/involuntary; controllable/uncontrollable; ordinary/catastrophic; delayed/immediate; natural/man-made; old/new; necessary/unnecessary; and occasional/continuous. There can be a high correlation between some of the pairs listed. In this sub-section we focus on the first three. The fourth pair has been discussed above in the context of latency. Aspects related to the last pair are discussed below.
7. In estimating the value of pain and suffering, the researchers mapped symptoms and activity restrictions to the various health outcomes identified in epidemiological and clinical studies.
8. See Takeuchi *et al.* (2008) for a recent example in which an effort is made to disentangle the two types of altruism in the context of child mortality using a societal perspective.
9. *www.oecd.org/dataoecd/16/21/39338429.pdf*

ANNEX 1.A1

Review of the Epidemiological and Economic Evidence

As noted above, part of the motivation for this study was the perception that environmental health risks for children are significant and distinct from that for adults. It is important, therefore, to review the epidemiological evidence on the relative importance of such risks. Moreover, robust measures of the value of health concerns for children based upon stated preference methods require the use of scenarios which reflect risks which are meaningful to respondents. Since the valuation of health end-points depends on quantification of the risk, it is necessary to know for which end-points there was solid epidemiological evidence.[1]

This Annex, which draws extensively, upon reports by Hunt and Arigoni Ortiz (2006a and 2006b), reviews the relationship between children's health and the environment, summarising the characteristics of the main health outcomes associated with children's environmental exposures. It also provides a review of the economic studies which have been undertaken which relate (sometimes indirectly) to the valuation of environmental health risks for children. Studies which relate to both morbidity and mortality are included in the review.

Review of the Epidemiological Studies

Mortality studies

Only a few epidemiological studies have focused on the association between child-mortality and environmental hazards, and a causal relationship has been found for at least some studies in the case of air pollution. Some studies have established a relationship between different environmental risk factors and potential chronic diseases such as cancer, but these studies are presented in the next section, which addresses morbidity.

Several epidemiological studies based on time-series data analysis have identified causality between exposure to specific air pollutants and mortality in children. For instance, Currie and Neidell (2005) found that carbon monoxide

(CO) had a significant effect on infant mortality in California (US). In Sao Paulo (Brazil), Conceicao *et al.* (2001) observed a significant association between respiratory mortality in children and daily levels of CO, sulphur dioxide (SO_2), and particulate matter (PM_{10}). Similarly, Lin *et al.* (2004) also showed a consistent relationship between exposure to PM_{10} and SO_2 and daily neonatal mortality with a short time lag in Sao Paulo (Brazil). These results confirmed those of previous studies on similar issues led in the US (Chay and Greenstone, 1999) and Brazil (Saldiva *et al.*, 1994).

Woodruff *et al.* (1997) evaluated the relationship between infant mortality and PM_{10} in the US, through analysis of a cohort of approximately four million infants between 1989 and 1991. The study focused on four infant death causes: sudden infant death syndrome with normal birth weight; and, respiratory deaths for normal birth weight and low birth weight infants; and all-cause mortality. The odds-ratio[2] for all-cause mortality for the high exposure group *versus* the low exposure group was 1.10; for sudden infant death syndrome, the odds-ratio was 1.26; for respiratory deaths in normal birth weight infants, the odds-ratio was 1.40, while for low birth weight infants, high exposure was not significantly associated with mortality from respiratory diseases. Woodruff *et al.* (1997) concluded that exposure to PM_{10} was associated with increased risk of post-neonatal mortality.

Morbidity studies

Many epidemiological studies have focused on the impact of air pollution on children's health. For instance, Gauderman *et al.* (2005; 2007) and McConnell *et al.* (2006) found that the proximity to major roads and freeways increased the prevalence of asthma and wheezing for children living in Southern California (US). Gauderman *et al.* (2007) found that local air pollution had detrimental and independent effects on lung functions, resulting in pronounced lung function deficit at the age of 18. Chauhan *et al.* (2003) found a positive association between high exposures to NO_2 and the severity of resulting asthma exacerbation in 8-11 year old children in the UK.

However, Penard-Morand *et al.* (2005) did not find any consistent positive relationship between NO_2 and asthma. The same study found that an increase in the exposure to PM_{10}, SO_2 and ozone was positively related to increased prevalence of asthma and allergic rhinitis. More recently, Brauer *et al.* (2007) used data from a Dutch birth cohort of children between 0 to 4 years of age, and they found a significant and positive association between traffic-related air pollution and asthma and wheezing, as well as with several types of respiratory infections (*e.g.* ear, nose, and throat infections; flu and serious colds). Other studies which find some evidence of a link between respiratory diseases and air pollution in include Segala *et al.* (2008), Hertz-Picciotto *et al.* (2007), Dales *et al.* (2006) Triche *et al.* (2006), Pierse *et al.* (2006) and Zhang *et al.* (2002). Exposure to air pollution was also found to be associated with low birth weight in several case studies (see for example Bobak and Leon, 1999; Dugandzic *et al.*, 2006; Bell *et al.*, 2007), and with developmental delays at age 3 (Perera *et al.*, 2006).

Although outdoor air pollution is still the focus of the largest number of studies (in particular, traffic-related air pollution), other environmental hazards have been considered in the literature, both in OECD and non-OECD countries. For instance, there is an increasing interest in the linkages between indoor air pollution (mainly as environmental tobacco smoke – ETS) and child-morbidity. Tanaka *et al.* (2007) studied the relationship between passive smoking at home and the prevalence of allergic disorders in Japanese schoolchildren; they estimated a relative risk[3] of 1.33 of incident asthma among 6-18 year old children. Lewis *et al.* (2005) and Crain *et al.* (2002) also found a significant association between ETS and childhood asthma in the UK and the US, respectively. Rauh *et al.* (2004) observed negative impacts of early exposure to ETS on mental development at 2 years of age in New York City (US), using data from a birth cohort. Similar results were obtained by Yolton *et al.* (2005), who determined an inverse relationship between exposure to ETS and cognitive and academic abilities among 6-16 year old US schoolchildren, even at low exposure levels.

Lead and other heavy metals have also received attention from researchers. Lead exposure was found to be associated with decreased cognitive performance in children by Lanphear *et al.* (2005), who used data from birth cohorts in the US, Mexico, Australia and the former Yugoslavia. As another example, mercury, especially methylmercury, has been consistently linked to impaired cognitive performance (Axelrad *et al.*, 2007) and damaged brain functions (Grandjean *et al.*, 1997). Arsenic exposure may have similar effects on children, as suggested in Rosado *et al.* (2007), Wang *et al.* (2007) and Wasserman *et al.* (2007).

Many recent studies have focused on chemicals and pesticides. For example, Gouviea-Vigeant *et al.* (2003) investigated the link between exposure to some chemicals (solvents, pesticides and petrochemicals) and childhood cancers in the US. They found that exposure to such chemicals may increase the likelihood of childhood leukaemia and other cancers (in particular brain and central nervous system cancers); however, it was not possible to find evidence of cancer from exposure to specific chemicals. Indeed, mixtures or groups of chemicals (*e.g.* pesticides, hydrocarbons and solvents) were more likely to affect children's health. Their analysis also showed that early-life exposure significantly increased the likelihood of cancer.

Zahm and Ward (1998) reviewed the epidemiological studies analysing the linkages between exposure to pesticides (for both adults and children) and several types of cancers (*e.g.* leukaemia, lymphoma and sarcoma). They found that exposure of children to pesticides resulted in greater risks of cancers, suggesting that children may be particularly sensitive to the carcinogenic effects of pesticides. Rudant *et al.* (2007) also found a significantly positive association between use of domestic pesticide (at the household level) and childhood blood diseases in France. Similar results were obtained by Menegaux *et al.* (2006) who investigated the impact of pesticide exposure on childhood leukaemia in France.

Only a few studies have analysed the effects of water pollution on children's health. Schwartz *et al.* (1997) investigated the linkages between drinking water turbidity and gastrointestinal illnesses in the US. They found that an increase in water turbidity consistently resulted in increased emergency visits and hospital admissions for gastrointestinal diseases. As another example, Xiong *et al.* (2007) examined the impact of fluoride in drinking water on liver and kidney functions in Chinese children. They found that drinking water fluoride levels above 2 mg/l can seriously damage liver and kidney functions in children. Finally, as mentioned above, elevated arsenic levels in drinking water were associated with impaired cognitive performance in China (Wang *et al.*, 2007) and in Bangladesh (Wasserman *et al.*, 2007).

Discussion

The review of the epidemiological literature highlights the emphasis on air pollution in epidemiological research, either because it is a high-priority issue in political agendas or because of data availability. Evidence from mortality studies is limited, compared to that from morbidity studies. Those studies which have been conducted in European countries have focused on morbidity, not mortality. Nonetheless, the evidence suggests that children are susceptible to exposure to environmental pollution, the health endpoints of most importance being air pollution-induced mortality and respiratory symptoms, and perhaps cancers associated with pesticide use.

Although the literature suggests a causal relationship between exposure to air pollution and mortality or morbidity in children, the complex interdependencies among variables should be borne in mind when interpreting the findings of at least some mortality studies. In addition, new "confounding" factors are still being identified. For example, Braga *et al.* (2000) investigated the potential confounding effect of respiratory epidemics on deaths associated with air pollution. They argued that controlling for influenza epidemics could result in an under-estimation of all respiratory epidemics. However they also concluded that the association between air pollution and respiratory-related deaths was still robust, even after having controlled for all types of respiratory epidemics.

Another type of confounding effect is the potential synergy of environmental pollution, either with other pollutants or with individual behaviour (*e.g.* smoking). Synergistic effects occur when the damage caused by two or more pollutants is greater than the effect caused by each individual pollutant acting alone. For instance, Lin *et al.* (2004) showed that the combined effects of PM_{10} and SO_2 on daily neonate mortality was stronger than the combined effects of the two pollutants on their own, suggesting a potential synergy between PM_{10} and SO_2. They also concluded that primary pollutants correlated strongly with each other, and that PM_{10} presented the highest correlations with other pollutants. Unfortunately there is no evidence which relates specifically to children.

Finally, it should be noted that only studies providing empirical support for the existence of a relationship between exposure to environmental pollution and adverse health effects on children have been presented here. It must be emphasised that other studies carried out on similar issues did not find any significant relationship. For example, Lewis *et al.* (2005) found no evidence of association between living near a main road and wheezing or asthma. Similarly, Gouveia and Fletcher (2000) did not establish causality between exposure to air pollutants and mortality in children under 5 years old (RR = 0.921 for NO_2 and respiratory mortality, and RR = 1.141 for CO and pneumonia).

Review of Valuation Studies

Given the relative paucity of economic studies provide estimates of WTP for risk reductions for children related to environmental exposures this review includes a discussion of both COI and WTP studies in two sub-sections. As with the review of the epidemiological literature the studies reviewed include both those which relate to morbidity and those which relate to mortality. And finally, those studies which estimate WTP for both children and adults are discussed in a final sub-section.

Cost of illness studies

The measurement of COI for children is particularly problematic. In particular, the value of the "lost productivity" component of COI for a child is particularly uncertain. Depending upon the nature of the health impact, it may refer to future earning losses (when the child is an adult) or to parental productivity losses (when parents stay at home to care for their sick children, *i.e.* when the parents act as caregivers). In principle, it could therefore result in COI for children that is lower than (or equal to) equivalent COI for adults.

Aggregation of COI to derive an estimate of the social benefits of a risk reduction is usually done on the basis of an environmentally attributable fraction (EAF) model, in which EAF is defined as "the percentage of a particular disease category that would be eliminated if environmental risk factors were reduced to their lowest feasible levels" (Smith *et al.*, 1999). The EAF is therefore a composite value that is computed as the product of the incidence of a risk factor, multiplied by the relative risk associated with that risk factor (Landrigan *et al.*, 2002). Using that approach, social costs are computed as follows:

$$Costs = disease\ rate \times EAF \times population\ size \times cost\ per\ case.$$

"Disease rates" are estimated by incidence or prevalence rates (depending upon information availability); and "cost per case" represents discounted lifetime expenditures ("cost of illness"). Although the determination of disease rates and EAF may be subject to uncertainty, the estimation of "cost per case" is even more controversial. A selection of recent studies estimating COI values for specific health outcomes or aggregated COI is presented here.[4]

Respiratory diseases

Weiss *et al.* (2000) assessed the costs of childhood asthma in the US in 1994. The total estimated costs of childhood asthma in 1994 were EUR 2.86 billion. Direct medical expenses were estimated to be EUR 1.75 billion and accounted for 62% of total costs. 80% of indirect costs (EUR 0.85 billion) were attributable to lost work productivity through disability.

Schramm *et al.* (2003) calculated the cost of illness of atopic asthma and seasonal allergic rhinitis (SAR) in Germany. They estimated the average annual cost of SAR to be EUR 1080 per child and EUR 1530 per adult. When adding the costs of severe asthma, total annual costs for the two health outcomes were estimated at EUR 7860 for a child and EUR 9207 for an adult. For children, 60 to 78% of the expenditures were direct costs, while 58% of adults' expenditures were indirect costs. The authors also concluded that these costs were increasing with the severity of atopic asthma and/or SAR.

Waterborne diseases

Lorgelly *et al.* (2008) assessed the cost of illness of gastroenteritis in children in the UK. The average cost for a child was estimated to be between EUR 85 and EUR 202 per episode. Based on the prevalence of this disease in the UK, the study concluded that gastroenteritis annually costs EUR 13 million to society as a whole.

Dasgupta (2004) assessed the value of damages from contaminated water supplies in India, to derive total costs of illness. The average cost of treatment of waterborne diseases was estimated at EUR 8 for a child, EUR 5 for an adult and EUR 7 for an elderly person. Wage loss due to illness was estimated to be EUR 3.5 per household. This led to an annual cost of illness of EUR 108 per household. Given that there were 150 748 households in urban Delhi, this led to an annual total cost of EUR 16.28 million for the whole population of Delhi.

Cognitive and developmental delays

Grosse *et al.* (2002) evaluated the economic benefits of reducing children's exposure to lead in the US. Discounted lifetime earnings were estimated at EUR 646 000 for each 2 year-old (using a 3% discount rate). Given that there were approximately 3.8 million 2 year-old children in the US in 2002, the total benefits of reducing childhood lead exposure ranged between EUR 98 billion and EUR 285 billion.

Korfmacher (2003) assessed the benefits of eliminating lead poisoning in children in New York State (US). Healthcare benefits (*i.e.* direct treatment) were estimated to be EUR 2.7 million and increased potential earnings EUR 693 million (applying a 3% discount rate). Although the healthcare costs of lead poisoning in New York State were quite significant, these values probably under-estimated the true costs because some of the most costly impacts of lead (*e.g.* osteoporosis, hypertension, stroke and neonatal mortality) could not be quantified when the study was undertaken.

Similarly, Stefanak *et al.* (2005) evaluated the costs of childhood lead poisoning in Mahoning County, Ohio (US). Screening and treatment costs were estimated to be almost EUR 112 000 per child. They also assessed the future costs for the cohort of lead-poisoned children (with blood level greater than 10 mg/dl) age 12-71 months in 2002 to be EUR 1.4 million, using a 3% discount rate.

Trasande *et al.* (2005) calculated the cost of illness of exposure to methylmercury in the US, with a particular focus on the impacts on the developing brain. They estimated that lost productivity associated with methylmercury toxicity cost EUR 7.8 billion per year, applying a 3% discount rate. Of this total, the study concluded that EUR 1.2 billion was attributable to mercury emissions from US power plants.

Miller *et al.* (2006) estimated the costs of early-life exposure to ETS and developmental delays, in New York City (US). They estimated the costs of early intervention services per year due to ETS to be EUR 88 million per year for all New York City births, based on a 3% annual discount rate.

Nevin *et al.* (2008) estimated the monetary benefits of preventing childhood lead poisoning in the US by replacing old windows with lead-safe windows. The benefits per child from improved lifetime earnings were estimated to be EUR 18 934 for pre-1940 housing and EUR 7 758 for 1940-59 housing. This analysis did not take into account potential ancillary health benefits associated with the reduction of lead exposure in children (*e.g.* avoided medical costs of treatment and avoided special education in later life associated with attention deficit hyperactivity disorders).

Multiple health endpoints

Carabin *et al.* (1999) described the costs of illness of three common infections in toddlers: colds, diarrhoea and vomiting. They followed a cohort of 273 toddlers attending day care centres in Quebec, Canada. Total direct costs were estimated to be almost EUR 73 per child, while indirect costs were estimated to be EUR 129.

Landrigan *et al.* (2002) estimated the costs of paediatric environment-related diseases in the US. They focused on four major childhood diseases: lead poisoning (EAF = 100%), asthma (EAF = 30% – range: 10-35%), childhood cancers (EAF = 2, 5 and 10%) and neurobehavioral disorders (EAF = 10% – range: 5-20%). The present value was calculated using average annual earnings for full-time and part-time employees, labour force participation rates, estimates of annual home production loss, and a real discount rate of 3%. They estimated that:

- lead poisoning costs were EUR 43.3 billion;
- asthma costs were EUR 1.8 billion;
- cancer costs were EUR 0.27 billion; and
- neurobehavioural disorders costs were EUR 8.2 billion.

Total annual costs were estimated to be EUR 49 billion, which represented 2.8% of total US health care costs at that time.

Massey and Ackerman (2003) estimated the costs associated with five major environment-related health problems that significantly affect children: cancer, asthma, lead poisoning, neurobehavioral disorders and birth defects. Total costs for one year were estimated to be EUR 3 billion. When applying an EAF, their estimates ranged from EUR 0.5 billion to EUR 1.4 billion per year for Massachusetts alone. Discounting of nonmonetary future events was not included in the calculations.

Davies (2005) assessed the cost of environmental diseases that affect children in Washington State (US), also based on the EAF approach. Again, the discount rate used in the calculations was not specified. Cost estimates are presented in Table 1.A1.1. The total costs of these childhood diseases were estimated at EUR 1 675 million, of which EUR 1 429 million were indirect costs.

Table 1.A1.1. **Costs of Selected Childhood Diseases in Washington State**

(2006 EUR million)

Disease	Cost estimate
Child asthma	EUR 44
Childhood cancer	EUR 10-14*
Lead exposure	EUR 1340
Birth defects	EUR 3.8-5
Neurobehavioral disorders	EUR 64.7-273*

* Different methods were used to estimate these costs, hence a range of values is provided.

Source: Davies (2005).

Hutchings and Rushton (2007) evaluated the economic burden of childhood diseases in Europe. Based upon the EAF approach, they estimated the costs of illness associated with cancer, asthma, neurodevelopmental disorders and lead poisoning. Total costs were estimated to be above EUR 16 billion with EUR 174 million for cancer, EUR 3 billion for asthma, EUR 3 billion for neurodevelopmental disorders, and EUR 9.9 billion for lead poisoning. The authors highlighted that direct costs represented the major share of the total costs associated with childhood cancer and asthma. All costs except for lead poisoning were discounted at an annual rate of 3%.

The main findings from this review of COI studies are:

- Estimated productivity losses associated with childhood illnesses are generally greater than direct medical costs.
- Diseases presenting cognitive/developmental delays and/or neurobehavioural disorders generate extremely high costs, in particular with respect to other childhood diseases, such as cancers and asthma.
- The financial costs (direct and indirect costs) of childhood illnesses are very large, although even these do not account for intangible aspects, suggesting a potential under-estimation of the true costs.

Willingness to pay studies

As mentioned above, cost of illness values represent only the financial costs of a disease, and do not include intangible costs, such as pain and suffering, or the inability to enjoy leisure activities. Willingness to pay (WTP) studies provide values that account for intangible aspects of disease, because they measure individual preferences, which include all sources of utility and causes of disutility to the individual.

WTP values to avoid a given risk can be obtained either from revealed preference studies (based on observed purchasing behaviour) or from stated preference studies (based on hypothetical behaviour). Revealed preference studies use indirect methods to value the monetary amount required to accept a variation in the risk level. They assume that individuals reveal their preferences through consumption and expenditures which are related to health impacts. This is done by using information available on different markets, such as the labour market, the housing market, and the safety products market. The "hedonic" method and the "averting behaviour" method are revealed preference techniques.

Stated preference approaches estimate the *ex ante* valuation of a variation in individual welfare related to the variation of the status of individuals exposed to a particular health risk. These studies present people with a hypothetical scenario (via telephone, postal or individual survey), and ask them about their maximum WTP to compensate for a variation in their well-being. These studies ultimately provide estimates of WTP values for a reduction in health risk, or analogously, willingness-to-accept (WTA) values for an increase of health risk.[5]

Stated-preferences techniques (the contingent valuation method, the conjoint analysis methods[6]) can be applied to value a reduction in mortality risk. The WTP value obtained (*i.e.* the WTP to reduce mortality probability) is then used to derive the value of a statistical life (VSL).[7] However, stated-preferences techniques are not specific to mortality risk valuation, and can be also used to value morbidity endpoints.

Only a few studies have so far dealt with the valuation of reducing health risks for children, and most of these were not specific to the environmental context. However, since most of the studies are valuing personal safety questions, they still contribute to a better understanding of the value parents place on children's health. In addition, some of the health outcomes valued could also be associated with environmental degradation (even though they were not stated as such in the associated surveys), and corresponding WTP could therefore be used in environmental policy-making. A review of the limited evidence available is proposed below.[8]

Mortality studies

Joyce *et al.* (1989) measured the impact of air pollution on neonatal mortality rates, using a health production function, and focussing on the WTP of mothers to reduce air pollution levels. The marginal WTP for prenatal care ranged between EUR 2 and EUR 7, depending on individual characteristics. The marginal WTP for neonatal care was higher, between EUR 29 and EUR 198, suggesting a higher WTP for younger infants. From these WTP values, Dickie and Nestor (1998) derived estimates of infants VSL, ranging between EUR 77 000 and EUR 2.6 million.

Carlin and Sandy (1991) calculated the implicit value of a young child's life as revealed by the decisions of the mother about using a child car safety seat. The data came from a survey implemented in 1985 and were used in a utility maximisation approach. The value of a child's life was derived from the mother's probability of purchasing and properly using a car seat. Fatality risk reductions were considered, along with the time and money costs of raising a child to the age of 18. The VSL of a child under the age of five was estimated to be EUR 942 000.

Blomquist, Miller and Levy (1996) estimated the implied values of reducing fatal and non-fatal injuries risks for different road user populations: adults, children and motorcyclists. They incorporated time and disutility costs associated with car seat belt and motorcycle helmet use. The data were obtained from a 1983 survey, which has included parents with children under the age of five. The VSL for a child ranged between EUR 5.16 million and EUR 9.22 million, while the value to reduce child non-fatal injury was EUR 218 000. These values compared with equivalent values for adults (VSL of EUR 3.47 million and EUR 99 000 for a non-fatal injury) and for motorcyclists (VSL of EUR 2.38 million and EUR 75 000 for a non-fatal injury).

Mount, Weng, Schulze and Chestnut (2001) examined family automobile purchases, to estimate the amount of money spent on safety, and then to derive the VSL of different age groups (children, adults and the retired). They applied a hedonic price function on data from a 1995 survey (aggregated data). Central estimates derived suggested that children had a VSL of EUR 11.6 million, while adults had a VSL of EUR 11.3 million and the retired persons had a VSL of EUR 8.2 million.

Jenkins, Owens and Wiggins (2001) estimated the parental values of reduced fatality risk to children, by examining the market for child bicycle helmets. The value of reducing mortality risk was computed for 5-9 and 10-14 year-old children. Data from a survey were used in a utility maximisation model. The estimated VSL for helmet users varied between EUR 1.6 million and EUR 3.4 million (5 to 9 years); and EUR 1.4 million and EUR 3.3 million (10 to 14 years), according to different assumptions.

Takeuchi *et al.* (2006) conducted a contingent valuation survey in Japan, to estimate the parental WTP to reduce child mortality. The median WTP to reduce annual child mortality by 1% was 7 500 yen, while the median WTP to reduce annual child mortality by 5% was 11 000 yen. Based on the first value, they derived the VSL for a child of 980 million yen.

Morbidity studies

Viscusi, Magat and Huber (1987) implemented a contingent valuation survey in the US to estimate the individual WTP to prevent the risk of injury associated with two injuries: poisoning from insecticide and poisoning from toilet bowl cleaner. Injuries were proposed, depending upon whether the respondent had young children or not. WTP estimates are presented in Table 1.A1.2.

Table 1.A1.2. **WTP to Prevent Injuries Associated with Pesticides**
(2006 EUR)

Reduction of risks from insecticide:
● Skin poisoning: EUR 1 101 (individuals without young children)
● Inhalation: EUR 1 276 (both subsamples)
● Child poisoning: EUR 2 555 (individuals with young children)
Reduction of risks from the toilet bowl cleaner:
● Eye burns: EUR 545 (individuals without young children)
● Chloramine gassings: EUR 815 (both subsamples)
● Child poisoning: EUR 902 (individuals with young children)

Source: Viscusi *et al.* (1987).

Reductions of risks from insecticides were therefore valued more than injuries from toilet bowl cleaners. In particular, reducing child poisoning risk from insecticide products was valued almost three times more than reducing child poisoning from toilet bowl cleaner, which was presented in the survey as less risky than the former. Moreover, the WTPs to reduce risks to children were greater than the WTPs to reduce similar risks to adults.

Agee and Crocker (1996) estimated the benefits associated with children morbidity risks related to a low-level lead exposure. The study inferred the parents' WTP to reduce the risk of neurological impairments for children due to exposure to lead, both from parents who chose chelation as treatment for their child and from parents who did not choose chelation as treatment. The WTP of parents who chose chelation was EUR 138 per child, while the WTP of parents who did not choose chelation was EUR 14 per child. The overall mean WTP was estimated to be EUR 21 per child. Aggregated benefits for a 1% reduction in child body lead burden (over the number of US metropolitan households in 1984) ranged from EUR 216 million to EUR 2 billion. The study also noted that the parental *ex ante* WTP for a 1% reduction in child body lead burden exceeded the estimated cost-of-illness associated with the same reduction.

Liu *et al.* (2000) carried out a contingent valuation study in Taiwan to estimate a mother's WTP for preventing herself and her child from getting a cold. The mean WTP to prevent the child from getting a cold was EUR 51, while the mean WTP to prevent the mother from getting a cold was EUR 33. The mother's WTP to prevent her child from suffering a cold was approximately twice as large as her WTP to prevent herself from getting a cold of comparable duration and severity.

Dickie and Gerking (2001) implemented a contingent valuation survey to estimate the parental WTP to reduce skin cancer from solar radiation exposure, for their children and for themselves. Both melanoma and non-melanoma skin cancer risks were considered. WTP for a 1% point reduction in non-melanoma skin cancer risk was estimated at EUR 2.84 for the child and EUR 1.15 for the parent, again showing that parents were willing to pay more to reduce non-melanoma skin cancer risks to their children than to themselves.

Agee and Crocker (2001) estimated the annual WTP to increase their own and children's health, as well as the parental WTP to reduce their child's daily exposure to environmental tobacco smoke. The study focused on smoking parents and analysed parents' consumption of tobacco products and their assessment of their children's exposure to environmental tobacco smoke. The WTP for a 1% reduction in child exposure to tobacco smoke was EUR 9. The WTP for a 10% improvement in child health status was EUR 404, while the same WTP for the parent was EUR 222. These results suggested that parents valued their children's health twice as much as their own health.

Dickie and Messman (2004) implemented a stated-preference study to evaluate the parents' WTP to avoid acute illnesses. They found that WTP for avoiding episodes was less for parents than for children (Table 1.A1.3).

Table 1.A1.3. **WTP to Avoid Acute Illnesses**

(2006 EUR)

Mean WTP to avoid one symptom for one day: EUR 45
Mean WTP to avoid seven days of one symptom: ● For the child: EUR 134-313 ● For the parent: EUR 89-147
Mean WTP to avoid one-week incident of acute bronchitis: ● For the child: EUR 357 ● For the parent: EUR 179

Source: Dickie and Messman (2004).

Accounting for the endogeneity of behavioural responses to illness (*e.g.* use of medical care and absence from work or school), Dickie and Brent (2002) estimated that the mean WTP to avoid one day of symptom was EUR 84 for children and EUR 31 for adults.

Maguire, Owens and Simon (2004) measured the value of reducing babies' exposures to pesticide residues. They used hedonic methods and analysed data from observed consumption behaviour in the baby food market. They inferred the consumers' premium for organic baby food, and found that parents were willing to pay EUR 0.09-013 per jar more for organic food than for conventional varieties, *i.e.* an annual price premium of EUR 66 (600 jars × 0.11). This is approximately 16-27% more than traditional baby food. This premium could be interpreted (at least in part) as a desire to avoid pesticide residues in baby food.

Amin and Khondoker (2004) assessed the parental WTP to avoid an episode of diarrhoea in a contingent valuation survey in India, focusing on children between 5 and 7 years. The median WTP for male children was EUR 0.64, whereas the median WTP for female children was EUR 0.48, *i.e.* 34% lower than the WTP for male children.

Braun Kohlová and Scasny (2006) implemented a contingent valuation survey in the Czech Republic to estimate the WTP to reduce selected respiratory diseases: severe and mild acute bronchitis, acute laryngitis and acute asthma. They focused on children living in Teplice and Prachatice (Czech Republic). The results suggested that WTP varies according to severity, not according to duration. The WTP for an asthma attack lasting for one day (EUR 43) is significantly higher than the WTP for a mild bronchitis lasting for five days (EUR 38), and the WTP for a laryngitis requiring three days of hospitalisation plus five days at home (EUR 64) is higher than for a severe bronchitis lasting for ten days (EUR 39); the pair-wise differences except for severe bronchitis and asthma attack were significant at the 0.05 level. For comparison, the mean WTP for reducing mild bronchitis in adult was EUR 21, that implies a marginal rate of substitution between child and adult adverse health outcome of 1.85.

Mansfield *et al.* (2006) observed the averting behaviour of parents to protect their children from exposure to ozone. They based their analysis on a sample of 231 children, between 2 and 12 years old, living in the US. The mean parental WTP for a one-day reduction in restricted time outdoors was EUR 31.

Mead and Brajer (2005) which evaluated the aggregated health benefits to children of reducing air pollution in China. They used both COI and WTP values. In addition, when no child-specific value was available, they used adult values instead (Table 1.A1.4).

Table 1.A1.4. **Health Costs of Air Pollution in China**
(2006 EUR)

Health outcomes	Average total costs
Cold	EUR 24 million
Acute bronchitis	EUR 210 million
Chronic bronchitis	EUR 446 million
Asthma	EUR 87.5 million
Asthma-related hospital admission	EUR 471 million
Paediatric outpatient visit	EUR 55 million
Emergency room visit	EUR 8 million
Total	**EUR 1.3 billion**

Source: Mead and Brajer (2005).

Valuation Studies for both Children and Adults

Some empirical studies have shown that people believe that, *ceteris paribus*, a programme that protects young people is better than one which protects old people. Examples include Lewis and Charny (1989), where people stated they preferred saving the life of a 35-year-old rather than the life of a 60-year-old.[9] Tsuchiya *et al.* (2003) offer three reasons for favouring the young over the old: i) the young have longer life expectancies; ii) the young are more productive; and iii) the old have had a greater share of expected life years. That is, other things being equal, a given health programme should favour the young, either because it delivers greater benefits due to the difference in time/age existing between young and old populations (larger benefits for young adults given their larger expected remaining lifespan), or because young people have lived less life and therefore "deserve" the health improvement more than older people.

All of these arguments are, of course, equally valid when comparing adult and child values. However, in this case there are likely to be other factors (*e.g.* parental altruism, risk perceptions), which play a role in explaining any apparent differences in preferences. For instance, in the aforementioned person trade-off study by Lewis and Charny (1989), in addition to the differences between age groups amongst adults, they found even stronger preference for risk reductions for 5 year-olds, relative to 70 year-olds. They also found a slight preference for risk reductions for 8 year-olds over 2 year-olds.

Other economic studies have estimated the WTP to reduce health risks associated with different causes for children and adults. Although the evidence is mixed, most of the studies concluded that the WTP to reduce mortality risks to children was greater than the WTP to reduce similar risks to adults. For instance, Liu *et al.* (2000) estimated the WTP to avoid an episode of "cold". The mothers' WTP to prevent their child from having a cold was almost twice the WTP for themselves. Based on a study of automobile safety, Mount *et al.* (2001) estimated the VSL of different age groups (children, adults and the retired). They found that the VSL of a child was quite similar or slightly larger than that of adults but greater than that of an elderly person. Jenkins *et al.* (2001) estimated the VSL for a child according to different age categories: ages 5 to 9 and ages 10 to 14. The results showed that the VSL for a 5- to 9 years-old is higher than the VSL for a 10- to 14 years-old, suggesting a greater risk aversion towards the youngest.

Blomquist *et al.* (1996) have estimated the implied values of reducing fatal and non-fatal injuries risks for different road user populations: adults, children and motorcyclists. They found that the VSL for a child is greater than the VSL for an adult, reflecting the idea that parents value the life of their children more than their own. Liu *et al.* (2000) evaluated a mother's WTP for preventing herself and her child from a minor disease (a cold). They found that the mother's WTP for her child is approximately twice as large as her WTP to prevent herself from getting a cold of comparable duration and severity.

Similarly, Dickie and Ulery (2002) calculated parental WTP to avoid acute illnesses and found that WTP for avoiding episodes was less for parents than for children. The value parents were willing to pay to avoid acute illnesses in their children was about twice the value for themselves. Dickie and Gerking (2001) estimate the parental WTP to reduce skin cancer from solar radiation exposure, both for their children and for themselves. The results showed that parents are willing to pay twice as much to reduce non-melanoma skin cancer risks to their children than to themselves.

Agee and Crocker (2001) estimated the annual WTP to increase "own" and "children" health services, as well as the parental WTP to reduce their child's daily exposure to environmental tobacco smoke. They found that parents valued their children's health twice as much as their own health. More recently, Agee and Crocker (2007) found that smoking mothers were willing to pay USD 144 to improve their own health by 25%, while they were willing to pay USD 262 for a comparable improvement in their child's health.

Viscusi *et al.* (1987) estimated the WTP to prevent the risk of injury associated with household pesticides. The results showed that respondents were willing to pay almost three times as much (on average) to avoid child poisonings from insecticides than to avoid poisoning from toilet bowl cleaner. They also found that the WTP to reduce risks to children was greater than the WTP to reduce any other risks considered in the survey. Similar results were found in a study on hazardous household cleaning products carried out by Evans and Viscusi (1991). Higher WTP values were found for the reduction of child poisoning risks, as respondents with children were willing to pay on average USD 1.31 more per bottle for the reduction of child poisonings, in comparison to the reduction of pesticide inhalations.

Hammitt and Haninger (2010) estimated the VSL for children and adults in the United States based on WTP to reduce fatal-disease risks associated with exposure to pesticides through food consumption. The results indicated that WTP to reduce risk to one's child is systematically greater (USD 12-USD 15 million) than the WTP (USD 6-USD 10 million) to reduce one's own risk. The study also provides a rich body of evidence on issues such as latency, context, and the effect of the assumed household allocation.

Discussion

Overall, this literature reviewed for the VERHI project suggests that:

- WTP values in general exceed corresponding cost of illness values, suggesting the importance of intangible aspects of illness over direct and indirect costs of illness;[10]
- values for reducing child *mortality* are in general greater than values for *morbidity* outcomes; and
- parents are in general willing to pay more to reduce health risks to their children than to themselves.

Notes

1. Because it is difficult to value child health endpoints associated with parental exposure to environmental hazards (since it can be considered as ancillary effect of parent's own health effects associated with that environmental exposure), studies that refer to children's health outcomes associated with parental exposure during gestation (*i.e.* prenatal exposure) were not included in the review, and the focus was placed on direct post-natal exposures to environmental hazards.
2. The "odds ratio" (OR) represents the risk of occurrence of a health endpoint in one group, divided by the risk of it occurring in another group.
3. The "relative risk" (RR) is the risk of an event occurring (or of developing a disease), relative to exposure. Relative risk is a *ratio* of the *probability* of the event occurring in the exposed (PE) group *versus* the non-exposed group (PNE): RR = PE/PNE.

4. For comparison purposes, all reported cost figures have been converted into 2006 EUR, using PPP exchange rates, unless specified otherwise.
5. The notions of WTP and WTA are firmly grounded in the theory of welfare economics and correspond to notions of "compensating" and "equivalent" variations. WTP and WTA should not, according to theory, diverge very much. In practice, they do appear to diverge, often substantially – with WTA being greater than WTP. Hence the choice of WTP or WTA may be of importance when conducting CBA. For more details see OECD (2006b).
6. Conjoint analysis, also known as choice modelling, gathers a number of different techniques: conjoint choice experiment, contingent rating, contingent ranking, and paired comparisons.
7. VSL is also known as the "value of a prevented fatality".
8. For comparison purposes, all cost figures from studies presented in this report have been converted into 2006 EUR, using PPP exchange rates, unless specified otherwise.
9. In addition, Cropper *et al.* (1994) applied a "person trade-off approach" to compare saving lives at different ages. They found that saving one 30-year-old is perceived to be equivalent to saving eleven 60-year-olds. Johannesson and Johansson (1997) asked a sample of individuals about their choice between saving lives now and in the future. They found that saving five 50-year-olds or thirty-four 70-year-olds is judged equivalent to saving one 30-year-old. In addition, this study revealed that the age of the respondent has no effect on his/her choice, which means that both young and old adults give priority to saving the life of the youngest. Some studies provide evidence on a "senior death discount" (*i.e.* the VSL for the elderly should be lower than that of adults below 70, because older people appear to attach a lower WTP to reduce mortality risk). For instance, Tsuge *et al.* (2005) implemented a survey in Japan and found that the persons aged above 70 tend to have a lower WTP for the same risk reduction. This would imply a lower VSL for seniors. Krupnick (2007) undertook a review of 26 "stated preference" surveys, to assess the "senior death discount". His qualitative meta-analysis provided mixed results, because only half the studies supported the existence of a "senior discount" effect.
10. Stieb *et al.* (2002) and Rabl (2004) showed that intangible aspects represent a significant percentage of total health costs up to 90% for non-fatal cancers.

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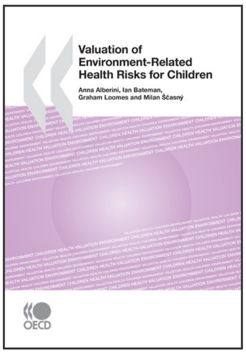

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