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USE OF EVALUATION TOOLS IN POLICY-MAKING AND HEALTH IMPLICATIONS FOR CHILDREN

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FOREWORD

This report was prepared in the context of the OECD Project on the "Valuation of Environment-related Health Impacts, with a Particular Focus on Children" (VERHI). This three-year project (2006-2008), funded by the European Commission under the 6th Framework Programme of Research (contract number SSPE-CT-2005-006529), seeks to improve the incorporation of environment-related health impacts on children in policy-making.

The OECD Secretariat was asked to provide two inputs related to the preparatory methodological work to be carried out during the first year of implementation of this Project. The first input consists of an overview of the use of economic evaluation tools in environmental and health policy-making in OECD member countries. This constitutes the first part of the following report.

The second input was originally supposed to report on the incorporation of environment-related health impacts in policy-making, with special focus on implications for children. However, due to limited evidence, a review of current environmental regulations is proposed here instead, in order to determine whether children's specific vulnerability to environmental hazards is actually being integrated in environmental policy-making. This review is provided in the second part of the following report.

USE OF EVALUATION TOOLS IN POLICY-MAKING AND HEALTH IMPLICATIONS FOR CHILDREN

1. Introduction

Public decision makers require estimates of the diverse consequences of policy interventions (*e.g.* social, economic, and financial impacts) in order to implement new policies or revise current policies and programmes. For that purpose, economic valuation of the costs and benefits of policies has become a central tool in many countries, in particular in the OECD region. Estimates of the full range of these impacts can be used for at least three purposes:

- Measuring the effectiveness of environmental policy and social programmes;
- Prioritising environmental policies and target groups; and,
- Setting optimal targets to improve environmental policy design.

In addition, empirical studies have stressed the need for better understanding of how to value the environmental health risks faced by people in general, and by children in particular. Reliable estimates of the environmental and health impacts of environmental policies are important as part of sound policy evaluations in order to help policymakers assess the economic efficiency of policies intended to reduce these adverse health impacts.

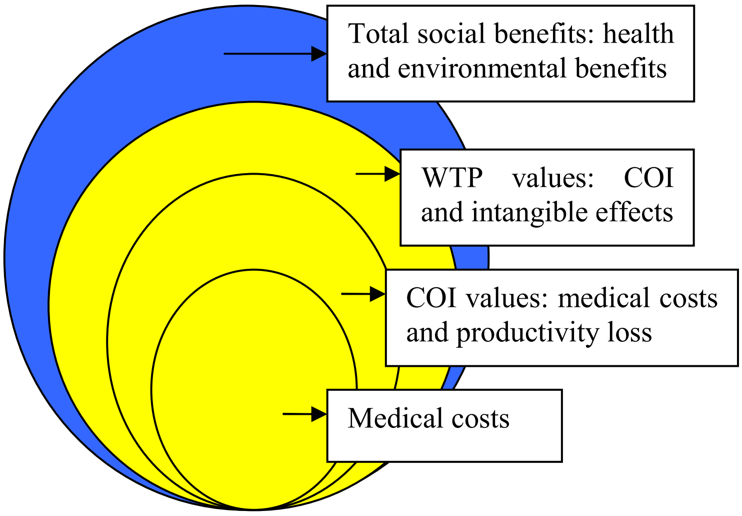
Many economic tools can be used to help policymakers determine the best policy options. Some of these tools involve quantification and/or monetisation of the likely impacts of environmental policy changes. All of them present advantages and disadvantages, and the choice of a specific tool will depend upon the purposes of the analysis required.

Environmental policies may lead to significant health benefits. More specifically, in some areas of environmental regulation (*e.g.* air pollution and water pollution), health benefits constitute the most important share of total benefits, including reduced risks of environment-related mortality and morbidity[1]. As such, appraisals of environmental policies often focus on the valuation of related health impacts. The methods used for the valuation of benefits to human health have raised many debates over the past years, but are not addressed directly here. However, monetisation of health benefits is not straightforward and different measures can be used to this end.

Health benefits are usually expressed in two forms: either as values of avoided costs of disease (*i.e.* costs of illness (COI) which include medical costs and productivity loss associated with illness) or as willingness-to-pay (WTP) values to avoid a given disease or to reduce a specific risk. Figure 1 represents the different types of benefits and benefit measures of environmental policies.

[1] It is important to note that benefits related to reduced mortality and chronic morbidity tend to dominate, accounting in general for more than 90% of total benefits (see for example US EPA, 1997). However, other impacts *(i.e.* not health-related) should not be disregarded.

Figure 1 – Benefits of environmental policies



The WTP to improve environmental quality can be defined as the maximum amount someone is willing to pay for the improvement of environmental quality while his/her level of welfare has not changed (*i.e.* the individual is just as well off with the improvement as without it). The methodologies underlying the estimation of COI and WTP differ widely: the COI is usually estimated *ex post*, whereas the WTP is generally estimated *ex ante* (for a more technical discussion, see OECD, 2006b). Moreover, COI reflects the medical costs of treatment, omitting the out-of-pocket expenses faced by the individual.

As WTP values encompass direct (medical costs) and indirect costs of illness (*i.e.* lost productivity) as well as intangible aspects (*e.g.* pain and suffering, impossibility of leisure or domestic activities when sick, etc.), they offer a better representation of individual preferences over a chance of illness *ex ante*. WTP estimates are considered as the most complete indicator of the value of illness and therefore as the most reliable estimate to be used in policy-making.

Finally, WTP estimates are preferred over COI values because of some methodological problems associated with the COI approach. For instance, since the COI method relies on the human capital approach, some categories of populations (*e.g.* unemployed, housewives, pensioners, students) are not considered as "workers" since they do not "produce" anything in the market. Moreover, COI estimates approximate a market value of reduced health risk under two conditions: (1) the direct costs of morbidity must reflect the economic value of goods and services used to treat illness; and (2) a person's earnings must reflect the economic value of lost work time, productivity and leisure time. Because of the distortions that exist on the labour and the medical markets, these assumptions do not routinely hold.

WTP to reduce mortality risks can be used to derive the value of a change in the risk of morbidity or mortality (the value of a statistical life (VSL)). However, when WTP estimates are lacking, COI estimates associated with a specific illness can constitute a good starting point, although they only account for the direct and indirect effects of the disease on people.

Valuing the health effects of environmental policies is in general integrated into a risk impact assessment, intending to evaluate the potential impacts of a given environmental policy. Based upon epidemiological, toxicological and clinical studies, policymakers determine acceptable levels of exposure to sources of environmental pollution. Disparities and similarities in the various approaches currently being applied in OECD countries are highlighted below.

Estimating the value of health impacts becomes even more complicated when the case of children is considered. Indeed, many recent epidemiological surveys have highlighted the specific vulnerability of children to environmental hazards. While mortality rates for children are, of course, much lower than for adults, there are certain causes of deaths for which children are **relatively** more vulnerable (c.f. the figures indicated in bold in Tables 1 and 2)[2]. As shown in Tables 1 and 2, the proportion of deaths in children compared with adults is greater for diseases of the nervous system, diseases of the respiratory system (more particularly from asthma), and from endocrine disorders (*i.e.* diseases that could result from exposure to specific environmental risks[3]), than for some other causes of death.

However, most environmental policies currently in place are based upon information related to adult populations and do not account for risk differences (*i.e.* exposure and susceptibility differences) between adults and children. Based on recent epidemiological and economic work undertaken in this area (see OECD, 2006c), one important observation is that children should not be considered as “adults” in environmental policymaking. Not considering the specific vulnerability of children to environmental hazards may lead to inefficient policies, and thus, to important losses in social welfare.

[2] Tables 1 and 2 provide standardised rates of mortality for selected causes for which some association with environmental factors has been shown.

[3] Morbidity is as relevant as mortality when considering the impact of the environment on health. However, there does not exist (to our knowledge) a global database providing the number of children affected by environment-related illnesses in the OECD countries. Such data is broadly available for adults.

Table 1. Standardised rates of mortality associated with selected causes (per 100,000), in European OECD countries

	Asthma			Pneumonia			Diseases of respiratory system			Diseases of circulatory system		
Countries	0-14 years	25-64 years	Ratio	5-14 years	25-69 years	Ratio	0-14 years	25-69 years	Ratio	5-14 years	25-69 years	Ratio
Austria	0.13	1.05	0.124	0	1.07	0	0.68	8.17	0.083	0	71.04	0
Czech Republic	0.1	0.86	0.116	0.51	6.8	0.075	2.3	15.92	0.144	0.68	134.5	0.005
Finland	0	0.63	0.000	0.16	6.57	0.024	0.36	13.26	0.027	0.46	91.03	0.005
France	0.07	0.97	0.072	0.07	1.94	0.036	0.74	8.29	0.089	0.54	50.08	0.011
Germany	0.1	1.29	0.078	0.28	2.9	0.097	1.02	11.46	0.089	0.6	78.86	0.008
Greece	0	0.15	0.000	0.18	1.4	0.129	1.45	12.19	0.119	0.63	86.63	0.007
Hungary	0	1.31	0.000	0.42	3.21	0.131	2.14	20.56	0.104	1.24	203.36	0.006
Iceland	0	0	0.000	0	0.93	0	0	9.3	0	0	63.35	0
Ireland	0	0.91	0.000	0.19	7.99	0.024	1.51	18.62	0.081	0.18	86.14	0.002
Luxembourg	0	2.82	0.000	0	2.54	0	0	16.95	0	0	80.49	0
Netherlands	0.1	0.21	0.476	0.25	2.97	0.084	0.96	10.45	0.092	0.96	70.3	0.014
Norway	0.12	1.57	0.076	0.16	1.65	0.097	0.7	10.17	0.069	0.32	58.54	0.005
Poland	0.08	1.24	0.065	0.51	5.17	0.099	1.95	13.5	0.144	0.93	157.99	0.006
Portugal	0.07	0.77	0.091	0.37	5.44	0.068	2.5	15.82	0.158	1.53	75.98	0.020
Slovakia	0	0.65	0.000	0.47	9.67	0.049	6.21	19.39	0.320	1.5	173.05	0.009
Spain	0.09	0.59	0.153	0.13	2.64	0.049	1.32	13.24	0.100	0.95	56.1	0.017
Sweden	0	0.77	0.000	0	3.02	0	0.66	9.29	0.071	0.33	61.92	0.005
Switzerland	0	0.4	0.000	0	1.71	0	0.65	6.38	0.102	0.23	46.64	0.005
United Kingdom	0.26	1.41	0.184	0.18	5.63	0.032	1.89	18.88	0.100	0.61	84.82	0.007

Source: WHO Europe mortality database, http://www.euro.who.int/eprise/main/WHO/InformationSources/Data/20011017_1

Table 2. Standardised rates of mortality associated with selected causes (per 100,000), in European OECD countries

	Nervous system disorders			Endocrine disorders			Lung cancer			Malignant neoplasm			Skin cancer		
Countries	0-14 years	25-69 years	Ratio	0-14 years	25-69 years	Ratio	0-14 years	25-69 years	Ratio	0-14 years	25-69 years	Ratio	0-14 years	25-69 years	Ratio
Austria	1.98	6.05	0.327	0.88	11.04	0.080	0	28.42	0	2.26	117.04	0.019	0	2.52	0
Czech Republic	4.48	9.79	0.458	0.59	4.58	0.129	0.05	38.62	0.001	3.36	164.5	0.020	0	2.5	0
Finland	1.14	9.91	0.115	0.97	5.31	0.183	0	17.1	0	3.2	90.93	0.035	0	1.91	0
France	2.54	8.81	0.288	1.36	6.84	0.199	0.01	33.7	0.000	2.78	138.91	0.020	0.01	1.89	0.005
Germany	1.96	6.68	0.293	0.78	6.57	0.119	0.01	27.91	0.000	2.32	122.35	0.019	0	1.8	0
Greece	1.28	4.92	0.260	1.16	2.74	0.423	0.07	28.42	0.002	3.29	103.21	0.032	0	0.91	0
Hungary	3.02	9.44	0.320	1.37	9.68	0.142	0.08	68.21	0.001	4.98	231.77	0.021	0	2.69	0
Iceland	5.23	10.39	0.503	0	1.99	0	0	24.73	0	4.5	102.69	0.044	0	2.36	0
Ireland	5.53	8.22	0.673	0.93	4.56	0.204	0	23.58	0	1.93	118.59	0.016	0	2.28	0
Luxembourg	2.38	4.98	0.478	2.63	4.24	0.620	0	24.07	0	0	113.73	0	0	3.71	0
Netherlands	2.73	6.52	0.419	1.47	7.8	0.188	0	32.19	0	2.46	125.93	0.020	0	3.27	0
Norway	3.08	7.75	0.397	1.9	6.78	0.280	0	24.2	0	3.21	110.75	0.029	0	4.03	0
Poland	2.66	7.46	0.357	0.79	6.68	0.118	0.03	47.63	0.001	3.96	172.66	0.023	0.02	2.29	0.009
Portugal	3.45	6.29	0.548	1.16	10	0.116	0	20.32	0	4.39	120.42	0.036	0	1.04	0
Slovakia	6.86	9.5	0.722	0.46	7.67	0.060	0	32.48	0	3.64	171.1	0.021	0	2.84	0
Spain	3.09	5.82	0.531	0.89	4.79	0.186	0.02	29.44	0.001	3.28	120.65	0.027	0.02	1.28	0.016
Sweden	1.59	6.45	0.247	0.79	6.16	0.128	0	17.37	0	3.16	92.62	0.034	0	2.96	0
Switzerland	1.64	6.42	0.255	1.12	5.01	0.224	0	21.96	0	2.31	100.19	0.023	0	2.43	0
United Kingdom	2.97	8.95	0.332	1.51	4.85	0.311	0	25.56	0	3.03	119.68	0.025	0.02	2.2	0.009

Source: WHO Europe mortality database, http://www.euro.who.int/eprise/main/WHO/InformationSources/Data/20011017_1

This paper focuses on priority-setting practices in OECD countries in the environment and health domains. The first section therefore proposes a brief description of the most commonly used decision-making tools in the two areas. This description is based on a recent OECD survey as well as a review of secondary literature. The specific case of children's environmental health is then considered. An examination of how children's specific vulnerability to environmental hazards is accounted for in policy-making leads to the conclusion that children-specific values are not used yet in policy-making, mainly due to scarce data and valuation challenges. In order to better understand how children are currently protected from environmental risks, a review of the environmental legislation targeting children introduced in OECD countries follows. A summary of the discussions appears at the end of the paper.

2. Economic evaluation

The purpose of this Section is to review the economic evaluation tools that are currently being used in environmental and in health policymaking. However, before dealing with these specific policy areas, it could be useful to briefly introduce the different instruments that are more generally in place to assess the effectiveness of public policy in OECD countries.

2.1. Economic evaluation of public policies in OECD countries

In order to assess the use of economic evaluation approaches in policymaking, we draw upon the findings from a recent OECD survey (OECD, 2004a)[4] as well as a review of secondary literature.

Based on available information, regulatory impact assessment (RIA) appears to be the evaluation tool most commonly used in OECD countries, but practice in this area appears to be quite mixed. Many differences can be observed between countries, more specifically related to the definition and use of the RIA approach. For instance, many OECD countries require a RIA for major regulations. However, in some countries RIA is required in all cases, while it is required only in selected cases in other countries. As another example, even though RIA can be used for both *ex ante* and *ex post* evaluation of policies, it is mainly being used at present for *ex ante* policy evaluation, except in Australia, Canada, Germany, the Netherlands, Switzerland and the UK.

Results are summarised in Table 3. It is clear that there is still considerable variation in approaches to policy appraisal, and countries with formal mandatory RIA are still a minority. Table 3 also highlights that there is no prescribed methodology to be specifically used in policy appraisal and evaluation. Many economic methods are being used across countries.

Nonetheless, OECD countries have tended to use RIA on a more regular basis over the last few years. By the end of 2000, 14 countries had comprehensive RIA programmes in place, and another 6 were using RIA for at least some regulations (OECD, 2002). According to Jacob *et al.* (2004), RIA implementation varies from 0% (in Italy) to 100% (in the UK). There is a high degree of implementation of RIA process in countries with a long tradition of conducting RIA and with strong policy commitment at high policy levels (for example in the US and the UK).

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[4] Although this survey does not specifically focus on environmental policy-making, it provides insights on current valuation practices.

Table 3. Current Use of Economic Evaluation Tools in OECD Countries

COUNTRIES	TYPE OF ANALYSIS	COST-BENEFIT ANALYSIS	DISCOUNT RATE
Australia	Regulatory impact statement	Yes for all cases	3%
Austria	Fiscal analysis	Yes in selected cases Benefits in all cases	No standardised rate
Canada	Socio-economic impact analysis	Yes for all cases	10% (real)
Czech Republic		Yes for all cases	To be determined soon
Denmark	Regulatory impact assessment	Yes for major construction projects	3%
European Commission	General impact analysis	No prescribed methodology	4% (real)
Finland	Regulatory impact assessment	No prescribed methodology	
France	General impact analysis	Yes for major regulations	4% for t<30 years then 2%
Germany	Regulatory impact assessment	Costs in all cases	4% (real)
Greece	Environmental impact assessment	No unified guidelines	
Hungary	Socio-economic analysis	Yes	Real interest rate
Iceland	Fiscal analysis	Yes	5%
Ireland	Qualitative regulation check	No	5% (real)
Italy	Regulatory impact assessment	No prescribed methodology	
Japan	Benefit test	Yes in selected cases	
Korea (South)	Regulatory impact assessment	Yes for major regulations	Equivalent to price increase rate
Luxembourg		Not employed	No discounting
Mexico	Regulatory impact assessment	Yes	
Netherlands	General impact assessment	Yes for major regulations	
New Zealand	Regulatory impact statement	Yes	5-7%
Norway	General impact assessment	Yes	2.5% (real)
Poland	Regulatory impact assessment	Yes for significant regulations	
Portugal	Fiscal analysis	Costs for major regulations	
Slovak Republic			5%
Spain			5%
Sweden	Cost analysis	Costs for all cases	4%
Switzerland	Regulatory impact assessment	Yes in selected cases	Varies between cantons
Turkey	Environmental impact assessment	No	Interest rate on debt finance
United Kingdom	Regulatory impact assessment	Yes for all cases	3.5% for t<30 years then declines
United States	Regulatory impact assessment	Yes for major regulations	3-7%

Source: OECD (2004a and 2006b).

From Table 3, it can also be seen that cost-benefit analysis (CBA) is unequally applied in policy evaluation work among OECD countries. Although the assessment of costs and benefits of the different policy options is often an important part of the RIA, it is not systematically made. In addition, its definition and use differ across countries: some countries cover all costs and benefits, while others cover only selected costs and benefits (*e.g.* health and environmental benefits, financial costs, etc.). Moreover, some countries estimate the costs and benefits of each regulatory proposal, while other countries estimate the costs and benefits for major regulations only (or for selected regulations). In general, CBA is mandatory if expected impacts are large (OECD, 2004a). For example, in Canada, all regulatory proposals must undergo a CBA when the estimated present value of cost is greater than C$50 million. In Korea, a regulation with potential annual costs of more than 10 billion Won must be subject to a CBA. In the US, major regulations with an anticipated economic impact of US$100 million or more[5] are required to quantify and monetise costs and benefits.

The practice of quantification also differs widely across countries. In some countries, such as the US, a detailed quantitative analysis is required for economically significant rules only: to the extent possible, all economic costs and benefits are quantified and monetised; when they cannot be quantified, a qualitative description is included. In other countries, such as the Netherlands, the analysis is rather more selective and more of the qualitative type, in particular when a broad range of impacts is considered (Jacobs *et al.*, 2004).

Despite having many positive aspects, RIA is not yet recommended in all OECD countries. Wider use of formal assessment processes (such as RIA) should facilitate not only more effective identification of the relevant policy options but also the achievement of these objectives at lower cost.

The OECD survey (OECD, 2004a) highlights various ways of increasing the *ex ante* policy assessment effort, including:

- Internal capacity-building: training of people responsible for policy evaluation/appraisal should help reduce resources allocated to policy evaluation. It should also increase the experience among units/division within the same departments and across departments, which would also reduce the costs associated with evaluation tasks.

- Stakeholder consultation: involving targeted stakeholders early in the assessment process allows for "buy-in" by the right stakeholders and increases the probability of affecting final decisions. Only relevant stakeholders should be involved in the policymaking process. Successful initiatives have been undertaken in Australia, Canada and the UK.

- Clear communication: in order to increase trust in both the approach and the results, communication to stakeholders and to the public should be transparent and done as early as possible. The involvement of industry and other stakeholders from the start of the appraisal process may also help avoid misperceptions of economic evaluation.

- Base *ex ante* assessments on previous ex post evaluations: in order to increase policy efficiency and effectiveness, *ex ante* assessments should draw upon conclusions and lessons learned from *ex post* evaluations of previous policies.

[5] The expected economic impact that triggers a CBA is not the net benefits but rather is calculated as the summed value of benefits and costs.

- Guidance document: clear guidance documents should be prepared and disseminated across departments to provide details on how to carry out high quality CBA (or CEA). These guidelines should be updated regularly according to the empirical evidence that is available.

In short, economic evaluation does not yet play a major role in policy evaluation in general. However, its use could become more important in specific areas, such as environment and health. These opportunities are examined in more detail in the following sections.

2.2. Economic evaluation in environmental policy-making

Many decision-making tools can be used to guide policy makers in the environmental field[6]. However, these tools are not substitutable and the relevance of a particular technique over another should be based on the nature of the issue to be addressed. This Section presents a brief overview of economic evaluation in environmental policy making.

Theoretical background

Over the years, various appraisal techniques have emerged in the environmental field. Because environmental economics is grounded in welfare economics, the tools used in environmental policy-making to compare different outcomes are defined according to welfare theory and therefore should satisfy the criterion from which the greatest benefit should be given to the greatest number. CBA has been developed according to these foundations and relies on the arguments of consumer sovereignty and maximisation of consumer's satisfaction (*utility*). When CBA is used for policy appraisal, it places a monetary value on all costs and benefits of the policy in question. As benefits and costs are expressed in the same unit (*i.e.* money), they are directly commensurable and easily comparable. If benefits are larger than costs, the policy is judged to be worth implementing. Hence, the policy is economically efficient and "socially" desirable, which means that its implementation will increase social welfare.

Arguably, the most difficult part of a CBA is to estimate the benefits of a policy. As the individual is assumed to be the best judge of his/her own welfare, individuals' preferences must be known in order to determine the impacts of a policy on their welfare. Preferences are either elicited through consumption choices and behaviour (*revealed preferences*) or through surveys (*stated preferences*)[7]. From these techniques, individuals' preferences are obtained by finding out the monetary amount an individual is willing to pay for an increase of his/her level of utility (*i.e.* a measure of the willingness to pay), or the compensation amount he/she is willing to accept for tolerating a decrease in his/her level of utility (*i.e.* a measure of the willingness to accept). The WTP is therefore considered as a measure of individuals' utilities. In order to determine aggregate social welfare, analysts sum individual WTP.

CBA is a very flexible approach. It provides a consistent framework for deciding when interventions are desirable. However, the assumptions underlying the CBA approach have been subject to many criticisms (see for example Sen, 1979). An alternative to CBA has therefore been proposed: the cost-effectiveness analysis (CEA).

CEA is a general approach, in which the benefits of policy (*consequences* or *effects*) are usually expressed in physical terms (*e.g.* number of deaths, number of cases of illness avoided, etc.)[8] instead of in

[6] A detailed presentation of these tools can be found in the Chapter 18 of OECD (2006a).

[7] A detailed presentation of both approaches can be found in OECD (2006a).

[8] CEA is quite distinct of cost-utility analysis – a special case of CEA where the benefits are expressed in terms of QALYs and which are presented in Section 2.3.

monetary terms, and costs are expressed in monetary terms. The objective of CEA is to choose the intervention that supplies a unit of effect at the lowest cost or alternatively, or that achieves the most effect per unit of cost (Dollar, Euro, etc.). However, in order to be able to compare two alternative treatments, treatment effects should be of the same kind across alternatives. For example, a CEA can compare different ways of diagnosing cancer or different treatments to reduce asthma episodes, but CEA cannot compare both effects at the same time (*i.e.* comparing treating asthma with diagnosing cancer). In environmental policy-making, CEA is mainly used to evaluate environmental health interventions (OECD, 2006a). A comparison of CEA and CBA is provided later.

Use of economic evaluation in environmental policy-making

The use of economic evaluation tools in environmental policy-making vary largely across OECD countries (OECD, 2004a). In some countries, environmental questions have been integrated in conventional RIA procedures, whereas other countries have developed separate environmental appraisal procedures (OECD, 2004a). For instance, in Australia and the UK, there have been recent attempts to broaden the scope of RIA to include environmental and other sustainability issues. Although the Netherlands have some experience with environmental impact assessment, it became part of the RIA process only recently in this country (OECD, 2004a).

At the European level, the European Commission (EC) has played a key role in supporting the use of economic decision-making tools when developing new regulations, especially in the environmental field. The EC has recently launched a new *Impact Assessment* procedure for all major policy proposals in order to improve the quality and coherence of its policies (EC, 2005)[9]. This approach provides information on the tradeoffs between the economic, social and environmental dimensions of forthcoming decisions. By allowing a full appraisal of the potential environmental costs and benefits of all major Commission proposals, as well as of the costs and benefits of specific environmental measures, this helps promote integration of environmental concerns in other policy areas.

Concerning the methodology prescribed for environmental policy appraisal, OECD countries do not apply a common approach. A survey made by Risk and Policy Analysts (RPA, 1998) examines the extent to which CBA and CEA are used in policymaking in 15 European countries and in Australia, Canada, Japan and the US. Overall, they found that 13 countries use CBA or CEA or both in environmental policymaking. More specifically, CBA is used in 9 countries as a part of environmental regulation while CEA is applied in 12 countries in regulatory assessment. In a few countries (*e.g.* Australia, Canada, the UK and the US), quantified analysis is required in policymaking, meaning that environmental and health impacts of environmental regulations have to be quantified and monetised as far as possible.

The RPA survey also highlights two facts. First, CBA is sometimes considered by policy-makers as a more thorough method of policy analysis than CEA. For this reason, CBA are more likely to be carried out than CEA, for example in Spain and Sweden, although CEA is used while CBA is not used at all in some countries (*e.g.* Japan and Denmark). Second, many OECD countries prefer to rely on less controversial methods, such as cost-minimisation analysis (whose objective is to determine the policy which minimises the costs of policy's objectives and which does not consider the benefits of the policy at all) or cost-compliance analysis (which focuses on the financial costs of a policy on the different industrial sectors), than CBA.

[9] The Communication *COM(2002)276 Final* from the Commission on impact assessment originally set up the framework. It is available at: http://eur-lex.europa.eu/LexUriServ/site/en/com/2002/com2002_0276en01.pdf

The environmental impact assessment (EIA) approach has also been developed in order to systematically evaluate the environmental impacts of a regulation[10]. EIA is therefore a useful input to the decision-making process, providing decision-makers with information about the environmental consequences of their actions, although it only considers the benefit side. Although the use of EIA and SEA in the development of policy for key sectors is rapidly evolving, there is no consensus yet on a standardised SEA methodology. This may explain the limited use of such methods in some countries. Moreover, because EIA and SEA only consider the environmental benefits of environmental policies (leaving aside health benefits), they can be considered as "partial" analysis and a more "complete" analysis (such as the one proposed by a CBA) would usually be preferable.

In some countries (*e.g.* the Netherlands and Finland) other evaluation tools, such as multi-criteria analysis (which quantifies both costs and benefits of a policy but does not put a monetary value on the benefits) and checklists are used, sometimes to supplement the information provided by the CBA. These tools are used in environmental policymaking in order to provide "quick indicators" of the potential impacts of a policy, but not to provide an indication of the relative significance of these impacts.

The evidence therefore tends to suggest that, even though there has been significant progress in the use of economic evaluation in policy appraisal, the use of such approaches is still quite limited. Two major obstacles to more systematic *ex ante* policy assessment have been identified:

- Limited resources: undertaking CBA or CEA is resource intensive (both in terms of time and money). Countries therefore tend to build upon assessments done for other contexts.

- Availability of data: data collection is a factor affecting the use of CBA and CEA. A great deal of reliable and available information is needed to carry out *ex ante* assessments. In the absence of reliable existing data, countries therefore again tend to transfer results from studies undertaken in other contexts (benefit transfer).

There is a clear trend towards the use of empirically based regulation and decision making. However, in general, the cost-benefit criterion (*i.e.* maximisation of net benefits) is not seen to be a single, sufficient basis for decisions. Other criteria are also of importance in policymaking, such as the distributional effects or the competitive impacts of a policy.

When CBA is applied in policy evaluation, the costs and benefits have to be quantified and monetised to the extent possible. To this end, governments often recommend specific values to be used for the monetisation of health impacts of environmental policies. Examples of such values when valuing the health impacts of air pollution (more specifically particulate matter (PM) pollution) are listed in Table 4, for different OECD countries.

[10] The EIA terminology usually refers to individual projects (*e.g.* creation of a motorway, an airport or a factory). When this assessment is applied to decision-making process (*e.g.* the appraisal of programmes and policies), it is generally referred to as "strategic environmental assessment" (SEA).

Table 4. Recommended values for PM-related health impacts (mean estimates in 2006 US$)[11]

Health impacts	EC (US$)	US (US $)	AUS (US$)	NZ (US$)	CAN (US $)
Mortality impacts					
Adult VSL	$1,786,640	$6,120,000	$2,643,150	$3,658,071	$5,902,989
Adult VOLY	$107,198		$129,375	$252,686	
Children VSL	$2,679,960				
Morbidity impacts					
Chronic bronchitis	$169,731	$331,000	$381,806		$381,235
Respiratory hospital admission	$1,787			$4,407	$2,829
Cardiovascular hospital admission	$1,787	$21,104		$5,876	$6,272
Consultation	$47				
Restricted activity day	$116	$106		$113	$74
Minor restricted activity day	$34	$48.48			$37
Use of respiratory medication	$1				
Asthma symptom day	$34				$37
Asthma attacks			293505		
Discount rate	4% (real)	3-7%	3%	5-7%	10% (real)
Source	AEA (2005a)	ABT Ass. (2005)	BTRE (2005)	Bicknell (2001)	Lang et al. (1996)

In order to have the most reliable and complete estimate of the value of a given health impact, economics recommend the use of WTP measures. They are indeed the only means to correctly integrate all types of effects: direct, indirect and intangible. However, this practice is not usually applied in OECD countries. As mentioned before, many criticisms have brought into question the reliability of the methods used to obtain WTP values (revealed and stated preferences techniques, and more particularly, the contingent valuation method (see for example Hausman, 1993)). As such, there is a general lack of trust in those values.

By contrast, cost-of-illness (COI) values are typically preferred over WTP values because there is less uncertainty concerning their calculation methods. Although COI values do not include the intangible effects of a disease such as pain and suffering, they are often recommended when valuing the health benefits of environmental policies. More governments likely recommend using COI values than WTP values.

Another issue when valuing the health impacts of environmental pollution is related to “latency”. Indeed, most environmental risks are characterised by a latency period (*i.e.* a delay between the exposure to a given environmental risk factor and the onset of ill-health). Analyses typically account for this lag by discounting the benefits over the latency period. Examples of discount rates currently being applied in environmental policymaking are presented in Tables 3 and 4 (see also OECD, 2006b). These data show that there is great variation across countries in discounting practices, with some countries applying constant

[11] National currencies have been converted into 2006 US dollars in using power purchasing parities. Source: OECD, http://www.oecd.org/dataoecd/61/56/1876133.xls.

rates and others applying hyperbolic discount rates (*e.g.* UK and France) in order to better account for the long-term nature of environmental effects.

2.3. Economic evaluation in health policy-making

Health economics consists of the application of economic tools, theories and concepts to the topics of health and health care. Health economics is mainly concerned with issues related to the allocation of (scarce) resources to improve survival, quality of life and equity. Various approaches have been developed to analyse and evaluate resource allocation in the health sector. More particularly, methods of economic evaluation have been developed to address the question of efficient resource allocation. The theoretical background and the use of economic evaluation in health policy-making are examined in more detail in the following Sections.

Theoretical background

A number of practical concerns (*e.g.* unrealistic assumptions, limited guidance for "real world" decisions) and ethical issues (*e.g.* placing a monetary value on the benefits of health care, taking into account the influence of income and/or ability to pay on policy choices) associated with conventional welfare economics have led to the rejection of this theory as a basis for economic evaluation in the health domain.

Indeed, the theoretical framework of economic evaluation in the health area is better described as an "extra-welfarist" approach. Sen was one of those who criticised welfarism and its notion of "social utility". Sen rejected the idea that all decisions, behaviours, or moral principles were driven by the notion of "utility". Sen qualified welfarism as "a limiting approach" because it rejected the relevance of non-utility information. Non-utilitarian motivations play an important role in decision-making, maybe more important than utility. An important set of non-utility information relates to individual "capabilities" (*i.e.* a person being able to do certain things). These were the bases of what would then be referred to as "extra-welfarism": by contrast with welfarism that bases individual welfare on the utility obtained from goods and services, extra-welfarism complements this approach by including other characteristics of people, such as moral principles, motivation, etc. Culyer (1989) and others have since promoted Sen's approach (Sen, 1979) and its application to the health area. As stated in Culyer (1989), health has been taken as the "*proximate maximand*". Health is not considered as a commodity. Rather, health is generating utility for individuals, directly (good health can be linked to well being) and indirectly (better health enables access to various things) (Culyer, 1989). The objective is therefore to maximise health (and not utility), subject to a budget constraint (*i.e.* limited resources), and the task consists in the measurement of changes in health (not changes in utility). The ideal outcome measure has to combine the two important dimensions of health care: length of life and quality of life.

In this paradigm, economic evaluation is perceived as a source of information that can help in the allocation of resources for medical care, as well as towards the development of new medical devices. All the costs related to each method of treatment are considered (and related to the consequences), in terms of increases in the length and/or quality of life. Cost-effectiveness therefore becomes the important criterion of selection. New and existing therapies and/or medical interventions/treatments are evaluated in terms of effects (or benefits) and costs.

However, in standard cost-effectiveness analysis (CEA), the effects of policy are not expressed in monetary terms, but rather in physical terms (*e.g.* number of deaths, number of asthma cases avoided, etc.), while costs are expressed in monetary terms.

CEA is a technique that allows for the comparison of the relative value of various clinical strategies. It consists in ranking the different alternative programmes according to their cost-effectiveness (CE) ratio. For example, if a new strategy is compared with current practice (the "low-cost alternative"), the calculation of the cost-effectiveness ratio (CE ratio) becomes:

$$CE\ ratio = \frac{\text{cost}_{new\ strategy} - \text{cost}_{current\ practice}}{\text{effect}_{new\ strategy} - \text{effect}_{current\ practice}}$$

The result might be considered as the "price" of the additional outcome purchased by switching from current practice to the new strategy. If the price is low enough, the new strategy is considered to be "cost-effective". The most cost-effective policy is the one with lowest cost per life saved or the lowest cost per case of illness averted.

A different concept for cost effectiveness evaluation is based on the net "health benefit" estimation[12]. The (incremental) net benefit approach has been developed to handle uncertainty in CEA and statistical problems associated with the estimation of incremental cost-effectiveness ratio. The net benefit framework starts from the premise that the cost-effectiveness ratio provides only partial information to decision-makers who also need to judge whether the additional effect is worth the additional cost. This approach provides a decision rule (associated with a measure of "net-benefit") that the new therapy should be implemented only if the net-benefits are positive.

Two types of net-benefit measures have been proposed:

- A net-monetary-benefit (NMB) measure which is calculated by subtracting the additional cost from the additional effect valued in dollars:

  $$NMB=\lambda.E-C.$$

  where λ is the maximum WTP per unit of health gain, E are the incremental effects (benefits) and C are the incremental costs. When the benefits are expressed in terms of QALY, then NMB is defined as the following:

  $$NMB=QALY.\ \lambda-C$$

- A net-health-benefit (NHB) measure which is calculated by subtracting the additional cost valued in effect units from the additional effect:

  $$NHB=E-C/\lambda.$$

The net benefit approach directly involves the WTP parameter into the cost effectiveness estimation. This measure has the advantage of including an estimate of the degree to which patients value treatment. However, it is necessary there to know the WTP of the patient, so this approach must also rely on revealed or stated preferences techniques (with their inherent problems). A subsequent problem can rise as QALYs and WTP are not generally based on the preferences of the same population, which means that they are also not consistent with each other.

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[12] For more details on the net benefit approach, see Hoch *et al.* (2002).

Although CEA is more broadly appealing to health economists than the traditional welfarist approach, there is a major problem with CEA: since benefits are expressed in physical terms, it is difficult to compare between the cost-effectiveness of treatments in different disease areas with different clinical outcome measures. In order to address this issue, cost-utility analysis (CUA) has been developed.

CUA is a special case of CEA where effects are expressed as quality-adjusted life years (QALYs). A QALY is an index measure combining quantity and quality of life, ranging from 0 (health equivalent to death) to 1 (perfect or excellent health). It allows for the estimation of the “cost per QALY gained”, which can be estimated for any treatment of any type of disease. In other words, CUA allows for the comparison of different interventions in different medical areas. CUA is therefore an essential tool for resource allocation, to help decide on expenditures and prioritise within the budget.

CUA and QALYs have received increased interest in the medical and health field because QALYs are perceived as being neutral and as considering outcomes equitably. Indeed, within the WTP approach, choices are heavily influenced by income and ability to pay. However, QALYs are also influenced by income but income is not explicitly accounted for in the CUA.

CUA is the most commonly used approach relative to the other techniques of evaluation in the health domain. A good example is the growing number of “league tables” of different health care interventions that have been created recently[13]. However, how are the cost-effectiveness ratios obtained from CUA actually used in health policy-making? The next section attempts to answer that question.

Use of economic evaluation in health policy-making

Many health economic studies have assessed the use of economic evaluation tools in health priority setting. The main result is that economic approaches to health priority-setting decisions seem to have a limited impact in practice. Indeed, according to Drummond (2004 and 2001), Hutubessy *et al.* (2003), and Hoffmann *et al.* (2000) among others, cost-effectiveness information has been mainly used in Australia (since 1992, to decide which pharmaceuticals should be reimbursed from public funds), in Canada (also used for drug reimbursement schemes), and in a few European countries (mainly in the UK and the Netherlands) in selected cases, and not on a systematic basis.

Beyond these examples, the actual use of CEA-based information to guide priority setting in health care remain rather limited. *A number of potential reasons may account for this situation*, including methodological and institutional barriers. Examples of methodological barriers could include the following:

- Different perspectives (perspective of the society, of health care providers, or of a third-party) or different elicitation procedures (visual analogue scale, standard gamble, time trade-off, etc.[14]) lead to systematically different valuation of the same health states (Hauck *et al.*, 2004; Edgar, 1997).

- It may be difficult to generalise the results to other settings (Hauck *et al.*, 2004; Drummond, 2001). Indeed, benefit transfer from one setting to another is not straightforward and should be applied with great caution. Many parameters have to be adjusted to account for heterogeneity between the two settings. In the context of health care, transfer of results from one country to

[13] See for example the web-based registry of published CUA developed at the Harvard school of Public Health. Access to the database: http://www.tufts-nemc.org/cearegistry/index.html.

[14] The different techniques of elicitation of health-related quality of life measures, such as QALYs, are presented in the Annex.

another is likely to be even more tendentious because health care systems, disease incidence and relative prices and costs are usually quite different.

- Many uncertainties undermine the estimation of the costs and benefits (consequences) (OECD, 2005; Hauck *et al.*, 2004; Drummond, 2004). As such, valuation of effectiveness parameters may not be reliable and lead to wrong decision-making.

- CEA and CUA may not take into account equity, fairness and distributive considerations in a satisfactory manner (Lawrence *et al.*, 2006; OECD, 2005; Drummond, 2004; Sassi, 2003; Donaldson *et al.*, 2002, Edgar, 1997). Although QALYs are thought to promote equity in allocating resources to those who would otherwise lose most, many studies suggest that QALYs discriminate against people for whom treatment would not lead to a large gain in QALYs. Moreover, in the QALY approach, saving one person having a prognosis of 5 years in good health is equivalent to saving five people having a prognosis of one year of good health, which may not be consistent with social welfare maximisation.

- CEA and CUA do not account for other important effects, such as the impacts on society, on the environment, or more generally on the economy (Lawrence *et al.*, 2006; Coast, 2004; Drummond, 2004; Donaldson *et al.*, 2002). As such, important decisions cannot rely solely on results from CEA or CUA.

- CEA and CUA do not address practical public finance concerns (Donaldson *et al.*, 2002; Drummond, 2001; Hoffmann *et al.*, 2000). Indeed, results from economic evaluations in health care hardly provide answers to the questions raised in priority setting, such as where will the additional resources needed to move from treatment A to treatment B come from? The approach applied in a CEA gives the impression that a more cost-effective treatment can replace a less cost-effective one at no additional (gross) cost (Donaldson *et al.*, 2002). Decision makers facing tight budget constraints may not be able to shift financial resources from one treatment to another.

- Issues related to time and timing may also undermine the reliability of the results obtained from a CEA or a CUA. First, it seems preferable to undertake an economic evaluation of a specific treatment before the widespread use of this treatment, otherwise measures can be biased. Indeed, once a treatment is used and widely applied, patients or medical doctors become used to it and may only highlight its positive sides if asked to evaluate its effectiveness (Hauck *et al.*, 2004). Second, empirical evidence suggests that experience and valuation of health states are not stable over time. More specifically, a given state may be considered to be of lower quality if it is expected to last for a longer period. However, a patient affected by a chronic illness for several years is likely to value this specific health state more than a person who has no experience with it (Nord, 1989).

In addition to the above methodological barriers, many institutional issues curb the use of economic evaluation in health care. The main important limitation is the limited knowledge about formal methodologies to be used in this context (*i.e.* mainly CEA and CUA). Many surveys (Coast, 2004; Drummond, 2001; Hoffmann *et al.*, 2000) have shown that decision-makers do not exactly know how these methods work, and therefore may not understand the practical relevance of the results provided. They also find that the concepts behind the QALY-based approaches are difficult to understand and to apply (Coast, 2004). As a consequence, decision makers do not generally trust the results provided by economic evaluations and health economic studies.

Nor is the objective of health decision makers solely to maximise the health output (Coast, 2004). Decisions in health care generally depend on a complex interplay of political, social and economic factors (Drummond, 2004; Donaldson *et al.*, 2002; Hoffmann *et al.*, 2000) [15]. Therefore, CEA and CUA approaches may be considered by policy makers as being somewhat disconnected from the real world.

Finally, some surveys tend to suggest that economic evaluation is not widely used in health policy making because of the low involvement of decision makers (Hoffmann *et al.*, 2000). A greater involvement of policy makers in health economic studies and economic evaluations would probably result in better understanding of the method and in a better trust and acceptance of the results provided by the study. The integration of policy makers early in the evaluation process could significantly impact the practical use of economic evaluation in priority setting.

In conclusion, although there is a growing interest in economic evaluation among health care decision makers, results from economic evaluations are not widely used at the moment. Economic considerations in health policy making are generally regarded as important and the level of acceptance is high (Drummond, 2004; Hoffmann *et al.*, 2000). However, methodological and institutional barriers limit the actual use of economic decision making tools in the health area. Improving the quality of the economic studies, educating high-level policy makers and medical students, as well as involving decision makers early in the evaluation process could increase the use of both CEA and CUA.

2.4. Elements of comparison

The objective of evaluation is to determine the policy option that will enable the achievement of predefined objectives (*e.g.* reducing air pollution by 30%, reduce mortality or morbidity, etc.). This policy option should be, as far as possible, cost-efficient; in other words, it is preferable that the net benefits of the policy be maximised; otherwise the policy would lead to potential losses in social welfare. Therefore, in order to determine the most efficient policy option to achieve one specific goal, costs and benefits of a given policy have to be quantified and compared.

Arguably, the most challenging step is the estimation of the benefits (most people can agree on the definition and estimation of costs). Benefit valuation is more problematic as it necessitates a comprehensive definition of what the benefits are and different measures of benefit can be applied, those measures being obtained by different means. CBA, CUA and CEA propose different approaches for obtaining different types of benefits. They should not be seen as replacements for one another, but complements. Comparing these three methodologies could help determine which approach is the most relevant for policy evaluation (at least in the environment and health contexts).

When considering *benefit* valuation, CBA seems the most complete approach. Indeed, it allows for the accounting of all types of benefits (environmental, health, social, economic, etc.) while CEA and CUA only focus on health benefits. Although health benefits can represent an important gain of environmental policies, environmental policies also have other impacts. Ignoring these other impacts is a serious flaw in CUA and CEA.

In addition, benefits and costs estimated in a CBA are commensurable and directly comparable, as they are both expressed in monetary terms. However, in a CEA, costs are evaluated (as in a CBA), but benefits are not monetised (but rather expressed in physical units) and thus not directly commensurable with costs. Similarly, a CUA will provide (for a given intervention) a monetary figure for the costs and a number of “QALY gained” for the benefits. As money and QALY gained are not comparable, a CUA will rank interventions according to their cost-per-QALY ratio. Therefore, in terms of commensurability

[15] The same is true for environmental policy, and the same critic can apply to CBA.

between costs and benefits, only CBA allows for a direct comparison of all the costs and benefits of a policy.

If we now focus only on *health* benefits, CBA allows for the comparison of all health benefits (whether in mortality or in morbidity terms). Similarly, CUA (which uses an index as an outcome measure (the QALY)), is able to compare between, for example, an intervention to treat asthma and another intervention intending to reduce headache. Therefore, CUA is also able to compare health benefits all together. However, CEA will not be able to compare the health benefits of different interventions, because the benefits are generally expressed in different units. The only exception could be when CEA considers interventions for reducing mortality and benefits are expressed in terms of death avoided. When comparing an intervention reducing mortality and another one reducing morbidity, only CBA and CUA can directly compare the benefits. Table 5 summarises these considerations.

Table 5. Commensurability in CBA, CUA and CEA

	CBA	CUA	CEA
Commensurability between benefits and costs	Yes	No	No
Commensurability between all health benefits	Yes	Yes	No (except mortality)
Commensurability between health benefits and other benefits	Yes	No	No

Another important advantage of CBA over CEA and CUA is that only CBA can identify the most cost-efficient policy option, and determine whether a given policy would be worthwhile and increase social welfare. CUA and CEA can only identify cost-effective interventions; they cannot say whether the intervention will increase social welfare. In addition, CEA and CUA cannot handle the question of efficient allocation of resources, while CBA allows for prioritisation of resource allocation. CEA does not easily account for distributive and fairness issues while CBA can do this. OECD (2006a) illustrates how distributional considerations can be integrated into a standard CBA in order to account for equity and fairness. Finally, it should be noted that WTP can be used to measure many different kinds of outcomes, whether they are related to health, the environment, or the economy.

However, CUA has two advantages over CBA. First, in general, a CBA is not commonly applied to compare different policy instruments at the same time (for example to determine the best instrument to tackle air pollution), whereas a CUA can compare different treatments simultaneously and determine which one is the most cost-effective. Second, a single CBA cannot determine the highest priority area, *i.e.* a CBA cannot say whether the government should prioritise actions to reduce air pollution, water pollution, or protection of biodiversity. In order to set priorities in one specific area (*e.g.* environment or health), many CBA have to be undertaken, which could be very costly. By contrast, in a CUA-based approach, a single analysis can determine the area(s) where interventions will be most cost-effective.

Given the many drawbacks of CEA and CUA, decisions based on these approaches may neither reflect society's nor decision makers' objectives, nor their health priorities. As such, the best (but still not perfect) approach seems to be CBA.

2.5. Conclusions

CBA is a powerful tool that can provide useful insights to policy makers relative to efficient resource allocation in any field of public intervention, which is not the case of alternative approaches (*e.g.* CEA and CUA). Such practice should be encouraged and more widespread across Ministries and Departments than is actually the case.

In many OECD countries, CBA is not applied even for major environmental or health regulations that can have important impacts on the environment, health or the economy. In particular, in environmental policy-making, decisions about what standards to apply or "acceptable levels of risk" are based upon decisions that do not involve the use of economic evaluation, but rather involve decisions based on scientific information provided by technical experts (*e.g.* epidemiologists, engineers, etc.). Even though this practice is not ideal, the situation may be even worse when considering children, given the great number of differences between adults and children and the reliance on adult-based studies. This issue is addressed in next section.

3. Children's health in environmental policymaking

3.1 Current situation

The objective of this Section is to discuss whether current environmental policy-making considers the specific vulnerability of children to environmental hazards. To support this discussion, a questionnaire was administered by the OECD Working Party on National Environmental Policy in 2006. This questionnaire reviewed the means by which environmental health impacts are incorporated in policy-making and policy evaluation. Based upon the responses received[16], it is clear that environmental health impacts of policies are not being systematically assessed by using economic evaluation tools (see previous discussion). Moreover, most countries do not account for differences in age within the population when undertaking policy evaluation, suggesting that children are usually being considered as adults in policy-making.

These findings raise two key issues. The first is that considering children as adults when evaluating an environmental policy may result in poor recommendations. Indeed, recent epidemiological studies (see Tamburlini, 2006) raised the risk differences between adults and children:

- exposure differences: children are exposed to high levels of risk more often than adults and they can thus experience a larger effective dose than adults; and,
- susceptibility differences: children's bodies are still developing and respond differently than adults to the same apparent levels of exposure.

Therefore, exposure to toxic substances may lead to totally different health outcomes, depending upon whether this exposure concerns a child or an adult, the age of the subject, etc. Because of these differences, children and adults should not be treated in the same way in policy-making.

Nevertheless, current environmental policies are typically based on adults' susceptibilities and responses to environmental hazards. Existing values used for monetisation of environment-related health impacts focus on adult populations. For instance, in OECD countries applying cost-benefit analysis and alternative approaches for policy evaluation, the VSL of a child is the same as that of an adult. The CBA of EC CAFE Programme is an exception to this (Table 6).

Table 6. Summary of mortality valuation data for the CBA of CAFE (2005€)

Mortality	Median values	Mean values
Infant mortality	€1,500,000/death	€4,000,000/death
Value of a statistical life	€980,000/death	€2,000,000/death
Value of a life year	€52,000/year	€120,000/year

Source: AEA Technology Environment (2005b)

[16] Twelve countries and the European Commission have completed the questionnaire to date.

Moreover, current environmental policies mostly use scenarios that often do not match well with environmental scenarios. For example, in some countries, transport-related values are adjusted on an ad-hoc basis to reflect an environmental context. As such, there is concern that the continued use of existing estimates from unrelated scenarios that do not take population and the context factors into account may result in misguided cost-benefit analyses, and then to misallocation of resources, especially when environmental policies with significant implications for children are under consideration.

The second underlying problem raised by the questionnaire is that child-specific values are rare, mainly because of the methodological issues associated with this type of valuation. Undertaking economic valuation in the context of children's health and the environment is indeed challenging. The first reason is the lack of data. Regarding epidemiological evidence, there is relatively weak evidence of a causal link between exposure to environmental pollution and mortality or morbidity impacts in children, except in the context of air pollution (Hunt, 2006a). For instance, it will be difficult to estimate the benefits of reducing children's exposure to pesticides, although their impacts on children's health are generally acknowledged. The scope of available dose-response functions specific to children is unfortunately narrower than that for adults. Using adult data may therefore be considered as a second-best alternative.

On the economic side, empirical evidence is also missing. Scapecchi (2006) and Hunt (2006b) provide reviews of the literature on children's health. Only a few studies deal with the valuation of benefits to children, while the literature on adults is quite extensive (see for example OECD, 2006a). This scarcity of economic values precludes an evaluation of the health impacts of existing environment-related health policies. It may also lead to inappropriate policy decisions. Finally, the paucity of research in this area results in limited empirical evidence to inform policy analysts on the most appropriate valuation method when considering children.

Examples of **obstacles** associated with implementing a survey to estimate the willingness to pay (WTP) to reduce environmental health risks to children could include[17]:

- *Third party elicitation*: the lack of well-defined preferences and budget constraint precludes asking children directly about their preferences. Third party elicitation (relying on parents) is therefore recommended, but it may be affected by altruism. It has also direct implications on the survey sampling, as only parents could be interviewed[18];

- *Latency*: the delay between exposure and the onset of illness or death (*latency*) that characterises many environmental health risks is a major concern for children, because of their particular vulnerability to environmental pollutants and their longer lifespan. As such, it is important to ensure that the issue of latency is well understood by the survey respondents when they have to state a value for an event that will occur in the future;

- *Low probabilities*: mortality risks for children are generally associated with very low probabilities. Given that the average respondent may not be familiar with the concept of probability, careful communication of risk and probabilities is necessary to ensure the reliability of estimates;

[17] For a more detailed description and discussion of the main issues associated with the valuation of environmental health risks to children, see OECD (2006c).

[18] Existing stated-preference studies valuing benefits to children's health focus on parents, and do not generally interview a representative sample of the population.

- *Familiarity with the valuation task*: unfamiliarity with this sort of decision-making, the cognitively demanding nature of the task, and uncertainty associated with future health events make it difficult to elicit preferences from parents;

- *Risk perception*: involuntariness of risk may have substantial impacts on the WTP for a given risk reduction. This makes context particularly important; and

- *Preferences for risk reductions for adults and children*: empirical evidence suggests that a programme protecting young adults is generally preferred over a programme that protects older people, because it delivers greater benefits due to the difference in time/age existing between these two populations (larger benefits for young adults given their larger expected lifespan). Differences in preferences may be even greater when the programme in question reduces risks to children.

Other practical aspects have to be handled during the design of WTP surveys. For instance, the type of benefit measure must be chosen. Indeed, two types of measures can be estimated: the value of a statistical life (VSL) or the value of a life year (VOLY). Arguments can justify the use of both approaches (see OECD, 2006a). The choice between the two possible outcomes will generally depend on the health outcome to be valued.

A decision has also to be made on the type of risk reduction to be valued. Two "goods" can be valued: mortality risk reductions or extensions in life expectancy. Most of the literature has previously focused on reducing mortality risks to adults. Some authors have argued that in certain environmental contexts it is preferable to value extensions in life expectancy (ExternE, 1999).

After the implementation of the survey and the procurement of benefit estimates, a sensitivity analysis (in which the parameters of the key variables vary) is highly recommended to be performed to ensure stable results. In particular, different discount rates have to be tested as the influence of this factor on the estimates is well known. The estimates may then have to be translated to be interpreted and used in policy-making.

Empirical studies have found **ways to overcome** some of the challenges listed above. For instance, previous work by Hammitt and Liu (2004) and more recently by Alberini and Chiabai (2007) has shown that with appropriate survey design, latency issues can be adequately addressed. In that work, people understood the difference between immediate and latent risks and were able to provide reliable estimates.

Concerning preference elicitation, most of previous studies relied on parents as it appears to be the most relevant approach. Various theoretical economic models indeed suggest that parents' choices are the appropriate proxies for children's preferences and constitute a reliable source of information (Viscusi *et al*., 1987). Even though the level of altruism is likely to remain a major concern, this approach has the advantage of asking the persons who are actually directly affected by the risk reduction and who have the interests of the child at heart.

Risk perception issues and difficulties in understanding low probabilities may be overcome by good communication at the beginning of the questionnaire. Respondents in a "stated preference" study cannot simply be told about their risks and about the risk reductions delivered by a policy or private product or action: the literature is in agreement that they must be educated using visual aids. In previous literature, mortality risks have been communicated to respondents using pie charts, risk ladders, grids of squares, and set of dots. Corso, Hammitt and Graham (2001) compare visual aids when the risks are very small. Under these circumstances, it may be useful to remind respondents of a comparable population of reference (for example, 1 in million means one person in a large city) or to aggregate small risks over a specified length of time. For example, if it is difficult to show a reduction of 1 in 10,000 a year, it might be easier to consider 10 years, and to show (1 in 10,000 times 10, which is 1 in 1000). This strategy was used successfully by Alberini *et al.* (2004) and Krupnick *et al.* (2002) in the Canada and the US, because these visual aids are generally appreciated by the respondents.

Regarding the valuation approach, both "stated" and "revealed preference" techniques have been applied in the past, to derive WTP to reduce health risks to children. However, the choice between the different approaches will highly depend upon the health outcome, the type of risk reduction and the benefit to be valued.

The lack of empirical evidence does not allow for the provision of precise recommendations. In order to provide methodological guidance, as well as empirical results, that would be useful to policy-making, design and reform, the OECD Environment Directorate is currently coordinating empirical surveys in three OECD Member countries (Italy, the Czech Republic and the United Kingdom). Preliminary results should be available by mid-2008[19].

In the meantime, existing values should be used in policy-making. Despite many drawbacks, most economic analyses will have to rely on adult values for children's health effects. The US EPA recommends that "*with few child health valuation studies available, analysts may need to rely on transferring adult benefits to children until more information becomes available*" (US EPA 2003, pp. 1-6). Further research, in particular child-specific epidemiologic and economic studies, has to be supported in order to design effective policies towards children.

3.2 Review of environmental policies and initiatives

The previous section suggests that economic evaluation of environmental policies does not fully account for children's specific vulnerability to environmental risks. One question therefore remains, that is whether the correct amount of protecting children from environmental health risks is being provided.

Recent literature suggests that common tort law may be helping to increase the protection of children from harmful substances, such as chemicals or pesticides. Indeed, many courts encourage the collection of valuable scientific data that can be used to protect children. For example, the US Supreme Court has condemned the negligent testing and design principles of an herbicide producer who did not assess the health impacts this product posed, in particular to children's health (Klass, 2005).

Regulatory legislation and initiatives targeted at children to reduce their exposure to adverse environmental risk factors are rather unequal among OECD countries. Many countries have national policies in place that include children and that pay attention to children's particular vulnerability to environmental hazards. Based upon WHO work (WHO, 2004 and 2005) and reviews of the secondary "grey" literature (*e.g.* Spady et al., 2006; Armstrong, 2006), this Section provides an overview of the legislation and policy initiatives on children's environmental health implemented in OECD countries. It focuses on selected environmental hazards that particularly affect children, *e.g.* air pollution, chemicals and hazardous substances, radiation, food safety, noise, water and soil pollution.

This overview is based upon limited evidence from different sources. A clear distinction has to be made between environmental policies explicitly targeted at children's health (*e.g.* pesticides regulations), policies targeted at the general population, but where exposure and/or susceptibility of children is particularly acute, and policies targeted at children which cannot be considered as being specifically

[19] For more information on the VERHI project, visit http://www.oecd.org/env/social/envhealth/verhi.

environmental (*e.g.* noise-related policies). The review also considers other initiatives that have been undertaken at the national and supra-national level.

Air quality

Most laws that regulate air quality in OECD countries do not explicitly consider children. Although air quality legislation has been implemented in Europe, Japan, Korea, Australia, New Zealand, Mexico and Canada, none of these laws and regulations explicitly concerns air pollution and its impacts on children's health, or children special needs. Recently, however, some European Commission (EC) Directives, such as the *Air Quality Framework Directive* (96/62/EC) or CAFE, have singled out children and the resulting effects pollutants have on this vulnerable population group, but none of the EC Directives explicitly mentions children, even as a susceptible population.

In the US, the situation is different as children are explicitly included as part of a "vulnerable" population in the *Clean Air Act*. In addition, the recent revisions (December 2005) to the *National Ambient Air Quality Standards* specifically mention a number of sub-populations as potentially susceptible to health effects as a result of particulate matter (PM) exposure, including people with existing heart and lung diseases, older adults and children.

Similarly, in the *Impact Statement for the National Environment Protection (Air Toxics) Measure Guidelines* prepared by the Australian government, health risk assessment is explicitly stated as having to take into account particularly vulnerable populations, such as children and the elderly. However, the *National Environment Protection (Ambient Air Quality) Measure* does not consider vulnerable populations at all.

The unique child-oriented air quality law or regulation from an OECD country dealing with air quality is state of California's Bill 25 which imposes specific requirements "*relating to the protection of infants and children from environmental health hazards*".

Chemicals and other dangerous substances

Chemicals and hazardous substances

Although chemicals can severely affect children's health, many laws or regulations introduced in most OECD countries do not explicitly consider children. For instance, although the new EU chemical policy on *Registration, Evaluation, Approval and restrictions of Chemical Substances* (REACH) takes into consideration the assessment of human health, it does not specifically set children aside as a vulnerable population.

Interventions introduced in North America generally better take into account children's susceptibility to exposure to chemical substances. For instance, the US *Kid Safe Chemicals Act* (2005) limits children's food-related exposures to lead, mercury and other chemical hazards (excluding pesticide residues). This Act is an extension of the *Toxic Substances Control Act,* but proposes a totally different process for evaluating chemical safety – actually quite similar to that of REACH. The KSCA would require manufacturers to provide sufficient evidence of child health and environmental safety before being allowed to distribute a chemical in a consumer product. If no data on safety and health exists which can be evaluated, the product containing the chemical is not allowed to be produced or marketed.

Also in the US, the *Draft Final Guidelines for Carcinogen Risk Assessment* and the *Supplemental Guidance for Assessing Cancer Susceptibility from Early-life Exposure to Carcinogens* (initiated in 2003) call for consideration of susceptible populations and/or life stages. More specifically, the development of these two documents raises and requires the explicit consideration of childhood cancer risks. In addition, a pilot programme was implemented (1999) to research some chemical toxicity issues as they relate to children: the Voluntary's Children Chemical Evaluation Programme (VCCEP)[20].

In Canada, some valuable research has been conducted in the area of children's environmental health, in particular on persistent organic pollutants (POP) contamination and the effects of contaminants in the Great Lakes. In addition, the Canadian *Hazardous Products Act* (1969, revised in 1985) explicitly accounts for children's needs and requires that detailed attention must be paid to several products directly reated to children, *e.g.* pacifiers, toys, nipples on feeding bottles, etc.

Although Japanese policies on the use of chemical substances intend to protect the whole population, the Tokyo Metropolitan Government has been particularly proactive in creating guidelines in 2002 to protect children's health from exposure to chemicals. Specific environmental risk factors were addressed according to the level of children exposure and the importance of their health effects: paint, indoor air, plant pesticide spraying and food contamination. Campaign initiatives were also developed in order to raise community's awareness of the potentially harmful effects of chemicals on children's health.

Pesticides

Children's health can be affected by pesticides according to three modes of exposure: industrial use (crop spraying), pest management (in schools and other public areas), and residues in food (particularly fruits and vegetables). This Section summarises public interventions related to industrial use and pest management (*e.g.* Integrated Pest Management programmes). Pesticide residues in food are considered later (in the section on food safety).

Concerning exposure to pesticide from industrial use and pest management, only few laws explicitly consider children. For instance, in the US, the *Federal Insecticide, Fungicide, and Rodenticide Act* (1947, amended in 2004) mentions children in a section dealing with pesticide packaging but it does not consider children's needs directly.

As another example, some Canadian municipalities have developed initiatives to protect children's health from pesticides. Some school boards have therefore implemented policies to reduce chemical exposure of children, *e.g.* pest control programmes based on integrated pest management principles. The *Pest Control Products Act* was recently revised to require any new pesticides (and those needing re-evaluation) to be assessed according to new, more child-protective, approaches. It also suggested applying extra safety factors to protect children. A new amendment to the Pesticide Act will explicitly include children and pregnant women as particularly vulnerable populations.

In other OECD countries such as Australia, Japan, Korea, New Zealand and the EU, children's specific vulnerability to pesticides is not explicitly considered in existing legislation.

Heavy metals

1) Lead

Lead poisoning is particularly dramatic when it affects children, because it can result in severe neurodevelopmental impairments. Lead contained in ambient air was totally phased out in the 1980s in Europe and the US by effective gasoline and fuel policies. However, lead poisoning continues to pose a major risk to children's health through ingestion of lead-containing material such as paint, soil and toys.

[20] More information on the VCCEP can be found at: http://www.epa.gov/HPV/vccep/

Lead-reducing legislation greatly varies across OECD countries. For instance, in the US, different laws and regulations are currently in place to reduce children's and the general population's exposure to lead: the *Lead-based Paint Poisoning Prevention Act* (1970), the *Lead Contamination Control Act* (1988), the *Safe Drinking Water Act* (1976, amended in 1986), the *Residential Lead-based Paint Hazard Reduction Act* (1992), the *Toxic Substances Control Act* (1976). *Executive Order 13045* and the *Toxic Substances Control Act* have highlighted the specific vulnerability of certain groups of children at higher risk for lead poisoning (the poor and those living in deteriorating housing). US legislation to reduce children's exposure to lead supports research to identify the various sources of lead contamination and requires to account for the findings from this research in a programme to reduce environmental health risks to children.

By contrast, European and Canadian lead legislation is less extensive than in the US. Although there are a number of EU Directives regulating lead, these are not child-specific (with the exception of Directive 92/85/EEC related to occupational health which specifically mentions pregnant women). In Canada, lead is no longer found in paint and gasoline. This has clear positive health impacts on children and the general population. However, the Canadian legislation is not explicitly targeted at children.

2) Mercury

Because of its toxicity to developing cells, methyl mercury contaminating fish or shellfish and inorganic mercury are pollutants of particular concern for very young children and women of child-bearing age.

However, for the most part, existing mercury legislation is not directly targeted at children, but considers only the general population. For example, some regulations limit the amounts of permissible mercury emissions from power plants, and there are various European and US laws that limit the amount of, or ban, mercury found in toys, batteries, mercury switches, and mercury contained in fever thermometers.

One exception is the *Utility Mercury Reductions Rule* initiated in 2004 in the US which intends to reduce mercury emissions from oil- and coal-fired electric utility steam generating units. At the EU level, restrictions on the marketing of certain measuring devices containing mercury (COM/2006/69 final) pertain specifically to children's environmental health, but do not explicitly mention children in the text.

The main mercury legislation that focuses on the needs of children and pregnant women deals with food safety. For example, the European Food Safety Authority provides recommendations for the reduced consumption of fatty fish to women of child-bearing age, pregnant and breastfeeding women and children. Similar guidelines have been published in other OECD countries. These generally provide specific advice for pregnant women and young children, such as Japanese, Australian, and New Zealand guidelines which particularly emphasise mercury in food fish and provide a cautionary statement about reduced fish consumption during pregnancy (http://www.foodstandards.gov.au/foodmatters/mercuryinfish.cfm).

Toy safety

In the last two decades, many countries have introduced laws and regulations dealing with the chemical risks posed by children's toys. At the EU level, Directive 88/3/378 (1988) was the first to set out limit values for harmful substances contained in toys, such as arsenic, mercury, chromium, cadmium, lead, etc. Directive 1998/485/EC (1998) makes explicit reference to child toys and child care goods and requires that children should be protected from the harmful effects of chemicals listed in Directive 88/3/378. Recently, Directive 2005/84/EC amended Council Directive 76/769/EC, placing a permanent restriction on the marketing and use of phthalates in toys and childcare articles. It also requires affixing a warning notice to soft PVC toys intended for children under three years of age which could be put in the mouth. This

Directive was transposed into national legislation on 11 July 2006 and came into operation on 16 January 2007.

In the US, the *Hazardous Substances Act* and its amendments, notably the *Child Protection and Toy Safety Act* (1969) and the *Small Part Standards* (1980), became the primary vehicle to protect children from dangerous substances and toys. These laws and regulations have since been supplemented by the *Consumer Product Safety Act* (1972). Federal toy safety regulations appear in the Code of Federal Regulations, Title 16-Commercial Practices, also known as 16 CFR.

In Canada, safety requirements for toys, equipment and other products for use by a child in learning or play are specified in the *Hazardous Products Act* and the associated *Hazardous Products (Toys) Regulation*s. Restricted toys can only be advertised, sold or imported if they meet specific safety requirements, as defined in the *Hazardous Products (Toys) Regulations*. In 2005, the *Children's Jewellery Regulation*s came into force, allowing the import, advertisement or sale of jewellery for children less than 15 years of age only if the jewellery does not contain more than 600 mg/kg total lead and 90 mg/kg migratable lead.

Other OECD countries have product safety regulations which set limit values for some chemical and harmful substances in a variety of products. However, laws that specifically deal with the chemical risks that toys can pose to children are rare.

<u>Child-resistant packaging of chemical and toxic substances</u>

Child-resistant packaging and labelling regulations were introduced in most OECD countries to protect children from acute exposure to harmful chemical and toxic substances which generally results in poisoning. For instance, in Europe, many Directives on packaging propose standards for constructing lids and caps that could not be easily opened by children. In the US, the *Poison Prevention Packaging Act* (1970) deals with childproofing containers. Similarly, the *Hazardous Substances (Packaging) Regulations* (2001) introduced in New Zealand specifically refer to child resistant packaging, and the *Medicines Regulations* (1984) requires the use of strip or blister packaging which is "*reasonably resistant to young children*" for medicines such as paracetamol, aspirin, and antidepressants. The Australian *Therapeutic Goods Act* (1989) was amended in 2004 to broaden the list of therapeutic goods that must provide child resistant containers.

In Canada, requirements for child-resistant packaging for chemical products began in 1973, when the *Hazardous Products (Hazardous Substances) Regulations* (re-named the *Consumer Chemicals and Containers Regulations* in 1988) made under the *Hazardous Products Act* were amended to introduce such packaging for certain corrosive drain cleaners and liquid wood furniture polishes and cleaners. Over the years, the legislation was amended to expand the use of child-resistant packaging to other chemical and toxic substances, such as cyanoacrylate adhesives, turpentine, pine oil, methyl alcohol, petroleum distillates and certain other products containing sodium peroxide.

Radiation

Radiation can be defined as either ionising or non-ionising. Ionising radiations mainly takes the form of cosmic rays, X-rays or radon. Non-ionising radiations come from several sources, such as UV radiation, radio frequency, electromagnetic fields (EMF) from power lines and microwaves. In the rest of this section, only laws and regulations related to non-ionising radiations are considered.

<u>UV radiation</u>

To limit high doses of UV radiation, most OECD countries have prepared guidelines to protect children. For example, national policies on UV exposure prevention have been put in place in the Czech Republic, Spain, Finland and France. Spain and France specifically forbid the use of commercial solariums for children.

Education campaigns have also been used to help teach children behavioural choices that can reduce environmental health risks. UV radiation exposure is a major issue in some countries, in particular in Australia and New Zealand. In New Zealand, the Ministry of Health established in the 1980s a UV radiation programme which established a nationwide network of solar UV radiometers in major population centres to collect UV levels continuously during daylight hours over the summer months.

Another example was the implementation in 2004 of the *Sun Smart Programme* of the anti-cancer Council of Victoria, Australia, which made significant achievements. It raised awareness of skin cancer issues, educated parents on the dangers of UV radiation and encouraged a behavioural change towards sun-related lifestyle. Modelled on *Sun Smart*, the US EPA *Sun Wise Schools Programme* educates children to protect themselves from excessive exposure to ultraviolet radiation.

In Canada, Health Canada shares a joint initiative with Environment Canada called the *UV Index and Sun Protection Programme*. This Programme educates students about the UV index and how to use it as a tool to minimize the risk to their health from solar ultraviolet radiation. This programme provides kits for parents and teachers (*UV Index Resource Kits*). In addition, Canada has phased out the production of all ozone-depleting chemicals, and has taken steps to control the use of ozone-depleting substances through regulations that are part of the *Canadian Environmental Protection Act* (CEPA).

Electromagnetic fields

Most OECD countries have no legislation on electromagnetic fields (EMF) and no guidelines directed to the public are available. Two pieces of legislation should however be highlighted. At the EU level, Recommendation 1999/519/EC limited the exposure of the general public to electromagnetic fields (0 Hz - 300 GHz). The Recommendation set out a system of basic restrictions and reference levels for overall public exposure, given that it is the task of the Member States to ensure that adequate health protection measures are taken. Although the Recommendation invokes the precautionary principle, children are mentioned only peripherally.

More recently, the European Commission asked the competent scientific committee to update the existing scientific advice on health risks associated with the exposure to electromagnetic fields in the light of new evidence on exposure and research on health effects. The preliminary report was finalised by November 2006. Children specific sensitivity to radio frequency fields is explicitly mentioned, although no epidemiological evidence to date.

Food safety

One important objective of food safety regulations is to protect public health. Although children may be severely affected by food contaminants, policies implemented in most OECD countries do not account for children specific vulnerability to food risks.

By contrast, several EU laws and regulations deal specifically with pesticide contamination of infant and baby foods and formulae. These Directives show a greater consideration of children's needs for protection from harmful pesticide residues in food and food products. For example, Directives 96/5/EC, 98/36/EC, and 2003/13/EC establish nutritional requirements and pesticide standards for cereal-based foods and baby foods for infants and young children. Infant formula criteria, set by the EU Directive 2004/43/EC, restrict the use of particular toxins and pesticides in food for infants and young children.

More recently, the Directive 396/2005 recognises children's vulnerabilities to harmful environmental exposures, especially combined exposures and the range of health impacts that may result. It therefore required the consideration of vulnerable groups, such as children and the unborn, when setting standards or thresholds.

Unlike similar regulations introduced in most OECD countries, the US *Food Quality Protection Act* (FQPA, 1996) established provisions requiring the considerations of children in setting health standards. The FQPA amended the *Federal Insecticide, Fungicide, and Rodenticide Act* (FIFRA) and the *Federal Food Drug, and Cosmetic Act* by fundamentally changing the way EPA regulates pesticides. Some of the major requirements included stricter safety standards, especially for infants and children, and a complete reassessment of all existing pesticide tolerances. The FQPA explicitly considers vulnerable populations like children than most other pesticide legislation.

Analogously, the Canadian *Food and Drugs Act* (1985) makes direct reference to considerations relating to children's health and safety. Although the focus of this legislation is on food directly, some regulatory details also address children's issues in relation to product rather than environmental safety, child-resistant packaging for example.

Noise

Legislation has been introduced in several OECD countries to avoid, prevent or reduce the harmful effects (including annoyance) of environmental noise. However, only a few laws and regulations addressing the issue of noise apply to children. In the US and the EU, some laws regulate the noise level in workplaces, and mention children as a group that should not be exposed to noise. National initiatives implemented in Austria, the Czech Republic, Finland and Hungary should particularly be emphasised because they include special levels for children exposure in residential areas, playgrounds, schools and nurseries.

Drinking water regulations

Laws and regulations ensuring a safe drinking water have been introduced in all OECD countries for a long time in order to protect consumers' health. Despite the high vulnerability of children to unsafe drinking water, most of the sanitation and drinking water legislation passed in OECD countries does not directly mention children or consider them as a vulnerable population.

The only exception is the US *Safe Drinking Water Act* (1974) and further amendments (in 1996) which required children to be considered in setting health and safety standards.

Soil quality

Different policies are contributing to soil protection given the interface between the earth, air and water. But to ensure an adequate level of safety, soil protection or soil quality laws and regulations have been introduced in many OECD countries. Examples include the *Comprehensive Emergency Response Compensation and Liability Act* (CERCLA) introduced in 1980 in the US, the *Canadian Soil Quality Guidelines* (1996), the EU *Thematic Strategy for Soil Protection* nor the Framework Directive (COM(2006)232), New Zealand *Soil Conservation and Rivers Control Act* (1941), and the Korea, *Soil Environment Conservation Act* (1995, amended in 2003). However, none of these makes explicit mention of children, or of their specific exposures and health impacts to soil pollution.

One exception is related to the definition of environmental quality standards for dioxin in soil in Japan. When the Ministry of the Environment set these standards, it used the standard for infants' amount of food digested as a default value.

3.3 Other initiatives

This Section reviews other initiatives implemented at the national or supra-national levels, relating to regulation or to other types of public activities (*e.g.* research-related programmes). General legislation requiring the consideration of children's specific vulnerability to environmental hazards when designing laws and regulations is presented. Legislation at the national and supra-national levels is then considered. Other types of initiatives undertaken by OECD countries to increase the knowledge on children's health and the environment close the Section.

National general legislation

US

Two US laws have had a major influence on the way children's environmental health is considered in policy-making. The first one is the *Executive Order 13045* (EO 13045), according to which federal agencies must (1) make it a high priority to identify and assess environmental health risks and safety risks that may disproportionately affect children; and (2) ensure that policies, programmes, activities and standards address disproportionate risks to children that result from environmental health risks or safety risks. EO 13045 created a new dynamic in requiring that children's vulnerabilities be considered when designing policies. Another consequence of EO 13045 was the creation of the Interagency Forum on Child and Family Statistics that produces an annual compendium of the most important indicators of US children's well being (http://www.childstats.gov/).

The second important piece of US legislation explicitly mentioning children is the Maryland House Bill 313 which stresses the need to assess the impacts that State's policies, programmes and activities may have on children's health. Bill 313 raised awareness (for example in improving public education around children's environmental health) and developed key related initiatives, such as a survey of mercury in schools, an analysis of the State's infrastructures for protecting children's environmental health, and preparing a report providing indicators of children's environmental health, etc.

These two laws call for the consideration of children's unique characteristics in policy-making and, on the basis of their respective provisions, should result in increased protection of children from environmental hazards.

Australia

The *National Environmental Health Strategy* (http://enhealth.nphp.gov.au/strategy/nehs/index.htm) recognises the specific vulnerability of children, and explicitly recommends that "*All governments and agencies should take into account children's special vulnerability to environmental hazards when developing and reviewing their Environmental Health Programmes*".

Mexican Environmental Health Action Programme (2001-2006)

Quantitative environmental health targets were established under the 2001-2006 *Environmental Health Action Programme*. These include: 30% reduction of respiratory illnesses due to exposure to atmospheric pollution and 60% reduction of those due to children's exposure to indoor pollution; 15% reduction of average population exposure to atmospheric pollutants; 70% reduction of incidence of high child blood lead levels; and a guarantee of access to safe drinking water by 70% of the population. For some of these targets, however, baselines have yet to be established.

Korean Environmental Health Plan (2006-2015)

The Korean *Environmental Health Plan* is a 10 year comprehensive plan whose objective is to halve the population exposed to environmental pollution by 2015. Part of this *Plan* addresses the protection of children from environmental pollution. Examples of actions and measures undertaken could include: conducting health impact assessment of infants and children; measuring mercury concentrations in fish and shellfish to provide advisory for daily maximum intake for children and others; regulating or banning hazardous products contained in children's goods; and measuring chemical substances in playgrounds.

Supra-national initiatives

EC initiatives

The major EC initiative focusing on the protection of children (and other vulnerable groups as well) is the SCALE initiative, *i.e.* the *European Environment and Health Strategy* (COM(2003)338 FINAL). This *Strategy* proposes a framework to ensure the protection of the society as a whole, and more specifically of vulnerable groups (children, pregnant women, the elderly, etc.). The objective of the *Strategy* is to better understand environmental hazards, in order to design efficient policies to respond to this emerging issue. The first phase of the *Strategy* focuses on priority adverse health effects, such as childhood respiratory diseases, asthma and allergies, and childhood cancers. In addition, the European Research Area has been established to support research on environmental health.

In addition to environmental legislation, the EC action programme and legislation on health has generated a new dynamics in health policy. The programme on public health related to 2003-2008, as well as initiatives on food safety, health impact assessment and radiation protection, complete the EC picture.

World Health Organisation initiatives

The *Children's Environmental Health Action Plan for Europe* (CEHAPE) was adopted at the Fourth Ministerial Conference on Environment and Health (2004) on "The future for our children"[21]. It highlights the main commitments on children's environmental health and proposes actions to reduce or eliminate the exposure of children to environmental risk factors that are particularly relevant for but not sufficiently addressed, in general environmental policies. It focuses on four regional priority goals: safe water and adequate sanitation, injury protection and adequate physical activity, clean outdoor and indoor air, and reduction of chemicals in the environment. Fifteen environmental risk hazards are considered, including air pollution, water supply and sanitation, food contamination, and UV radiation.

In order to help countries identify priority actions, various implementation tools are proposed. These include criteria to set priorities (*e.g.* economic criteria, the number of children at risk, the nature and severity of the health effect), appropriate mechanisms and institutional frameworks (*e.g.* the creation of a task force to involve the stakeholders at all levels of the government) to ensure the implementation of CEHAP at the national level. Also discussed are information, education and communication tools to change the knowledge base, the attitudes and beliefs of individuals about environmental health and risks faced by children.

Children's environmental health indicators are also an essential tool to monitor the status of children's health (compared to their environment). These indicators measure the improvement of related health outcomes, the reduction of environmental exposure and the implementation of child-specific environmental policies.

[21] For more details, see http://www.euro.who.int/childhealthenv/policy/20020724_2.

National CEHAPs have already been implemented or planned to be soon implemented in several European countries, such as Austria, Belgium, Finland (in preparation), Norway (in preparation), Sweden (proposal), Turkey (under approval) and the UK (in preparation). In some countries, *National Environmental Health Action Plans* (NEHAP) are planned to be soon revised to include a special section on children, as been done in France and Germany, where children have been included as "vulnerable populations". This will also be the case for Greece, Hungary, Italy, the Netherlands, Poland and Slovakia.

A series of case studies was done by the World Health Organisation (WHO, 2004) on the planning and the implementation of environment-related actions to improve children's health. Different risk factors were covered, such as indoor and outdoor air pollution, unsafe water and inadequate sanitation, food contamination, and hazardous chemicals. Several associated health effects were also addressed: asthma and allergies, injuries, respiratory diseases, neurodevelopmental disorders, cancer and waterborne diseases. These case studies highlighted children-related areas of concern where actions have been taken to reduce children exposure to harmful environmental risk factors. This information will soon be integrated into a web-based database on actions and policies related to children's environmental health.

Research-related initiatives

Many countries have implemented other types of initiatives (mainly research-related) to better understand the relationship between the environment and children's health. These programmes may in the end provide relevant inputs for more sound (and evidence-based) policy-making.

One example is children's longitudinal cohort studies. Such studies are being implemented, for instance in Australia, France, the US and more recently, in Germany, to determine how environmental factors affect children's health and development. Children are followed from birth until they are legally considered as adults (18-20 years old). Similarly, in Japan, a cohort of school-aged children and a case control study of infants have been carried out to establish the relationship between local pollution from diesel motor vehicles and respiratory ailments.

A second example is the increasing development of indicators to evaluate the state of children's environmental health. These indicators are being developed to inform and help policy-makers appreciate the effectiveness of current environment and health policies planned to reduce adverse health effects, thereby facilitating priority setting. So far, only a few sets of children's environmental health indicators are available, due to the lack of suitable data. These include the Child Health Indicators of Life and Development project and the SCALE initiative, both launched by the European Commission; the Children's Health and the Environment Indicators in North America, undertaken by the Commission for Environmental Co-operation; and the Global Initiative on Children's Environmental Health Indicators, involving many countries and international organisations, led by the WHO[22].

3.4 Conclusions

The above considerations suggest that, despite a large number of policies and initiatives undertaken in OECD countries to protect children's health from environmental hazards, the situation can still be improved. Most environmental legislation does not ultimately account for children's specific susceptibility to environmental hazards. Several pieces of legislation on chemicals consider children in the context of requirements for warning messages or child-resistant caps on containers used to store toxic substances. Children are sometimes mentioned in regulations and laws as a "susceptible population" among the elderly and pregnant or breast-feeding women. However, children's needs are not generally regarded as the key

[22] More details on the Global Initiative on Children's Environmental Health Indicators can be found in OECD (2004b).

driver of policy-making. Three noteworthy exceptions are the *Food Quality Protection Act,* the *Safe Drinking Water Act* and the proposed *Kid Safe Chemical Act* initiated in the US in which provisions require explicit consideration of children in setting health standards.

Several reasons could explain the absence of child-specific environmental health legislation. One predominant reason may be assumptions made about how current environmental health policies targeting the general population actually protect children. Indeed, in many countries, children's vulnerability to a given environmental hazard is considered to be similar to that of adults. The lack of methodological and regulatory guidance may also be invoked. There is in general a lack of awareness of the importance to recognise children's unique characteristics in policy-making.

Further measures should be implemented to better protect children from adverse health effects of the environment. As these policy tools have a direct influence on children, they should be reviewed. In addition, *ex post* evaluations of current policies should be more systematically developed in order to determine their efficiency and how they can better integrate societal concerns for children.

4. Summary

Empirical evidence suggests that the use of economic evaluation tools in environmental policymaking is still limited and unequal across OECD countries. Member countries do not generally have the same requirements and do not recommend a common approach. For instance, CBA is used for environmental policymaking purposes in a minority of countries. Only a few countries (Australia, Canada, the UK and the US) require quantified and monetised (as far as possible) impact analysis. On the health side, methodological and institutional barriers limit the use of economic decision making tools in priority setting. Although CBA and CEA are not widely applied, these tools are considered nevertheless important in many countries.

Concerning European countries, many regulations derive from EC Directives. The EC is clearly a major "driver" toward increased use of economic evaluation tools, and more specifically of CBA, in environmental policymaking. European countries can often rely on EC analyses and do not need to undertake their own CBA or CEA studies when transposing EC directives into national policies. Countries then have to find the most cost-effective way to achieve the objectives defined by the EC.

One of the main objectives of environmental policies is to reduce adverse health effects of environmental degradation. This goal is complementary to the main objective of health policy, which is the improvement of the health of the population. However, most national environmental policies focus on monitoring the environmental situation. Although such monitoring is of high importance, it should not concentrate all resources. Estimating the health impacts of environmental policies is, as shown above, of high relevance for policy-making and should therefore be explicitly undertaken as well.

In addition, environmental health policies are not developed and implemented equally among OECD countries. Some countries (*e.g.* Australia and the US) have been more proactive than others. Thanks to initiates from the EC and the WHO in this area (respectively the European Environment and Health Strategy and the NEHAPE initiative), actions and plans related to environmental health are also emerging.

A key area in which regulatory intervention is very limited relates to children's health and the environment. The actual legislation undertaken in most OECD countries does not explicitly mention children and their specific vulnerability to environmental risk factors. Some laws that specifically target children do not treat the specific issues that distinguish children from adults.

However, this caveat has been recognised, and many countries are now actively engaged in research and information gathering, primarily focusing on children's health and the environment. It is also anticipated that current reviews of major regulations would result in greater consideration of susceptible populations, in particular children. Following the US *Executive Order 13045* and the *Australian Environmental Health Strategy* (Commonwealth Department of Health and Aged Care, 1999), the special vulnerability of children to environmental risk factors is being more explicitly mentioned and taken into account when developing and reviewing environmental health policies and programmes.

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